



Annexes - Shadow Prices Handbook

Report

Delft, March 2010

Author(s):

Sander de Bruyn
Marisa Korteland
Agnieszka Markowska
Marc Davidson
Femke de Jong
Mart Bles
Maartje Sevenster



Publication Data

Bibliographical data:

Sander de Bruyn, Marisa Korteland, Agnieszka Markowska, Marc Davidson, Femke de Jong,
Mart Bles, Maartje Sevenster
Annexes - Shadow Prices Handbook
Delft, CE Delft, March 2010

Environment / Effects / Costs / Prices / Economy / Analysis / Measure / Method

Publication number: 10.7788.25b ENG

CE-publications are available from www.ce.nl

Commissioned by: Ministry of Housing, Spatial Planning and Environment, thermPhos,
Stichting Stimular.

Further information on this study can be obtained from the contact person Sander de Bruyn.

© copyright, CE Delft, Delft

CE Delft
Committed to the Environment

CE Delft is an independent research and consultancy organisation specialised in developing structural and innovative solutions to environmental problems. CE Delfts solutions are characterised in being politically feasible, technologically sound, economically prudent and socially equitable.



Contents

Annex A	General pricing principles	5
A.1	Price levels	5
A.2	Discount rates	5
A.3	Income elasticity	6
A.4	Risk premium	6
Annex B	Abatement cost estimation	7
B.1	Climate change	7
B.2	Ozone depletion	11
B.3	Acidification and photo-oxidant formation	13
B.4	Particulate formation	15
B.5	Human toxicity	19
B.6	Eutrophication	21
B.7	Final waste	22
B.8	Noise	23
B.9	Noise abatement costs	24
Annex C	Damage cost estimation	27
C.1	Introduction	27
C.2	Climate change	27
C.3	Acidification, photo-oxidant formation and particulate formation	33
C.4	Human toxicity	46
C.5	Eutrophication	49
C.6	Ozone depletion	50
C.7	Ionising radiation	56
C.8	Noise	59
C.9	Land use	61
Annex D	From per-pollutant values to weighted environmental themes	65
D.1	Abatement costs	65
D.2	Economic damage costs	67
Annex E	Valuation methods, benefit transfer and uncertainty	71
E.1	Introduction	71
E.2	Valuation methods for environmental goods	71
E.3	Value transfer	74
E.4	Uncertainty	78
Annex F	Modelling in ReCiPe	83
F.1	Introduction	83
F.2	Concept of the Cultural Theory	83
F.3	The three perspectives	84
F.4	Overview of choices for the perspectives (in ReCiPe)	84
Annex G	Comparison of NEEDS and ReCiPe	87
G.1	Introduction	87



Annex H	Physical impact indicators	89
H.1	Introduction	89
Annex I	Data ‘shelf life’	95
I.1	Introduction	95
I.2	New policy targets	95
I.3	New abatement cost curves and economic trends	95
I.4	New methods and values for damage valuation	95
Annex J	Full list of weighting factors	97
J.1	List of damage costs	97
J.2	List of abatement costs	114



Annex A General pricing principles

A.1 Price levels

In Annexes B and C the abatement and damage costs for each of the respective impact themes are reported at the price level of the year they were determined in the literature in question. These prices were then converted to 2008 prices using the European consumer price index. Consequently, all reported prices are in €₂₀₀₈.

This Harmonised Index of Consumer Prices (HICP) is compiled across all the countries of the European Union according to a standard methodology. It can also be determined for the Eurozone as a whole. It is then a weighted average of the price indices of all the member states who have adopted the Euro. Table 1 reports the HICP values for the years 2001-2008 for the Eurozone. These values have been applied to all European-based data expressed in Euros. The trend in the consumer price index in the Netherlands for 2001-2008 is very similar to that of the Eurozone, so these data were adopted for this country as well.

Table 1 HICP values (Eurozone)

Year	HICP compared with previous year	Value of Euro relative to 2000 level
2001	0.023	1.023
2002	0.023	1.046
2003	0.021	1.068
2004	0.021	1.091
2005	0.022	1.115
2006	0.022	1.139
2007	0.021	1.164
2008	0.033	1.202

Source: ECB, 2009.

A.2 Discount rates

In the literature on abatement and damage cost calculation, discount rates are used to assign a present value to costs, benefits and damages occurring in the future. In cases where we have made our own calculations, a discount rate of 2.5% has been taken, a value often employed in the literature for calculating certain abatement costs. For damage costs, a 3% discount rate is generally adopted in the European literature. This is important, because health damage does not always occur directly (as with acute mortality and morbidity), but sometimes only up to several decades later. Because a (slightly) higher discount rate is used for estimating damage costs, the damages may turn out *lower* than if a 2.5% discount rate had been used.

In this project, though, we have opted not to carry out any conversion to a common discount rate of 2.5%. This is because this would require a very substantial effort, for in that case we would have to determine, for each individual case of illness, the time sequence between dose and effect. Any gains from having slightly more accurate data would not weigh up against the effort expended on deriving them. In this context we note that the difference in value of a damage related to an illness that would occur 10 years post-



emission equals approximately 5%. In other words, if we were to calculate the net present value using the figure of 3%, we would obtain a value 5% lower than if we calculated with 2.5%. On top of this, we would also have to make a separate correction for the impacts of demography, economic growth and income elasticity with respect to environmental quality.

A.3 Income elasticity

In economic terms, environmental impacts lead to ever costlier damages as time elapses. This is because, as personal income grows, the environment is increasingly highly valued because of the positive income elasticity of demand for environmental quality. In the ExternE project a positive income elasticity of 0.85% was assumed, based on a review of the literature. As a result, ExternE takes a so-called uplift factor of 1.7% on an annual basis (2% economic growth combined with 0.85% income elasticity). Combined with the 3% discount rate, this means a net effective discount rate in the ExternE project of 1.3% (3% minus 1.7%). Because every SCBA should include specific discussions of income elasticity of demand for environmental quality as well as economic growth projections, precise shadow prices should therefore in principle also be calculated separately in each SCBA, obviously in accordance with the economic growth scenarios employed in that particular analysis.

A.4 Risk premium

In this study the calculations have not been adjusted using any sort of 'risk premium'. There are a variety of reasons for working with such a premium, depending on the type of risk involved, but it is debatable whether its use is required in the case of environmental damages. For a discussion of this issue the (Dutch) reader is referred to the 'SCBA Guidelines for Environmental Policy' (CE, 2007b) and RMNO (2008). In 2009 CPB and PBL worked on a review of the use of risk premiums in environment-inclusive SCBAs (see RWS, 2009). These developments have not been incorporated in our calculations. When our shadow prices are used in an SCBA, they will therefore need to be separately corrected for any preferred risk premium. We obviously realise this is by no means optimal, but the shadow prices presented here are to be seen more as cost estimates than as precise calculations of the value in question for use in an SCBA.



Annex B Abatement cost estimation

This annex describes the methods adopted in this study for estimating abatement costs.

B.1 Climate change

B.1.1 Target and projections

Under the Kyoto Protocol the industrialised world has committed itself to achieving a 5.2% reduction in its greenhouse gas emissions relative to 1990 by the years 2008-2012. Individual countries have different targets, however. The EU as a whole has accepted a reduction target of 8%. Within the EU this target then panned out to a 6% reduction target for the Netherlands, translating to an emissions level of 206 Mt CO₂-equivalents in the period 2008-2012.

Since 2005 the European Union's emissions trading scheme (EU ETS) for greenhouse gases has been in force for CO₂ emitters in two sectors: (heavy) industry and power generation (Directive 2003/87/EC). As of 2012, the aviation emissions due to flights to and from European destinations will also fall under the scheme.

The international community is currently engaged in preparations for agreement on *post*-Kyoto targets. In anticipation of this next phase, the European Commission has recently set new targets for the EU: a 20% emissions reduction in 2020 relative to 1990. If other developed nations are willing to accept similar cuts, the EU has expressed its willingness to accept a 30% reduction target.

In the European context, the Netherlands now has an obligation to cut its carbon emissions by 16% by 2020, relative to 2005, to the extent that these emissions are not regulated by the EU ETS. These emission cuts do not necessarily have to take place in the Netherlands, though, as use can be made of the Clean Development Mechanism (CDM), under which reductions may also be secured in developing countries. These cuts may then be included along with domestic efforts to meet the cited target.

Following earlier pledges to set climate and energy targets for 2020, the Dutch government, too, has stated (in its programme 'Clean and Efficient'; Werkprogramma Schoon en Zuinig; VROM, 2007a) that aggregate Dutch carbon emissions are to be reduced by 30% relative to 1990 by 2020, preferably within a European framework. This 30% reduction target equates to an emissions cap of approximately 150 MtCO₂-eq. in the year 2020 (VROM, 2007b).

B.1.2 Abatement costs

For the theme of climate change, marginal abatement costs are governed by three factors:

- a the CO₂ trading price under the EU ETS;
- b the costs associated with the Clean Development Mechanism (CDM);
- c the marginal costs of reducing the domestic emissions of non-EU ETS sectors.



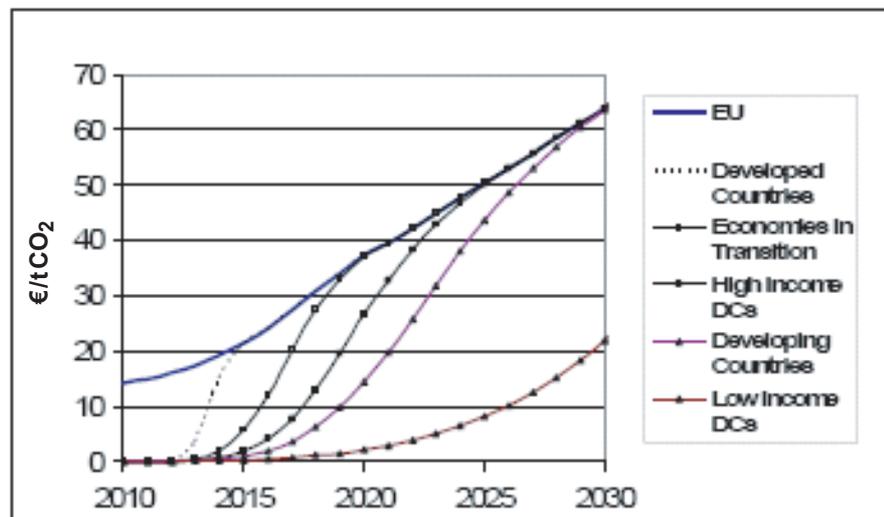
On face value, the highest costs might be expected to arise in the third category. However, the marginal abatement costs are governed by more than just the marginal costs of domestic emission cuts of non-EU ETS sectors, as the government is free to make greater use of the CDM should these costs prove too high. The cost of emissions reductions via the CDM are generally far lower than those achieved at home. Whether or not the government considers costs are rising too sharply depends to an extent on the marginal abatement costs in sectors participating in the EU ETS (i.e. the CO₂ trading price).

In 2007 the ‘Clean and Efficient’ programme (VROM, 2007a) was reviewed by the Netherlands Energy Research Centre (ECN) in collaboration with the Netherlands Environmental Assessment Agency (MNP). To make dependence on European policy explicit, two scenarios were thereby distinguished: ‘EU high’ and ‘EU low’. In the first, the emissions reduction target is 30% and the CO₂ trading price € 50/t (2007 prices) in 2020. In the second these figures are 20% and € 20/t. In the ‘EU high’ scenario the country needs to secure domestic emissions cuts of 50-67 MtCO₂-eq. in 2020 relative to the reference scenario, that is, *exclusive of* any emission cuts achieved abroad via emission allowances acquired under the EU ETS and CDM schemes. In the ‘EU low’ scenario the required cuts are 25-37 Mt in 2020.

According to the ‘Options Document on Energy and Emissions 2010/2020’ published by ECN and MNP (2006), the marginal abatement costs of emissions cuts of 25-37 and 50-67 MtCO₂ are around € 20 and € 50 per tCO₂, respectively.

These cost estimates can be compared the results of an impact assessment by the European Commission (EC, 2007: 36). This document states that long-term stabilisation of climate change at 2°C above pre-industrial levels can be achieved at abatement costs ranging from € 15/tCO₂ in 2010 and € 65/tCO₂ in 2030 (Figure 1).

Figure 1 Variation in carbon price across regions and time (EC, 2007: 36)



Based partly on this assessment, we consider the € 20-50/tCO₂ projected by ECN and MNP a plausible figure for the marginal abatement costs.

It should be noted that these values are, on average, lower than the central value of € 50/tCO₂ (2002 price level) argued for in the 2002 update (CE, 2002a). This value was based on the policy package set out in the Netherlands



‘Climate Policy Implementation Plan’ (Uitvoeringsnota Klimaatbeleid), which proposed a ‘basic’ and an ‘additional’ set of measures for securing the country’s Kyoto targets. In identifying policy measures, the government at the time adopted a cut-off point of € 68 (150 guilders) per tCO₂-eq. (1998 prices). In the light of the current European post-Kyoto targets, the EU ETS and its influence on policies in non-participant sectors, we consider the government’s earlier points of departure and thus the earlier analysis to be no longer valid for use in future policy evaluation exercises.

B.1.3 International comparison

The IMPACT handbook (CE, 2008b) provides a summary of the figures obtained in a number of studies that have estimated the external costs of CO₂ emissions. These are reproduced in Table 2.

Table 2 Estimated CO₂ abatement costs (€/tCO₂) cited in various studies

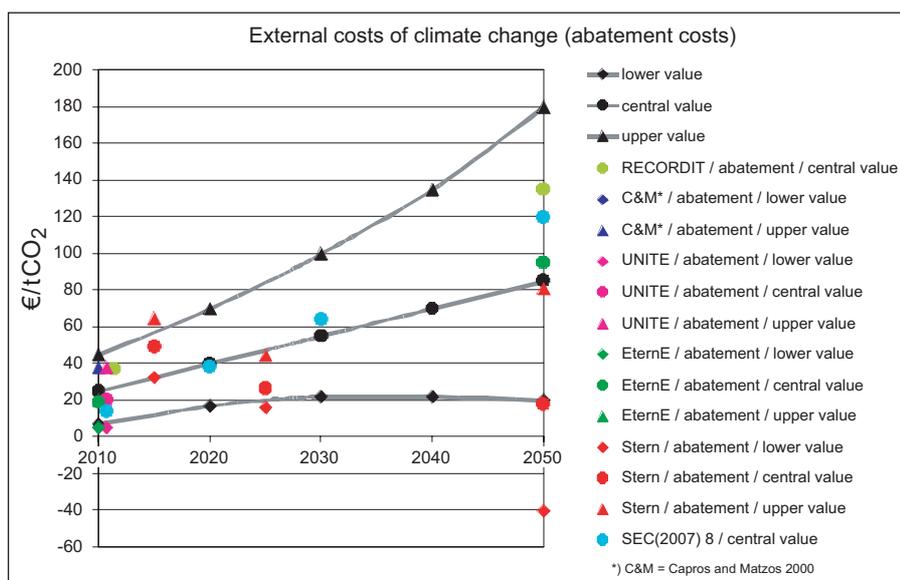
Source	Year of application	Abatement costs (€/tCO ₂)			Reference for abatement costs
		Min	Central	Max	
RECORDIT, 2000/1	2010		37		Kyoto target Long-term IPCC 50% reduction target
	2050		135		
Capros and Mantzos, 2000	2010	5		38	Kyoto target: lower value based on trading with countries outside EU, upper value on situation without trading outside EU
UNITE, 2003	2010	5	20	38	Based on Capros and Mantzos, 2000
INFRAS, 2004	2010		20		Kyoto target Long-term IPCC 50% reduction target
	2050		140		
ExternE, 2005	2010	5	19	20	Kyoto target Stabilisation at 2°C temperature rise
	2050		95		
Stern, 2006	2015	32	49	65	Average abatement costs
	2025	16	27	45	
	2050	-41	18	81	
SEC, (2007) ⁸	2010		14		Stabilisation at 2°C temperature rise
	2020		38		
	2030		64		Linear extrapolation based on 2020-2030 data
	2050		120		

Source: CE, 2008b.

These results can also be depicted graphically (Figure 2).



Figure 2 Estimated CO₂ abatement costs (€/tCO₂) cited in various studies



Source: CE, 2008b.

Within the NEEDS project, avoidance costs were assessed according to two scenarios (NEEDS, 2008a):

1. Realistic. In this scenario, the following assumptions are used. Marginal abatement costs are related to Kyoto targets for 2010, 20% reduction in CO₂ emissions in 2020 and subsequently an annual increase in costs with a rate 3% p.a is assumed. Marginal abatement cost curves (MACCs) for the -20% target vary widely and depend very much on which instruments are eligible for use (e.g. permitted 'import' of emission reduction via the Clean Development Mechanism (CDM) or Joint Implementation (JI)). It was consequently assumed that future policy will be defined in such a way that marginal abatement costs will increase by 3% p.a. This result is in line with the recommendations of HEATCO (2006) and Watkiss et al. (2005b) studies. With these MACs, the 2°C target will most likely not be secured. According to the meta-model of Kuik (Kuik et al. 2008), in this scenario we would end up at about 450 ppm CO₂, or roughly 2.8°C warming.
2. Ambitious. In this scenario, values from the meta-analysis model developed by Kuik et al. (2008) in the CASES project were used. If we take 365 ppm to be the goal of the 2° target, the meta-model yields results of approx. 100 €/tCO₂ for 2025 and 200€/t for 2050 (in 2000 level of prices). Because of the fact that this value for 2025 is considerably higher than that the values provided in the recent JRC report (2007), it has been proposed to keep on using the value from Kuik et al. (2008) for 2050, but to interpolate exponentially between 2010 and 2050, which brings the values for 2020 closer to the JRC and NEEDS estimates.

These assumptions lead to the values reported in Table 3.

Table 3 Recommended values of CO₂ abatement costs according to two scenarios (€₂₀₀₈ per tCO₂), discounted to year of emission, price level 2008

Scenario/Year of emission	2010	2015	2020	2025	2030	2035	2040	2045	2050
Realistic	25	29	31	35	37	40	54	71	83
Ambitious	25	33	50	55	80	94	119	158	214

Source: Based on CASES, 2008.



It can be noted that the central estimates cited in the IMPACT handbook review (see Table 3) for the period up to 2030 converge with the values proposed in the NEEDS project in the realistic scenario (and are lower than the values estimated for the ambitious scenario, while post-2030 values proposed in IMPACT lie below both trends defined with the two scenarios of NEEDS).

B.1.4 The values adopted in this study

More than anything else, the uncertainties regarding CO₂ abatement costs reflect the question of what targets the Dutch government will eventually be setting for national emission cuts. If, post-Copenhagen, an EU-wide decision is taken to go for a 30% reduction, the European target will be the same as the Netherlands' and the abatement costs will be € 0.05/kg CO₂. If the EU sticks to its 20% target and the Netherlands to its 30% target, the costs may rise very sharply indeed, for the emissions regulated via the EU ETS (covering some 70% of Dutch emissions) lie more or less beyond Dutch jurisdiction. Although the Netherlands may in principle adopt a more stringent emissions cap, this may lead to competitive disadvantages for Dutch industry *within* the EU, support for which could not be counted on. For this reason the additional 10% reduction (from 20 to 30%) will have to be secured in other sectors, and consequently at higher cost.

Because it makes little sense for the Netherlands to get ahead of international developments, we propose taking a figure of € 0.025/kg CO₂ for abatement costs. The upper and lower bounds are then € 0.02/kg CO₂ and € 0.05/kg CO₂.

B.2 Ozone depletion

B.2.1 Targets

Under the Montreal Protocol on Substances that Deplete the Ozone Layer, concluded on 16 September, 1987, to which the Netherlands is also a signatory, international agreements were laid down to protect the integrity of the stratospheric ozone layer. The aim of the protocol is to ban all substances that damage this layer. It has been in force since 1 January, 1989.

On 1 October, 2000 the European Union elaborated and tightened the Montreal Protocol under EC Regulation 2037 (amended in EC Regulations 2038 and 2039) on Substances that Deplete the Ozone Layer. Under the terms of this legislation chlorofluorocarbons may no longer be sold in EU member states, nor existing installations refilled with CFCs, even if these are from existing stocks.

At the national level, the production, trade and use of CFCs and halons are regulated under the Decree on Ozone-Depleting Substances (Besluit ozonlaagafbrekende stoffen, 2003), which came into force in August 2003. It covers the use of CFCs in refrigeration and air conditioning plant, in the manufacture of insulation materials and in solvents and cleaning agents, and the use of halons in fire extinguishing equipment. This decree replaced an earlier decree on the same issues (Besluit inzake stoffen die de ozonlaag aantasten, 1995).

Under the Decree on Electric and Electronic Equipment (Besluit beheer elektrische en elektronische apparatuur), which came into force on 13 August, 2004, producers and importers of white and brown goods (TV's, computers, video recorders and, for example, hair dryers) are obliged to dispose of their products responsibly in the waste phase (i.e. immediately the consumer disposes of them).



Under the Decree on Disposal of White and Brown Goods (Besluit verwijdering wit- en bruingoed)¹ all trade in cooling and refrigerating plants containing (h)CFCs has been banned since 1 January, 1999. Since then, those trading in used goods and equipment, like second-hand shops, may no longer buy or sell fridges and freezers containing these chemicals. The ban covers both domestic and international trade, with no exceptions whatsoever permitted and no firm or company eligible for exemption. Nor are there any provisions for financial compensation, in any form. This means that the fridges and freezers that consumers deliver to municipal recycling centres, second-hand shops, repair shops and high-street retailers may not be resold. Transactions between private citizens are still permitted, however. With this trade ban the government seeks to prevent fridges and freezers being shipped to Africa and Eastern Europe, which lack the capacity to dispose of CFCs in an environmentally responsible manner.

The Regulation on Collection of CFCs and Halons (Inzamelingsregeling CFKs en halonen), which prescribes environmentally benign collection and processing of these chemicals, was adopted on 23 August, 2002. It builds on agreements between the government (the Environment ministry, VROM) and the Dutch Association of Refrigeration and Air Conditioning Companies, NVKL, and gives owners of fridges and other cooling equipment a discount on environmentally responsible disposal of CFCs and halons. It was published in the Government Gazette no. 167 on 2 September, 2002, coming into force two days later.

B.2.2 Abatement costs

For those situations where emissions are not prohibited, we have derived a rough indication of the shadow price from the costs of processing under the Regulation on collection of old CFC and halon stocks.

Under this regulation, parties collecting these chemicals can charge € 5 per kilogram CFC/halon. For costs incurred over and above this figure they are eligible for a subsidy, up to a maximum of € 5/kg CFC/halon. If the costs exceed € 10/kg (incl. VAT), the minister may reject such an application.

Besides the CFC/halon collection scheme, there is also the waste disposal fee laid down in the Decree on Disposal of White and Brown Goods. In the case of large white goods, the aim of the legislation is to control emissions of CFCs and metals like arsenic, copper, mercury, lead and chromium. The waste disposal fee for 'standard' large white goods is € 5, while that for fridges and freezers is € 17. This difference of € 12 per fridge/freezer can be allocated exclusively to the avoidance of CFC emissions. In the Decree it is estimated that around 180 tonnes of (H)CFCs are recovered annually from some 450,000 units, which equates to about 0.4 kg (H)CFC per unit. Combined with the earlier figure of € 12 per fridge/freezer, this yields a shadow price of € 30/kg for (H)CFCs. As CFC-11 is the main focus of this (H)CFC recovery scheme, we have adopted a shadow price of € 30/kg CFC-11.

¹ Adopted on 21 April, 1998 (no. 97.000.3292, Government Gazette 238) under the title Vaststelling van regels voor het na gebruik innemen en verwerken van wit- en bruingoed (Besluit verwijdering wit- en bruingoed).



B.3 Acidification and photo-oxidant formation

B.3.1 Targets and projections

On the basis of the Gotenburg Protocol, in 2001 the EU established the so-called NEC Directive, laying down national emission ceilings (NEC) for the following air pollutants: sulphur dioxide (SO₂), nitrogen oxides (NO_x), ammonia (NH₃) and non-methane volatile organic compounds (NMVOC). For the Netherlands, the ceilings for 2010 are 50, 260, 128 and 185 kt/y, respectively, for SO₂, NO_x, NH₃ and NMVOC.

In 2009 the European Commission will be proposing new emission ceilings which in 2020 may no longer be exceeded. The targets for that year reported in Table 4 are based on the latest status of discussions on desired European emission trends and the respective contributions of individual member states (cf. CE, 2008a; see also IIASA, 2007; 2008).

Table 4 Targets and projected emissions for 2020 (kt/y)

	NO _x	SO ₂	NH ₃	NMVOC
NEC target, 2010	260	50	128	185
Anticipated NEC target, 2020 (CE, 2008a)	186	35	119	143
Emission ceiling for 2020 according to TSAP ambition level (GCN, 2008 based on IIASA, 2006, 2007)	186-223	35-50	118-123	145-164
Projected emission trend in <i>Global Economy</i> scenario ² (ECN, 2006)	279	83	147	182
Projected emission trend in <i>Global Economy</i> scenario under scheduled policy (GCN, 2008)	205.0	51,2	142.8	-

B.3.2 Abatement costs

Nitrogen oxides (NO_x)

At the time the Options Document (ECN/MNP, 2006) was drawn up, in the absence of additional policy NO_x emissions were projected to stand at 279 kt in 2020. Compared with the policy target of 186 kt NO_x for that year, this means a gap of 93 kt NO_x. According to the Options Document, the marginal abatement costs of this additional emission reduction are approximately € 50/kg NO_x.

We do not anticipate the marginal abatement costs actually reaching this figure of € 50/kg NO_x, however. According to the Options Document, around 70 kt NO_x can be avoided at marginal costs of only a few Euros per kg, and over 80 kt NO_x at marginal costs of around € 10/kg. It has often transpired in the past that long-term technological trends are underestimated, which means the abatement cost curve rises too sharply in the vicinity of the policy target. In addition, there is synergy with other policy targets (such as climate policy) so that the actual costs for NO_x may prove to pan out rather lower. A better measure of the marginal abatement costs is therefore provided by the latest cost estimates of the measures required to secure these targets. According to ECN (2009), in the case of NO_x emissions the upcoming update of the Decree on Emission Standards for Combustion Plant B (Besluit Emissie Eisen Stookinstallaties B) (BEES B) can make an important contribution to securing

² Global Economy is one of the four economic scenarios for Europe up to the year 2040 elaborated by the Netherlands Bureau for Economic Policy Analysis (CPB, 2003). For a translation of these scenarios to emissions, see ECN, 2006.



the NEC target. The average cost effectiveness is less than € 5/kg avoided NO_x emission, but varies substantially from sector to sector. For industry, power generators and refineries the costs will be around € 9/kg NO_x emission avoided in 2020. The Netherlands Environmental Assessment Agency (MNP) has also estimated the cost effectiveness of additional NO_x abatement measures (MNP, 2007a) and arrives at values varying from net benefits to around € 10/kg costs (e.g. Euro 6 NO_x emission standards for light vehicles as of 1-1-2014). Apart from a tightening of BEES B, the agency's report on major air pollution in the Netherlands (MNP, 2008) cites as the principal potential measure 'tightening of NO_x emissions trading from 40 to 20 g NO_x/GJ in 2020', with a cost effectiveness of € 1-2/kg (ECN/MNP, 2006). In the Netherlands Emission Guidelines for Air (NeR), finally, an indicative reference value of € 4.60/kg NO_x emission is cited for the cost effectiveness of abatement measures.³ Based on these data, we estimate the marginal abatement costs of securing the NEC target in 2020 at between € 5 and € 10/kg NO_x, essentially equivalent to the € 7/kg NO_x cited in the 2002 update (CE, 2002a).

Sulphur dioxide (SO₂)

At the time the Options Document (ECN/MNP, 2006) was drawn up, in the absence of additional policy SO₂ emissions were projected to stand at 83 kt in 2020. Compared with the government's target of 35 kt SO₂ in 2020, this means a gap of 48 kt. According to the Options Document, the marginal abatement costs of this additional emission reduction are approximately € 25/kg SO₂.

As in the case of NO_x, we do anticipate marginal abatement costs actually rising this high. According to the Options Document, around 37 kt SO₂ can already be avoided for a few Euros marginal costs per kg. Beyond this point, though, the marginal abatement costs soon rise. Here, too, we anticipate technological developments bringing down costs.

According to ECN (2009), the cost effectiveness of the measures implied by the BEES B update is € 7-8/kg SO₂. MNP (2007a) calculates figures of € 0.5-3.5/kg for 'switch from oil to gas firing in the chemical industry' and € 13/kg for 'lowering of sulphur content of red diesel for mobile equipment, inland shipping and fisheries'. The database of the Austrian International Institute for Applied Systems Analysis (IIASA), which provides the EU with the calculations underpinning each set of NEC Directives, cites a price of around € 5-6/kg for one of the most effective measures ('use of low-sulphur diesel oil - stage 2 (0.045% S)'). The Netherlands Emission Guidelines for Air (NeR), finally, cites an indicative reference value of € 2.30/kg SO₂ emission as the cost-effectiveness of abatement measures.⁴

Based on the available data, our expert judgement is that the marginal abatement costs for securing the NEC target in 2020 will be € 5-10/kg SO₂. This figure is substantially higher than the € 2.50/kg SO₂ cited in the 2002 update (CE, 2002a). The difference is due specifically to the substantially tougher NEC Directive for 2020.

³ NB: Although these values were reported in the most recent Guidelines (February 2009, unchanged since December 2006 with respect to Section 2.11), they date from the mid-1990s.

⁴ See previous footnote.



Ammonia (NH₃)

At the time the Options Document (ECN/MNP, 2006) was drawn up, in the absence of additional policy NH₃ emissions were projected to stand at 147 kt in 2020. Compared with the government's target of 119 kt NH₃ in 2020, this means a gap of 28 kt. According to the Options Document, the marginal abatement costs of this additional emission reduction are approximately € 10/kg NH₃.

It should be noted, however, that emission cuts of 22 kt NH₃ can be achieved at € 4/kg NH₃. Similarly, analysis of the marginal abatement cost curve for NH₃ reduction based on the SCBA for the acidification ceilings yielded a value of € 3.7/kg for reducing the first 21 kt. The additional measures for securing the target reviewed by MNP have a cost effectiveness of € 2/kg NH₃ ('tightening of low-emission application of manure to grassland') and € 3/kg NH₃ ('changes in dairy cattle feed').

Based on the available data, we have taken an approximate figure of € 4/kg NH₃ for the marginal abatement costs. This value is lower than the € 11/kg NH₃ cited in the 2002 update (CE, 2002a) for the two themes of acidification and smog taken together. The difference is due specifically to the NEC Directive for 2020 being relaxed (119 kt) compared with the 100 kt target for 2010 set in the 4th National Environmental Policy Programme (NEPP4).

NMVOG

At the time the Options Document (ECN/MNP, 2006) was drawn up, in the absence of additional policy NMVOG emissions were projected to stand at 182 kt in 2020. Compared with the government's target of 143 kt in 2020, this means a gap of 39 kt. The Options Document provides too few measures to secure such a reduction. Around 12 kt can be reduced at a cost of up to around € 8/kg NMVOG and another 16 kt by means of measures of which the cost is as yet unclear. In MNP (2007a) the cost effectiveness of the additional measures is estimated to be around € 5/kg NMVOG.⁵ The National Emission Guidelines for Air (NeR), finally, cites an indicative reference value of € 4.60/kg VOC emission for the cost effectiveness of abatement measures.⁶

We estimate a figure of around € 5/kg for the marginal costs of NMVOG reduction. This figure is substantially higher than the € 0.90/kg NMVOG cited in the 2020 update (CE, 2002a). The difference is due specifically to the NEC Directive for 2020 being more stringent (143 kt) than the target for 2010 set in NEPP4 (163 kt).

B.4 Particulate formation

B.4.1 Targets

For the theme of 'particulate formation' it is above all the pollutants PM₁₀, PM_{2,5}, NO_x and SO₂ that are important. The targets and abatement costs for NO_x and SO₂ have already been discussed under 'acidification'.

National policy targets with respect to airborne particulates derive from European air quality standards, which were transposed into national legislation in the Air Quality Act (*Wet luchtkwaliteit*), adopted in 2007. On 11 June, 2008 a new EU directive (2008/50/EC) came into force. Although leaving the current

⁵ Based on Figure 3.1, p.35: a reduction of around 20 kt NMVOG for about € 100 million.

⁶ See footnote 3.



standards for PM₁₀ and NO₂ unchanged⁷, it does provide member states scope for postponing compliance with these standards if they can demonstrate they are taking sufficient efforts to improve air quality. In addition, new European standards have been introduced for PM_{2.5} (Article 15.1), setting limits on annual average concentrations and average urban background.⁸ Meanwhile, the Netherlands has been granted a derogation by the EU, which means the deadline for meeting the air quality standards for PM₁₀ has been postponed to 2011. Implementation of the EU PM_{2.5} directive in Dutch legislation has been largely rounded off (PBL, 2009).

While standards are in place for *concentrations* of airborne particulates, then, there are currently no limits on particulate *emissions*. No emission ceiling has been set under the present NEC Directive (for 2010). The European Commission has stated its intention to include such a ceiling in its next review of the NEC Directive. It is anticipated that national emission ceilings for PM_{2.5} will be set for 2020⁹ (see Table 5).

Table 5 Targets and projected emissions in 2020

	PM ₁₀	PM _{2.5}
Anticipated NEC target, 2020 (CE, 2008a)	-	16
Emission ceiling for 2020 according to TSAP ambition (GCN, 2008, based on IIASA, 2006)		16
Projected emission trend in <i>Global Economy</i> scenario ¹⁰ (ECN, 2005)	47	
Projected emission trend in <i>Global Economy</i> scenario under scheduled policy (GCN, 2008)	32.2	13.9

B.4.2 Abatement costs

As Table 5 shows, total PM_{2.5} emissions in 2020 are anticipated to be lower than the national target. When it comes to particulate *concentrations*, too, according to the latest estimates it is expected that the government's intended policies will suffice to comply (largely) with the then valid air quality standards (PBL, 2009).¹¹ In this context the limits for PM_{2.5} are deemed less stringent than the current limits for PM₁₀, which means that if policy-makers succeed in securing the latter from 2011 onwards, the new PM_{2.5} limits will automatically be met by 2015. It should be noted, though, that when it comes to PM_{2.5} concentrations (and the implied policy challenge) there are

⁷ The daily average limit for PM₁₀ is 50 µg/m³, which may be exceeded on no more than 35 days per calendar year. The limit for the annual average concentration is 40 µg/m³.

⁸ By 2015 the annual average limit of 25 µg/m³ must be complied with throughout the Netherlands and the average urban background concentration may nowhere exceed 20 µg/m³. A 20% reduction target has been set for PM_{2.5} concentrations between 2010 en 2020.

⁹ Owing to political developments, the NEC Directive review process has been subject to delay. One reason is that the outcome of the Copenhagen climate talks is being awaited because of the relationship between climate policy and airborne particulate levels, which may be influenced either positively or negatively, depending on the kind of CO₂ measures promoted.

¹⁰ *Global Economy* is one of the four economic scenarios for Europe up to the year 2040 elaborated by the Netherlands Bureau for Economic Policy Analysis (CPB, 2003). For a translation of these scenarios to emissions, see ECN, 2006.

¹¹ PBL (2009) is based on the latest understanding, with due allowance also being made for the impact of the current economic downturn. Earlier studies painted a slightly less rosy picture. In MNP (2007b), for example, the anticipation was that it should on average be feasible to keep PM_{2.5} concentrations down to 25 or 20 µg/m³, but that additional national policy would be needed to tackle the urban background. The PM_{2.5} targets were to be more stringent than those for PM₁₀.



considerable with respect to emissions, the chemistry of the particles involved, their health effects and measurement (PBL, 2009).

It should be noted, though, that compliance with the national target does not necessarily mean that air quality standards (daily limits, MTRs) will not be locally exceeded. If they are, there will still be a need for abatement measures. However, the abatement cost method cannot be used in the context of addressing local impacts.

Although the national air quality standards will probably be respected and the anticipated national emission ceiling not be exceeded, the daily limit value for PM_{10} is presently constraining economic activity in the Netherlands, holding back construction projects, for example. National air quality policy has therefore been focused on improving air quality, among other things via at-source measures like subsidies on particle traps for diesel vehicles and at-source measures in agriculture and industry. Under the National Cooperative Air Quality Programme (NSL), one of the cornerstones of government policy,¹² national, provincial and local governments collaborate to secure European air quality standards. The essence of the programme is that development projects are only given the go-ahead if their impact is sufficiently offset by emission abatement measures. In this framework it has been agreed that, besides action in the transport and farming sectors, industry must also make its contribution to emission cuts.¹³ At the heart of the Particulate Action Plan for Industry (Actieplan fijnstof industrie; VROM, 2008b) is the obligation to use Best Available Techniques (BATs). European reviews of BATs, the so-called BAT reference documents, or BREFs, have been included in the Netherlands Emission Guidelines for Air (NeR) for use as a baseline by the competent authority in licensing procedures.¹⁴

In the new NeR (2009) particulates are designated a priority pollutant. Controlling emissions thereof is regulated via the general provisions in force for airborne dust. The target is to reduce ambient levels to a maximum of $5 \text{ mg}/\text{Nm}^3$, with several exemptions being granted for specific production processes. Cloth filters are still deemed the Best Available Technique for controlling particulate emissions (both PM_{10} and $PM_{2.5}$).

At the same time, though, the SenterNovem (2009) includes the suggestion that there is no obligation to implement measures costing more than $\text{€ } 2.30/\text{kg}$ avoided emission. This reference value is out-of-date and not keyed to particulates as we now understand the term. VROM (2008b: 39) notes that “this value was drawn up in the mid-1990s and embodies the understanding of particulates as it stood over 10 years ago, when PM_{10} was not yet an issue”. Ideally, then, the cost effectiveness of abatement measures for PM_{10} or $PM_{2.5}$ should not be compared with this reference value, which proves to yield an underestimate.

A series of studies have examined the actual costs of countless technical options for reducing emissions in various sectors and come to values in excess of $\text{€ } 2.30/\text{kg}$ (TNO, 2002; MNP, 2005; 2007b; CE, 2008c; VROM, 2008b). The results encompass a very broad range from around $\text{€ } 10/\text{kg}$ to $\text{€ } 275/\text{kg}$.

¹² Against the background of the impact of at-source measures in transport, industry and agriculture, in the NSL it is decided what additional measures are required.

¹³ It should be noted that the Action Plan does not yet specify which industrial sectors will be involved, nor the respective cuts that will be needed (ECN, 2009).

¹⁴ Under the IPPC Directive (European Directive 96/61/EC) EU member states are obliged to regulate large industrial polluters via an integrated permit based on Best Available Techniques (BAT).



For an alternative estimate of abatement costs we must look at the marginal costs of the most expensive measures that need to be taken to secure the statutory air quality targets. Immediately, though, we run up against a practical problem, for there is no straightforward method available for converting marginal changes in emissions (in kg) to resultant concentrations (in $\mu\text{g}/\text{m}^3$). This means existing kilogram-based cost estimates for individual measures cannot be used to calculate abatement costs, because there is no way to relate these directly to the policy target (a concentration) and it is therefore very hard to set a cut-off point in the array of potential measures, while it is hard to convert policy-geared cost estimates¹⁵ to abatement costs per kg.

If we endeavour, nonetheless, to establish a rough, conservative cost estimate based on current policy initiatives, we come to a figure of € 50/kg for PM_{10} . According to the cited Particulate Action Plan, this is the average cost effectiveness of achieving a residual emission of $5 \text{ mg}/\text{Nm}^3$ in industry, assuming the use of cloth filters (the NeR's BAT). By and large, this is a more effective means of emissions reduction than the measures already implemented in the transport sector, the cost effectiveness of which is generally between € 50/kg and € 250/kg (VROM, 2008b)¹⁶, which is probably worse than for the measures taken in agriculture, where the overall cost effectiveness of combined air scrubbers in large poultry and pig farms is estimated at around € 30-50/kg (MNP, 2007a; VROM, 2008c).¹⁷ The costs of reducing $\text{PM}_{2.5}$ emissions are likely to be about the same, or slightly higher (cf. VROM, 2008c). Given the latest understanding that the targets for PM_{10} are more stringent than those for $\text{PM}_{2.5}$ (PBL, 2009), for $\text{PM}_{2.5}$, too, we base ourselves on the abatement costs for PM_{10} .

At the same time there is a good reason for nonetheless sticking to the NeR abatement cost estimate of € 2.30/kg, however, as this is not an implausible value when considering the daily limits (MTRs) for PM_{10} and NO_x . The MTRs are the same for both pollutants¹⁸, while there is good reason to assume that € 2.30 per avoided kg NO_x is a reasonable figure for the contribution of NO_x to the theme of human toxicity, as will now be argued.

For NO_x emissions, a shadow price of € 5-10/kg NO_x was cited in Section B.3.2. However, NO_x contributes not only to the theme of human toxicity, but also to eutrophication, photo-oxidant formation and acidification. Based on the shadow price for NMVOC and the identical equivalence factor for NO_x with respect to the theme of photo-oxidant formation, we estimate the contribution of NO_x at around € 5. In the case of acidification and eutrophication, the contribution is around € 2 and € 3, respectively, with both these values subject to some uncertainty. Based on the total figure for NO_x of around € 10/kg, however, we do not anticipate the shadow price for the theme of PM formation being much in excess of a few Euros per kg.

¹⁵ See, for example, MNP (2007a; 2007b) and VROM (2008c).

¹⁶ MNP (2007a), for example, cites a figure of € 275/kg for the cost effectiveness of measures to meet the Euro VI emission standards for heavy road vehicles. CE (2008) estimates the cost effectiveness of promoting use of particle traps at around € 150-210/kg.

¹⁷ If we exclude several outliers, VROM (2008c) cites figures of up to around € 40/kg as a ballpark figure for the cost effectiveness of current strategies for PM_{10} reduction.

¹⁸ According to the ReCiPE characterisation factor, kg for kg NO_x contributes about half as much as PM_{10} itself.



As things stand at the moment, and based on current understanding, it is not feasible to calculate any precise figure for PM₁₀ en PM_{2.5} abatement costs. We have adopted a value of € 2.30/kg, but indicate in the shadow price tables that the current costs of abatement are higher, by also citing € 50/kg as an indicative value. We have elaborated two sets of weighting factors (sets 1a and 2), though, to illustrate that the value adopted for particulates has a major impact on weighting factors.¹⁹

B.5 Human toxicity

B.5.1 Targets and abatement costs

Although numerous substances contribute to the theme of ‘human toxicity’, there are no national ceilings in force for emissions of any of them. Instead, there are maximum tolerable risk levels (MTRs), i.e. ambient concentrations in air and water, that may not be exceeded.

The shadow price method cannot be used directly for local impacts, however. In order to calculate shadow prices for pollutants like heavy metals for which national emission targets are lacking, we here derive such prices by relating the values found for PM₁₀ to other pollutants via their relative MTRs. Thus, a pollutant for which the MTR is ten times lower than that for PM₁₀ is assigned a ten times higher shadow price.

In contrast to the other environmental theme, then, in this case equivalence factors have not been derived from ReCiPE. Given the premises of the abatement cost method, we feel it would be more appropriate in this case to proceed from standing government limits, i.e the MTRs.

A correction is made for the decay time or half-life of the pollutant (T) and the ratio between the specific mass of air (F=1.273) and water (F=1) (cf. VROM, 1994). Since the shadow prices are determined in comparison to PM₁₀, the shadow price for a pollutant p becomes: $(T_p/T_{pm10}) * (F_p/F_{pm10}) * MTC_{pm10}/MTC_p) * 2,30$ Euro/kg.

Table 6 provides a summary of the shadow prices of the toxic impacts of emissions to air and water of pollutants relevant to the present study. The maximum tolerable concentrations (MTCs, limit values) have been taken from RIVM (2009b).

¹⁹ If a value of € 2.30/kg is taken, the impact of SO₂ is ascribed largely to acidification, with very little remaining for PM_{2.5} (cf. Section 6.5).



Table 6 Shadow prices of toxic impacts of emissions to air and water (€₂₀₀₈/kg emission)

Pollutant	MTC	T	F	Shadow price
<i>Emissions to air</i>	(µg per m ³ , l)			
CO	10,000	8	1.273	0.01
Benzo(a)pyrene	0.001	8	1.273	92,000
Particulates (PM ₁₀)	40	8	1.273	2.30
Dioxins (ng TEQ)	0.000001	8	1.273	92,000,000
As	0.5	8	1.273	184
Cd	0.5	8	1.273	184
Co	0.2	8	1.273	460
Cr	0.0025	8	1.273	36,800
Cu	10	8	1.273	9
Hg	0.15	8	1.273	613
Ni	0.25	8	1.273	368
Pb	0.5	8	1.273	184
Zn	100	8	1.273	0.92
Fluoride	0.05	8	1.273	1,840
<i>Emissions to water</i>				
As	32	1,000	1	282
Cd	2	1,000	1	4,517
Co	3.1	1,000	1	2,914
Cr	84	1,000	1	108
Cu	3.8	1,000	1	2,377
Hg	1.2	1,000	1	7,528
Ni	6.3	1,000	1	1,434
Pb	220	1,000	1	41
Zn	40	1,000	1	226
Fluoride	1,500	35	1	0.21

Finally, a comparison can be made with the charges for effluent discharges, levied since the Surface Water Pollution Act (Wet verontreiniging oppervlaktewateren) came into force in 1970. The revenues from these charges, based on the 'polluter pays' principle, are used to cover the costs of effluent treatment. Rates differ from district to district, varying in 2009 between € 50 and € 70 per 'pollution unit'. As there is little difference in per-kilo treatment costs for the various heavy metals, the same figures have been adopted for most metals (Commissie Integraal Waterbeheer, 1998). Any relationship with toxicity is thus virtually lacking, although we have adopted a ten times higher charge for the former 'black list' pollutants arsenic, cadmium and mercury. Consequently, we have not used these charges for calculating shadow prices.



Table 7 Charges for emissions to water

Emission	'Pollution unit' per kg	Charge (€/kg)
Arsenic	1	600
Cadmium	1	600
Mercury	1	600
Chromium	0.1	60
Copper	0.1	60
Lead	0.1	60
Nickel	0.1	60
Silver	0.1	60
Zinc	0.1	60

B.6 Eutrophication

The focus of government policy in this area is on reducing emissions from households, industry and agriculture. It is mainly the last of these sectors, with its production of slurry and manure and use of artificial fertilisers, that is responsible for eutrophication. The costs this sector needs to expend are therefore representative of the shadow price for eutrophication.

The use of fertilisers and minerals on agricultural holdings is regulated under the Fertiliser Act (*Meststoffenwet*), with its attendant 'usage norms' and subsidiary regulations. The European Nitrate Directive was implemented through an amendment to the Act dated 1 January, 2006. The usage norms lay down the maximum phosphate and nitrogen dressings that may from that date be applied to agricultural crops. Every holding in the country is subject to three kinds of usage norm:

1. for manure, expressed in kg nitrogen, laying down the maximum amount of manure that may be applied annually per hectare of farmland;
2. for total nitrogen, laying down the maximum amount of nitrogen that may be applied annually per hectare, with only the active nitrogen from manure and other organic fertilisers being counted and artificial fertiliser also included;
3. for total phosphates, laying down the maximum amount of phosphate that may be applied annually per hectare, again including manure and other fertilisers, both organic and artificial.

Under the Fertiliser Act, the charges to be paid by livestock holders exceeding the former 'loss norms' for phosphate and nitrogen have been replaced by administrative fines for exceeding the usage norms, as follows:

- € 7/kg nitrogen in excess of the manure usage norm;
- € 7/kg nitrogen in excess of the total nitrogen usage norm;
- € 11/kg phosphate in excess of the total phosphate usage norm;
- € 3.50/kg nitrogen in excess of the nitrogen usage norm if a fine is also imposed for exceeding the manure norm;
- € 5.50/kg phosphate in excess of the phosphate usage norm if a fine is also imposed for exceeding the manure norm;
- € 11/kg phosphate and € 7/kg nitrogen for quantities for which no disposal route can be attested.

Based on these charges, we have adopted figures of € 11/kg and € 7/kg, respectively, for the shadow prices of phosphate (PO₄) and nitrogen (N). Given the higher fines in force today, these values are higher than those given in the 2002 update (CE, 2002a).



B.7 Final waste

Since 1 January, 1996 there has been an outright ban on landfilling waste that can be incinerated or recycled. In this context ‘recycling’ is taken to include the use of unprocessable waste of low or zero toxicity as a base layer for new roads. The material ultimately remaining after the various processing steps, which is not amenable to further incineration or recycling, is known as ‘final waste’.

The National Waste Management Plan (Landelijk afvalbeheerplan, LAP) sets out government policy for managing all waste materials to which the terms of the Environment Control Act (Wet milieubeheer) apply. These include policies on ‘useful application’ of these materials and on landfilling and incineration thereof. The recently published draft LAP for 2009-2012 (VROM, 2008a) sets out the following general policy goals with respect to waste:

- Waste prevention, that is to say: delinkage from economic growth of growth in the total amount of waste generated.
- Reduction of the environmental burden associated with ‘waste management’ itself. In principle, this means as much waste as possible must find a useful application, that only waste for which this is not feasible may be disposed of, with only non-combustible waste eligible for landfill.
- From a life cycle policy perspective, reduction of the environmental impact of integral supply chains (raw materials extraction, production, use and waste management, including recycling). Among other things, this means that when seeking to reduce the environmental impact of the waste phase the entire chain must be taken into account, and that impact reduction efforts may not lead to environmental impacts being shifted to other links in the chain.

These objectives have been translated into quantitative targets, including:

- An increase in useful application of aggregate waste streams from 83% in 2006 to 85% in 2015. This can be achieved by further encouragement of at-source waste separation and increased downstream separation of streams. This will facilitate product reuse and material reuse and recycling.
- Maintaining, or improving on, the 95% figure achieved in the Netherlands in 2006 for useful application of construction and demolition waste, despite the substantial increase in the volume of this type of waste projected for the coming years, from 24 Mt in 2006 to 31 Mt in 2021.
- Maintaining, or improving on, the 90% figure achieved in the Netherlands in 2006 for useful application of industrial waste, despite the projected increase in the volume of this type of waste, from 16 Mt in 2006 to 18 Mt in 2021.
- Reducing landfill of combustible ‘residual’ household waste, from 1.7 Mt in 2007 to 0 Mt in 2012.

In formulating these targets, the government makes no distinction between the various types of final waste, based on toxicity, for example. For this reason, in this study we do not provide separate shadow prices for the various categories.



B.7.1 Abatement costs

One of the more tangible impacts associated with landfilling final waste is land use, as embodied in the landfill site itself. In contrast to the situation for other environmental impacts, for land use there is a real market price available. By using going land prices and the costs of operating a landfill site, it should therefore be possible to derive a suitable shadow price for final waste. This approach yields too low a value, however. Society puts a higher price on land used permanently for landfill than on land used for other, less irreversible ends. This is because from the perspective of sustainable development future options should be restricted as little as possible.²⁰ Landfilling of waste entails a risk of toxic leakage that is hard to assess, moreover.

The above considerations explain why the government is taking action to *prevent* final waste arising that is (far) costlier than the price currently charged by landfill operators.

The latest draft of the LAP (VROM, 2008a), like the previous one (VROM, 2002), includes a provision that the minimum standard for processing non-hazardous process-specific industrial waste (with the exception of those specific waste streams covered by other sectoral plans) is useful application, unless such application is unfeasible because of the nature and composition of the waste or the additional costs associated with it are substantially higher than the costs of waste disposal. 'Substantially' is defined as more than 150% of the landfill tariff, including landfill tax.

In 2008 there were two landfill rates in force: a low tariff of € 14.56/t for waste with a specific weight over 1,100 kg/m³ and a high tariff of € 88.21/t for the remainder. In 2007 the average cost of incinerating household residual waste and similar commercial residual waste was about € 120/t. The cost of landfill, including the associated environmental tax, is roughly the same (VROM, 2008a).

From the above we derive a shadow price for industrial final waste of € 180/t. It should be noted, though, that for certain specific toxic waste streams far higher figures may apply.

B.8 Noise

Noise levels are frequently expressed in dB(A), which, as an approximation, makes due allowance for human perception of noise and corrects for frequency. In policy circles, noise is generally expressed in L_{den}, a measure of the number of decibels spread over the entire year, with 'forfeits' of 5 and 10 dB, respectively, being added for noise during evenings and nights. In addition, there is a 'rail bonus' (of 5 dB) and an 'aircraft forfeit' (of 5 dB), because railway noise is often experienced as less of a nuisance and aircraft noise as more of a nuisance than road traffic noise.

²⁰ As a comparison: the fact that land has a certain price does not yet imply that another country may simply pay that amount to buy a tract of land and add it to its own territory.



B.8.1 Policy framework

Since the late 1970s the Noise Nuisance Act (*Wet geluidshinder*) has provided the legislative framework for Dutch government policy on noise. It contains an extensive set of provisions to prevent and control noise nuisance caused by industry, road and rail traffic and so on. For noise-sensitive roadside sites there is in principle a 'preference limit' of 48 dB(A), but in certain situations exemptions may be granted, with higher values of 53 dB(A) up to 68 dB(A) being tolerated. The maximum exemption value is the noise nuisance tolerated after permission being granted by mayor and aldermen, the provincial executive or the minister of the Environment ministry (VROM). There may then be no further deterioration of the situation.

The present Noise Nuisance Act is under pressure because autonomous growth of road traffic may lead to an uncontrolled rise in noise levels (the 'enforcement gap'). This is because the only situation in which current legislation prescribes that action be considered is when there are interventions in or around transport infrastructure. This means no action is taken in situations of traffic *growth*, which obviously involve a risk of increased noise levels. In the new legislation currently being drawn up, so-called 'noise production ceilings'²¹ are therefore to be introduced.

The 'efficiency criterion'

In the current situation, *Rijkswaterstaat*, the Directorate-General for Public Works and Water Management, uses a so-called 'efficiency criterion' to establish what kind of measures (and on what scale) are to be deemed cost-effective in preventing exceedance of standards. For this purpose the criterion contains reference costs for abatement measures, which are then compared with the estimated noise reduction. If these costs are below a certain threshold (€ 3,000) per dB-dwelling reduction, the measure is deemed efficient. In calculating the number of dB-dwellings, allowance is also made for the original noise level, under the assumption that a decrease from 70 to 69 dB brings a greater subjective improvement than from 51 to 50 dB. A reduction of 1 dB at a noise level of 70 dB therefore counts as 2.5 dB, while a 1 dB reduction at 51 dB only counts for 1 dB.

B.9 Noise abatement costs

Our proposal is to derive the abatement costs for noise from the current *Rijkswaterstaat* efficiency criterion. Although under the new Noise Nuisance Act a single statutory (new) efficiency criterion is to be laid down, this has not yet been set.

From the above we derive an (unweighted) shadow price of € 3,000 per dB-dwelling. It would be preferable, though, to have the shadow price depend on the degree of nuisance and the type of noise involved. There is also the question of what thresholds should be adopted above which this shadow price holds.

²¹ These are equal to current noise levels at roadside reference points plus a margin. Future noise levels may not exceed these 'production ceilings'.



We recommend the following thresholds²²:

- 50 dB for road traffic noise;
- 55 dB for railway noise; and
- 45 dB for aircraft noise.

The weighted shadow price for noise (€ per dB-dwelling) is then given by:

$$€ 3,000 \times [1 + 0.05 * (L_{den} - threshold)]$$

The value of € 3,000 adopted per dB-dwelling can be interpreted as an investment sum, viz. the investment the government is willing to make to reduce the nuisance to below the threshold. Assuming a 25-year depreciation period for real estate and a risk-free interest rate of 2.5%, this translates to an outlay of € 162 per annum. Taking the average household to comprise 2.3 persons, this yields estimated abatement costs of € 70 per dB per year.

²² Based on the Miedema and Oudshoorn (2001) noise nuisance curve, which shows that for a given noise level people experience greatest nuisance from aircraft noise and least from railway noise. Cf. Nijland and Van Wee (2008).





Annex C Damage cost estimation

C.1 Introduction

This annex provides an extensive account of the methods employed in valuing the damage due to relevant pollutants for the various environmental themes.

C.2 Climate change

Climate change impacts are due mainly to emissions of so-called greenhouse gases, viz. carbon dioxide, (CO₂), nitrous oxide (N₂O) and methane (CH₄). Other substances contributing to global warming include refrigerants (hydrofluorocarbons) and high-altitude aircraft emissions (water vapour, sulphate, soot aerosols and nitrous oxides).

In estimating the damages associated with climate change, several problems are encountered: the impacts concerned are global in nature and not evenly distributed across the planet, and there are uncertainties about the atmospheric lifetime of CO₂. In other words, an additional emission of CO₂ results in impacts occurring far away, both in time and space.

C.2.1 Methodology

Damage cost estimates for climate change have been developed since the early 1990s, mainly through so-called Integrated Assessment Models (IAM). These models combine developments in economic growth with impacts from climate change. The models typically include a cost assessment of direct impacts, but exclude certain indirect effects (termed “socially contingent effects” by Watkiss et al. (2005a), such as the incidence of wars due to climatic stress).

Climate change damages include a broad range of effects related to temperature rise, such as changes in global precipitation, sea level rise, increased risk of extreme events such as drought and severe storms, and in the longer term possibly alteration of ocean currents. These effects may lead to various impacts associated with social costs, which can be summarised as follows (based on Watkiss et al., 2005a):

- Sea level rise may lead to loss of both dry land and wetlands. These impacts can be measured in terms of the costs of protection, which are relatively easy to assess. Another category of costs related to this type of impact is the cost of human migration which depends on various social and political factors (and thus a ‘socially contingent effect’) and is not captured by most valuation models.
- The impact of climate change on energy use will depend on the range and magnitude of temperature changes and is a combination of increase and decrease in demand for heating, whereby falling demand for heating in winter may be offset by rising demand for air conditioning in summer.
- Agricultural impacts are related to changes in cultivated area, in crop types and in yields resulting from changes in temperature and precipitation. In addition to these direct changes, there may be effects related to adaptive abilities and changes in demand and trading patterns, which depend on socio-economic factors.
- There will be impacts on water supplies: in some areas water shortages will be exacerbated by climate change, for example. There is therefore a



potential for increased water scarcity, the costs of which may be very high and highly socially contingent.

- Health effects include decreased cold stress in winter and increased heat stress in summer. To some extent at least, these direct effects will cancel out. Additional impacts include increased incidence of certain parasitic diseases like malaria. Indirect effects on human health include impacts related to changes in food production (especially declining food production in tropical/subtropical countries).
- Ecosystem and biodiversity impacts are the most complex and difficult to evaluate. Potential impacts include an increased risk of extinction of certain vulnerable species. Certain isolated systems like coral reefs are particularly at risk.
- Extreme events such as heat waves, droughts, storms and cyclones may not be linearly dependent on temperature change and the impact of such events is also very hard to assess. Damages will depend on the location and timing of the event and on adaptive responses and are thus also partly socially contingent.
- Major events, including potential catastrophic effects such as loss of the West Antarctic ice sheet, loss of the Greenland ice sheet, methane outbursts, instability or collapse of the Amazon Forest, changes in ocean currents and Indian monsoon transformation, are extremely hard to assess.

These impacts in turn influence the endpoints human health, ecosystems and capital goods. However, they may also result in other social costs, such as migration, which are included in most IAM (e.g. in the FUND model).

Controversial aspects of these models that lead to discrepancies among the results of the various studies include:

- the time horizon adopted (how far into the future are impacts considered?);
- the treatment of risk and uncertainty;
- the underlying emission and economic growth scenarios;
- the discount rate used to account for damages occurring in the (distant) future;
- the dependence of damage valuation on income level (e.g. similar damage suffered by a rich person is assigned a higher monetary value than that suffered by a poorer person);
- whether or not equity weighting is being used (see Box 1).

Box 1: Equity weighting

Most Integrated Assessment Models assume that the value to be assigned to damages depends on income level, which means the Value of a Statistical Life (VSL) is set proportional to GDP, for example. Equity weighting, by contrast, attaches greater weight to damages occurring in low-income regions than in high-income regions. This corresponds with the theory of declining marginal utility of consumption: the higher an agent's income, the less welfare loss they will suffer from the same absolute loss of income, i.e. the same absolute loss of income causes a greater loss of welfare to the poor than to the rich (based on NEEDS (2007c). See also Chapter 5.

As IAMs do not normally differentiate the damages across the various endpoints as defined in this study, we can only give total estimates for the damage. Moreover, we focus below only on the damage due to CO₂, translating this into damage due to other GHG emissions using equivalence factors at the endpoint level.



C.2.2 Total damage costs for CO₂

Over the last few years the term ‘Social Cost of Carbon’ (SCC) has been gaining currency and a growing number of studies are concerned with monetary assessment of SCC. This SCC can be interpreted as the total discounted value of future costs and benefits related to emission of one additional unit of CO₂. Tol (2008) provides a meta-analysis of 211 studies on SCC, and arrives at a mean value of \$ 23 per tonne of carbon at a 3% discount rate, equivalent to approximately \$ 6.3/tCO₂ (or about € 5/tCO₂). The range of estimates reported in Tol’s paper is very broad, however: from less than zero to over \$ 2,000/tC.²³

The damage cost approach shows that the external costs of GHG emissions rise over time, as the negative effects of global warming become ever severer as the global temperature rises. As a result, the literature on the damage cost approach generally gives a range of values that can be used in tools like SCBA if a project results in CO₂ emissions for a longer period of time (see Section 5.6).

In the NEEDS and CASES projects²⁴, damage costs were based on the results of the FUND model (for a model description, see Box 2), as reported in Table 8.

Table 8 Recommended values of damage costs for CO₂ (Euro₂₀₀₈²⁵ per tCO₂), discounted to the year of emission, without equity weighting

Emissions in decade	2000-09	2010-19	2020-29	2030-39	2040-49	2050-59	2060-69	2070-79	2080-89	2090-99
Damage costs	8	13	16	18	21	33	30	38	48	54

Source: CASES 2008.

Box 2: The FUND model

FUND is an Integrated Assessment Model (IAM), a computer model of economic growth with a controllable externality of greenhouse warming effects developed by Professor Richard Tol (IVM VU Amsterdam and Economic and Social Research Institute, Dublin). The model distinguishes 16 major regions of the world and runs from 1950 to 2300 in time steps of one year. The period 1950-1990 is used for model calibration, while the period 1990-2000 is based on observations. The climate scenarios for the period 2010-2100 are based on the EMF14 Standardised Scenario, which lies somewhere between the IPCC’s IS92a and IS92f scenarios. The model estimates marginal damages from emission of one extra tonne of carbon (and other greenhouse gases). The climate impact module includes the following categories: agriculture, forestry, sea level rise, cardiovascular and respiratory disorders related to cold and heat stress, malaria, dengue fever, schistosomiasis, diarrhoea, energy consumption, water resources and unmanaged ecosystems. The impacts of climate change are monetised. If people die prematurely due to temperature stress or have to migrate because of sea level rise, these effects are evaluated using approximate valuation factors from literature. For example, the Value of a Statistical Life (VSL) is set at 200 times annual per capita income. The value of

²³ In the literature the (damage or abatement) costs related to carbon dioxide emissions are typically expressed in dollars or Euros per tonne of carbon dioxide (CO₂) or per tonne of carbon (C). Costs per t C translate into costs per t CO₂ by dividing by a factor 44/12 = 3.667.

²⁴ NEEDS: New Energy Externalities Developments for Sustainability, European Commission research project implemented during the period 2004-2008, part of the ExternE series; CASES: Cost Assessment for Sustainable Energy Systems, European Commission research project implemented during the period 2006-2008.

²⁵ The original values reported in Euro₂₀₀₀ per tonne have been recalculated using HICP indicator (see Annex A.1).



emigration is set at 3 times per capita income, while the value of immigration is 40% of the per capita income in the host region. The monetary value of 4 million USD was taken for loss of one square kilometre of dryland in OECD countries in 1990, and was assumed proportional to GDP per km². FUND uses Ramsey-style discounting, which is a combination of the consumption growth rate, risk aversion and the pure rate of time preference (P RTP). P RTP is assumed at three different levels: 0, 1 and 3%. The effective discount rate used even for a specific P RTP varies over time and region, since per capita consumption growth rates vary over time and by region.
(Based on NEEDS (2007c).)

The IMPACT handbook (CE, 2008b) reviews a number of other studies on CO₂ damage costs, the results of which are summarised in Table 9 and Figure 3.

Table 9 Estimated damage costs of climate change according to various studies (€/tCO₂)

Source	Year of application	Damage costs (€/tCO ₂)			Comments
		Min	Central	Max	
ExternE, 2005	2010		9		
Watkiss et al., 2005b	2000	14	22	87	Results based on damage costs only
	2010	17	27	107	
	2020	20	32	138	
	2030	25	39	144	
	2040	28	44	162	
	2050	36	57	198	
Watkiss et al., 2005b	2000	14	22	51	Results based on comparison of damage and abatement costs
	2010	16	26	63	
	2020	20	32	81	
	2030	26	40	103	
	2040	36	55	131	
	2050	51	83	166	
Tol, 2005		-4	11	53	Based on studies with P RTP = 1%
Stern, 2006*	2050		71		Business-as-usual scenario
	2050		25		Stabilisation at 550 ppm
	2050		21		Stabilisation at 450 ppm
DLR, 2006		15	70	280	Based on Downing, 2005

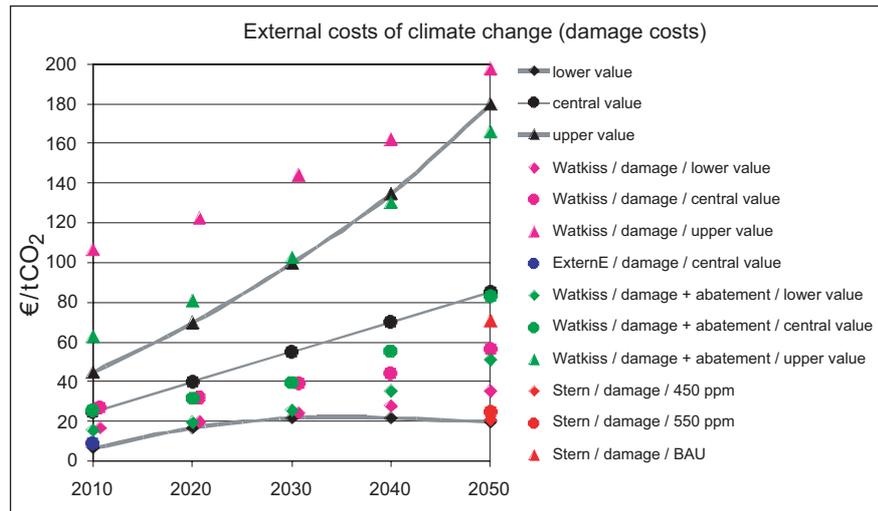
Source: CE, 2008b.

* See Box 3.

It may be noted that the values of damage costs recommended within the NEEDS project converge with the lower bound of the values proposed in Watkiss et al. (2005b).



Figure 3 Estimated damage costs of climate change according to various studies (€/tCO₂)



Box 3: The Stern Review

The Stern Review deserves special mention, not for its scientific merits but for its significant political impact. The report, which discusses the effects of climate change and global warming on the world economy, was issued on October 30, 2006 by economist Lord Nicolas Stern of Brentford for the British government. Its main conclusion is that the benefits of strong, early action on climate change considerably outweigh the costs. Stern proposes that one percent of global gross domestic product (GDP) *per annum* should be invested in order to avoid the worst effects of climate change, and that failure to do so could risk global GDP being up to twenty percent lower than it otherwise might be. For modeling damages, the Stern Review uses the PAGE model (one of the Integrated Assessment Models).

The Stern Review has been criticised by many economists. Most critiques relate to modeling details and assumptions, especially the assumed rate of discounting, which is very low compared with most other studies (Stern adopted a pure rate of time preference PRTP of 0.1% while in the NEEDS project, for example, a PRTP of 1% is assumed). The Stern estimates of damages related to GHG emissions are consequently much higher than those reported in most other studies. Tol (2008), for instance, concludes that the Stern Review is an outlier and that its impact estimates are pessimistic even when compared to grey literature and other estimates using low discount rates.

Despite the criticism, the Stern Review remains the most influential and most widely known and discussed report on the economics of climate change to date. Even some of Stern's adversaries admit the Stern review is 'right for the wrong reasons' (e.g. Arrow, 2007; Weitzman, 2007).

(Based on Tol (2008) and Wikipedia.)

C.2.3 Comparison with abatement costs

From comparing the estimates of CO₂ external costs based on the damage and avoidance cost approaches (based on CE, 2008b), the following conclusions can be drawn:

- Estimates based on damage costs tend to be *lower* than those based on abatement costs, certainly in the short term.
- The spread in estimates of short-term external costs among different studies is smaller for avoidance costs than for damage costs.
- The central values calculated in recent studies for long-term (i.e. 2050) damage and avoidance cost all tend to be in the same range: € 50-100/tCO₂. The claim by Stern (2006) that the damage costs are higher than



the avoidance costs, which also appears to be the underlying assumption for the EU strategy aimed at stabilising global warming at 2°C above pre-industrial levels, can be neither confirmed nor rejected on the basis of these recent estimates.

- Both damage costs and avoidance costs are expected to rise over time.

C.2.4 Conclusion and recommended values

We here adopt the approach followed in the IMPACT study based on a literature review of the various estimates for CO₂. In this approach the abatement costs are used for the time frame up to 2020, with damage costs being used thereafter. The reason for using abatement costs in the short term at least is based on the notion that current environmental policies obviously impose stricter targets than one would expect based on damage costs. The average marginal abatement cost of € 25/tCO₂ for a 20% reduction is far above the median damage costs based on Tol (2008). The reason is that politicians obviously put a higher value on preserving the climate than economists would advocate. This may be for various reasons, such as omissions in the damage estimates (ignoring indirect effects), a lower time preference on the part of politicians than estimated by economists, moral imperatives such as ‘global stewardship’, etcetera. As yet, however, policies have only been formulated up to the year 2020. For emissions occurring after that date, IMPACT bases itself on damage costs for estimating longer-term impacts.

The recommended values for CO₂ shadow prices derived in this way are reported in Table 10 and Figure 4, specified for different years of application.

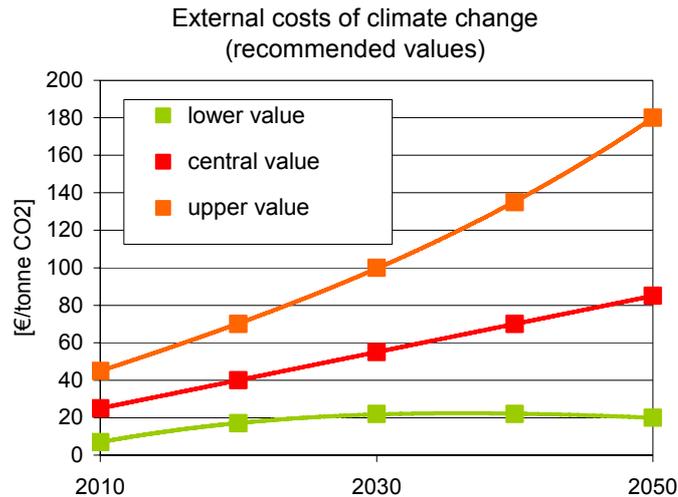
Table 10 Recommended values for external costs of climate change (€/tCO₂), expressed as single values for a central estimate and lower and upper values

Year of application	Central values (€/tCO ₂)		
	Lower value	Central value	Upper value
2010	7	25	45
2020	17	40	70
2030	22	55	100
2040	22	70	135
2050	20	85	180

Source: CE, 2008b.



Figure 4 Recommended values for external costs of climate change (€/tCO₂), expressed as single values for a central estimate and lower and upper values



C.3 Acidification, photo-oxidant formation and particulate formation

This section deals with the damages due to acidification, photo-oxidant formation and airborne particulates, established using the Impact Pathway Approach and based largely on estimates from the NEEDS project. Description of the methodology draws extensively on NEEDS (2008a).

C.3.1 Methodology

The group of so-called classical air pollutants includes sulphur dioxide (SO₂), nitrogen oxides (NO_x), particulate matter (PM), ammonia (NH₃) and non-methane volatile organic compounds (NMVOC). NMVOC and NO_x contribute to the formation of tropospheric ozone and ozone is therefore also sometimes referred to as a classical pollutant.

A state-of-the-art method for calculating the external costs associated with emissions of classical pollutants is the Impact Pathway Approach developed within the framework of the ExternE²⁶ series of projects, as described in Chapter 5. The ExternE methodology aims to cover all relevant (i.e. non-negligible) external effects identified via the Impact Pathway Approach.

In this methodology local, Europe-wide and hemispheric modelling were used to simulate environmental quality status and estimate differences between specific cases of pollution and the reference situation. The exposure-response models were compiled and critically reviewed by the ExternE expert groups. Physical impacts were evaluated in monetary terms. For some of the impacts, market prices could be used to evaluate damages. For non-market goods, values were derived using stated and revealed preference methods (cf. Section 5.3).

²⁶ ExternE (External costs of Energy) is a series of research projects initiated by the European Commission aimed at estimating the socio-environmental damages associated with energy conversion.



With respect to particulate matter (PM), damage cost estimates performed within the NEEDS project are available for PM_{coarse} and $PM_{2.5}$, not for PM_{10} , which comprises both fractions. Based on current emission data for the Netherlands/Europe²⁷, we have calculated the fractions of $PM_{2.5}$ and PM_{co} in PM_{10} . Subsequently, the price to be assigned to PM_{10} was determined by weighting the prices of PM_{co} and $PM_{2.5}$ with their respective shares in PM_{10} emissions.

C.3.2 Health-related impacts

Health impacts are endpoints which can be modelled using the IPA. Two crucial elements of this approach are definition of concentration-response functions (CRF) and monetary valuation of health impacts.

Within the NEEDS project, a set of CRFs for PM and ozone and corresponding monetary values have been proposed. These functions are the most important and reliable concentration-response functions used in the ExternE series of projects for valuing the health effects associated with emissions of classical pollutants.

It should be noted that according to the recommendations of the NEEDS project experts, human health impacts have only been defined for particulate matter (primary as well as secondary) and ozone.²⁸ Impacts due to emissions of SO_2 , NO_x and NH_3 are factored in after chemical transformation with reactants leading to an increase of concentration of secondary particulate matter (SIA, secondary inorganic aerosols). In the scientific community there is considerable debate on whether SIA has the same toxicity as primary particles, with no consensus yet emerging. In the NEEDS project it was therefore assumed that the damage due to SIA should be the same as for primary particles.

The general approach to estimating the effects of PM (or ozone) on morbidity uses the relative risk found in epidemiological studies, expressed as a percentage change in endpoint per (10) $\mu\text{g}/\text{m}^3$ PM_{10} (or $PM_{2.5}$) and links this with (i) the background level of the health endpoint in the target population, expressed as new cases per year per unit population, (ii) population size and age, and (iii) the relevant pollution increment, expressed in $\mu\text{g}/\text{m}^3$. The results are then expressed as extra cases, events or days per year attributed to PM (ExternE, 2005). Within the Ecosense model, uniform breakdown into age groups (Age Group Functions, AGF) and risk groups (Risk Group Functions, RGF) have been assumed for the whole of Europe, based on NEEDS (2007b).

Even though the risk of death due to diseases related to air pollution is lower than the risk of milder effects, reduced life time expectancy (YOLL) has the highest weight in the calculation of the total value of damages related to air pollution (see Table 11).

²⁷ Depending on the dataset.

²⁸ These toxic impacts cover the bulk of the toxic impacts associated with these pollutants. However, NO_x also has a toxic effect other than through SIA. In this study this is taken into account in Section 4.6, using equivalence factors.



Table 11 Overview of -sponse functions for PM and ozone and corresponding monetary values

Core Endpoints										
	Pollutant	Risk group (RG)	RGF value	Age Groups (AG)	AGF value	CRF [1/(µg/m ³)]	phys. Impact per person per µg per m ³ [1/(µg/m ³)]	unit	Monet Val per case or per YOLL [Euro]	External costs per person per µg per m ³ [1/(µg/m ³)]
primary and SIA < 2.5, i.e. Particle < 2.5µm										
Life expectancy reduction - YOLL	PM2.5	all	1.000	Total	1	6.51E-04	6.51E-04	YOLL	40,000	2.60E+01
chronic netto Restricted activity days (netRADs)	PM2.5	all	1.000	MIX	1	9.59E-03	9.59E-03	days	130	1.25E+00
Work loss days (WLD)	PM2.5	all	1.000	Adults_15_to_64_years	0.672	2.07E-02	1.39E-02	days	295	4.10E+00
Minor restricted activity days (MRAD)	PM2.5	all	1.000	Adults_18_to_64_years	0.64	5.77E-02	3.69E-02	days	38	1.40E+00
primary and SIA < 10, i.e. Particle < 10µm										
Increased mortality risk (infants)	PM10	infants	0.002	Total	0.009	4.00E-03	6.84E-08	cases	3,000,000	2.05E-01
New cases of chronic bronchitis	PM10	all	1.000	Adults_27andAbove	0.7	2.65E-05	1.86E-05	cases	200,000	3.71E+00
Respiratory hospital admissions	PM10	all	1.000	Total	1	7.03E-06	7.03E-06	cases	2,000	1.41E-02
Cardiac hospital admissions	PM10	all	1.000	Total	1	4.34E-06	4.34E-06	cases	2,000	8.68E-03
Medication use / bronchodilator use	PM10	Children meeting PEACE criteria - EU average	0.200	Children_5_to_14	0.112	1.80E-02	4.03E-04	cases	1	4.03E-04
Medication use / bronchodilator use	PM10	asthmatics	0.045	Adults_20andAbove	0.798	9.12E-02	3.27E-03	cases	1	3.27E-03
Lower respiratory symptoms (adult)	PM10	symptomatic_adults	0.300	Adults	0.83	1.30E-01	3.24E-02	days	38	1.23E+00
Lower respiratory symptoms (child)	PM10	all	1.000	Children_5_to_14_years	0.112	1.86E-01	2.08E-02	days	38	7.92E-01
Ozone [µg/m³] - from SOMO35										
Increased mortality risk	SOMO35	Baseline_mortality	0.0099	Total (YOLL = 0.75a/case)	1	3.00E-04	2.23E-06	YOLL	60,000	1.34E-01
Respiratory hospital admissions	SOMO35	all	1.000	Elderly_65andAbove	0.158	1.25E-05	1.98E-06	cases	2,000	3.95E-03
MRAD	SOMO35	all	1.000	Adults_18_to_64_years	0.64	1.15E-02	7.36E-03	days	38	2.80E-01
Medication use / bronchodilator use	SOMO35	asthmatics	0.045	Adults_20andAbove	0.798	7.30E-02	2.62E-03	cases	1	2.62E-03
LRS excluding cough	SOMO35	all	1.000	Children_5_to_14_years	0.112	1.60E-02	1.79E-03	days	38	6.81E-02
Cough days	SOMO35	all	1.000	Children_5_to_14_years	0.112	9.30E-02	1.04E-02	days	38	3.96E-01

Abbreviations: Risk Group, RG: group within the general population with a handicap; RGF value: share of RG within the general population; Age group, AG: groups distinguished by different age cohorts; AG value: share of different age cohorts; CRF: concentration-response function; YOLL: Years of Life Lost; RAD: Restricted Activity Days; SIA: Secondary Inorganic Aerosols; SOMO35: sum of ozone means over 35 ppb; WLD: Work Loss Days; MRAD: Minor Restricted Activity Days; LRS: lower respiratory symptoms.

Table constructed for the whole of Europe.

Source: NEEDS (2008a), based on NEEDS (2007b).

Based on the CRFs and monetary valuation (see Chapter 5), a set of values for the external costs per tonne of pollutant has been proposed after modelling a 15% reduction of emissions of each pollutant in different regions using the EcoSense model.²⁹ The results are available for emissions occurring in 39 European and non-European countries and five marine regions and also include emission-weighted average values for the EU-27. The receptor domain covers the whole of Europe. In addition to modelling the impact of emissions occurring in Europe and adjacent areas, a Northern Hemispheric Model was employed to estimate the external costs of impacts on human health outside Europe caused by emissions of classical pollutants in Europe. These values were derived by applying the same CRF and monetary valuation of impacts as within Europe, regardless of whether the impacts take place in America or Asia, say. In the tables below we first give the values of impacts in Europe and then also the values derived from the Northern Hemispheric Model.

For some substances, different values are given for low height of release, unknown height of release and high height of release. These numbers were estimated using the EcoSense model by using different sets of source-receptor matrices (SRM), using the procedure described below.

²⁹ <http://EcoSenseWeb.ier.uni-stuttgart.de>.



The SRM were derived by simulating a 15% emission reduction in each sub-region. This was done in two ways, providing two sets of SRM, viz.:

- for pollutants from all sources, i.e. all SNAP sectors (i.e. including transport, industry, domestic firing systems, but also combustion plants);
- for pollutants (primary particles, SO₂ and NO_x) from SNAP sector 1 (combustion in power plants) only.

Since the first SRM set provides an average value for all emissions, these values have been used for the estimates with unknown height of release. The second SRM set corresponds to releases above 100 m. Further processing of the results allowed €/t values to be derived corresponding to low height of release for primary particulate matter, i.e. below 100 m.

Summarising, if the height of release is unknown, as is often the case in LCA data that includes all a product's life cycle stages, the corresponding values of unknown height of release should be used. If the height of release is approximately known, the values referring to low height of release should be taken for emissions with a release height below 100 m, and the values referring to high height of release should be taken for emissions above 100 m. For emissions from power plants, the results according to high height of release should be used (NEEDS, 2008a).

An Excel tool developed within the NEEDS project permits user-defined assumptions regarding year of emission. The values are provided in two formats: discounted to the year of emission and discounted to the year 2000. In Table 12 and Table 13 we report the estimated external costs associated with the various specific pollutants for emissions in the year 2008, discounted to the year of emission for the EU-27 and the Netherlands, respectively.³⁰ The original values from the Excel tool, which are given in Euros of 2000, have been adjusted to 2008 prices using the Harmonised Index of Consumer Prices (cf. Annex A).

Table 12 External costs related to human health per tonne of specific classical air pollutants released in a specific year and discounted to that year (Euro₂₀₀₈): EU-27 average

Pollutant	External cost 2008 at given height of release		
	Low	Unknown	High
Height of release			
NH ₃	13,040.71	13,040.71	13,040.71
NVMOC	802.80	802.80	802.80
NO _x	7,689.11	7,689.11	5,659.27¹⁾
PMco ³¹	2,026.23	1,821.93	673.01
PM _{2.5}	35,616.50	33,574.64	1,6856.42
SO ₂	8,347.69	8,347.69	7,671.08

Notes: Values in bold indicate that the values for low/high height of release differ from those for unknown height of release.

Source: CASES, 2008.

³⁰ 'Discounted to the year of emission' means that all non-negligible health impacts that may occur in future years due to emission of a given substance are discounted back to the year of emission. With some substances, like PM, the impact lasts for a relatively short time and is assumed to persist for one year, while the impact of other substances, like CO₂ and heavy metals, may prevail for several hundred years.

³¹ PMco (PM coarse) is particulate matter with a diameter between 2.5 and 10 µm.



Table 13 External costs related to human health per tonne of specific classical air pollutants released in a specific year and discounted to that year (Euro₂₀₀₈): values for emissions from the Netherlands

Pollutant	External cost 2008 at given height of release		
	Low	Unknown	High
NH ₃	23,110.50	23,110.50	23,110.50
NMVOG	1,670.79	1,670.79	1,670.79
NO _x	9,093.71	9,093.71	7,113.44
PMco	3,854.17	3,842.77	1,957.73
PM _{2.5}	65,341.78	64,535.87	29,708.46
SO ₂	14,114.75	14,114.75	10,595.05

Note: Values in bold indicate that the values for low/high height of release differ from those for unknown height of release.

Source: CASES, 2008.

As can be seen in the tables, the height of release is especially relevant in the case of particulate matter, where impacts for high height of release may be over 50% lower than in the case of unknown or low height of release. Such differences are due to different types of emission characteristic for different heights of release. Transport and domestic firing systems are characterised by a low height of release, where the density of receptors is higher, while high emissions from power plant stacks diffuse over a much greater area, with on average a far lower population density.

Table 14 reports the estimates obtained in the NEEDS project by Northern Hemispheric Modeling (NHM), i.e. for the impacts of pollutants emitted within Europe to other regions. These numbers give only a rough estimate and are the same for emissions occurring anywhere in Europe at any height of release, so in Table 14 in the first column we show general numbers that apply to both the EU-27 and the Netherlands. The two last columns give the total impact on human health from the emissions occurring in the EU-27 and in the Netherlands, after summing the impact within Europe and outside Europe (Table 14 reports only the figures for unknown height of release). As can be seen, for certain pollutants like NH₃ and PPMco, the additional impact associated with non-European receptors is very small, below 1% of the total figure, while for others, especially NMVOG, this additional impact is relatively high, constituting about 20-40% of the total impact value. These differences are due to physical characteristics of the specific substances and, in particular, the likelihood of being transported over long ranges.

Table 14 Values (€/t of emissions) for Northern Hemispheric Modelling (NHM) and total external costs for human health (Euro₂₀₀₈): values for emissions from the EU-27 and the Netherlands

	Average estimates for NHM	Total EU-27	Total Netherlands
NH ₃	3.73	13,044.44	23,114.23
NMVOG	491.66	1,294.46	2,162.45
NO _x	180.15	7,869.26	9,273.86
PMco	2.88	1,824.81	3,845.65
PM _{2.5}	216.92	33,791.57	64,752.80
SO ₂	382.53	8,730.22	14,496.28

Source: Own calculations based on CASES, 2008.



The values reflecting the impact of classical pollutants on human health change over time because of two factors: 1) an ‘uplift factor’ related to economic growth, which means that VOLY and other factors forming the basis for valuing the CRF functions for human health are expected to grow in real terms (according to the assumptions made within the NEEDS project, the uplift factor equals 1.7% until 2030 and 0.85 thereafter), and 2) a change in background pollutant emissions, which in the Ecosense model is assumed to occur in the year 2015. For emissions between 2000 and 2014, the model is based on background emissions for 2010, while for later years predicted background emissions for the year 2020 are used (NEEDS, 2008a). The latter factor has a different impact on different values (so no systematic increase or decrease of all estimates can be observed).

C.3.3 Impacts on agricultural crops

Within the NEEDS project, the impacts of air pollution on crops have been divided into impact of SO₂, acidification of agricultural soils due to NH₃, SO₂ and NO_x, impact of ozone and effects of nitrogen deposition (NEEDS, 2008a).

Impact of SO₂

The CRF function for SO₂ assumes that yields will increase with SO₂ concentrations from 0 to 6.8 ppb (part per billion on a molecular level) and decline thereafter. The function is used to quantify changes in crop yield for wheat, barley, potato, sugar beet and oats and is defined as:

$$y = 0.74 \cdot [\text{SO}_2] - 0.055 \cdot [\text{SO}_2]^2 \quad \text{for } 0 < [\text{SO}_2] < 13.6 \text{ ppb}$$

$$y = -0.69 \cdot [\text{SO}_2] + 9.35 \quad \text{for } [\text{SO}_2] > 13.6 \text{ ppb}$$

with y = relative yield change; and
 $[\text{SO}_2]$ = SO₂ concentration in ppb.

Acidification of agricultural soils

For acidification effects, an upper-bound estimate of the amount of lime required to balance atmospheric acid inputs on agricultural soils across Europe has been estimated. Ideally, the analysis of liming would be restricted to non-calcareous soils, but this refinement has not been introduced given that even the upper-bound estimate of additional liming requirements is small compared with other externalities. The additional lime required is calculated as:

$$dL = 50 \text{ kg/meq} \cdot A \cdot dDA$$

with dL = additional lime requirement in kg/year;
 A = agricultural area in ha; and
 dDA = annual acid deposition in meq/m²/year.

Impact of ozone

For the assessment of ozone impacts, a linear relationship between yield loss and the AOT 40 value (Accumulated Ozone concentration above a Threshold of 40 ppbV) calculated for the crop growing season (May to June) has been assumed. The relative yield change is then calculated using the following equation together with the sensitivity factors given in Table 15:

$$y = 99.7 - \text{Alpha} \cdot \text{AOT40crops}$$

with y = relative yield change; and
 Alpha = sensitivity factors.



Table 15 Sensitivity factors for different crop species

Crop species	Sensitivity factor
Rice	0.4
Tobacco	0.5
Sugar beet, potato	0.6
Sunflower	1.2
Wheat	1.7

Fertilisation effects from nitrogen deposition

When it comes to nitrogen there is also a beneficial effect, in the sense that nitrogen is an essential plant nutrient, applied by farmers in large quantities to their crops. Deposition of oxidised nitrogen on agricultural soils is thus beneficial (assuming the dosage of any fertiliser applied by the farmer is not excessive). The reduction in fertiliser requirement is calculated as:

$$dF = 14.0067 \text{ g/mol} \cdot A \cdot dDN$$

with dF = reduction in fertiliser requirement in kg/year;
 A = agricultural area in km²; and
 dDN = annual nitrogen deposition in meq/m²/year.

C.3.4 Monetary valuation of crop losses

Crop losses are assessed in monetary terms using the prices of the crops damaged by air pollution. Table 16 summarises the prices per tonne used within the NEEDS project for assessing crop damage due to air pollution.

Table 16 Updated prices of major crops used within the NEEDS project (€/t)

	Updated price per tonne	Source
Sunflower	273	FAOSTAT (€ ₂₀₀₁)
Wheat	137	IFS (€ ₂₀₀₃)
Potato	113	FAOSTAT (€ ₂₀₀₁)
Rice	200	IFS (€ ₂₀₀₃)
Rye	99	FAOSTAT (€ ₂₀₀₁)
Oats	132	FAOSTAT (€ ₂₀₀₁)
Tobacco	2,895	IFS (€ ₂₀₀₃)
Barley	93	IFS (€ ₂₀₀₃)
Sugar beet	64	FAO (€ ₂₀₀₂)

Source: ExternE, 2005.

It should be noted that prices have fluctuated significantly in recent years, with those for major crops like wheat and potato rising until 2008 and subsequently falling. As future crop prices remain uncertain, in this project we have opted to stick to the values used in the NEEDS project.

Average values for the impact of acidification and ozone on agricultural crops due to emissions of NO_x, SO₂, NH₃ and NMVOC generated in 2008 at unknown, low and high height of release based on the results of the NEEDS project are reported below in Table 17 and Table 18. Table 17 gives the values estimated for the EU-27, Table 18 those for the Netherlands. According to the Ecosense calculations, the values for EU-27 are the same for unknown and low height of release thus they are reported in one column.



Table 17 External costs related to agricultural crop damage per tonne of specific classical air pollutants (Euro₂₀₀₈), EU-27 average

Pollutant	External cost 2008 at given height of release	
	Unknown/low	High
NH ₃	-219.93	-219.93
NMVOOC	227.14	227.14
NO _x	394.19	216.32
SO ₂ primary particles	-32.45	-22.83
SO ₂ secondary particles*	-15.62	-9.61

Notes: Values in bold indicate that the values for unknown/low height of release differ from those for high height of release.

* For SO₂ two models have been used, in order to include the impact of both primary and secondary particles. To model sulphate formation a Windrose Trajectory Model (WTM) was used (ExternE, 2005).

Source: CASES, 2008.

Table 18 External costs related to agricultural crop damage per tonne of specific classical air pollutants (Euro₂₀₀₈), values for the emissions from the Netherlands

Pollutant	External cost 2008 at given height of release		
	Low	Unknown	High
NH ₃	- 200.01	- 200.01	- 200.01
NMVOOC	461.76	461.76	461.76
NO _x	- 266.69	-266.69	- 187.48
SO ₂ primary particles	- 41.04	- 41.04	-69.70
SO ₂ secondary particles	-70.91	-72.11	-74.51

Note: Values in bold indicate that the values for low/high height of release differ from those for unknown height of release.

Source: CASES, 2008.

Note that in some cases negative values are reported to account for the fact that up to a certain concentration certain substances (specifically, nitrogen and sulphur dioxide) have a beneficial impact: they act as fertilisers. Negative values can thus be interpreted as benefits.

With regard to the development of the values over time, up to the year 2014 the values are the same in constant prices; from 2015 they differ because of the different background concentrations assumed in the Ecosense model from the year 2015 onwards.

C.3.5 Impacts on buildings and capital goods

Air pollutants emitted by the burning of fossil fuels have a serious impact on buildings. The most important pollutants in this category are sulphur and nitrogen compounds, including secondary pollutants and particulates (primarily SO₂, NO₂, O₃ and their reaction products). The effects include loss of mechanical strength, leakage and failure of protective coatings due to the degradation of materials. Within the two-step approach adopted in ExternE, the exposure-response functions link the ambient concentration or deposition of pollutants to the rate of material corrosion, and the rate of corrosion to the time of replacement or maintenance of the material. Performance requirements determine the point in time at which replacement or maintenance is deemed to become necessary. This point is given in terms of critical degradation (ExternE, 2005).



In EcoSenseWeb a CRF has been implemented for the following materials:

- limestone;
- sandstone;
- natural stone, mortar, rendering;
- zinc and galvanised steel;
- paint on steel;
- paint on galvanised steel;
- carbonate paint.

The monetary values used for evaluating impacts to building materials are based on replacement and repair costs and are reported in Table 19.

Table 19 Monetary values used for the evaluation of impacts to building materials

Materials (maintenance costs per m ²)	Monetary value (€ ₂₀₀₀)
Galvanised steel	Country-specific (17-55)
Limestone	299
Mortar	33
Natural stone	299
Paint	13
Rendering	33
Sandstone	299
Zinc	27

Source: NEEDS, 2008a.

Average values for the impact of NO_x and SO₂ emissions generated at unknown, low and high heights of release in 2008 on buildings and materials are reported below in Table 20 and Table 21. Table 20 gives the values estimated for the EU-27, Table 21 those for the Netherlands.

Table 20 External costs of damage to buildings and materials per tonne of specific classical air pollutants (Euro₂₀₀₈), EU-27 average

Pollutant	External cost 2008 at given height of release		
	Low	Unknown	High
NO _x	84.13	85.33	85.33
SO ₂	344.92	311.27	277.62

Note: Values in bold indicate that the values for low/high height of release differ from those for unknown height of release.

Source: CASES, 2008.

Table 21 External costs of damage to buildings and materials per tonne of specific classical air pollutants (Euro₂₀₀₈), values for the Netherlands

Pollutant	External cost 2008 at given height of release		
	Low	Unknown	High
NO _x	84.13	99.75	98.55
SO ₂	344.92	543.21	592.49

Note: Values in bold indicate that the values for low/high height of release differ from those for unknown height of release.

Source: CASES, 2008

These values remain unchanged regardless of assumptions about the year of emission.



C.3.6 Impacts on ecosystems

Within ExternE, the environmental impact of air pollution on biodiversity has been estimated for emissions of SO₂, NO_x and NH₃. This impact is associated with acidification and eutrophication of soils. An approach using the measure 'potentially disappeared fraction' (PDF), i.e. biodiversity losses due to acidification and eutrophication, was used (NEEDS, 2008a).

Acidification is caused mainly by emissions of sulphur oxide (SO_x), nitrogen oxides (NO_x) and ammonia (NH₃) and the attendant deposition of acidifying substances like H₂SO₄ as well as a range of sulphates. Eutrophication due to airborne pollutants is due mainly to NO_x and NH₃.

Concentration-response function

For any given land use type, a certain average number of plant species can generally be established. If the soil becomes polluted due to deposition of acidifying and eutrophying substances, the number of species present and thus biodiversity are reduced. Hence, a delta PDF per deposition can be calculated.

In EcoSense the following information is used to model the loss of biodiversity due to SO₂, NO_x and NH₃ emissions, using the following data:

- Values of PDF per deposition of N and S on natural soils are taken from Ott et al. (2006); see Table 22.
- Depositions per 50 x 50 km² grid cell are available from regional dispersion modelling.
- In addition, for each grid cell the share of natural soil is available.
- Finally, a (country-dependent) 'pressure index' is used to account for differences in soil sensitivity.

Table 22 PDF per deposition of N and S on natural soil

Air pollutant	Deposition increase in kg/m ² * year on natural soil (10 mol/ha)	Average PDF of natural land for the Netherlands	PDF * m ² * year per kg deposition
Reference Value (Background Level)	--	0.746429	--
SO _x	6.4 * 10 ⁻⁵	0.74654	1.73
NO _x	4.6 * 10 ⁻⁵	0.746867	9.52
NH ₃	1.7 * 10 ⁻⁵	0.74687	25.94

Source: NEEDS, 2008a.

Monetary valuation

Within the NEEDS project, the evaluation has been updated according to an average WTP for Europe from Kuik et al. (2008) of €0.47/PDF/m² (2004 prices) rather than the value of € 0.45/PDF/m² used previously in ExternE, which was based on minimum restoration costs. Since the new value is based on WTP studies, it has been adopted here as more suitable for external cost calculation.

The approach described above is designed to be applicable on a European scale and will not therefore reflect very specific local conditions, e.g. the occurrence of species that are unique to a single location. However, the values proposed may serve as a first approximation.



Table 23 and Table 24 report estimated values of the external costs associated with the impact of the specific pollutants on biodiversity, for emissions generated at unknown/low and high height of release (for this category the values for unknown and low height are the same) occurring in the year 2008, discounted to the year of emission. Table 23 gives the values estimated for the EU-27, Table 24 those for the Netherlands.

Table 23 External costs of impacts on biodiversity loss per tonne of specific classical air pollutants emitted in 2008 (€₂₀₀₈), EU-27 average

Pollutant	External cost 2008 at a given height of release	
	Unknown/low	High
NH ₃	4,688.22	4,688.22
NMVOOC	-97.35	-97.35
NO _x	1,295.54	1,190.98
SO ₂	253.58	186.28

Note: Values in bold indicate that the values for high height of release differ from those for unknown/low height of release.

Source: CASES, 2008.

Table 24 External costs of impacts on biodiversity loss per tonne of specific classical air pollutants emitted in 2008 (€₂₀₀₈), values for the Netherlands

Pollutant	External cost 2008 at a given height of release	
	Unknown/low	High
NH ₃	4,859.77	4,859.77
NMVOOC	-87.30	-87.30
NO _x	1,479.46	1,442.16
SO ₂	453.18	1,002.30

Note: Values in bold indicate that the values for high height of release differ from those for unknown/low height of release.

Source: CASES, 2008.

As can be seen in the tables, the impact of NMVOOC (non-methane volatile organic compounds) on biodiversity is on average beneficial, resulting in negative damage values.

Development of the values related to biodiversity loss over time is conditional on the same factors as values for human health impact, i.e. on uplift factor (always positive) and on a change of background emissions assumed in the Ecosense model after the year 2014 (this factor causes an increase of some of the values and a decrease of others).

C.3.7 Impacts not covered in the damage estimates

Classical air pollutants may have two other impacts which are not included in most estimates, nor in this handbook. These are:

- impacts on cultural heritage;
- impacts on visibility.

It should be noted that the value of these impacts is probably very small compared to the total estimates of damages due to these pollutants. However, if a SCBA is oriented specifically towards these impacts (or if these impacts are to be deemed large), additional estimates may be provided. Some rough estimation procedures for this purpose are now given.



Cultural heritage

It is worth noting that the NEEDS estimates cited above do not capture any historical or artistic values that can be attached to certain buildings, so that the reported values can be regarded as a lower bound reflecting only use values related to 'regular' buildings and materials. To date, no database exists that could link existing relevant valuation studies with all the historical and artistic objects across Europe. An estimated figure for such values can be derived from empirical studies focusing on specific objects and monuments, or at least on the specific categories of such objects.

Maintenance costs for historical buildings are much more variable than the maintenance costs for 'ordinary' houses. The range of Willingness-to-Pay values for preserving cultural heritage reported in different studies reviewed in ExternE (2005) is very broad: from €1 to €86 a year (this range reflects different methodologies, however, and covers both individual and household WTPs).

Rabl (1999) has proposed a method for calculating damage values for cultural heritage, whereby he claims that for a typical case the amenity loss can be assumed to be approximately equal to soiling costs (for a zero discount rate). Hence, the total damage costs can be estimated to be twice the value of the cleaning costs. The key assumptions in this approach are: 1) the amenity loss is restored by renovation (cleaning and repair); 2) people minimise the total cost; and 3) the decision to clean or repair a given object is made by the people suffering the amenity loss. The proposed function relates the soiling damage at receptor location i to the concentration of total suspended matter (TSP):

$$S_i = a * P_i * \Delta TSP$$
$$\text{and } a = b * 2$$

where S_i is annual soiling damage at receptor location i , P_i is the number of people at location i , ΔTSP is a change in annual average concentration of Total Suspended Particles in $\mu\text{g}/\text{m}^3$, a is WTP per person per year to avoid soiling damage of $1 \mu\text{g}/\text{m}^3$, and b stands for cleaning costs per person per year due to a concentration of $1 \mu\text{g}/\text{m}^3$ of TSP.

Visibility

Visibility impacts are associated with a reduction in visual range due to the presence of pollutants, especially NO_2 , in the atmosphere. In Europe this issue has received relatively little attention, but in the United States there has been a comprehensive study on the topic (ABT, 2000), with the goal of estimating a general relationship between improvement in visibility and the average value placed on such improvements. A plausible relationship was found regarding recreational value in national parks. In addition, a relationship between visibility changes in residential areas and household values for these changes was established. Both relationships were assessed with respect to relative rather than absolute visibility ranges.

Annual values per household for visibility improvements depend very much on location. For example, for a 20% improvement in visibility, values for recreational use were estimated in the range USD11-19 (1999 prices), and values for residential areas in the range of USD24-278 (1997 prices) (ABT, 2000).

Based on this information, it is impossible to relate visibility impacts to a certain dose (e.g. tonne) of specific pollutants. Using values for relative visibility changes based on the US studies in Europe would be inappropriate,



because of major uncertainties in benefit transfer. ExternE (2005) suggests that in the absence of comprehensive European studies on this topic, policy measures with discernible effects on visibility should be assessed in qualitative terms only.

C.3.8 Summarising tables

Table 25 and Table 26 report the total damage costs associated with classical pollutants. In these tables, the values reflecting impacts of specific pollutants emitted in 2008 on human health (including Northern Hemispheric Modelling, NHM), crops, materials and biodiversity have simply been summed. Table 25 gives the values estimated for the EU-27, Table 26 those for the Netherlands.

Table 25 Total impact of the specific pollutants (€₂₀₀₈ per tonne emission), discounted to year of emission 2008, EU-27 average

Pollutant	External cost 2008 at a given height of release		
	Low	Unknown	High
NH ₃	17,512.73	17,512.73	17,512.73
NMVOG	1,424.25	1,424.25	1,424.25
NO _x	9,643.11	9,644.31	7,332.05
PPMco	2,029.12	1,824.81	675.89
PPM _{2.5}	35,833.42	33,791.57	17,073.35
SO ₂	9,274.64	9,247.00	8,485.06

Note: Values in bold indicate that the values for low/high height of release differ from those for unknown height of release.

Source: Tables in Sections C.3.2 - C.3.6, own calculations.

Table 26 Total impact of the specific pollutants (€₂₀₀₈ per tonne emission), discounted to year of emission 2008, values for the Netherlands

Pollutant	External cost 2008 at a given height of release		
	Low	Unknown	High
NH ₃	27,773.99	27,773.99	27,773.99
NMVOG	2,536.91	2,536.91	2,536.91
NO _x	10,570.76	10,586.38	8,646.82
PPMco	3,857.05	3,845.65	1,960.61
PPM _{2.5}	65,558.70	64,752.80	29,925.38
SO ₂	15,223.47	15,379.52	12,428.16

Note: Values in bold indicate that the values for low/high height of release differ from those for unknown height of release.

Source: Tables in Sections C.3.2 - C.3.6, own calculations.

C.3.9 Emissions from transport

The emissions presented in this handbook are averages for an entire country, viz. the Netherlands, and may not be representative of true damages at the local level, owing to differences in:

- population density: in densely populated areas damages will obviously be greater than in more sparsely populated areas;
- the pollution source: this is especially relevant for particulate emissions, as transport emissions, emitted at lower ambient levels, will be more harmful than those from point sources higher up.

Within the NEEDS project, no specific values for transport emissions have been estimated. Such estimates have been produced in several other projects, though, for example in the IMPACT handbook (CE, 2008b) based on various HEATCO studies. Here, we reproduce the estimates from the IMPACT study for



human-health effects due to emissions of PM_{2.5}; see Table 27 for the estimates for emissions in the Netherlands. The value of € 65,558.70 estimated in the NEEDS project for low height of emission (and transport can be assigned to this category) would be substantially higher if population density and emissions source are taken into account.

Table 27 Air pollution costs in €/tonne of pollutant for road, rail and waterways (€₂₀₀₈ per tonne pollutant) from HEATCO and NEEDS

Pollutant	PM _{2.5} (exhaust)			PM ₁₀ (non-exhaust)		
	Urban metropolitan	Urban	Outside built-up areas	Urban metropolitan	Urban	Outside built-up areas
The Netherlands	507,759.83	163,925.30	99,268.55	203,103.93	65,498	39,659.35
Low height of emission (NEEDS)	65,558.70					

* Urban metropolitan: cities with over 0.5 million inhabitants.

** Urban: smaller and medium-sized cities with up to 0.5 million inhabitants.

Source: CE, 2008b.

The figures in Table 27 need some explanation, specifically for comparison with the NEEDS estimates:

- Within the NEEDS project it is assumed that primary and secondary particles are equally harmful per unit mass, irrespective of particle composition. HEATCO treats the various particle fractions and sources as follows: (1) nitrates as equivalent to 0.5 times the toxicity of PM₁₀; (2) sulphates as equivalent to PM₁₀ (or 0.6 times PM_{2.5}); (3) primary particles from power stations as equivalent to PM₁₀; (4) primary particles from vehicles as equivalent to 1.5 times the toxicity of PM_{2.5}.
- Within NEEDS a general dispersion model is used for PM emissions, while HEATCO presents values differentiated by urban and interurban traffic situations.

The values used in NEEDS and HEATCO for VOLY are largely the same, however. Within HEATCO, for acute exposure a value of € 60,500 per year of life lost (YOLL) is taken and for chronic mortality € 40,300, both very similar to the values used in NEEDS (€ 60,000 and € 40,000, respectively).

In addition to the detrimental impact of air pollutants on human health, external costs of transport include congestion, noise and accidents. For estimates of these effects, the reader is referred to the IMPACT handbook and/or source publications cited within the IMPACT handbook.

C.4 Human toxicity

Human toxicity potential (HTP) is an indicator used to compare the relative importance of toxic emissions. The HTP is a characterisation method used to weight emissions, such as those listed in life-cycle inventories and toxic release inventories. HTP is a screening-level risk indicator that accounts for both toxicity and the potential to result in exposure, but this does not generally account for regional differences or variations in population density. Within the HTP indicator, over 300 substances are classified according to the potential damage they can cause.



In Section C.3. we established a shadow price for PM_{2.5}. Since the underlying damage under the heading ‘human toxicity’ relates by definition to human toxic impacts only, one would anticipate being able to use the characterisation factors (in DALY) to establish the potential damages from other pollutants, too. This would yield a relatively simple approach that would, from a methodological point of view, be transparent. However, this choice depends crucially on the reliability of the particular endpoint factors being used. Moreover, since there are several small differences in the background documentation regarding population size, age cohorts and years of reference used in both the estimation for PM_{2.5} and the characterisation factors used in the ReCiPe project, this approach will not yield entirely consistent estimates for HTP. In this handbook we have therefore chosen to add our own estimates for a selected number of additional toxic pollutants, to extend the coverage of our damage estimates. These additional estimates were also developed in the ExternE projects.

C.4.1 Methodology

Within the NEEDS project, damage costs have been established for several toxic pollutants, viz. heavy metals, formaldehyde and dioxins. Country-specific results used in the NEEDS project regarding the inhalation pathway for heavy metals (As, Cd, Cr, Ni, Pb) have been calculated in the ESPREME project (ESPREME, 2007), with country-specific values regarding ingestion being calculated for As, Cd and Pb in the NEEDS project (Fantke, 2008). The Excel tool developed in NEEDS also includes values for mercury, formaldehyde and dioxins based on various studies. These are so-called generic values, expressed directly as ‘Euro per tonne’. As these are European averages, they are applicable to all the countries of Europe and any height of release.

The value for Cr-VI is derived from the value for Cr. It is assumed that Cr-VI is the only toxic form of chromium and that this accounts for approx. 20% of environmental chromium. Hence, the monetary value for Cr-VI is around 5 times that for Cr. Consequently, damage costs for either Cr or Cr-VI must be used, and not both.

CRF for inhalation of heavy metals can be found in ESPREME (2007), Spadaro and Rabl (2008) and MethodEx (2006). Country-specific external costs associated with inhalation of heavy metals are included in the EcoSense model.

For As, Cd and Pb there has been a parameterisation of new WATSON model runs including ingestion of food and drinking water due to emissions into air. This is described in Fantke (2008). The CRF functions associated with As, Cd and Pb are summarised in Table 28 and Table 29.



Table 28 Physical endpoints of human health effects due to ingestion of food, as used in the parameterisation procedure (Fantke 2008)

Pollutant	End-Point	Risk Group	Expos. Time	Unit Risk	Severits Measure			Costs per Case	
					Fatality	YOLL	YLD		IQ Points
-	Name	-	(years)	(risk/kg _{intake})	-	-	-	(€/case)	
As	Skin cancer	85%	All	70	7.85E-01	5.18	0.16	0	213,520
As	Bladder cancer	85%	All	70	3.91E+00	3.94	0.31	0	169,660
As	Cardiovascular mortality	100%	All	35	2.35E+02	12.80	1.00	0	512,000
As	Still birth (babies)	100%	All	1	2.86E+02	0.00	1.28	0	51,200
Cd	Osteoporosis	100%	All	35	6.26E+02	0.00	0.28	0	11,200
Cd	Renal dysfunction	100%	All	35	3.13E+01	0.00	0.64	0	25,600
Pb	Anaemia	100%	All	1	1.32E+02	0.00	0.64	0	25,600
Pb	IQ points loss (children)	100%	0-1 years	1	1.15E+03	0.00	0.00	1	8,600

Source: Fantke, 2008.

Table 29 Physical endpoints of human health effects due to ingestion of drinking water, as used in the parameterisation procedure

Pollutant	End-Point	Risk Group	Expos. Time	Unit Risk	Severits Measure			Costs per Case	
					Fatality	YOLL	YLD		IQ Points
-	Name	-	(years)	(risk/kg _{intake})	-	-	-	(€/case)	
As	Skin cancer	85%	All	70	7.85E-01	5.18	0.16	0	213,520
As	Bladder cancer	85%	All	70	3.91E+00	3.94	0.31	0	169,660
As	Cardiovascular mortality	100%	All	35	2.35E+02	12.80	1.00	0	512,000
As	Still birth (babies)	100%	All	1	2.86E+02	0.00	1.28	0	51,200
Cd	Osteoporosis	100%	All	35	6.26E+02	0.00	0.28	0	11,200
Cd	Renal dysfunction	100%	All	35	3.13E+01	0.00	0.64	0	25,600
Pb	Anaemia	100%	All	1	2.74E+02	0.00	0.64	0	25,600
Pb	IQ points loss (children)	100%	0-1 years	1	1.15E+03	0.00	0.00	1	8,600

Source: Fantke, 2008.

C.4.2 Monetary valuation

The monetary valuation of morbidity and mortality is based on multiplication of the YOLL (years of life lost) and the YLD (years lost due to disability) by the corresponding monetary value of a life year lost, i.e. € 40,000 per VOLY. The monetary valuation of IQ loss is based on Spadaro and Rabl (2008) and is € 8,600 per IQ point.

Table 30 and Table 31 provide a summary of monetary values per tonne of specific pollutants. These are values for the external costs associated with the impact of these pollutants on human health, for emissions occurring in the



year 2008, discounted to the year of emission. It should be noted that these values are relevant only for emissions of the substances to the atmosphere. As indicated in the previous section, however, the impact of some of these substances (As, Cd, Pb) has been evaluated based on models describing ingestion via food and water (i.e. not only via inhalation). Table 30 reports the values estimated for the EU-27, Table 31 those for the Netherlands. These values remain the same regardless of the height of release.

Table 30 Values of external costs associated with human health impacts per tonne of specific pollutants (€₂₀₀₈), EU-27 average

Pollutant	External cost 2008
Cd	115,149.41
As	728,378.29
Ni	3,164.38
Pb	382,726.15
Hg	11,002,445.72
Cr	18,224.47
Cr-VI*	91,122.34
Formaldehyde	275.06
Dioxin	5,09E+10

* The values for Cr and Cr-VI should be used separately and not both.

Source: CASES, 2008.

Table 31 Values of external costs associated with human health impacts per tonne of specific pollutants (€₂₀₀₈), values for the Netherlands

Pollutant	External cost 2008
Cd	126,863.64
As	811,080.11
Ni	5,366.44
Pb	407,702.57
Hg	11,002 445,72
Cr	33,461.19
Cr-VI*	167,305.94
Formaldehyde	275.06
Dioxin	5.09E+10

* The values for Cr and Cr-VI should be used separately and not both.

Source: CASES, 2008.

C.5 Eutrophication

Eutrophication is the nutrient enrichment of the aquatic environment (eutrophication of freshwater bodies) or terrestrial environment (eutrophication of soils).

C.5.1 Eutrophication of freshwater bodies

Nutrients that are limiting to the yield of aquatic biomass are phytoplankton (algae), but also duckweed. The growth of algae is governed mainly by the availability of phosphorus and nitrogen. As a result, additional influx of P and N may lead to increased algal growth, causing a chain of adverse ecological effects. The relevant substances include phosphorus and nitrogen compounds emitted to water and soils as well as ammonia (NH₃) and nitrogen oxides (NO_x) emitted to air. In temperate and subtropical regions of Europe, freshwaters



are typically limited by phosphorus, while in marine waters it is usually nitrogen that limits production of algal biomass.

Within the ReCiPe project, CARMEN was used to evaluate the fate factor for aquatic eutrophication due to nutrients emitted in Europe. It is thereby assumed that N is the limiting nutrient in all coastal waters and P in all freshwater bodies. In ReCiPe it is only a damage factor for eutrophication of freshwater due to the phosphorus emissions that is estimated. The resultant figures are reported in Table 32.

Table 32 Fate factors and eutrophication potentials (EP) for eutrophication of freshwaters

Emission type	Compartment	Fate factor yr/km ³	EP freshwater
Manure P	Soil	1.72x10 ⁻⁹	0.050
Fertiliser P	Soil	1.83x10 ⁻⁹	0.053
P from sewage treatment plants	Freshwater	3.44x10 ⁻⁸	1

Source: Own calculations based on ReCiPe 2008 (Goedkoop et al., 2009).

Monetary valuation

The potentially disappeared fraction (PDF) per m³ freshwater has been combined with the average WTP taken from (Kuik et al., 2008). The average WTP for terrestrial ecosystems was estimated at € 0.55/PDF/m². This can be translated to a valuation for freshwater ecosystems by using the terrestrial and freshwater species density from ReCiPe (Goedkoop et al., 2009). This would imply a value of € 0.03/PDF/m² for freshwater ecosystems. The resultant figures are reported in Table 33.

Table 33 Monetary valuation of eutrophication of freshwaters

Emission type	Compartment	PDF.yr.m ³ /kg	€/kg
Manure P	Soil	2.81	€0.09
Fertiliser P	Soil	2.99	€0.09
P from sewage treatment plants	Freshwater	56.3	€1.78

Source: Goedkoop et al., 2009.

C.5.2 Eutrophication of soils

Within ExternE, the environmental impact of air pollution on biodiversity due to emissions of SO₂, NO_x and NH₃ have been estimated. The impact is related to acidification and eutrophication of soils. An approach using the potentially disappeared fraction (PDF), i.e. biodiversity losses due to acidification and eutrophication is used (NEEDS, 2008a). See also Annex C.3. To divide the impacts of NO_x and NH₃ between soil acidification and eutrophication, we took the approach set out in Annex D.2.

C.6 Ozone depletion

Stratospheric ozone filters the UV-B radiation out of incoming sunlight, thus preventing UV damage at the planet's surface. Ozone is continually formed and destroyed by natural processes that depend on stratospheric chemical conditions and incoming sunlight. In recent decades, however, ozone destruction has increased due to anthropogenic emissions of chlorine- and bromine-containing molecules. These ozone-depleting substances (ODS) act as



catalysts in the ozone destruction reaction and as such are preserved and therefore have the potential to destroy many ozone molecules during their stratospheric residence time. The impacts of UV-B radiation on human health are discussed in Section C.6.2, impacts on nature and capital in Section C.6.3.

C.6.1 Methodology

The best-known group of anthropogenic ODP are probably the chlorofluorocarbons (CFCs), used mainly as aerosol propellants and for refrigeration. Other significant sources include hydrochlorofluorocarbons (HCFCs), halons (brominated chlorofluorocarbons), carbon tetrachloride (CCl₄), methylchloroform (CH₃CCl₃) and methylbromide (CH₃Br). After peaking in the early 1990s, total emissions of ODP had halved by the early 21st century and are expected to return to approximately natural levels by the middle of the century (WMO, 2003).

At the midpoint level, the relative potential of an ODS to destroy stratospheric ozone (Ozone Depleting Potential; ODP) is expressed in kg CFC-11-equivalents per kg ODS (based on WMO, 2003; Goedkoop et al., 2009).

Table 34 Midpoint characterisation factors of several ozone-depleting substances

Ozone Depleting Substance	kg CFC-11-equivalent/kg ODS
CFC-10	0.73
CFC-11	1
CFC-113	1
CFC-114	0.94
CFC-115	0.44
CFC-12	1
CH ₃ CCl ₃	0.12
Halon 1001	0.38
Halon 1201	1.4
Halon 1202	1.3
Halon 1211	6
Halon 1301	12
Halon 2311	0.14
Halon 2401	0.25
Halon 2402	6
HCFC-123	0.02
HCFC-124	0.02
HCFC-141b	0.12
HCFC-142b	0.07
HCFC-22	0.05
HCFC-225ca	0.02
HCFC-225cb	0.03
CCl ₄	0.73
CH ₃ Br	0.38

Source: Goedkoop et al., 2009.



Since ozone depletion is not part of the NEEDS project, no damage cost estimates were available for each specific ODS. Given that such estimates are also lacking in the literature, we opted to base the human health effect estimates on the ReCiPe (Goedkoop et al., 2009) methodology.³² Below, we briefly describe the calculation adopted in ReCiPe to obtain human health impact endpoints (in DALY/kg ODS).

In order to calculate the estimated damages, the cumulative ODP of each ODS is first translated into the Effective Equivalent of Stratospheric Chlorine (EESC), which expresses cumulative halogen loading in parts per trillion. The marginal effect of ODS emissions is captured by the fate factor, which expresses the additional chlorine release per kg ODS (ppt/kg CFC-11-eq.) based on the time-integrated effect of the ODS over its stratospheric residence time. The final damage estimate for the environmental theme can then be calculated by multiplying the fate factor by a damage factor, which depends on the specific theme and is described in the appropriate Sections below.

To obtain the shadow price for a specific substance, the endpoints need to be valued appropriately. In the case of human health effects, these were obtained by using a standard value for a VOLY and multiplying this by the endpoints for each substance. For capital effects, Hayashi et al. (2006) multiplied the endpoint damage for different crops by the estimated producer cost (based on NEEDS, 2008a).

C.6.2 Health-related impacts

UV-B radiation damages biological tissue. In humans, it leads to different kinds of skin cancer (carcinoma) and reduces the opacity of the lens of the eye (cataract). In ReCiPe the endpoint characterisation factors were calculated assuming marginal ODS reductions in 2003 and integrating health impacts until 2044 (when EESC is expected to reach pre-1980 levels and additional marginal emissions will not lead to additional human health loss). Here, we have assumed that marginal emission changes in 2008 result in damages similar to those in 2003. The effect of a change in UV-B radiation on human health was calculated using the AMOUR model (RIVM, 2007) which takes latitude, population density, skin-type distribution, age distribution and altitude into account. The resulting damage factor is expressed in DALY per unit change in EESC (in ppt). These factors can then be translated into characterisation factors in DALYs per tonne CFC-11-equivalent for each group of ODS by multiplication by the fate factor.

Table 35 Endpoint characterisation factor per ODS group

ODS group	Characterisation factor (DALY/t CFC-11-eq.)
CFCs	1.76
CCl ₄	3.30
CH ₃ CCl ₃	4.41
Halons	2.64
HCFC	3.65
CH ₃ Br	4.72

Source: Goedkoop et al., 2009.

³² Note that, compared with the methodology preferred in the remainder of this report, ReCiPe cost estimates yield lower values, as ReCiPe adheres to a stricter threshold for accepting scientific evidence on which estimates are based.



In order to calculate the endpoint damages per tonne of actual ODS emission (rather than CFC-11-equivalent), the grouped characterisation factors (Table 35) need to be multiplied by the characterisation factor for each specific substance at the midpoint (Table 34). This gives the endpoint characterisation factor in DALYs per emitted kg ODS (Table 36, column 2). Note that these reflect global damages.

Table 36 Endpoint characterisation factors for human health effects and external cost per ODS

Ozone Depleting Substance	Characterisation factor (DALY/kg ODS)	External cost, undiscounted (€ ₂₀₀₈ /kg)	External cost, discount rate 2.5% (€ ₂₀₀₈ /kg)
CFC-10	2.41E-03	132.60	51.87
CFC-11	1.76E-03	96.84	37.88
CFC-113	1.76E-03	96.84	37.88
CFC-114	1.65E-03	91.03	35.61
CFC-115	7.74E-04	42.61	16.67
CFC-12	1.76E-03	96.84	37.88
CH ₃ CCl ₃	5.29E-04	29.12	11.39
Halon 1001	1.00E-03	55.20	21.59
Halon 1201	3.70E-03	203.36	79.55
Halon 1202	3.43E-03	188.83	73.87
Halon 1211	1.58E-02	871.54	340.94
Halon 1301	3.17E-02	1743.08	681.88
Halon 2311	3.70E-04	20.34	7.96
Halon 2401	6.60E-04	36.31	14.21
Halon 2402	1.58E-02	871.54	340.94
HCFC-123	7.30E-05	4.02	1.57
HCFC-124	7.30E-05	4.02	1.57
HCFC-141b	4.38E-04	24.10	9.43
HCFC-142b	2.56E-04	14.06	5.50
HCFC-22	1.83E-04	10.04	3.93
HCFC-225ca	7.30E-05	4.02	1.57
HCFC-225cb	1.10E-04	6.02	2.36
CCl ₄	2.41E-03	132.55	51.85
CH ₃ Br	1.79E-03	98.69	38.61

Source: Goedkoop et al., 2009.

To obtain shadow prices in €/kg, this CF needs to be multiplied by a Value of a Life Year (VOLY). As discussed in Section 5.3, we use a VOLY of € 40,000, corrected for inflation and with an uplift factor to the year 2008. The resulting shadow prices per kg ODS are listed in column 3 of Table 36.

Note, however, that ozone depletion damages are not discounted in ReCiPe. The effects of UV exposure on human health are cumulative, i.e. the incidence of skin cancer cases resulting from increased ozone depletion will increase over time and then decrease again. It may therefore be more appropriate to discount future damages over time. To approximate the distribution of future damages resulting from present-day ozone depletion, we assumed a peak in health damages 40 years after ODS emission (Goedkoop et al., 2009), with half the cases occurring before this peak and half thereafter. The increase from now until the peak, as well as the subsequent decrease were assumed linear, resulting in a triangular distribution of health damages over time. These damages were discounted at 2.5 % and then monetarised using the same VOLY



as for the undiscounted values. The resulting discounted damage costs are reported in column 4 of Table 36.

C.6.3 Nature- and capital-related effects

UV-B radiation also affects non-human biological tissue. Specifically, an increase in radiation associated with ozone depletion will lead to lower Net Primary Production in ecosystems and lower crop yields in agricultural settings. Damages to nature and capital per kg ODS emission have been estimated by Hayashi et al. (2006), who expanded the LIME model to incorporate a damage function for ozone-depleting substances. Similarly to the ReCiPe methodology, a fate factor was determined that translates the ODP of each ODS into EESC and calculates a corresponding increase in UV-B radiation at different latitudes.

For crop damages, the fate factor was then multiplied by a damage factor reflecting crop reduction per unit UV-B increase to yield an endpoint characterisation factor for different ODS (in kg crop yield reduction per kg ODS). Producer cost estimates for specific crops were taken from NEEDS (2008a) and multiplied by endpoint characterisation factors to obtain a figure for the external costs per kg ODS.

Similarly, reduced timber yield was calculated by multiplying the fate factor by a damage factor reflecting decreased Net Primary Production per unit UV-B increase. The resulting endpoint characterisation factor was multiplied by producer cost estimates from Japan to obtain external cost per kg ODS.

The external costs reported in Hayashi (2006) were expressed in Yen₂₀₀₀/kg ODS. Here, these values were converted to Euro₂₀₀₀ by using the average exchange rate over 2000 (99.3 Yen = 1 Euro), and corrected for inflation and with an uplift factor to the year 2008. The final external cost in €/kg emitted ODS corresponds with our endpoint for agricultural and timber productivity and is reported in Table 37.



Table 37 External costs for social assets per Ozone Depleting Substance

Ozone Depleting Substance	Characterisation factor (DALY/ kg ODS) ³³	External cost, timber (€ ₂₀₀₈ /kg ODS)	External cost, crops (€ ₂₀₀₈ /kg ODS)
CFC-10	n.a		
CFC-11		1.14	0.11
CFC-113		1.22	0.12
CFC-114	1.00	1.14	0.11
CFC-115	0.60	0.69	0.06
CFC-12		1.20	0.11
CH ₃ CCl ₃	0.07	0.08	0.01
Halon 1001	n.a		
Halon 1201	n.a		
Halon 1202	n.a		
Halon 1211		2.89	0.27
Halon 1301		16.76	1.58
Halon 2311	n.a		
Halon 2401	n.a		
Halon 2402	5.04	5.76	0.54
HCFC-123		0.00	0.00
HCFC-124		0.02	0.00
HCFC-141b		0.10	0.01
HCFC-142b		0.07	0.01
HCFC-22		0.05	0.00
HCFC-225ca	0.02	0.02	0.00
HCFC-225cb	0.02	0.03	0.00
CCl ₄		1.10	0.10
CH ₃ Br		0.01	0.00

Source: Hayashi et al., 2006.

C.6.4 Discussion

The most important health effect of ozone depletion is an increased incidence of skin cancer. Cancer is regarded as a terrible disease that most people are likely to want to avoid more than any other. Hence they may well cite a higher WTP to avoid living with a cancer than with most other diseases. This premium value for a VOLY for cancer-related diseases was not taken into account here, but may increase the external cost for ODS. Our figures may therefore represent something of an underestimate. On the other hand, human health effects resulting from ODS emissions reflect global damages: the extent and location of ozone depletion are independent of the location of emission. By taking the EU VOLY of € 40,000 independent of the location of the damages, an overestimate of the true damage costs may occur, as WTP for additional life years may be correlated with GDP.

³³ The LIME model estimates do not include capital estimates for each ODS treated in ReCiPe. However, health damage estimates are reported for most substances. Since the underlying fate factor used for health and capital effects is the same, the damage ratio between different substances is the same for capital effects and health effects. Where available, the health damage ratio between a specific substance and CFC-11 was calculated. Subsequently, the CFC-11 endpoint characterisation factor for crop (timber) damages was multiplied by this ratio to obtain external cost estimates for these substances. This calculation was applied to CFC-114, CFC-115, CH₃CCl₃, Halon 2402, HCFC-225ca and HCFC-225cb.



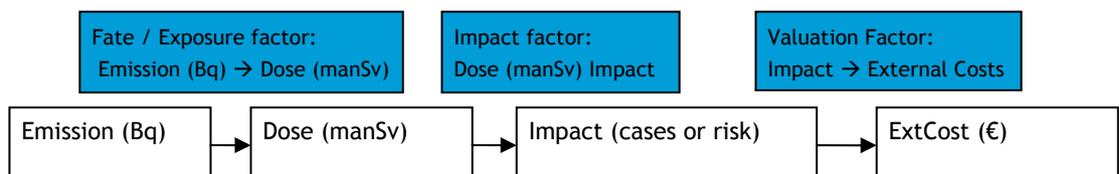
The estimated external costs for health effects exceed those for capital damages by around two orders of magnitude. A similar situation was observed for the group of classical pollutants, where impacts on human health are by far the most costly compared with any other endpoint. This is not that surprising when it is taken into account that valuation of a statistical human life (measured using VSL or VOLY) that is lost due to the impact of a given pollutant is always higher than the valuation of the capital damages caused by the same dose of the pollutant. Within capital damages, the external cost estimates for timber production is about one order of magnitude greater than the combined crop damages. The difference between crops and timber may be explained by the fact that Hayashi et al. used Japanese producer costs to estimate global timber damages. As these costs are likely to be higher in Japan than the average global figure, they may be an overestimate.

Note that increased UV-B radiation may also cause material degradation and ecosystem damage. However, monetary valuations per unit of ODS are not yet available for these environmental themes. Hence, the external costs presented here form an underestimate of the true external costs.

C.7 Ionising radiation

The subatomic particles and electromagnetic waves radiating from certain materials carry enough energy to detach electrons from other atoms or molecules, a process called ionisation. When living tissue is exposed to ionising radiation, it may suffer DNA damage, leading to apoptosis or genetic mutation, which may eventually lead to the development of cancers as well as hereditary defects passed on to subsequent generations. The amount of ionising radiation resulting from radionuclide emissions is measured in Becquerel (Bq), which expresses the number of nucleus decays per second. In NEEDS (2008a) the following simplified approach has been adopted to calculate the external costs of radionuclide emissions.

Figure 5 Scheme of assessment of exposure, physical impact and external costs due to release of radionuclides (from Needs 2008)



C.7.1 Health-related effects

The fate and exposure factors used in NEEDS have been calculated using the methodology set out in UNSCEAR (1993, 2000), whereby radionuclide emissions (in Bq) are related to the 'equivalent radiation dose' at the population level. This equivalent dose is expressed in man-Sievert (manSv), which is calculated by multiplying the amount of absorbed radiation (in J/kg) by a 'quality factor' that depends on radiation type (e.g. photons vs. alpha particles) and a factor that takes into account the exposed part of the body, the duration and level of irradiation. The resulting combined fate and exposure factors in manSv/PetaBecquerel (PBq; 10^{15} Becquerel) are listed in Table 38. As can be seen from these data, the human radiation exposure associated with emissions depends on the medium to which the radionuclide was emitted.

Table 38 Endpoint damages to human health caused by radionuclide emissions



Radionuclide	Emitted to	Dose (manSv/PBq)	Fatal cancers (cases/PBq)	Non-fatal cancers (cases/PBq)	Hereditary defects (cases/PBq)
Aerosols, radioactive, unspecified	Air	2,000	100	240	20
Carbon-14	Air	92,270	4,614	11,072	923
Carbon-14	Water	1,000	50	120	10
Cesium-137	Air	7,400	370	888	74
Cesium-137	Water	98	4.90	11.76	0.98
Hydrogen-3, Tritium	Air	4.1	0.21	0.49	0.04
Hydrogen-3, Tritium	Water	0.85	0.04	0,10	0.01
Iodine-129	Air	64,000	3,200	7,680	640
Iodine-131	Air	20,300	1,015	2,436	203
Iodine-131	Water	63,438	3,172	7,613	634
Iodine-133	Air	0	0	0	0
Iodine-133	Water	0	0	0	0
Iodine-135	Air	0	0	0	0
Krypton-85	Air	0.214	0.01	0.03	0.00
Krypton-85	Water	0	0	0	0
Krypton-85m	Air	0	0	0	0
Noble gases, radioactive, unspecified	Air	0.43	0.02	0.05	0.00
Radon-222	Air	2,5	0.13	0.30	0.03
Thorium-230	Air	30,000	1,500	3,600	300
Thorium-230	Water	0	0	0	0
Uranium-234	Air	8,000	400	960	80
Uranium-234	Water	198	9.90	23.75	1.98
Uranium-235	Air	0	0	0	0
Uranium-235	Water	0	0	0	0
Uranium-238	Air	7,000	350	840	70
Uranium-238	Water	1,963	98	236	20
Strontium-90	Water	4.7	0.24	0.56	0.05
Rubidium-106	Water	3.3	0.17	0.40	0.03
Lead-210	Air	1,000	50	120	10
Polonium-210	Air	1,000	50	120	10
Radium-226	Air	600	30	72	6

Source: CASES, 2008.

The health impacts of radiation absorption may manifest themselves in the form of fatal and non-fatal cancers and hereditary defects. It is estimated that each manSv equivalent radiation dose leads to 0.05 cases of fatal cancers, 0.12 cases of non-fatal cancers and 0.01 cases of hereditary defects (see NEEDS, 2008a). For each of these, the expected number of cases per unit emission are shown in Table 38, columns 3-5, for each of the relevant radionuclides.



The valuation of these impacts was based on the number of DALYs per cancer. For fatal cancers, the resulting YOLL (15.95) was multiplied by a VOLY of € 40,000 and the Cost of Illness (COI; € 481,050) was added, summing to € 1.12 million. For non-fatal cancers, the COI of € 481,050 was used. For hereditary effects, a standard value of statistical life (VSL) was taken, summing to € 1.5 million per case.

The external cost per unit emission was calculated by multiplying the disease-specific valuations by the expected number of diseases, shown in Table 38. As described in Annex A, in NEEDS an uplift factor is applied to account for the positive income elasticities of demand (1.7 % until 2030, 0.85% thereafter), and a discount factor of 3% until 2030 and 2% thereafter. Importantly, the radiation emitted by a certain substance changes over time, depending on its half-life. This should be corrected for in the uplift and discount factors, which in NEEDS was only done for Rn-222, H-3 and C-14 (the most prevalent emissions associated with nuclear fuel cycles). The resulting Net Present Values of emissions in the year 2008 are listed in Table 39.

Table 39 External costs of radionuclide emissions

Radionuclide	Emitted to	€ ₂₀₀₈ /PBq NPV 2008
Aerosols, radioactive, unspecified	Air	3.54E+08
Carbon-14	Air	1.92E+09
Carbon-14	Water	1.29E+07
Cesium-137	Air	1.31E+09
Cesium-137	Water	1.74E+07
Hydrogen-3, Tritium	Air	7.02E+05
Hydrogen-3, Tritium	Water	1.51E+05
Iodine-129	Air	1.13E+10
Iodine-131	Air	3.59E+09
Iodine-131	Water	1.12E+10
Iodine-133	Air	5.17E+05
Iodine-133	Water	0.00E+00
Iodine-135	Air	0.00E+00
Krypton-85	Air	3.79E+04
Krypton-85	Water	0.00E+00
Krypton-85m	Air	0.00E+00
Noble gases, radioactive, unspecified	Air	7.61E+04
Radon-222	Air	1.99E+04
Thorium-230	Air	5.31E+09
Thorium-230	Water	0.00E+00
Uranium-234	Air	1.42E+09
Uranium-234	Water	3.50E+07
Uranium-235	Air	1.16E+09
Uranium-235	Water	1.27E+08
Uranium-238	Air	1.24E+09
Uranium-238	Water	3.48E+08
Strontium-90	Water	8.32E+05
Rubidium-106	Water	5.84E+05
Lead-210	Air	1.77E+08
Polonium-210	Air	1.77E+08
Radium-226	Air	1.06E+08

Source: Needs, 2008a.



C.7.2 Nature- and capital-related effects

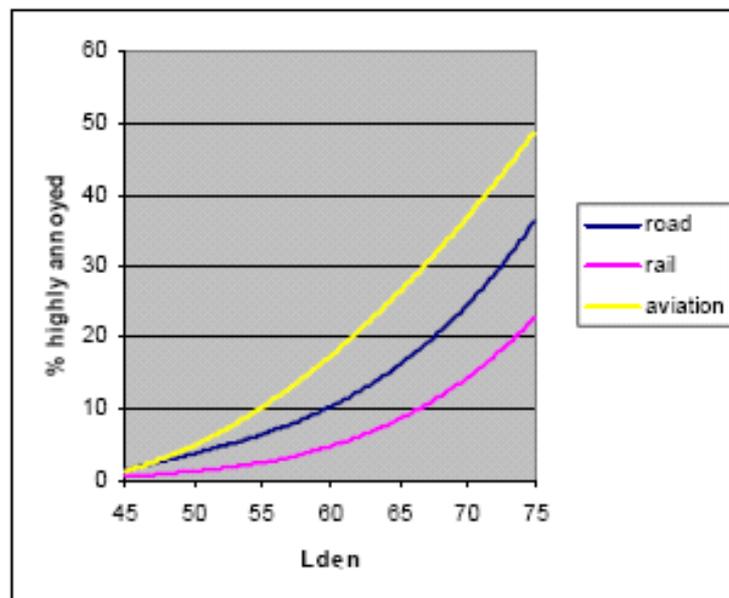
Radiation exposure also affects non-human organisms, and has a detrimental effect on social assets (e.g. it may cause malfunctioning in electronic equipment). No monetary valuation of these effects was available from the literature, and the external costs presented here are therefore an underestimate of the true costs.

C.8 Noise

Noise is often expressed as L_{den} [in dB(A)], a measure of the number of decibels during the year, with a 'penalty' for the evening and night. However, perceptions of noise and noise annoyance depend not only on the sound, but also on such characteristics as its pitch and predictability and personal factors like noise sensitivity and age.

The so-called Miedema curve (see Figure 6) shows that for any given noise level, a higher percentage of people are annoyed by road noise than by rail noise. Noise from aircraft is most annoying.

Figure 6 The Miedema curve for annoyance due to various forms of noise



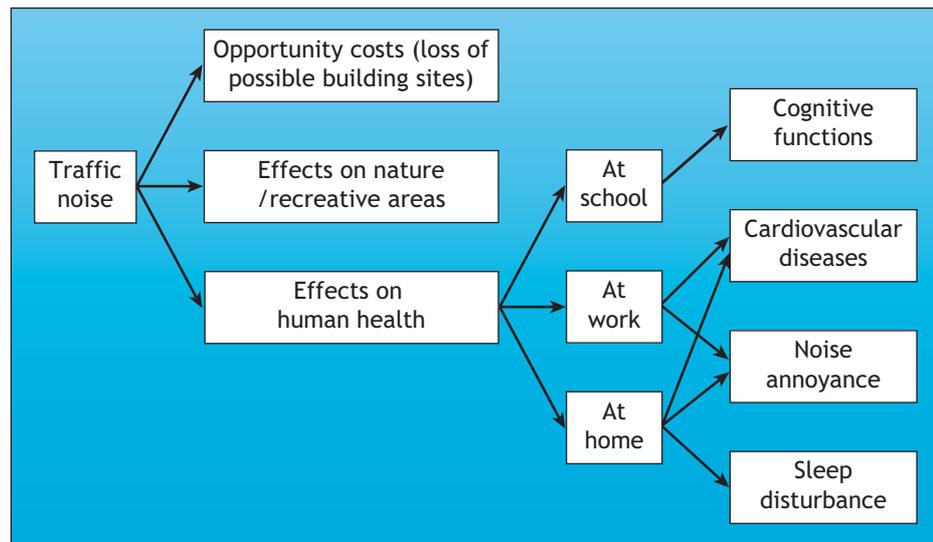
C.8.1 Methodology

As set out in Figure 7 below, the damaging effects of traffic noise are:

- loss of possible building sites due to government regulations;
- effects on ecosystems or recreation areas;
- effects on human health at school, at work or at home.

Most studies only value these last effects, which are also relevant for other types of noise (i.e. due to nearby industrial activities).

Figure 7 Damages associated with traffic noise



The two main impacts of noise on human health are (HEATCO, 2006):

- annoyance, reflecting the disturbance that individuals experience when exposed to noise;
- health impacts: mainly stress-related health effects like hypertension and myocardial infarction.

The value of annoyance caused by noise can be estimated using the stated preference (SP) or hedonic pricing (HP) method. In the first of these, people are asked about their behavior in a hypothetical situation; this is therefore a direct way to determine individuals' willingness-to-pay (WTP). In the hedonic pricing method, WTP is determined from analyses of the impact of noise on house prices.

C.8.2 Monetary valuation

The HEATCO project, funded by the EC, analysed the latest research to provide a consistent framework for monetary valuation of transport projects. This included the valuation of environmental issues like noise. The guidelines developed within the HEATCO project take annoyance and health impacts as two independent effects (assuming the health risk is not taken into account in people's perceived noise annoyance). To estimate the value of noise annoyance, stated preference surveys were carried out in five European countries. Benefit transfer was applied to derive national noise annoyance valuations. Quantifiable health costs were added to these values to derive a total value for noise. See Table 40.

Table 40 Values for noise exposure in 2002 PPP € prices (per year per person exposed) for the Netherlands

L _{den} dB(A)	Road	Rail	Aircraft
≥51	10	0	15
≥52	20	0	30
≥53	29	0	46
≥54	39	0	61
≥55	49	0	76
≥56	59	10	91
≥57	69	20	106
≥58	79	29	122
≥59	88	39	137
≥60	98	49	152
≥61	108	59	167
≥62	118	69	183
≥63	128	79	198
≥64	137	88	213
≥65	147	98	228
≥66	157	108	243
≥67	167	118	259
≥68	177	128	274
≥69	186	137	289
≥70	196	147	304
≥71	261	212	374
≥72	277	228	396
≥73	293	244	417
≥74	310	261	439
≥75	326	277	461
≥76	343	294	483
≥77	359	310	505
≥78	375	326	527
≥79	392	343	548
≥80	408	359	570
≥81	425	376	592

Source: HEATCO, 2006.

C.9 Land use

The main impact of land use is damage to ecosystems due to the effects of land occupation and transformation. This section explores the effects of occupation of a certain area of land for a certain period of time.

C.9.1 Methodology

To value ecosystems we adopted the NEEDS approach, which is based on work by NEEDS (2008a). This approach compares the species' abundance of a specific land use (S_i) to the species' abundance of a reference land use (S_{ref}). Species' abundance is measured as the number of vascular plant species per square metre. The reference land use is a composite of various land uses occurring in the Swiss lowlands.



The NEEDS approach uses the inverse of the relative species' abundance, which is termed the Potentially Disappeared Fraction (PDF):

$$PDF_i = 1 - \frac{S_i}{S_{ref}}$$

For valuing land use, we used the approach developed by Kuik *et al.* (2008) in the CASES project. In that project, PDF was defined in terms of Ecosystem Damage Potential (EDP). On the basis of 24 studies on ecosystem valuation, the average value per EDP per hectare per year was calculated, yielding a figure of €4,706 (in 2004 prices). This is the average global value (mainly for Europe and North America) and will therefore not reflect very specific local conditions. However, the value can serve as a first approximation.

In the ReCiPe project (Goedkoop *et al.*, 2009) the average PDF values for different land use types have been estimated based on data from the United Kingdom and Switzerland. These are given in Table 41 for the hierarchist perspective (cf. Annex F).

Table 41 Average PDF values for 18 land use types

Land use type	PDF (per m ² per year)
Monoculture crops/weeds	1.39
Intensive crops/weeds	1.33
Extensive crops/weeds	1.29
Monoculture fertile grassland	1.13
Intensive fertile grassland	0.92
Extensive fertile grassland	0.69
Monoculture infertile grassland	0.85
Extensive infertile grassland	0.44
Monoculture tall grassland/herb	1.36
Intensive tall grassland/herb	1.05
Extensive tall grassland/herb	0.75
Monoculture broadleaf, mixed forest and woodland	0.63
Extensive broadleaf, mixed and yew LOW woodland	0
Broad-leafed plantation	0.81
Coniferous plantations	0.91
Mixed plantations	1.10
Continuous urban	1.4
Vineyards	0.86

Source: Goedkoop *et al.* 2009.



C.9.2 Monetary valuation

Multiplying the PDF values cited in Table 41 by the monetary value of € 0.47 per PDF per m² per year (derived from the CASES project) yields the figures for external costs reported in Table 42.

Table 42 External costs for occupation of a certain area of land for different land use types (2004 prices)

Land use type	External costs (€ per m ² per year)
Monoculture crops/weeds	€ 0.65
Intensive crops/weeds	€ 0.63
Extensive crops/weeds	€ 0.61
Monoculture fertile grassland	€ 0.53
Intensive fertile grassland	€ 0.43
Extensive fertile grassland	€ 0.32
Monoculture infertile grassland	€ 0.40
Extensive infertile grassland	€ 0.21
Monoculture tall grassland/herb	€ 0.64
Intensive tall grassland/herb	€ 0.49
Extensive tall grassland/herb	€ 0.35
Monoculture broadleaf, mixed forest and woodland	€ 0.30
Extensive broadleaf, mixed and yew LOW woodland	-
Broad-leaved plantation	€ 0.38
Coniferous plantations	€ 0.43
Mixed plantations	€ 0.52
Continuous urban	€ 0.66
Vineyards	€ 0.40

Source: Goedkoop et al., 2009 and Kuik et al., 2008.

C.9.3 Towards weighing factors

To obtain a single figure for valuation of land use, we adopted the average values for Dutch land uses cited in CBS Statline and multiplied them by the external costs; see Table 43.

Table 43 Average valuation of land use for the Netherlands

Land use	Percentage in NL	Valuation (€2008 per m ² per year)
Intensive fertile grassland	29%	0.51
Intensive crops	39%	0.74
Continuous urban	15%	0.78
Monoculture broadleaf (mixed forest and woodland)	17%	0.35
Average NL		0.612

Source: CBS, Statline, own calculations.





Annex D From per-pollutant values to weighted environmental themes

As explained in Chapter 6, in this project we have developed three sets of weighting factors, based on the monetary values assigned to emissions of individual pollutants:

1. A weighting set based on abatement costs characterised at midpoint level.
2. A weighting set based on economic damage costs at endpoint level characterised at midpoint level.
3. A weighting set based on direct valuation of damage to health and biodiversity at endpoint level.

Because we have shadow prices for a limited number of pollutants only, use of characterisation factors is inescapable. These factors indicate the relative contribution of a given pollutant to a particular environmental impact. If we had shadow prices for all the 1,000 pollutants taken on board in an LCA, there would be no need to work with characterisation factors, because the individual shadow prices would themselves embody such characterisation. We could then use these shadow prices to calculate the importance of pollutant A compared with pollutant B. If we only have a shadow price for pollutant A, though, there is no option but to use characterisation factors.

In translating from monetary valuation to weighting sets, in practice two problems are encountered:

- a Multiple impacts: many pollutants impact simultaneously on several environmental themes, across which their shadow price needs to be allocated.
- b Implicit characterisation: the fact that the damage estimates for multiple pollutants within a given theme already express an implicit characterisation, which may deviate from the midpoint characterisation cited in ReCiPe. How to deal with such differences?

These problems only arise with the first two weighting sets. For each of these two sets we have sought the best means of tackling the issues, as described below.

D.1 Abatement costs

Problem B (implicit characterisation) was resolved by first of all proceeding from just one priority pollutant for each theme, generally the pollutant taken as the equivalence factor in the midpoint characterisation factors (e.g. SO₂ for acidification). In principle, it is this priority pollutant that steers the valuation process. Following calculation, though, we assessed whether the resultant characterisation is in line with the values assigned to the other pollutants. If, for the theme of acidification, for example, we use SO₂ to express the value of SO₂ equivalents, then that value also provides an implicit valuation of NO_x and NH₃ (the acidifying impact of which is also expressed in SO₂ equivalents in the ReCiPe midpoint characterisation factors). What we have thus done is subsequently assess whether the characterisation found was in line with the shadow prices calculated for the pollutants in question. In all cases there proved to be reasonable agreement. As an example: an SO₂ price of € 5/kg yields an NO_x price of € 9/kg, a value nicely within the government's range of € 5-10/kg NO_x.



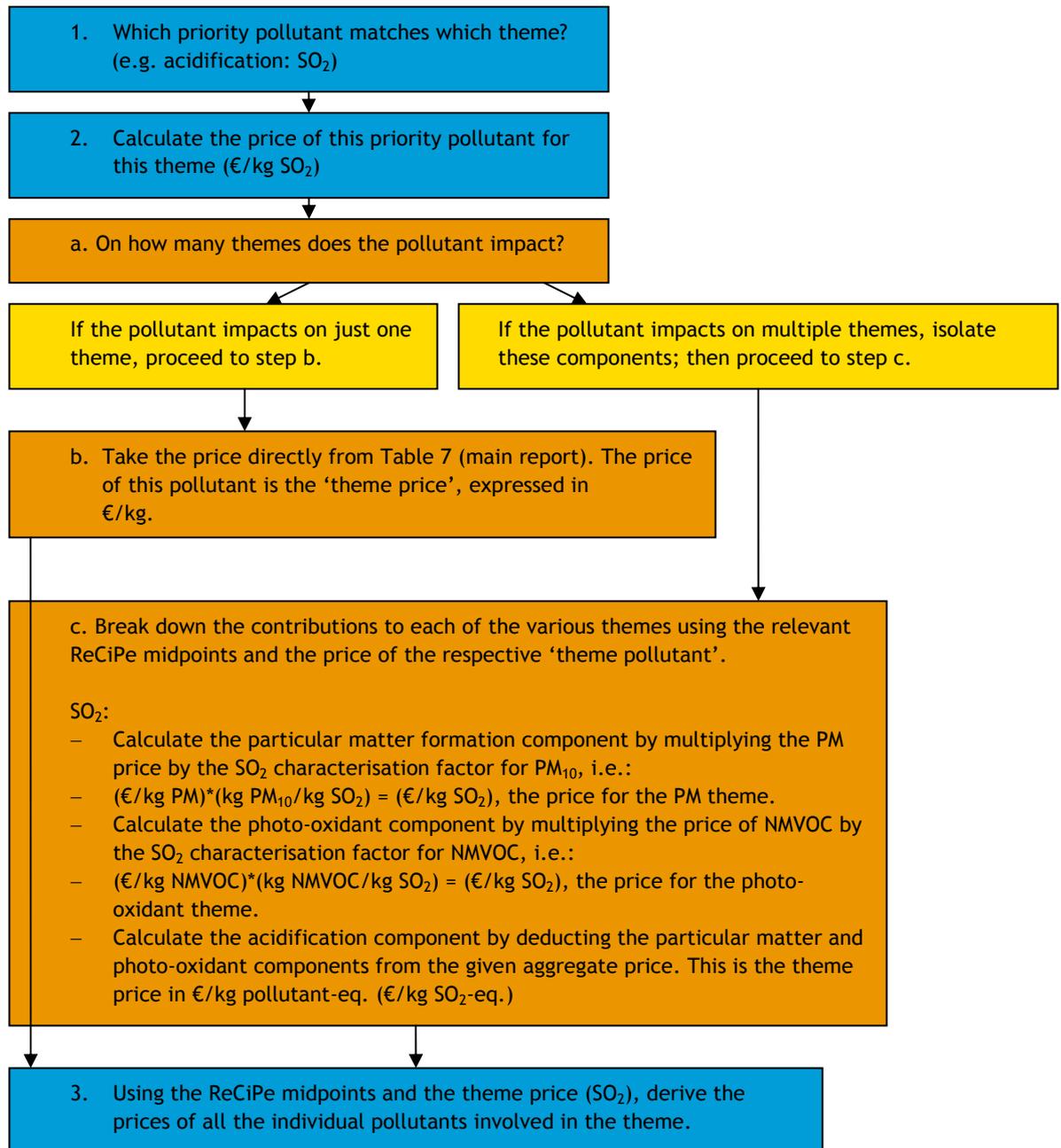
Problem A (multiple environmental impacts) proved relatively straightforward to resolve, as there are pollutants having an impact on just one environmental theme (PM₁₀ on human toxicity, for example, and NMVOC on photo-oxidant formation). By relating the valuation assigned to this environmental impact to the characterisation factor of a pollutant with multiple impacts, these multiple impacts can be duly allocated.

With both solutions, the implicit assumption is that the government is cognisant of each pollutant's relative contribution to the various individual environmental themes and consequently pursues pollutant-control policies that are not geared to pollution cuts as such, but to reducing their environmental impact.

The procedure adopted is set out in the schematic on the next page (Figure 8).



Figure 8 Schematic representation of procedure adopted for calculating abatement costs



D.2 Economic damage costs

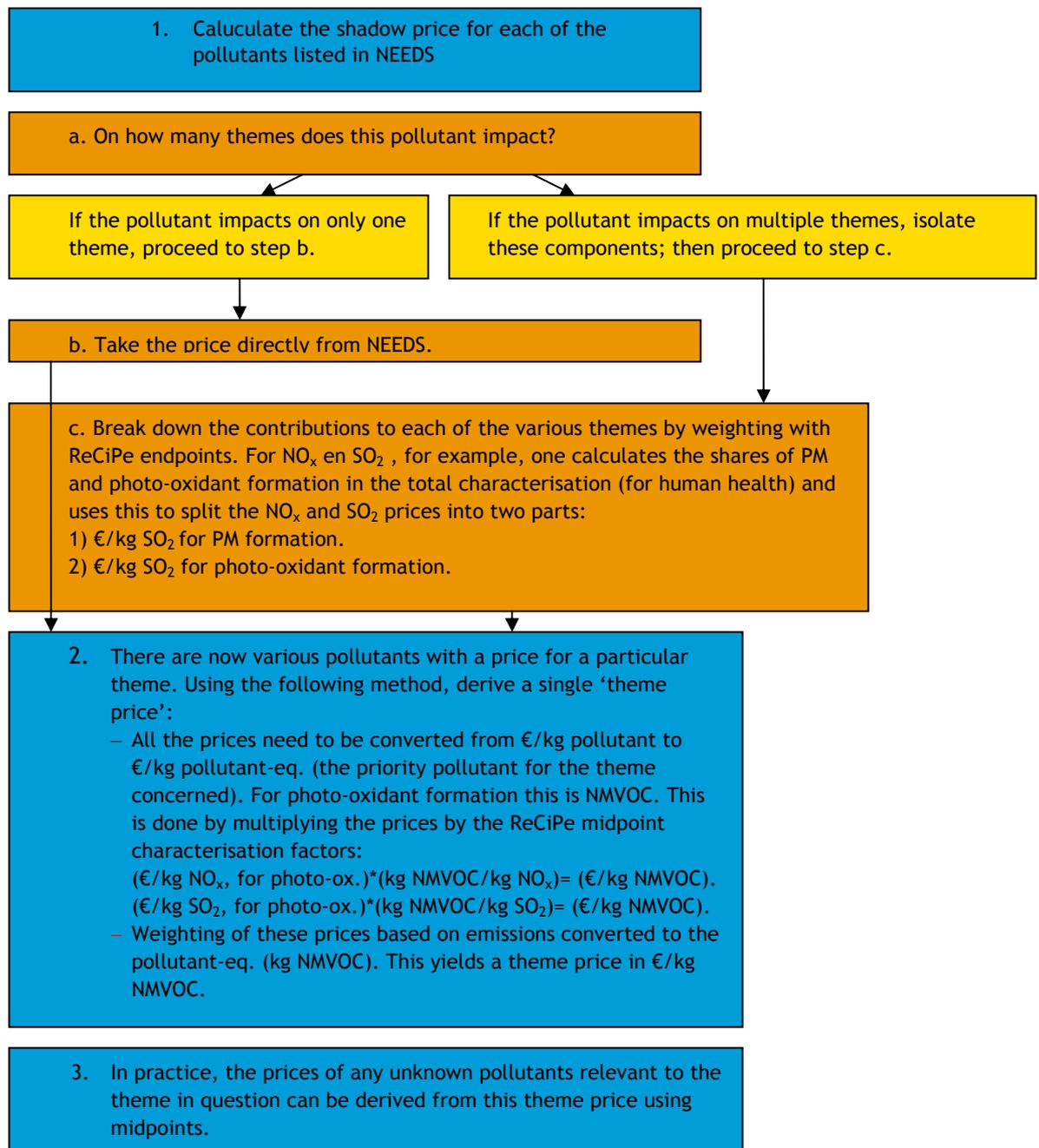
Problem B (implicit characterisation) was resolved by taking a weighted average of the damages occurring in the Netherlands. To this end we multiplied all the damages calculated in Annex C by the respective emissions occurring in the Netherlands and then divided the figure obtained by the emissions expressed in the relevant ReCiPe midpoint characterisation factor.

In resolving problem A (multiple environmental impacts) use was made of the ReCiPe endpoints. This is because these express, according to ReCiPe, how much each pollutant contributes to a particular endpoint. These were used to allocate the contributions of each pollutant across the midpoints.



The procedure adopted is shown in the following schematic (Figure 9).

Figure 9 Schematic representation of procedure adopted to calculate economic damage costs



NB. The 'theme-splitting' method used for the abatement costs cannot be used for the damage costs, because for the theme photo-oxidant formation you would then only take the NMVOC price from NEEDS, using this figure to derive prices for NO_x, SO₂, etc. The same holds for PM formation. However, if one knows the prices for both themes in which NO_x is involved (at endpoint level, human health: HH) and these are multiplied by the midpoint characterisation factors, one arrives at a different total value than the HH NO_x price from NEEDS. It is therefore better to split the HH price and ecosystem price from NEEDS using endpoint characterisation factors (i.e. weighting).

This method was applied under the assumptions set out below.



Climate change

The total damage costs for climate change (central value) are € 0.0250/kg CO₂-eq. for emissions in the year 2008 (cf. Annex C.1). It should be carefully noted that this figure increases for emissions emitted in later years.

Using ReCiPe endpoint characterisation factors, we split these total damage costs into damage to human health and damage to ecosystems. According to ReCiPe, 1 kg of CO₂ results in a loss of 1.46E-06 DALYs and 7.93E-09 species or 0.57 PDF/m².³⁴ Using the value of a DALY (€ 55,021.46³⁵) and a PDF (€ 0.55/PDF.m²³⁶), we find that 1 kg of CO₂ results in a damage of € 0.08 to human health and € 0.32 to ecosystems. The total climate change damage costs therefore needs to be subdivided into damage to human health and damage to ecosystems in the proportion of 1 to 4.1.

Acidification and photo-oxidant formation

In this impact category, we use damage costs per pollutant from NEEDS (cf. Annex C.3). As these costs differ for each pollutant, we use the method described in Section 4.5.1 to derive damage costs for each impact category, i.e. for:

- PM formation (PM₁₀-eq.).
- Photo-oxidant formation (NMVOC-eq.) and;
- Acidification of soils (SO₂-eq.).

Acidification of soils includes damages caused by NO_x, NH₃ and SO₂ to ecosystems and capital and land. The damages due to NO_x, SO₂, NH₃ and PM_{2.5} under the heading PM formation relate to human health effects. Photo-oxidant formation includes damages caused by NO_x, SO₂ and NMVOC to human health.

Eutrophication of freshwater

In this case we have used endpoint characterisation factors from ReCiPe and valued (aquatic) ecosystems based on the valuation for PDF.m² and the terrestrial and freshwater species density. Kuik et al. (2008) valued ecosystems using a value of € 0.47/PDF.m² in 2000 prices (€ 0.55/PDF.m² in 2008 prices; see Section 5.3.3). In the ReCiPe project (Goedkoop et al., 2009) terrestrial and freshwater species densities have been estimated as 1.38E-08 per m² and 7.89E-10 per m³, respectively. Using these factors, we arrive at a value of € 0.03 per PDF m³.

Eutrophication of soils

For this impact category we have used the damage costs from NEEDS (cf. Annex C.3). In NEEDS no distinction is made between damages due to acidification and to eutrophication of soils.

In NEEDS, the impact categories are damages to human health (not relevant here), ecosystems, crops and capital. For ecosystems, we have assumed that NO_x and NH₃ have an impact on both acidification and eutrophication, while SO₂ impacts on acidification only. For capital, we assume that the total impact can be classified under the heading acidification. For crops, we assume that the negative damages due to NO_x and NH₃ (only for the Netherlands, not for

³⁴ ReCiPE assumes a terrestrial species density of 1.38E-08 per m².

³⁵ This is the original value of VOLY from the NEEDS project recalculated with the uplift factor of 1.7% and multiplied by the HICP factor to get the value for 2008.

³⁶ This value has also been recalculated from the original value used in the NEEDS project, in the same way as VOLY (see previous footnote).



Europe) are due to their eutrophication impact, while SO_2 has an acidification impact.

To distinguish the eutrophication and acidification impacts of NO_x and NH_3 , we subtracted the acidification impact (using the damage cost of SO_2) from the total damage costs to arrive at the eutrophication impact. To deal with implicit characterisation, the method described in Annex D.2 was applied. Since ReCiPe includes aquatic eutrophication only, we have used the midpoint characterisation factors for marine eutrophication under the assumption that the relative damages of NO_x and NH_3 to marine water and soils are similar.

The damage costs of NMVOC are reported separately, because NMVOC affects ecosystems indirectly.

Human toxicity

Damage costs for each individual pollutant have been adopted from NEEDS (cf. Annex C.4). In NEEDS, the human toxicity of classical pollutants relates solely to air pollution. To arrive at a single damage cost for this impact category, we therefore use the ReCiPe midpoint characterisation factors for human toxicity via this route.

Radiation

Damage costs for each individual radionuclide have been adopted from NEEDS (cf. Annex C.6). To arrive at a single damage cost for this impact category, we have taken an (unweighted) average of the various radionuclides, as we have no data on the relative emission of each. The corresponding figure should therefore be regarded as a proxy. To compare: if we had calculated the shadow price using information from the ReCiPe endpoint factors, the corresponding value would be higher than using the route via NEEDS.



Annex E Valuation methods, benefit transfer and uncertainty

E.1 Introduction

In this annex we first describe the general methods in use for valuing environmental goods (Section E.2), then identify ways to transfer benefit estimates from one country to another (Section E.3) and finally give an assessment of uncertainty (Section E.4).

E.2 Valuation methods for environmental goods³⁷

The methods in use for valuing environmental goods can be divided into three categories:

- stated preference methods;
- revealed preference methods;
- production function approaches.

E.2.1 Stated preference methods

Stated preference approaches are based on surveys in which the respondents give direct answers about their Willingness to Pay (WTP) or Willingness to Accept (WTA) in relation to certain hypothetical changes in environmental quality. By far the most common method in this category is the **Contingent Valuation Method (CVM)**. Although CVM was first developed in the 1960s in the USA, it did not become widely used until the mid-1970s. CVM has been sanctioned for use in the USA in environmental damage claims, while in Europe many government agencies commission its use.

One crucial element of every CVM study is the design of the questionnaire and, in particular, a scenario describing the hypothetical situation of there being a market and consequently a price for the good in question (which is obviously not the case).

Respondents in CVM questionnaire are a random sample of the relevant population; this might be a national sample, for example, a sample of a local population or a sample of visitors to a recreational area.

These questionnaires have several other important design features, including the following:

- a People must be given a reason why they might be asked to pay for something that has so far been available free. For example, raising funds might be essential for preserving certain environmentally valuable areas.
- b The payment vehicle must be credible and non-controversial. For example, in most countries a tax would be credible but might be controversial if the benefits arising from preserving a certain environmental asset did not accrue to the entire population eligible for such a tax.
- c Respondents should be given adequate, unbiased information on the environmental good and its hypothetical market.
- d The question about WTP/WTA can be asked in various formats: as an open-ended question, using a payment card, or as a dichotomous choice in which

³⁷ This section is based on Hanley et al., 2001.



the respondents must accept or reject a given bid price. The average WTP/WTA for the sample can be calculated using a variety of statistical methods, depending on the question format.

- e Protest bids should be identified. Protest respondents are those who declare a zero value, not because they do not value the good but because in this way they protest against certain aspects of the survey, e.g. the payment vehicle or the way the questions are posed. Protest bidders are usually separated out before statistical analysis progresses.
- f Debriefing questions are a good way to check how well the respondents have understood the scenario and questions.
- g The main survey is usually preceded by focus group sessions, which help in designing an appropriate questionnaire.
- h The survey may be carried out by mail, telephone or in face-to-face interviews.

The sample average can be aggregated into a population mean/median. During econometric analysis the researchers investigate the relationship between WTP/WTA and various socioeconomic variables such as age and education level.

CVM can be applied in a very wide range of situations, in both a global and local context. It is capable of measuring both use and non-use values, which is a great advantage of all stated preference methods over revealed preference methods. However, CVM has also attracted plenty of criticism. The main points of critique can be summarised as follows:

- The values obtained using CVM are hypothetical; people do not always do what they say they would do. If respondents think they will actually have to pay, they are likely to underestimate their WTP and if they think that their answer is not linked to any actual charges but will influence the environmental change, they are likely to overestimate it. Development of CVM over time has contributed to improvements in questionnaire design, leading to diminishing over- and underestimation of actual WTP.
- In some studies there is a scoping problem, with WTP appearing to be insensitive to the scope of the environmental phenomena. For certain respondents WTP apparently has a symbolic character and reflects a general feeling that environmental goods should be protected rather than a surrogate price for the good described in the scenario. However, Carson (1997) has found that the majority of CVM studies pass tests of scope.
- CVM results are dependent on the information stated in the questionnaire. As many respondents may have never heard of the (e.g.) species described, the researchers, by implementing the survey, are possibly changing the preferences they want to measure.

There are two other stated preference methods in use:

- **Choice experiments/Choice modelling.** According to the characteristics theory of value, the value of a given good is best explained in terms of the characteristics or attributes of that good. For example, the value of a given forest can be broken down into the value of its different attributes, such as the number of plant/animal species in a particular area, or maintenance costs. Identifying the relevant attributes is usually carried out using focus groups. Different bundles of attributes are assembled using statistical methods. The respondents are asked to choose their preferred bundle of attributes, with one of the bundles describing the status quo. By including costs as one of the attributes, it is possible to calculate the marginal WTP in the sample for each attribute and the value of any combination of attributes.



- **Contingent ranking** is similar to choice experiments, with respondents again being presented with bundles of attributes, which they are asked to rank from the most to the least preferred. Using statistical analysis, the economists can then assess marginal WTP for changes in attributes and WTP for a particular bundle of attributes.

E.2.2 Revealed preference methods

In revealed preference (RP) approaches, the value people place on environmental goods is inferred from their behaviour in actual markets for related goods. The most popular methods in this group are the hedonic pricing, hedonic wages and travel cost methods.

- **The hedonic pricing method (HPM)** is based on information from real estate markets. In this method the environmental characteristics of homes are assumed to be important determinants of house prices, along with other factors like location and size. Environmental characteristics can include noise levels, air quality, scenic views and proximity to landfills. In HPM the researchers use regression analysis to estimate the marginal price of environmental attributes, which can be viewed as WTP for the environmental characteristics.
- **The hedonic wages method (HWM)** is based on information from job markets. In this method, the health risk or mortality risk associated with different jobs are assumed to be important determinants of wage. By means of regression analysis the marginal WTP for increased mortality risk can be estimated. This method is particularly useful for deriving the Value of a Statistical Life.
- **The travel Cost Method (TCM)** estimates WTP for outdoor recreation by looking at the expenditures people make to get to the recreational site. These expenditures include time and travel expenses. Typically, people are willing to spend an amount less than or equal to the value of the utility they derive from a recreational trip. In this method, information collected during a survey of visitors to a recreational site is used to create a demand curve for visits (typically, the higher the costs of making a trip, the fewer visits are made per year to the site). From the demand curve the consumer surplus (or changes therein) can be calculated as a measure of the site's value. The most controversial aspects of this method include substitute sites and the value of leisure time. The availability and costs of reaching substitute sites influences the frequency of visiting a given site. This problem can be at least partly resolved by including the costs of trips to these other sites in the travel cost equation or by using random utility models. Valuation of leisure time is difficult, because this depends on the respondent's occupation. For a self-employed person it makes sense to estimate the value of leisure time in terms of hourly earnings foregone, while for someone working on a fixed contract or for a retired person, such an approach would not make sense. Studies reveal, however, that leisure time values are generally positively related to income; commonly used fractions in literature include 33 and 43%.

E.2.3 Production function approaches

In production function approaches, changes in the quality or quantity of an environmental resource are valued by estimating the implications of these changes for outputs and prices of market goods/services, usually in terms of changes in consumer and producer surpluses. This class of methods includes dose-response models (more recently referred to as ecosystem function valuation models) that model the effects of an increase in the concentration of certain pollutants on agricultural crops, forests or fisheries, for example. In production function approaches, monetary values are derived from links to marketed outputs or avoided damage costs or using the replacement cost



approach. One example of the latter would be to value the pollution treatment capacity of a wetland by estimating the costs of providing a human-engineered sewage treatment system of the same capacity.

Ecosystem function valuation approaches are sometimes preferred by ecological economists who criticise the other methods described above as being unreliable. They argue that, wherever possible, different functions of ecosystems should be identified and valued using the production function approach and, if this is not possible, the effects should be described in qualitative terms.

E.3 Value transfer³⁸

Damage valuation should ideally be based on high-quality, primary valuation studies. Such studies are not always available, however, which means that researchers and policy-makers in their efforts to provide reliable estimates of environmental damages for a given region/country often have to refer to primary valuation studies carried out for other regions/countries. Such a procedure is referred to as benefit transfer or, more generally, value transfer and covers both time- and space-related adjustments. Value transfer may be regarded as the final step of the Impact Pathway Approach, required in cases where no relevant primary valuation studies exist.

E.3.1 Approaches to value transfer

There are two main approaches to value transfer:

1. Unit value transfer:
 - simple unit value transfer;
 - unit value transfer with adjustment for income differences.
2. Function transfer:
 - Value function transfer.
 - Meta-analysis.

Unit value transfer

In the unit value transfer approach, the unit value at the study site is assumed to be representative of the policy site. In option (a), no adjustment for differences in income or other characteristics is made, while in variant (b) there is adjustment for differences in income levels or in the costs of living between the two sites. In the function transfer approach, a value function is either estimated at the study site and transferred to the policy site (a), or estimated from several study sites using meta-analysis (b).

Simple unit transfer is the easiest approach for transferring value estimates from one site to another. This approach assumes that the well-being experienced by an average individual at the study site is the same as that experienced by an average individual at the policy site. Thus, we can directly transfer the value estimate, often expressed as mean willingness-to-pay (WTP) per household per year, from the study site to the policy site.

The obvious problem with this transfer is that individuals at the policy site may not value the environmental benefits or damages the same as the average individual at the study sites. There are two principal reasons for this difference. First, people at the policy site may differ from those at the study site in terms of income, education, religion, ethnic group or other socio-economic characteristics affecting their demand for recreation. Second, even

³⁸ This section draws extensively on NEEDS (2007a).



if individuals' preferences related to environmental quality at the policy and study sites were the same, the potential for substitution may not be.

The simple unit value transfer approach should not be used for transfer between countries with very different income levels and costs of living. In such cases, *unit transfer with income adjustments* has been applied. The adjusted WTP estimate at the policy site WTP_p can be calculated as:

$$WTP_p = WTP_s (Y_p / Y_s)^\beta \quad (1)$$

where WTP_s is the original WTP estimate from the study site, Y_s and Y_p are the income levels at the study and policy site, respectively, and β is the income elasticity of demand for the environmental good in question. The income elasticity for various environmental goods is typically less than 1 and often in the 0.4-0.85 range³⁹.

In case there is a lack of data on the income levels of the affected populations at the policy and study sites, Gross Domestic Product (GDP) per capita figures can be used as proxies for income in international value transfers. As using the official exchange rates does not reflect the true purchasing power of currencies, Purchasing Power Parity-adjusted rates should be used.

Even if PPP-adjusted GDP figures can be used to adjust for differences in income and cost of living in different countries, such a procedure will not be able to correct for differences in individual preferences, initial environmental quality, substitute sites and goods, and cultural and institutional conditions between countries (or even within different parts of a country).

Function transfer

Transferring the entire *value function* is conceptually more appealing than merely transferring unit values, because more information is effectively taken into account in the transfer. The value relationship to be transferred from the study site(s) to the policy site can be estimated using either revealed preference (RP) approaches like TC and HP methods or stated preferences (SP) approaches like the CV method and Choice Experiments (CE).⁴⁰ For a CV study, the value function can be written as:

$$WTP_{ij} = b_0 + b_1 G_j + b_2 H_{ij} + e \quad (2)$$

where WTP_{ij} is the willingness-to-pay of household i at site j , G_j is the set of characteristics of the environmental good at site j , and H_{ij} is the set of characteristics of household i at site j , while b_0 , b_1 and b_2 are sets of parameters and e is the random error.

To implement this approach, the analyst would have to find a study in the existing literature with estimates of the constant b_0 and the sets of parameters b_1 and b_2 . It would then be necessary to collect data on the two groups of independent variables, G and H , at the policy site, insert them in Equation 2, and calculate the households' WTP at the policy site.

³⁹ Within the NEEDS project, an income elasticity of 0.85 has been used in the uplift factor for temporal adjustment of values due to economic growth.

Note that formally, β this is the income elasticity of WTP, not of demand, and that there is no simple relationship between the two measures.

⁴⁰ These methods were described earlier in this Annex (E.2).



The main problem with the value function approach arises because of exclusion of relevant variables in the WTP (or bid) function estimated in a single study. When the estimation is based on observations from a single study on one or a small number of recreational sites or a particular change in environmental quality, a lack of variation in some of the independent variables usually prohibits inclusion of these variables.

Instead of transferring the value function from one selected valuation study, results from several valuation studies could be combined in a *meta-analysis* to estimate one common value function. Meta-analysis has been used to synthesize research findings and improve the quality of literature reviews of valuation studies in order to arrive at adjusted unit values. In a meta-analysis, several original studies are analysed as a group, with the result from each study being treated as a single observation in a regression analysis. If multiple results from each study are used, various meta-regression specifications can be used to account for such panel effects.

The meta-analysis enables evaluation of the influence of a wider range in characteristics of the environmental good, the features of the samples used in each analysis (including characteristics of the population affected by the change in environmental quality) and the modelling assumptions. The resulting regression equations explaining variations in unit values can then be used together with data collected on the independent variables in the model describing the policy site to construct an adjusted unit value.

E.3.2 Steps in transferring value estimates

The following steps can be distinguished in the process of transferring value estimates from study site(s) to a policy (or project) site:

1. Determining the transfer unit

2. For *mortality*, the unit values would be in terms of the value of prevented fatalities (referred to also as Value of a Statistical Life, VSL) or the value of life years lost (Value of a Life Year, VOLY).
3. For *morbidity*, it would typically be value per symptom-day (described in detail for different illnesses) for acute symptoms, and value per case for chronic diseases.
4. For certain *ecosystem services*, unit costs can be used, e.g. market prices for fish, or unit cost per tonne of carbon. Within the NEEDS project an aggregate measure of ecosystem losses (Potentially Disappearing Fraction, PDF) has been used and evaluated.
5. For *recreational use*, consumer surplus per activity day (i.e. per visitor per day) can be used, multiplied by an estimated change in the number of activity days (or visits) at the policy site.

1. Spatial transfer

If the policy site is considered to approximate the study sites well in all respects, *unit value transfer* can be used. If there are several equally suitable study sites to transfer from, they should all be evaluated and the transferred values calculated from a value range.

For unit transfers between countries, differences in currency, income and cost of living between countries can be corrected for by using Purchasing Power Parity (PPP), corrected exchange rates or ratio of income per capita at PPP (often approximated with GDP at PPP). Using only PPP-adjusted exchange rates would mean assuming zero income elasticity (this approach would assume the same valuation of resources across countries, regardless of income differences between them), while using GDP at PPP ratio as an adjustment



factor would imply assuming a unitary income elasticity (so that if GDP at PPP is two times higher in country B than in country A, WTP for environmental improvements would be doubled as well). As empirical evidence shows that income elasticity of WTP for environment-related values is less than 1 but not equal to zero, neither of these approaches would be correct. In the approach adopted throughout this handbook, in which we propose valuation of damages based on the concept of opportunity costs rather than absolute (intrinsic) value, it would be more appropriate to use GDP at PPP ratio combined with income elasticity, according to Equation (1) above. The income elasticity factor should be based on recommendations from literature devoted to the specific topic in question. Skipping the elasticity factor would mean assuming an income elasticity equal to 1, which might cause overestimation.

Unit value transfer, especially with adjustment for income level, can be recommended as the simplest and most transparent way of transfer both within and between countries. This transfer method has in general also been found to be just as reliable as the more complex procedures of value function transfers and meta-analysis (NEEDS, 2007a).

2. Temporal adjustment

The value estimate should be adjusted from the time of data collection to current currency using the Consumer Price Index (CPI) for the policy site country. This adjustment allows the impact of inflation to be factored out. In transferring values from a study site to another country, these values should first be converted to local currency in the year of data collection using PPP (Purchase Power Parity) corrected exchange rates in the year of data collection, and the local CPI then used to update to current currency values.

In addition, for valuation studies performed several years earlier, it may be recommended to adjust the values using an uplift factor, to correct for the increase in WTP due to growth of average income in real terms.

E.3.3 Uncertainty and transfer errors

There are two main sources of uncertainty and errors in value transfer:

- errors associated with estimation of the unit value/value function at the study site;
- errors associated with transferring the study site value(s) to the policy site.

The first type of error can be minimised by choosing the primary studies which use the best, state-of-the-art methodology. The second type of error arises through transferral of the primary estimates, both in space and in time. The results of validity tests of different value transfer procedures for different types of environmental goods have shown that individual transfer errors in spatial value transfer vary from a few percent to several hundred percent (NEEDS, 2007a). However, average transfer errors, for both national and international value transfer, seem to be about $\pm 20-40\%$. In many cases this would be an acceptable transfer error. However, sensitivity analysis should be performed to see if this interval for the estimated values would influence the outcome of the CBA or policy recommendations.



E.4 Uncertainty⁴¹

The methodology for assessing uncertainty of the NEEDS estimates of damage costs is based on lognormal distributions and geometric standard deviations (i.e. multiplicative confidence intervals). This choice is related to the fact that damage cost values according to the Impact Assessment Method used within the NEEDS project are a product of several factors, such as increase in concentration of a given pollutant, slope of the CRF, density of the receptors and a monetary estimate of a given endpoint.

The lognormal distribution of a variable z (here representing damage costs) is obtained by assuming that the logarithm of z has a normal distribution. Invoking the central limit theorem for the product z , one can say that the lognormal distribution is the ‘natural’ distribution for multiplicative processes, in the same way that the Gaussian distribution is ‘natural’ for additive processes. Although the lognormal distribution becomes exact only in the limit of infinitely many factors, in practice it can serve as a good approximation even for a few factors, provided the distributions with the largest spread are not too far from lognormal (NEEDS, 2008b).

For many environmental impacts the lognormal model for the result is quite relevant because the impact is a product of factors and the distributions of the individual factors are not too far from lognormality. For most situations of interest here one can assume independence of the distributions (e.g. for atmospheric dispersion, CRFs and monetary values), and thus one finds that the geometric standard deviation σ_{gz} of the product z is given by:

$$[\ln(\sigma_{gz})]^2 = [\ln(\sigma_{gx1})]^2 + [\ln(\sigma_{gx2})]^2 + \dots + [\ln(\sigma_{gxn})]^2 \quad (3)$$

For a lognormal distribution, the geometric mean μ_g is equal to the median. If a quantity with a lognormal distribution has a geometric mean μ_g and a geometric standard deviation σ_g , the probability is approximately 68% that the true value will lie within the interval $(\mu_g/\sigma_g, \mu_g\sigma_g)$ and 95% that it will be in the interval $(\mu_g/\sigma_g^2, \mu_g\sigma_g^2)$.

Below, we report the approximate confidence intervals for damage values calculated within the NEEDS project in three categories: classical pollutants, GHGs and trace pollutants.

1. Classical pollutants

Rabl and Spadaro (1999) have examined the uncertainties of each step of the impact pathway analysis for classical pollutants to estimate the uncertainties associated with the various components of the calculation. Table 44 reports their assumptions for the component uncertainties and the results for the damage costs for mortality. Because mortality accounts for over two-thirds of the damage costs of many pollutants, the uncertainty associated with this endpoint can be viewed as a good estimate for that associated with the sum total of impacts.

⁴¹ This description is based on NEEDS (2008b).



Table 44 Uncertainty of damage cost estimates per kg of pollutant for mortality

	Log-normal?	σ_{gi} PM	$\ln(\sigma_{gi})^2$	σ_{gi} SO ₂ via sulphates	$\ln(\sigma_{gi})^2$	σ_{gi} NO _x via nitrates	$\ln(\sigma_{gi})^2$
<i>Exposure calculation</i>							
Dispersion	yes	1.5	0.164	1.7	0.282	1.7	0.282
Chemical transformation	yes	1	0	1.2	0.033	1.4	0.113
Background emissions	no	1	0	1.05	0.002	1.15	0.02
<i>Total σ_g for exposure</i>		1.5	0.16	1.76	0.32	1.9	0.41
<i>ERF</i>							
Relative risk	no	1.5	0.164	1.5	0.164	1.5	0.164
Toxicity of PM components	?	1.5	0.164	2	0.48	2	0.48
YOLL, given relative risk	no?	1.3	0.069	1.3	0.069	1.3	0.069
<i>Total σ_g for ERF</i>		1.88	0.4	2.33	0.71	2.33	0.71
<i>Monetary valuation</i>							
Value of YOLL (VOLY)	yes	2	0.48	2	0.48	2	0.48
Total σ_g		2.78	1.04	3.42	1.51	3.55	1.61

Source: NEEDS, 2008b.

Table 44 shows sample calculations of geometric standard deviation σ_g . The relative contributions of the σ_{gi} to the total can be seen in the column $\ln(\sigma_{gi})^2$.

NEEDS (2008b) report to three significant figures only, to bring out the differences between these pollutants and the larger uncertainties of the secondary pollutants. But in view of the subjective and rather uncertain assumptions made, the authors believe it is best to simply sum the results by saying that **the geometric standard deviation of these damage costs equals approximately 3**. This means that for classical pollutants, the true values lie, with a 68% probability, within an interval between the central value divided by three and the central value multiplied by three.

2. GHGs

For GHGs, the analysis of uncertainty is based on Tol (2005), who reviews a large number of damage cost analyses. He gathered over 100 estimates for the marginal damage costs to form a probability density function. The function proves to be strongly right-skewed, with a median of \$ 3.8/tCO₂, a mean of \$ 25.4/tCO₂ and a 95% confidence level of \$ 95/tCO₂. The distribution is not exactly lognormal since it extends to negative costs; these results were excluded because the authors do not believe the average worldwide impact could be beneficial. NEEDS (2008b) conclude that, if negative costs are excluded, the distribution is not too different from a lognormal with its tail of high estimates of low probability. They therefore consider it reasonable to estimate the geometric standard deviation of greenhouse gas damage costs by assuming that the ratio of the 95% level and the median equals σ_g^2 , implying that

$$\sigma_g = \sqrt{95/3.8} = 5 \text{ for the damage costs of CO}_2, \text{ CH}_4 \text{ and N}_2\text{O.}$$



Thus, we can conclude that for GHG estimates based on Tol (2005), approximately 68% of estimates falls within an interval between the central value divided by five and the central value multiplied by five. For the mean estimate from Tol (2005) of \$ 25.4 this would imply the following 68% confidence interval: (5, 127).

3. Trace pollutants

Using the same assumption about lognormality of damage distribution, NEEDS (2008b) calculate geometric means for the trace pollutants. The results are shown in Table 45.

Table 45 Summary of geometric standard deviations σ_g for the damage costs

Pollutant	σ_g
As, Cd, Cr-VI, Hg, Ni, Pb	4
Dioxins	5

Uncertainty related to transfer to other regions

NEEDS (2008b) have also examined the uncertainties associated with the transfer of the individual components of the damage costs calculation (emissions, atmospheric modeling, dose-response functions and monetary valuation) to regions other than the EU. The results are expressed in terms of geometric standard deviations and listed in Table 46. To obtain the total uncertainty for a given region, the figures relevant to that region need to be combined with the geometric standard deviations of the damage costs for the EU-15

Table 46 Geometric standard deviations associated with the transfer of components of the damage cost calculation

Component of calculation	σ_g
<i>Transfer of technologies</i>	
CO ₂ emissions with CCS	1.3
Other emissions	^a
<i>Atmospheric modelling</i>	
If no data for effective deposition velocity v_{dep}	1.5
If no data for stack height	2
If no data for local population or no data for wind	3
Background concentrations for sulphate and nitrate formation	1.2
Background concentrations for O ₃ formation due to NO _x	2
Background concentrations for O ₃ formation due to VOC	1.3
<i>Modelling of ingestion dose</i>	
Toxic metals	2
<i>Exposure-Response Functions</i>	
PM, NO _x , SO ₂ , toxic metals	2
<i>Monetary values, non-market goods</i>	
WTP for goods other than health	2
WTP for health	
(GDP/cap)/(GDP/cap) _{ref} = 0.5	1.3
(GDP/cap)/(GDP/cap) _{ref} = 0.2	1.7
(GDP/cap)/(GDP/cap) _{ref} = 0.1	2.1

^a Depends on site.



For example, if the transfer is to a region where no data for the effective deposition velocity v_{dep} are available, where the health system and individual sensitivities are very different from the EU-15, and where the PPP-adjusted GDP/capita is 1/5 that of the EU-15, the data in Table 47 indicate that the total uncertainty for the damage cost of PM_{10} can be expressed as $\sigma_g = 4.3$, which is much larger than the $\sigma_g = 3$ in the EU-15. The calculations are based on Equation 1, earlier in the text.

Table 47 Example of estimation of uncertainty with transfers

Example for PM_{10}	σ_g	$\log(\sigma_g)^2$
In EU-15	3	1.21
No v_{dep} data	1.5	0.16
CRF	2	0.48
WTP in region with $(GDP/cap)/(GDP/cap)_{ref} = 0.2$	1.7	0.28
Total	4.3	2.13

As can be seen in the table, the total uncertainty for the damage cost of PM_{10} in the region is $\sigma_g = 4.3$, much greater than the $\sigma_g = 3$ in the EU-15. If local population data are lacking, the uncertainty will increase to $\sigma_g = 5$.

NEEDS (2008b) note that many if not most policy applications of ExternE concern choices where the detailed location of the installations is not known in advance; in such cases one needs typical values for a country rather than site-specific results.

The authors conclude that the estimation of uncertainties is difficult and replete with uncertainties of its own; it necessarily involves subjective judgment, and various readers might well come up with different assessments of the component uncertainties. However, the authors of the report believe that unless all the component uncertainties are systematically over- or underestimated, there will be compensation of errors: some may be higher, some lower, but overall, the sum in Equation 1 is not likely to change much.





Annex F Modelling in ReCiPe

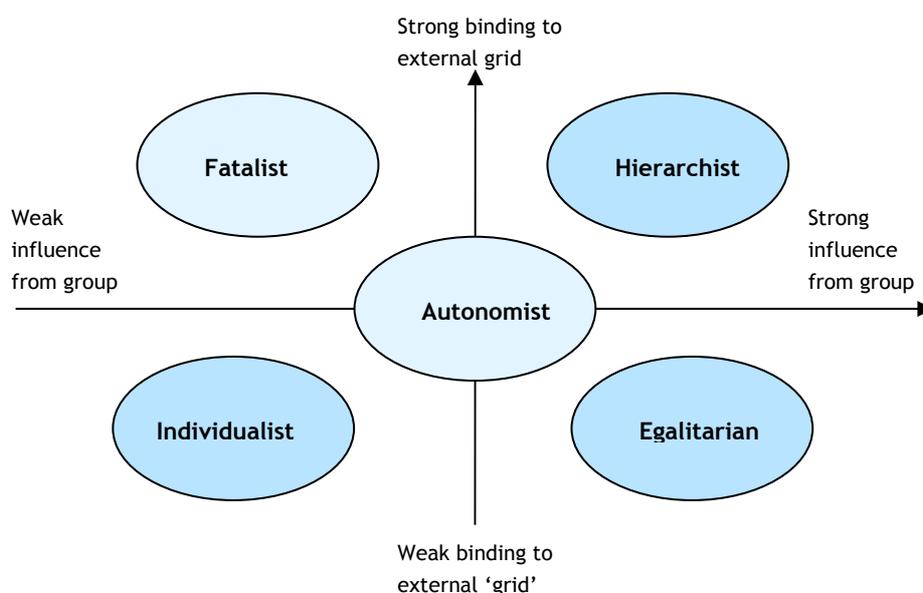
F.1 Introduction

The characterisation models used in the ReCiPe project are subject to uncertainty, since the modelled relationships reflect currently incomplete and uncertain knowledge of environmental mechanisms. Just as in Eco-indicator 99, it was therefore decided to group different sources of uncertainty and choices into a limited number of perspectives, according to the 'Cultural Theory' elaborated by Thompson, et al. (1990).

F.2 Concept of the Cultural Theory

Thompson et al. distinguishes five basic value systems by looking at the strength of the relations people have with their group and the degree to which an individual's life is circumscribed by externally imposed prescriptions (their so-called 'grid'); see Figure 10.

Figure 10 The five basic value systems according to Thompson et al.



The most important characteristics of these five 'archetypes' are:

1. Individualists lack strong links with either their group or their grid. They hold that all environmental limits are provisional and subject to negotiation.
2. Egalitarians have a strong link to the group, but a weak link to their grid. Relations between group members are often ambiguous and conflicts readily occur.
3. Hierarchists have strong links to both group and grid, both controlling others and being controlled by them. This hierarchy creates a high degree of stability in the group.
4. Fatalists have a strong link with the grid, but not with the group. These people act individually and are usually controlled by others.

- Autonomists are a relatively small group that escapes the manipulative forces of both groups and grids.

The last two archetypes cannot be used, because fatalists are guided by what others say and autonomists think completely independently.

F.3 The three perspectives

In ReCiPe the three perspectives can be summarised as follows:

- Individualist.** In this scenario only proven cause-effect relations are included and the short-term perspective is used. For human health issues age-weighting is applied. There is technological optimism with regard to human adaptation.
- Hierarchist.** Included in this scenario are facts backed up by scientific and political bodies. The hierarchical attitude is common in the scientific community and among policy-makers.
- Egalitarian.** This scenario uses the precautionary principle and the very long-term perspective.

F.4 Overview of choices for the perspectives (in ReCiPe)

Table 48 Overview of choices for environmental mechanisms (from pollutants to midpoint impact category)

Impact category	Individualist	Hierarchist	Egalitarian
Climate change	20-yr time-horizon	100 yr time-horizon	500 yr time-horizon
Terrestrial acidification	20-yr time-horizon	100 yr time-horizon	500 yr-time-horizon
Human toxicity	100-yr time-horizon Organics: all exposure routes Metals: drinking water and air only Only carcinogenic chemicals with TD ₅₀ classified as 1, 2A, 2B by IARC	Infinite time-horizon All exposure routes for all chemicals All carcinogenic chemicals with reported TD ₅₀	Infinite time-horizon All exposure routes for all chemicals All carcinogenic chemicals with reported TD ₅₀
Terrestrial ecotoxicity	100-yr time-horizon	Infinite time-horizon	Infinite time-horizon
Freshwater ecotoxicity	100-yr time-horizon	Infinite time-horizon	Infinite time-horizon
Marine ecotoxicity	100-yr time-horizon Sea + ocean for organics and non-essential metals. For essential metals only the sea compartment is included, i.e. excluding oceanic compartments.	Infinite time-horizon Sea + ocean for all chemicals	Infinite time-horizon Sea + ocean for all chemicals
Ionising radiation	100-yr time horizon	100,000-yr time-horizon	100,000-yr time-horizon

Source: Goedkoop et al., 2009.



Table 49 Overview of choices for environmental mechanisms (from midpoint to endpoint impact category)

Impact category	Individualist	Hierarchist	Egalitarian
Climate Change	Full adaptation; no cardiovascular risks; no malnutrition; low-range RR for natural disasters.	Mean adaptation; mean relative risk for all mechanisms; no diarrhea if GDP>6000 \$/yr.	No adaptation; high cardiovascular risks; high risk for disasters; high risk for malnutrition.
Climate Change	Dispersal of species assumed.	Dispersal of species assumed.	No dispersal of species assumed.
Terrestrial acidification	20 yr time-horizon.	100 yr time-horizon.	500 yr time-horizon.
Land occupation	Positive effects of land expansion are considered.	Fragmentation problem considered.	No positive effects of land expansion considered.
Land transformation	Max. restoration time is 100 yr.	Mean restoration times.	Maximum restoration times.
Fossil fuel depletion	Time horizon: 2030.	For coal, time-horizon: 2030; for all other fossil fuels: 2030-2080.	For coal, time-horizon: 2030; for all other fossil fuels: 2030-2080.

Source: Goedkoop et al., 2009.





Annex G Comparison of NEEDS and ReCiPe

G.1 Introduction

The work presented in this report builds to a major extent on earlier studies, in particular ReCiPe (Goedkoop et al., 2009) and NEEDS (2008a). In this section the main differences between the methods used in these two studies will be briefly explained.

G.1.1 Definition of damages

In defining damage ‘endpoints’, ReCiPe and NEEDS adopt different methodologies. In ReCiPe the impact of emissions is calculated with reference to three ‘impact themes’: human health, ecosystem quality and resource availability. For every emission, the impact on these themes is expressed in the same units: DALYs, annual loss of species and dollars per year, respectively. By using the same units, the impacts of different emissions on any one of these themes can be compared at ‘endpoint’ level.

Although NEEDS adopts similar impact themes (human health, loss of biodiversity, impacts on crops, and material damage), in this case no use is made of standardised ‘endpoints’. On the theme of ‘human health’, for example, not all impacts are converted into DALYs, with a number of specific consequences like lowering of IQ (for lead poisoning) or occurrence of hereditary defects (for ionising radiation) being taken into consideration, for instance. These damages are then assigned a value by determining the Willingness-to-Pay to avoid a certain degree of damage, for example, or the economic damage incurred through impaired health or treatment of a specific disease. Because different ‘endpoints’ are used, in principle the emission damages cannot be compared until they are monetised.

G.1.2 Geographical scope

There are also important differences between ReCiPe and NEEDS when it comes to geographical scope. NEEDS sets about its task by determining impacts at the regional, European level, and with the use of the EcoSense model produces estimates of damage costs for classical pollutants for each EU member state and for the EU as a whole. Damage estimates modelled with Ecosense are extended to adjacent regions like North Africa. If regional impacts cannot be established from the available data, NEEDS falls back on calculations at the global level. This holds for calculations of the damage costs associated with ionising radiation, for instance. NEEDS uses spatially differentiated databases (with 50 x 50 km grid cells) of population density, background concentrations and climatological conditions throughout Europe, which govern the relationship between emissions and physical impacts.

ReCiPe starts out by endeavouring to calculate damage costs at the *global* level (Goedkoop et al., 2009, p5). This is the case with emissions of greenhouse gases and ozone-depleting substances, for example, where the resultant damages are independent of emission location. The damage costs of emissions whose impact is governed by local or regional conditions are calculated at the regional level. Regional conditions affecting the nature and magnitude of impacts include climatological factors, background concentrations, hygienic factors and population density. These factors are relevant to the impact themes of acidification, eutrophication, photo-oxidant formation and human toxicity, for example, and the associated damages are therefore calculated at regional level. Although developed primarily for use in



European countries, these regional figures can in principle also be used for other economically advanced nations with a temperate climate. For classical emissions, ReCiPe uses average values of population density, background concentrations and climatological conditions for Europe as a whole.

G.1.3 Time frames and discounting

A further key difference is that the time frames adopted in ReCiPe have been made dependent on the perspective adopted. In ReCiPe, impacts are *not* discounted. In other words, an emission of CO₂ now leads to impacts on DALY and PDF up to the chosen time frame (cf. Annex F). In the NEEDS estimates these impacts *are* discounted. This explains the considerable differences between the damage values calculated according to the ReCiPe endpoints and the results obtained using the Integrated Assessment Models employed in the context of climate change studies.

G.1.4 Epidemiological thresholds

In deciding whether or not particular studies should be taken on board in developing damage estimates, both ReCiPe and NEEDS adopt a certain threshold. In ReCiPe, for example, the impact theme 'human environment' is not included, because the authors consider there to be too much uncertainty in the studies on this topic. ReCiPe also assesses the reliability of data, with criteria depending on the chosen perspective. The individualist perspective is the 'strictest', accepting only those studies in which impacts have been irrevocably established. The egalitarian perspective is the least 'strict', and is essentially congruent with the precautionary principle: damage impacts for which there is any reasoned evidence should be included. The hierarchist perspective is intermediate. Other differences between these perspectives relate to the time frame considered. The egalitarian perspective encompasses impacts occurring further into the future, while the individualist perspective is concerned above all with impacts on the present generation.

Although NEEDS weighs up the quality and availability of studies, this leads to a general decision about whether or not include the study in question. In this case no different perspectives are distinguished, as in ReCiPe. Given the fact that in NEEDS a selection is made from among the effects (in the case of lead poisoning, for example, such effects as anemia, renal dysfunction and high blood pressure are not included), but that this does not appear to be on as strict a basis as in the individualist perspective in ReCiPe, it can be roughly stated that the NEEDS approach is more in line with the hierarchist perspective identified in ReCiPe.



Annex H Physical impact indicators

H.1 Introduction

In this study we use a total of five indicators for physical impacts: YOLL, YLD, DALY, PDF and QALY, as described in this annex.

H.1.1 YOLL

One of the physical impacts of environmental pollution is premature death in a given population. Ozone, smog, toxic substances and ionising radiation all result in premature death. The years of life lost (YOLL⁴²) is a widely accepted indicator for this purpose, originating from health sciences.

YOLL correspond to the number of deaths multiplied by the standard life expectancy at the age at which death occurs. For a given cause, age and sex the following basic formula applies: $YOLL = N \times L$, where N is the number of deaths and L the standard life expectancy at age of death in years. The standard life expectancy used to calculate YOLL at each age is the same for deaths in all regions of the world. All non-health characteristics such as race, socio-economic status and occupation are excluded from consideration.

In the original variant, non-uniform age weights were applied, with less weight being given to years lived at younger and older ages⁴³. Age weighting reflects the social value of each year lost, and ranges from 0 at birth, peaking to 1.5 at age 25 and gradually declining thereafter (Murray, 1994). In addition, 3% time discounting is used. In recent publications, no age weights are applied.⁴⁴

In the ExternE project estimates of YOLL-impacts have been defined for various substances (see Section 2.3.5).

H.1.2 YLD

One commonly used indicator of morbidity is the years lost due to disability (YLD) for incident cases of the health condition. To estimate the YLD for a particular cause in a particular time period, the number of incident cases in that period is multiplied by the average duration of the disease and a weight factor that reflects the severity of the disease on a scale from 0 (perfect health) to 1 (dead). The disability weight depends on changes in economic productivity levels due to the illness (level of loss of functioning). The basic formula for YLD is: $YLD = I \times DW \times L$, where I is the number of incident cases, DW the disability weight (based on the judgment of a panel of experts) and L the average duration of the case until remission or death (in years). Disability weights are uniform for everyone living a year in a specified health state.

⁴² Also abbreviated to YLL.

⁴³ See e.g. Lopez and Murray (1998).

⁴⁴ Uniform age weights have been used in e.g. World Bank (2006b) and World Bank (2006a).



H.1.3 DALY

The DALY is a health gap measure that extends the concept of potential years of life lost due to premature death to include equivalent years of healthy life lost by virtue of individuals being in states of poor health or disability (Murray, 1996).

DALYs for a disease are the sum of the years of life lost due to premature mortality (YOLL) in the population and the years lost due to disability (YLD) for incident cases of the health condition. One DALY represents the loss of one year of equivalent full health (WHO, 2008).

In short, DALY can be defined as years of life lost plus years lost to disability:
 $DALY = YOLL + YLD$.

Box 4: Example of DALY calculations

With uniform age weights, death in infancy corresponds to 30 DALYs, and death at age 20 to around 28 DALYs. Thus, a disease burden of 3,000 DALYs in a population would be the equivalent of around 100 infant deaths or to approximately 5,000 persons living one year with blindness (DW 0.6) (GBD, 2006).

The DALY methodology has its roots in the QALY framework, developed in the 1970s (see Section H.1.5). In the 1980s the World Bank initiated a review of priorities for controlling specific diseases. The review generated findings about the comparative cost-effectiveness of interventions for most diseases important in developing countries, with the purpose of informing decision-makers in the health sectors of these countries. This process resulted in the publication of the first edition of Disease Control Priorities in Developing Countries (DCP1). This publication included an initial assessment of health status for low- and middle-income countries as measured by deaths from specific causes.

In 1992, the World Bank commissioned the so-called initial global burden of disease (GBD) study to provide a comprehensive assessment of the disease burden in 1990. The estimates, combined with analyses of the cost-effectiveness of interventions, were used during preparation of the World Development Report 1993 (World Bank, 1993).

Over the last decade the World Health Organization (WHO) has undertaken a new assessment of the global burden of disease for 2000-2002. It has also invested in improving the conceptual, methodological and empirical basis of burden of disease assessments. A comprehensive and updated version of these assessments is included in the WHO's 2006 publication (GBD, 2006). Global burden of disease statistics, including DALYs, YOLL, projections of mortality, etc., are accessible on the WHO's web pages (WHO, 2008).

Many countries and health development agencies have adopted the GBD approach as the standard for health accounting and for guiding determination of health research priorities, including Australia, India, Mexico, South Africa, Turkey and the United States, as well as the WHO itself (GBD, 2008).

Burden of disease and injury indicators (including DALYs) have been calculated for a set of major risk factors. One of the categories of risk factors is 'environmental risks', which include unsafe water, sanitation and hygiene, urban air pollution and indoor smoke from household use of solid fuels. Calculations do not cover any estimates related to specific pollutants.



H.1.4 PDFs

An indicator of biodiversity loss has been developed that is referred to as the Potentially Disappeared Fraction of selected species (PDF). For a given land use type, a certain number of vascular plant species is defined. If the land use type changes from one with a higher number of different species to one with fewer species, the number of species (biodiversity) is reduced. Hence, a 'delta PDF' can be calculated.

The PDF indicator was proposed in Eco-indicator 1999 for the assessment of acidification and eutrophication impacts. PDF measures the percentage of plants that is likely to disappear as a result of converting land use type. By extension, it can be assumed that the damage to plants and lower organisms is also representative of the damage to populations of higher animals (Eco-indicator, 2000).

Within the NEEDS project the PDF approach has been used for assessing acidification and eutrophication impacts on ecosystems. To be able to compare and evaluate specific land-use types, the species number of a set of land use types has been standardised for 1 m². This absolute species number is transformed into a relative number using the regional species richness of the Swiss Lowlands as a reference. Hence, the PDF values can be interpreted as the relative decline in biodiversity caused by a land use change from Swiss Lowland use to the respective land use category. For example, conversion of land from a Swiss Lowland state to conventional arable land would imply that 74% of all species would potentially disappear (NEEDS, 2006).

In the Ecosense model used within the NEEDS project for modelling and valuation of different impacts of pollution, the values of PDF per deposition are taken from a study by Ott et al. (2006), based on the damage model Natuurplanner. From the regional dispersion modelling, depositions per grid cell are available. For each 50 x 50 km² grid cell, the share of natural soil is available. Finally, a 'pressure index' (country-dependent) is applied which indicates the sensitivity of the soil (RIVM, 2000). This information has been used in EcoSense to model the loss of biodiversity due to SO₂, NO_x and NH₃ emissions.

This approach is designed to be applicable on a European scale. Consequently, it may not reflect specific local conditions, e.g. the occurrence of locally endangered species at a particular location. It can be used to make first approximations and, with the inclusion of this impact category, the importance of biodiversity and the Willingness-To-Pay to avoid its loss is demonstrated.

In the present project, PDF has been evaluated in monetary terms; see Section 5.3.3.

H.1.5 QALY

The indicator quality-adjusted life-year (QALY) takes into account the pain and suffering associated with disease and illness. One QALY is defined as 1 year in perfect health or its equivalent (Mauskopf and Morales, 2000). A health status index converts the concepts of health into quantifiable measures of an individual's perception of the impact of their health status on their general sense of well-being. Health states are classified based on the degree of restriction of mobility, social interaction, physical activity, pain or other symptoms a person may experience. The first classification of this kind was developed in Bush et al. (1972) based on a survey of 867 subjects in California. A more recent set of health states is the health utility index (HUI), for which



utility weights have been estimated.⁴⁵

In addition to health utility weights, to make due allowance for the estimated duration of each health state, the following assumptions need to be made in order to calculate QALYs lost as a result of illness (Mauskopf and Morales, 2000):

- 1 age at onset (e.g. 30 years);
- 2 remaining life expectancy (e.g. 46 years);
- 3 quality of life in the absence of the illness (e.g. 1);
- 4 discount rate (e.g. 3%);
- 5 number of different functional states (e.g. hospital days, bed days, house days);
- 6 morbidity for those dying from acute conditions;
- 7 pattern of utility loss for chronic conditions (e.g. constant utility loss);
- 8 life expectancy loss for chronic conditions (e.g. no life expectancy loss).

While QALYs are in principle universal, it is also possible to calculate with country-specific utility weights obtained in national samples. QALYs are not dependent on any socio-economic characteristics such as age or income.

QALYs have been used in health sector Cost-Effectiveness Analysis (CEA) for three decades (Sassi, 2006). They are most frequently used to assess the improvement in quality-adjusted life expectancy deriving from a specific health intervention relative to a situation in which either no intervention or a standard alternative intervention is provided. Thus, the health effects of certain interventions like vaccinations or preventing given products from reaching the consumer can be measured in QALYs. This approach is used, for example, by the American Food and Drug Administration (FDA) to evaluate programmes designed to improve health conditions.

Indicators like QALY and DALY are sometimes criticised on ethical grounds. Because of their design and how they are used in practice, one might draw the conclusion that disabled people are less entitled to scarce health resources for interventions that would extend their lives than people in normal health. This is in contrast with the basic principles of WHO (which, *nota bene*, endorses the DALY approach) (Arnesen and Nord, 1999).

Box 5: DALYs and QALYs

DALYs and QALYs are complementary concepts. QALYs are years of healthy life lived; DALYs are years of healthy life lost. To reflect the different burdens associated with different health states, QALYs use 'utility' weights, while DALYs use 'disability weights'. In a simplified example, if the utility of deafness is 0.67, the disability weight of deafness is $1 - 0.67 = 0.33$. Disregarding age weighting and discounting, and assuming a life expectancy of 80 years, a deaf man living for 50 years represents $0.67 \times 50 = 33.4$ QALYs gained and $0.33 \times 50 + 30 \times 1 = 46.6$ DALYs lost (based on Arnesen, 1999).

⁴⁵ Utility weights used in QALY calculation differ from disability weights used in DALY calculations in several respects. Disability weights refer to loss of functioning caused by a disease, while utility weights reflect the level of quality of life enjoyed in particular health states. The two types of weights are derived using different elicitation techniques and different groups of subjects. DALY calculations tend to be based on a universal set of standard weights based on expert valuations, while QALY calculations often rely on preference-based quality-of-life measures elicited from general population samples or groups of patients (Sassi, 2006).



Although QALYs and DALYs stem from the same broad conceptual framework, they are not interchangeable, as they are based partly on different assumptions and methodologies. Understanding the systemic differences between the two measures is important for enabling policy-makers to make sound judgements about the outcomes of specific health interventions (Sassi, 2006).





Annex I Data ‘shelf life’

I.1 Introduction

The present report provides a snapshot of the present state-of-the-art with respect to the valuation and weighting of emissions and environmental impacts. Among other things, it constitutes an update of the earlier results from 2002 (CE, 2002a). The main reason the shadow prices calculated in this previous study according to abatement costs needed to be updated was because the policy targets on which the shadow prices were based have meanwhile been superseded. Likewise, there will come a time that the results of the present study will once again have to be held up to the light. In this annex we consider the question of when such an update will be called for (cf. Section 7.2).

I.2 New policy targets

In the abatement cost method, shadow prices for selected emissions are calculated with reference to the intersection of the marginal abatement cost curve and the policy target, often depicted as a vertical line. As a result, any significant change in policy targets means that shadow prices need to be recalculated. If, for example, a new international climate treaty leads to tougher targets and a lowering of the emissions cap under the EU Emissions Trading Scheme, say, the shadow price of CO₂ will rise.

I.3 New abatement cost curves and economic trends

Besides the policy target, the other determining factor in the abatement cost method is the marginal abatement cost curve. This curve can change as a result of new understanding, changes in emissions due to changing economic activities, and technological trends. After all, over time emission abatement technologies become cheaper. This means that when sufficient information becomes available to prepare new marginal abatement cost curves the shadow prices should also be updated. A case in point would be publication of a new Options Document by ECN and PBL. Also, if the economic downturn that set in in mid-2008 proves to be structural rather than part of the business cycle, policy targets will be secured at lower cost.

I.4 New methods and values for damage valuation

In the case of damage costs, values have greater permanence because the underlying variables (e.g. dose-response functions, pollution dispersion characteristics and valuation of endpoints) will change relatively little over time. At the same time, though, the science of damage cost estimation does not stand still and new studies, or methodologies, may yield novel insights. This is particularly relevant to (a) use of a discount rate for calculating impacts occurring in the distant future (and then above all with respect to option values); (b) valuation of a human life in the context of environmental pollution; (c) valuation of option values and bequest values as an element of environmental valuation. In addition, those environmental themes for which a value has not yet been calculated using the impact-pathway approach (e.g. eutrophication) may in the future also become amenable to this route.





Annex J Full list of weighting factors

Based on the weighting factors provided in weighting sets 1 and 2 and the ReCiPe midpoint characterisation factors, extensive lists of damage and abatement estimates have been drawn up. These lists are provided in this final annex.

J.1 List of damage costs

Table 50, Table 51 and Table 52 report the damage costs of over 400 pollutants. In the table three categories are distinguished (air, soil and water), indicating on which compartment(s) the pollutant in question has an impact.

Shadow prices for emissions to air have been calculated using the weighting factors of weighting set 2 developed in the present report (cf. Chapter 6) and the ReCiPe midpoint characterisation factors. For certain pollutants like SO₂ and NO_x the valuation provided in NEEDS has been adopted unchanged.

The damage costs for emissions to soil and water have been calculated via direct valuation of ReCiPe endpoint characterisation factors (the method used for weighting set 3). For these compartments there were no values to adopt from NEEDS, which is concerned solely with emissions to air. Because direct valuation of ReCiPe endpoint characterisation factors is a less reliable method than method 2 (using NEEDS damage costs), these damage estimates are only approximate.

This list of damage costs is grounded in the assumption of a linear relationship existing between the contribution of the individual pollutant to the environmental theme (e.g. acidification) and the resulting damage.

Table 50 Damage costs for emissions to air (obtained via method 2 using NEEDS damage costs)

Substance	€/kg
1,4-Dioxane	8.67E-03
1-Butanol	6.12E-01
1-Butene	1.06E+00
1-Butene, 2-methyl-	7.58E-01
1-Butene, 3-methyl-	6.59E-01
1-Hexene	8.62E-01
1-Pentene	9.62E-01
1-Propanol	5.52E-01
1-Propanol, 3,3,3-trifluoro-2,2-bis(trifluoromethyl)-, HFE-7100	7.43E+00
2-Acetylaminofluorene	2.15E+00
2-Benzothiazolethiol	5.90E-03
2-Butanol	3.94E-01
2-Butanone, 3,3-dimethyl-	3.18E-01
2-Butanone, 3-methyl-	3.58E-01
2-Butenal	3.49E-01
2-Butene (cis)	1.13E+00
2-Butene (trans)	1.11E+00
2-Hexanone	5.63E-01
2-Hexene (cis)	1.05E+00



Substance	€/kg
2-Hexene (trans)	1.05E+00
2-Methyl pentane	4.13E-01
2-Methyl-1-propanol	3.54E-01
2-Methyl-2-butene	8.27E-01
2-Pentanone	5.40E-01
2-Pentene (cis)	1.10E+00
2-Pentene (trans)	1.10E+00
2-Propanol	1.85E-01
3-Hexanone	5.89E-01
3-Methyl-1-butanol	4.26E-01
3-Pentanol	5.89E-01
4-Methyl-2-pentanone	4.83E-01
Acetaldehyde	7.31E-01
Acetamide	1.30E-02
Acetic acid	9.56E-02
Acetic acid, methyl ester	5.81E-02
Acetic acid, propyl ester	2.77E-01
Acetone	9.57E-02
Acetonitrile	9.72E-03
Acetophenone	5.90E-03
Acrolein	1.27E+02
Acrylamide	8.03E+00
Acrylic acid	7.66E+00
Actinides, radioactive, unspecified	1.07E-01
Alcohol, diacetone	3.02E-01
Aldehydes, unspecified	2.26E+00
Allyl chloride	2.08E-01
Ammonia	2.78E+01
Aniline	7.99E-02
Aniline, p-chloro-, hydrochloride	7.29E-01
Anthracene	7.35E-03
Antimony	1.38E+02
Arsenic	8.11E+02
Barium	1.39E+01
Benzaldehyde	-7.42E-02
Benzenamine, 2-methoxy-5-nitro-	6.01E-02
Benzene	2.43E-01
Benzene, (epoxyethyl)-	4.66E-02
Benzene, 1,2,3-trimethyl-	1.25E+00
Benzene, 1,2,4-trichloro-	1.34E-01
Benzene, 1,2,4-trimethyl-	1.26E+00
Benzene, 1,2-dichloro-	2.41E-02
Benzene, 1,3,5-trimethyl-	1.36E+00
Benzene, 1,3-dinitro-	8.59E+00
Benzene, 1,4-dichloro-	2.06E-02
Benzene, 1-propyl-	6.24E-01
Benzene, 3,5-dimethylethyl-	1.30E+00
Benzene, ethyl-	7.18E-01
Benzene, hexachloro-	1.19E+01
Benzene, pentachloro-	3.05E+00
Benzidine	1.98E+00
Benzidine, 3,3'-dichloro-	1.24E+00
Benzo(a)pyrene	1.10E+00
Benzotrichloride	1.31E+01



Substance	€/kg
Benzyl chloride	8.38E-02
Beryllium	7.33E+03
Biphenyl, 4-amino-	7.45E-01
Bis(2-chloro-1-methylethyl)ether	1.45E-01
Bis(2-chloroethyl)ether	4.23E-01
Bromoform	2.62E-01
Butadiene	9.42E-01
Butadiene, hexachloro-	3.63E-01
Butanal	7.81E-01
Butane	3.47E-01
Butane, 1,1,1,3,3-pentafluoro-, HFC-365mfc	1.99E+01
Butane, 2,2-dimethyl-	2.37E-01
Butane, 2,3-dimethyl-	5.33E-01
Butane, nonafluoroethoxy, HFE-569sf2	1.48E+00
Butane, perfluoro-	2.22E+02
Butane, perfluorocyclo-, PFC-318	2.58E+02
Butanol, 2-methyl-1-	4.81E-01
Butanol, 2-methyl-2-	2.24E-01
Butanol, 3-methyl-2-	4.00E-01
Butene, 1,4-dichloro-2- (trans)	1.77E+00
Butyl acetate	2.65E-01
C.I. disperse yellow 3	1.67E-02
Cadmium	1.27E+02
Caprolactam	1.45E-02
Carbamic acid, ethyl ester	2.21E-02
Carbon dioxide	2.50E-02
Carbon dioxide, fossil	2.50E-02
Carbon dioxide, land transformation	2.50E-02
Carbon disulphide	9.80E-01
Carbon monoxide, biogenic	2.66E-02
Carbon monoxide, fossil	2.66E-02
Carbon-14	4.25E-01
Catechol	8.55E-03
Cesium-134	2.40E-02
Cesium-137	2.71E-02
Chlorendic acid	1.62E-01
Chlorine	4.31E+00
Chloroform	1.44E+00
Chloromethyl methyl ether	5.00E-01
Chromium	3.35E+01
Chromium VI	1.67E+02
Cobalt	8.90E+01
Cobalt-58	8.51E-04
Cobalt-60	3.34E-02
Copper	3.41E-01
Cumene	4.95E-01
Cyanide	1.74E+00
Cyclohexane	2.86E-01
Cyclohexanol	5.10E-01
Cyclohexanone	2.94E-01
Cyclohexene, 4-vinyl-	1.43E-02
Cyclohexylamine	2.93E-02
Cyclopentadiene, hexachloro-	3.08E+01
Decabromodiphenyl oxide	1.69E+01



Substance	€/kg
Decane	3.78E-01
Diethyl ether	4.38E-01
Diethyl ketone	4.07E-01
Diglycidyl resorcinol ether	1.27E-01
Dihydrosafrole	8.63E-03
Diisopropyl ether	3.92E-01
Dimethyl carbonate	2.46E-02
Dimethyl ether	2.11E-01
Dimethylcarbamyl chloride	4.46E+00
Dinitrogen monoxide	7.45E+00
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	5.09E+07
Dipropylthiocarbamic acid S-ethyl ester	2.83E-01
Dodecane	3.51E-01
Epichlorohydrin	2.64E+00
Ethane	1.21E-01
Ethane, 1,1,1,2-tetrachloro-	2.27E-01
Ethane, 1,1,1,2-tetrafluoro-, HFC-134a	3.58E+01
Ethane, 1,1,1,2-tetrafluoro-2-bromo-, Halon 2401	9.78E+00
Ethane, 1,1,1-trichloro-, HCFC-140	3.66E+00
Ethane, 1,1,1-trifluoro-, HFC-143a	1.12E+02
Ethane, 1,1,1-trifluoro-2,2-chlorobromo-, Halon 2311	5.48E+00
Ethane, 1,1,2,2-tetrachloro-	1.74E-01
Ethane, 1,1,2,2-tetrafluoro-, HFC-134	2.75E+01
Ethane, 1,1,2-trichloro-	8.70E+00
Ethane, 1,1,2-trichloro-1,2,2-trifluoro-, CFC-113	1.93E+02
Ethane, 1,1,2-trifluoro-, HFC-143	8.83E+00
Ethane, 1,1-dichloro-1-fluoro-, HCFC-141b	2.28E+01
Ethane, 1,1-difluoro-, HFC-152a	3.10E+00
Ethane, 1,2-dibromo-	3.03E+00
Ethane, 1,2-dibromotetrafluoro-, Halon 2402	2.76E+02
Ethane, 1,2-dichloro-	6.75E-01
Ethane, 1,2-dichloro-1,1,2,2-tetrafluoro-, CFC-114	2.87E+02
Ethane, 1,2-difluoro-, HFC-152	1.33E+00
Ethane, 1-chloro-1,1-difluoro-, HCFC-142b	6.05E+01
Ethane, 1-chloro-2,2,2-trifluoro-(difluoromethoxy)-, HCFE-235da2	8.75E+00
Ethane, 2,2-dichloro-1,1,1-trifluoro-, HCFC-123	2.72E+00
Ethane, 2-chloro-1,1,1,2-tetrafluoro-, HCFC-124	1.60E+01
Ethane, 2-chloro-1,1,1-trifluoro-, HCFC-133a	3.12E-01
Ethane, chloro-	4.93E-03
Ethane, chloropentafluoro-, CFC-115	2.02E+02
Ethane, fluoro-, HFC-161	3.00E-01
Ethane, hexafluoro-, HFC-116	3.05E+02
Ethane, pentachloro-	2.31E-01
Ethane, pentafluoro-, HFC-125	8.75E+01
Ethanol	3.93E-01
Ethanol, 2-butoxy-	4.76E-01
Ethanol, 2-methoxy-	3.02E-01
Ethene	9.85E-01
Ethene, chloro-	3.18E-01
Ethene, dichloro- (cis)	4.40E-01
Ethene, dichloro- (trans)	3.86E-01
Ethene, tetrachloro-	1.61E+00



Substance	€/kg
Ethene, trichloro-	3.24E-01
Ether, 1,1,1-trifluoromethyl methyl-, HFE-143a	1.89E+01
Ether, 1,1,2,2-Tetrafluoroethyl 2,2,2-trifluoroethyl-, HFE-347mcc3	1.44E+01
Ether, 1,1,2,2-Tetrafluoroethyl 2,2,2-trifluoroethyl-, HFE-347mcf2	9.35E+00
Ether, 1,1,2,2-Tetrafluoroethyl methyl-, HFE-254cb2	8.98E+00
Ether, 1,1,2,3,3,3-Hexafluoropropyl methyl-, HFE-356mec3	2.53E+00
Ether, 1,1,2,3,3,3-Hexafluoropropyl methyl-, HFE-356pcc3	2.75E+00
Ether, 1,1,2,3,3,3-Hexafluoropropyl methyl-, HFE-356pcf2	6.63E+00
Ether, 1,1,2,3,3,3-Hexafluoropropyl methyl-, HFE-356pcf3	1.26E+01
Ether, 1,2,2-trifluoroethyl trifluoromethyl-, HFE-236ea2	2.47E+01
Ether, 1,2,2-trifluoroethyl trifluoromethyl-, HFE-236fa	1.22E+01
Ether, 2,2,3,3,3-Pentafluoropropyl methyl-, HFE-365mcf3	2.75E-01
Ether, di(difluoromethyl), HFE-134	1.58E+02
Ether, difluoromethyl 2,2,2-trifluoroethyl-, HFE-245cb2	1.77E+01
Ether, difluoromethyl 2,2,2-trifluoroethyl-, HFE-245fa1	7.15E+00
Ether, difluoromethyl 2,2,2-trifluoroethyl-, HFE-245fa2	1.65E+01
Ether, ethyl 1,1,2,2-tetrafluoroethyl-, HFE-374pc2	1.39E+01
Ether, pentafluoromethyl-, HFE-125	3.73E+02
Ethyl acetate	2.06E-01
Ethyl acrylate	1.69E-02
Ethylene glycol	3.67E-01
Ethylene glycol monoethyl ether	3.80E-01
Ethylene oxide	2.08E-01
Ethyne	8.39E-02
Fertiliser, nitrogen	9.12E-01
Fluoranthene	3.90E-02
Formaldehyde	2.75E-01
Formic acid	3.15E-02
Heptane	4.86E-01
Hexane	5.02E-01
Hexane, 2-methyl-	4.04E-01
Hexane, 3-methyl-	3.58E-01
Hexane, perfluoro-	2.33E+02
HFE-227EA	3.85E+01
HFE-236ca12 (HG-10)	7.00E+01
HFE-263fb2	2.75E-01
HFE-329mcc2	2.30E+01
HFE-338mcf2	1.38E+01
HFE-338pcc13 (HG-01)	3.75E+01
HFE-347pcf2	1.45E+01
HFE-43-10pccc124 (H-Galden1040x)	4.68E+01
Hydrazine, methyl-	1.10E-01
Hydrocarbons, aliphatic, alkanes, cyclic	2.78E-01
Hydrocarbons, aromatic	2.59E-01
Hydrocarbons, chlorinated	1.65E+00
Hydrogen fluoride	5.49E+00
Hydrogen-3, Tritium	2.89E-05



Substance	€/kg
Hydroquinone	5.86E-02
Iodine-129	1.88E+00
Iodine-131	3.04E-04
Iodine-133	1.88E-05
Isobutane	3.02E-01
Isobutene	6.18E-01
Isobutyraldehyde	5.06E-01
Isopentane	3.99E-01
Isoprene	1.07E+00
Isopropyl acetate	2.07E-01
Krypton-85	2.82E-07
Lead	4.08E+02
Lead-210	3.04E-03
Maleic anhydride	8.38E-02
Manganese	6.61E+01
manure N	9.87E-01
m-Cresol	1.93E-02
Mercury	1.10E+04
Methacrylic acid, methyl ester	1.56E-01
Methane	6.25E-01
Methane, biogenic	6.31E-01
Methane, bromo-, Halon 1001	8.79E+01
Methane, bromochlorodifluoro-, Halon 1211	3.91E+02
Methane, bromodifluoro-, Halon 1201	6.49E+01
Methane, bromotrifluoro-, Halon 1301	6.48E+02
Methane, chlorodifluoro-, HCFC-22	4.92E+01
Methane, chlorofluoro-, HCFC-31	3.80E-01
Methane, chlorotrifluoro-, CFC-13	3.60E+02
Methane, dibromo-	3.85E-02
Methane, dibromodifluoro-, Halon 1202	5.09E+01
Methane, dichloro-, HCC-30	7.63E-01
Methane, dichlorodifluoro-, CFC-12	3.24E+02
Methane, dichlorofluoro-, HCFC-21	3.78E+00
Methane, difluoro-, HFC-32	1.69E+01
Methane, dimethoxy-	1.61E-01
Methane, fluoro-, HFC-41	2.30E+00
Methane, fossil	6.31E-01
Methane, iodotrifluoro-	1.00E-02
Methane, monochloro-, R-40	1.23E+00
Methane, tetrachloro-, CFC-10	1.88E+02
Methane, tetrafluoro-, CFC-14	1.85E+02
Methane, trichlorofluoro-, CFC-11	1.59E+02
Methane, trifluoro-, HFC-23	3.70E+02
Methanol	1.43E-01
Methyl ethyl ketone	3.67E-01
Methyl formate	2.66E-02
Molybdenum	3.18E+01
Molybdenum trioxide	1.84E+00
m-Phenylenediamine	6.71E-02
m-Xylene	1.09E+00
Naphthalene	1.69E-01
Nickel	5.37E+00
Nitrate	1.60E+00
Nitrilotriacetic acid	5.92E-04



Substance	€/kg
Nitrobenzene	2.58E+00
Nitrogen fluoride	4.30E+02
Nitrogen oxides	1.06E+01
Nitroglycerin	1.44E-02
N-Methylolacrylamide	1.12E-01
NM VOC, non-methane volatile organic compounds, unspecified origin	2.54E+00
N-Nitrosodiethylamine	1.31E+02
N-Nitrosodiphenylamine	1.34E-02
N-Nitrosodipropylamine	6.98E+01
N-Nitrosopiperidine	3.20E-01
Noble gases, radioactive, unspecified	2.63E-10
Nonane	4.07E-01
o-Cresol	2.44E-02
Octane	4.46E-01
o-Phenylenediamine dihydrochloride	2.46E-03
o-Toluidine hydrochloride	1.50E-02
o-Xylene	1.04E+00
PAH, polycyclic aromatic hydrocarbons	4.23E-01
Particulates	6.48E+01
p-Cresidine	4.75E-03
Pentanal	7.52E-01
Pentane	3.89E-01
Pentane, 2,3-dihydroperfluoro-, HFC-4310mee	4.10E+01
Pentane, 3-methyl-	4.71E-01
Pentane, perfluoro-	2.29E+02
PFC-9-1-18	1.88E+02
PFPME	2.58E+02
Phenol	1.91E-02
Phenol, 2,4,5-trichloro-	5.20E-02
Phenol, 2,4,6-trichloro-	1.59E-02
Phenol, 2,4-dichloro-	7.33E+00
Phenol, 2,4-dimethyl-	5.68E-02
Phenol, 2,4-dinitro-	1.35E+00
Phenol, pentachloro-	8.98E-01
Phosphorus	3.88E+02
Phthalate, dimethyl tere-	4.83E-02
Plutonium-238	1.34E-01
Plutonium-alpha	1.67E-01
Polonium-210	3.04E-03
Polychlorinated biphenyls	7.45E-01
Propanal	7.87E-01
Propane	1.73E-01
Propane sultone	1.06E+00
Propane, 1,1,1,2,2,3-hexafluoro-, HFC-236cb	3.35E+01
Propane, 1,1,1,2,3,3,3-heptafluoro-, HFC-227ea	8.05E+01
Propane, 1,1,1,2,3,3-hexafluoro-, HFC-236ea	3.43E+01
Propane, 1,1,1,3,3,3-hexafluoro-, HCFC-236fa	2.45E+02
Propane, 1,1,2,2,3-pentafluoro-, HFC-245ca	1.73E+01
Propane, 1,1,3,3-tetrafluoro-, HFC-245fa	2.58E+01
Propane, 1,2-dibromo-3-chloro-	2.21E+01
Propane, 1,2-dichloro-	5.51E+00
Propane, 1,3-dichloro-1,1,2,2,3-pentafluoro-, HCFC-225cb	1.60E+01



Substance	€/kg
Propane, 2,2-dimethyl-	1.70E-01
Propane, 2-nitro-	1.69E-01
Propane, 3,3-dichloro-1,1,1,2,2-pentafluoro-, HCFC-225ca	3.83E+00
Propane, perfluoro-	2.21E+02
Propargyl alcohol	4.02E-01
Propene	1.11E+00
Propene, 1,3-dichloro-	2.50E+01
Propene, 1-chloro-1-	7.70E-01
Propionic acid	1.47E-01
Propylene glycol	4.50E-01
Propylene glycol methyl ether	3.50E-01
Propylene glycol t-butyl ether	4.56E-01
Propylene oxide	8.75E-01
p-Xylene	9.96E-01
Pyrene	6.94E-02
Pyridine	2.19E+00
Radium-226	1.82E-03
Radon-222	4.85E-05
Safrole	2.31E-03
s-Butyl acetate	2.71E-01
Selenium	1.69E+02
Silver	8.01E+02
Sodium azide	6.50E-01
Styrene	2.05E-01
Sulphur dioxide	1.54E+01
Sulphur hexafluoride	5.70E+02
Sulphur oxides	1.10E+01
Sulphur, trifluoromethyl pentafluoride	4.43E+02
t-Butyl acetate	5.22E-02
t-Butyl alcohol	1.04E-01
t-Butyl ethyl ether	2.40E-01
t-Butyl methyl ether	1.79E-01
Thallium	1.16E+02
Thioacetamide	1.68E-01
Thiourea	1.39E-02
Thorium-230	9.10E-02
Tin	1.05E+00
Toluene	6.46E-01
Toluene diisocyanate	2.68E+02
Toluene, 2,4,6-trinitro-	1.02E+01
Toluene, 2,4-diamine	3.26E-01
Toluene, 2,4-dinitro-	2.97E+00
Toluene, 2,6-dinitro-	3.03E+01
Toluene, 2-chloro-	1.47E-01
Toluene, 2-ethyl-	8.86E-01
Toluene, 3,5-diethyl-	1.28E+00
Toluene, 3-ethyl-	1.00E+00
Toluene, 4-ethyl-	8.92E-01
Toluene, dinitro-	5.76E-01
Triethyl amine	1.63E-02
Undecane	3.78E-01
Uranium	1.64E-02
Uranium alpha	1.64E-02



Substance	€/kg
Uranium-234	1.94E-01
Uranium-235	4.25E-02
Uranium-238	1.64E-02
Vanadium	7.14E+01
Vinyl acetate	7.04E-02
Xenon-133	2.85E-07
Xylene	2.13E-02
Zinc	1.02E+01

Table 51 Damage costs for emissions to soil (obtained via method 3, with direct valuation of ReCiPe endpoint characterisation factors)

Substance	€/kg
2,4,5-T	3.40E+00
2,4-D	3.40E+00
Abamectin	1.06E+02
Acephate	4.66E+00
Acifluorfen	1.84E-02
Aclonifen	5.54E+00
Aldicarb	1.20E+03
Aldoxycarb	1.78E+02
Alpha-cypermethrin	1.98E+02
Aluminium phosphide	1.87E+03
Ametryn	1.01E+02
Amitraz	9.07E-01
Anthracene	3.26E+01
Antimony	3.32E+01
Arsenic	1.16E+01
Asulam	1.91E-01
Atrazine	1.78E+02
Azadirachtin	8.25E+01
Azinphos-methyl	2.36E+02
Barium	9.89E+00
Bendiocarb	2.86E+02
Benomyl	5.86E-01
Bentazone	2.45E+00
Beryllium	7.39E+02
Bifenox	8.04E-01
Bifenthrin	2.01E+02
Bitertanol	1.05E+00
Botran	1.53E+01
Bromacil	2.44E+00
Bromine	3.31E+02
Bromoxynil	1.88E+01
Bromuconazole	4.38E+01
Buprofezin	2.34E-01
Butylate	1.90E-01
Cadmium	1.13E+02
Captan	9.37E-01
Carbaryl	1.09E+01
Carbendazim	5.82E+01
Carbetamide	3.15E+00
Carbofuran	2.56E+02
Carboxin	8.10E+00



Substance	€/kg
Chlorfenvinphos	1.48E+02
Chloridazon	2.45E+00
Chlorimuron-ethyl	3.77E-03
Chlormequat	2.92E-02
Chlormequat chloride	7.36E-02
Chloropicrin	7.00E+02
Chlorothalonil	1.51E-01
Chlorotoluron	6.44E+00
Chlorpropham	8.91E-02
Chlorpyrifos	5.03E-01
Chlorsulfuron	2.34E-02
Chromium	5.40E-06
Chromium VI	3.17E-03
Clofentezine	6.71E-02
Clopyralid	9.40E+00
Cloquintocet-mexyl	4.64E-02
Cobalt	1.44E-20
Copper	4.24E-03
Cyanazine	5.83E-02
Cycloate	4.84E-01
Cycloxydim	3.53E-02
Cyfluthrin	1.26E-01
Cymoxanil	6.52E-02
Cypermethrin	5.29E+01
Cyromazine	5.24E+00
Daminozide	1.44E-03
DDAC	8.12E-01
Deltamethrin	6.71E-03
Desmedipham	3.24E-01
Desmetryn	1.19E+01
Diazinon	1.93E+00
Dicamba	2.09E-02
Dichlobenil	4.88E+00
Dichlorvos	2.40E-01
Dicofol	6.05E-01
Difenoconazole	5.16E+00
Difenzoquat	1.60E-02
Diflubenzuron	1.70E-02
Diflufenican	1.32E-02
Dimethipin	1.95E-01
Dimethoate	4.02E-02
Dimethomorph	2.62E+00
Dinoseb	5.61E+00
Dinoterb	2.43E+01
Diphenamid	8.36E-03
Dipropylthiocarbamic acid S-ethyl ester	3.42E-02
Diquat dibromide	1.69E+00
Disodium acid methane arsenate	4.28E+00
Disulfoton	7.87E-01
Dithianon	4.45E-02
Diuron	7.26E-02
DNOC	1.38E+01
Dodine	1.77E-03
Endosulfan	5.89E-02



Substance	€/kg
Endothall	1.74E-03
Epoxiconazole	1.74E+01
Esfenvalerate	3.09E+03
Ethephon	1.90E-01
Ethion	1.53E-01
Ethofumesate	2.16E+00
Ethoprop	4.29E+01
Etridiazole	5.44E+00
Fenamiphos	3.38E-01
Fenarimol	6.11E-01
Fenbuconazole	7.81E-02
Fenbutatin oxide	1.72E+01
Fenitrothion	1.22E-01
Fenpiclonil	9.60E+00
Fenpropathrin	2.84E-02
Fenpropimorph	7.26E-01
Fentin acetate	6.44E-01
Fentin hydroxide	4.07E-01
Fenvalerate	7.92E-02
Ferbam	1.67E+00
fertilizer P	9.40E-02
Fluazifop-P-butyl	8.92E-01
Fluazinam	5.32E+02
Fluometuron	9.41E-03
Fluorochloridone	4.36E+00
Flusilazole	2.31E-01
Flutolanil	5.61E-03
Folpet	5.20E-02
Fomesafen	1.92E-01
Fonofos	2.02E-01
Fosetyl-aluminium	1.45E-03
Fuberidazole	1.16E+01
Glufosinate ammonium	1.83E-01
Glyphosate	2.84E-05
Heptenophos	5.16E+00
Hexaconazole	3.44E-01
Hexazinone	7.15E-03
Hexythiazox	2.83E-02
Hymexazol	5.40E+00
Imazalil	2.40E-02
Imazaquin	1.02E-03
Ioxynil	8.28E+00
Iprodion	1.60E-03
Isofenphos	7.10E-01
Isoproturon	1.34E+02
Isoxaben	1.90E-02
Kresoxim-methyl	1.35E-02
Lactofen	1.87E+00
Lambda-cyhalothrin	6.04E+01
Lead	1.16E-01
Lindane	3.92E+00
Linuron	4.50E-01
Malathion	3.22E-04
Maleic hydrazide	5.20E-03



Substance	€/kg
Mancozeb	3.22E-05
Maneb	1.56E-03
Manganese	7.32E-02
manure P	8.88E-02
MCPA	1.47E-01
MCPB	5.07E-03
m-Cresol	2.09E-03
Mecoprop	1.40E+00
Mecoprop-P	5.00E-02
Mepiquat chloride	1.09E-02
Mercury	3.55E+02
Metalaxil	3.41E-02
Metamitron	1.14E+00
Metam-sodium	1.25E+03
Metazachlor	7.24E+00
Methabenzthiazuron	1.18E+01
Methamidophos	2.96E-01
Methidathion	9.02E-02
Methiocarb	1.09E-01
Methomyl	2.96E-01
Methoxychlor	1.18E-02
Metiram	1.57E-01
Metobromuron	1.99E+01
Metolachlor	2.69E-02
Metribuzin	2.77E-02
Metsulfuron-methyl	2.61E-03
Mevinfos	1.25E-02
Molinate	3.28E-01
Molybdenum	1.79E+00
Monolinuron	2.36E+01
Monosodium acid methanearsonate	5.08E+00
Myclobutanil	1.00E-02
Naled	6.88E-01
Napropamide	5.61E-03
Nickel	4.69E-02
Nicosulfuron	7.60E+00
Norflurazon	1.09E-02
Orbencarb	2.36E+00
Oryzalin	1.17E-02
Oxadixyl	1.88E+00
Oxamyl	1.06E-02
Oxydemeton-methyl	1.64E+01
Oxydiazon	4.27E-01
Oxyfluorfen	1.85E+00
Oxytetracycline	2.15E+02
Paclobutrazol	1.39E-02
Parathion	4.48E-02
Parathion, methyl	3.21E-01
Pendimethalin	4.07E-02
Phenmedipham	3.64E-05
Phorate	1.50E-01
Phosmet	1.33E-02
Phosphorus	6.72E+02
Picloram	1.96E-02



Substance	€/kg
Pirimicarb	3.20E+00
Pirimiphos methyl	1.03E-01
Prochloraz	2.14E-01
Procymidone	1.74E-02
Profenofos	1.14E-01
Prometryn	1.50E-02
Pronamide	8.20E-02
Propachlor	7.59E-03
Propamocarb	1.50E-02
Propamocarb HCl	1.90E+00
Propanil	1.74E-02
Propaquizafop	2.90E+00
Propargite	4.06E-02
Propene, 1,3-dichloro-	1.27E+00
Propham	1.25E-03
Propiconazole	1.05E-01
Propoxur	6.88E-02
Prosulfocarb	9.48E-01
Pyrazophos	2.05E-01
Pyridaben	2.19E+01
Pyridate	1.70E-02
Pyriproxyfen	5.83E-03
Quinmerac	3.34E-01
Quizalofop ethyl ester	1.14E-01
Resmethrin	2.38E-02
Rimsulfuron	3.06E+01
Rotenone	2.61E-01
Selenium	2.93E+02
Sethoxydim	4.30E-03
Silver	4.10E+01
Simazine	4.77E-01
Starane	6.04E-02
Sulprofos	5.24E-02
Tebufenozide	3.75E-02
Tebuthiuron	1.05E-02
Teflubenzuron	6.71E-01
Terbacil	2.02E-02
Terbufos	1.93E+01
Terbuthylazin	1.32E+01
Terbutryn	1.69E+00
Thallium	6.49E+00
Thiabendazole	1.59E-02
Thidiazuron	2.72E+01
Thifensulfuron-methyl	6.60E-02
Thiobencarb	1.45E-01
Thiodicarb	5.05E-02
Thiophanat-methyl	3.11E-03
Thiram	2.67E-02
Tin	7.65E-04
Tolclophos-methyl	1.39E-01
Tralomethrin	4.97E-01
Triadimefon	4.86E-02
Triadimenol	1.42E-02
Tri-allate	1.05E-01



Substance	€/kg
Triasulfuron	1.69E-01
Triazofos	1.29E+01
Trichlorfon	6.22E-03
Triclopyr	7.40E+00
Trifluralin	3.25E-01
Triforine	2.22E-01
Urea	1.31E-01
Vanadium	1.67E+00
Vinclozolin	1.04E-01
Zinc	2.55E-01
Zineb	5.43E-03

Table 52 Damage costs for emissions to water (obtained via method 3, with direct valuation of ReCiPe endpoint characterisation factors)

Substance	€/kg
1,4-Dioxane	6.57E-03
1-Butanol	1.02E-03
1-Octanol	9.02E-03
2-Benzothiazolethiol	4.44E-01
2-Butanol	1.36E-04
2-Butenal	6.28E-02
2-Ethoxyethyl acetate	3.38E-03
2-Propanol	5.54E-05
Acenaphthene	7.96E-02
Acetamide	1.49E-03
Acetic acid	2.08E-03
Acetone	4.22E-04
Acetonitrile	3.34E-03
Acetophenone	6.15E-03
Acrylamide	1.88E-01
Acrylic acid	1.79E-03
Acrylonitrile	2.26E-01
Allyl chloride	3.00E-02
Allylamine	1.10E-02
Aniline	4.43E-02
Aniline, p-chloro-, hydrochloride	1.02E-01
Anthracene	1.30E-01
Antimony	2.22E+01
Antimony-124	2.53E-02
Arsenic, ion	5.62E+02
Barium	1.56E+01
Benzaldehyde	1.20E-02
Benzenamine, 2-methoxy-5-nitro-	1.73E-02
Benzenamine, 4-methyl-	2.59E-01
Benzene	1.63E-02
Benzene, 1,2,4-trichloro-	3.19E-01
Benzene, 1,2,4-trimethyl-	3.32E-02
Benzene, 1,2-dichloro-	6.94E-02
Benzene, 1,2-dinitro-	3.97E+00
Benzene, 1,3,5-trimethyl-	1.03E-02
Benzene, 1,3-dichloro-	2.58E-02
Benzene, 1,3-dinitro-	2.44E+00
Benzene, 1,4-dichloro-	8.00E-02



Substance	€/kg
Benzene, 1,4-dinitro-	8.14E-01
Benzene, chloro-	5.81E-02
Benzene, ethyl-	7.84E-03
Benzidine, 3,3'-dimethyl-	5.57E-01
Benzyl chloride	5.07E-02
Beryllium	1.45E+01
Bis(2-chloro-1-methylethyl)ether	4.65E-02
Bis(2-chloroethyl)ether	3.27E-02
Bis(chloromethyl)ether	3.50E+02
Bromate	2.54E-01
Bromine	4.04E+00
C.I. direct blue 218	4.55E-04
C.I. disperse yellow 3	6.05E-02
Cadmium, ion	4.96E+00
Caprolactam	6.33E-04
Carbamic acid, butyl-, 3-iodo-2-propynyl ester	1.65E+00
Carbamic acid, ethyl ester	2.31E-03
Carboxylic acids, unspecified	1.37E-03
Catechol	5.04E-02
Cesium-134	4.44E+00
Cesium-137	5.16E+00
Chlorine	9.00E+00
Chloroacetic acid	5.40E-02
Chloroform	7.77E-01
Chromium VI	4.03E-02
Chromium, ion	2.84E-02
Cobalt	1.04E+00
Cobalt-58	1.26E-03
Cobalt-60	1.36E+00
Cumene	1.31E-02
Cumene hydroperoxide	1.09E-01
Cupferron	9.35E-02
Cyanide	7.54E-01
Cyclohexane	2.56E-03
Cyclohexanol	4.68E-04
Cyclohexene, 4-vinyl-	6.99E-02
Cyclohexylamine	1.86E-02
Decabromodiphenyl oxide	5.61E-05
Dibenzofuran	1.24E-01
Dicyclopentadiene	2.02E-02
Diethanolamine	1.17E-03
Dimethylamine	1.40E-03
Dipropylthiocarbamic acid S-ethyl ester	1.16E-01
EDTA	1.44E-03
Ethane, 1,1,1-trichloro-, HCFC-140	1.25E-02
Ethane, 1,1,2,2-tetrachloro-	1.46E-01
Ethane, 1,2-dibromo-	2.50E+00
Ethane, 1,2-dichloro-	6.67E-01
Ethane, chloro-	3.65E-03
Ethane, hexachloro-	4.58E+01
Ethanol, 2-butoxy-	3.38E-03
Ethanol, 2-methoxy-	5.96E-03
Ethene, 1,2-dichloro-	1.87E-03



Substance	€/kg
Ethene, chloro-	1.80E-01
Ethene, dichloro- (trans)	1.83E-01
Ethene, tetrachloro-	1.88E+00
Ethene, trichloro-	7.12E-03
Ethyl acrylate	3.60E-02
Ethylene diamine	2.26E-03
Ethylene glycol	2.95E-04
Ethylene oxide	7.68E-02
Fluoranthene	3.98E+00
Formaldehyde	2.14E-01
Formic acid	8.00E-04
Glyoxal	2.00E-03
Hydrazine	1.62E+00
Hydrocarbons, aromatic	9.22E-01
Hydrogen-3, Tritium	1.40E-05
Hydroquinone	4.20E-02
Iodine-129	3.09E+00
Iodine-131	1.55E-02
Lead	8.43E+00
Maleic anhydride	3.81E-07
Manganese	2.66E+01
Manganese-54	9.84E-03
m-Cresol	3.35E-02
Mercury	9.58E+02
Methacrylic acid, methyl ester	9.76E-03
Methane, dichloro-, HCC-30	3.87E+00
Methane, tetrachloro-, CFC-10	7.74E+02
Methanol	3.43E-04
Methyl acrylate	1.53E-02
Methyl ethyl ketone	2.42E-04
Molybdenum	4.89E+01
Molybdenum trioxide	4.26E-01
m-Phenylenediamine	2.01E-02
m-Xylene	1.35E-02
Naphthalene	1.22E-01
Nickel, ion	3.37E+00
Nitilotriacetic acid	9.05E-04
Nitrobenzene	7.62E-01
Nitroglycerin	5.59E-02
N-Methylolacrylamide	1.57E-02
N-Nitrosodiphenylamine	2.01E-01
o-Cresol	1.10E-02
o-Phenylenediamine	2.39E-02
o-Toluidine	2.51E-03
o-Xylene	1.45E-02
PAH, polycyclic aromatic hydrocarbons	1.29E-01
p-Cresidine	9.13E-03
p-Cresol	1.70E-03
Phenanthrene	1.47E-01
Phenol	2.27E-03
Phenol, 2,4,5-trichloro-	1.01E-01
Phenol, 2,4,6-trichloro-	3.63E-02
Phenol, 2,4-dichloro-	3.40E-02
Phenol, 2,4-dimethyl-	4.67E-02



Substance	€/kg
Phenol, 2,4-dinitro-	2.83E-01
Phenol, 2-nitro-	1.12E-01
Phenol, 4-nitro-	7.62E-03
Phosphate	1.80E+00
Phosphorus	1.44E+01
Phthalate, butyl-benzyl-	1.14E-01
Phthalate, dibutyl-	3.78E-01
Phthalate, dimethyl-	3.71E-03
Phthalate, dioctyl-	2.64E-01
p-Nitroaniline	5.79E-02
Polychlorinated biphenyls	4.73E+00
p-Phenylenediamine	1.94E-02
Propane, 1,2-dichloro-	4.54E+00
Propane, 2-nitro-	7.76E-02
Propene, 1,3-dichloro-	3.99E-01
Propylene oxide	1.82E-01
Pyrene	5.39E-01
Pyridine	6.31E-02
Radium-226	3.98E-03
Selenium	4.02E+02
Silver, ion	3.11E+01
Silver-110	1.55E-02
Sodium azide	6.62E-02
Sodium dimethyldithiocarbamate	1.16E+00
Sodium formate	2.28E-04
Styrene	2.65E-02
Sulphuric acid, dimethyl ester	7.32E-04
t-Butyl alcohol	2.25E-02
t-Butyl methyl ether	2.38E-03
Thallium	7.69E+01
Thiourea	5.31E-02
Tin, ion	3.42E-02
Toluene	6.19E-03
Toluene diisocyanate	4.72E+02
Toluene, 2,4,6-trinitro-	2.19E-03
Toluene, 2,4-diisocyanate	1.85E-03
Toluene, 2,4-dinitro-	4.52E-02
Toluene, 2,6-dinitro-	1.05E-01
Toluene, 2-chloro-	7.43E-02
Toluene, dinitro-	2.07E-01
Tribufos	7.41E+01
Tributyltin compounds	2.78E+02
Tributyltin oxide	2.78E+02
Triethyl amine	5.13E-04
Triethylene glycol	8.72E-06
Uranium alpha	7.04E-02
Uranium-234	7.48E-02
Uranium-235	7.04E-02
Uranium-238	7.04E-02
Vanadium, ion	1.70E+01
Vinyl acetate	1.19E-02
Xylene	7.20E-03
Zinc, ion	1.61E+00



J.2 List of abatement costs

The following table reports the abatement costs of over 400 substances. These costs have been calculated using the weighting factors of weighting set 1 developed in the present report (cf. Chapter 6) and the ReCiPe midpoint characterisation factors.

In the table three categories are distinguished: air, soil and water, indicating on which compartment(s) the pollutant in question has an impact. To obtain the damage costs of cadmium, for example, the costs of cadmium listed under 'air' and 'soil' must be summed.

This list of abatement costs should be used above all for weighting purposes. It is *not* the case that these costs represent the costs of technical measures to reduce emissions of a given pollutant. If they are used for valuation nonetheless, it should be borne in mind that this involves the following assumption: that government policy is both economically efficient and shaped in accordance with environmental relationship between pollutants (as set out in Section 4.3.3). As this is certainly not the case, due caution should be exercised if these implicit abatement costs are used for valuation purposes.

Table 53 Abatement costs for emissions to air

	€/kg
1,4-Dioxane	9.66E-01
1-Butanol	5.25E+00
1-Butene	9.10E+00
1-Butene, 2-methyl-	6.50E+00
1-Butene, 3-methyl-	5.65E+00
1-Hexene	7.40E+00
1-Pentene	8.25E+00
1-Propanol	4.74E+00
1-Propanol, 3,3,3-trifluoro-2,2-bis(trifluoromethyl)-, HFE-7100	7.43E+00
2-Acetylaminofluorene	2.39E+02
2-Benzothiazolethiol	6.58E-01
2-Butanol	3.38E+00
2-Butanone, 3,3-dimethyl-	2.73E+00
2-Butanone, 3-methyl-	3.08E+00
2-Butenal	3.89E+01
2-Butene (cis)	9.70E+00
2-Butene (trans)	9.55E+00
2-Hexanone	4.83E+00
2-Hexene (cis)	9.05E+00
2-Hexene (trans)	9.05E+00
2-Methyl pentane	3.55E+00
2-Methyl-1-propanol	3.04E+00
2-Methyl-2-butene	7.10E+00
2-Pentanone	4.63E+00
2-Pentene (cis)	9.45E+00
2-Pentene (trans)	9.45E+00
2-Propanol	1.59E+00
3-Hexanone	5.05E+00
3-Methyl-1-butanol	3.66E+00
3-Pentanol	5.05E+00
4-Methyl-2-pentanone	4.14E+00
Acetaldehyde	1.67E+01



	€/kg
Acetamide	1.45E+00
Acetic acid	8.20E-01
Acetic acid, methyl ester	4.99E-01
Acetic acid, propyl ester	2.38E+00
Acetone	1.14E+00
Acetonitrile	1.08E+00
Acetophenone	6.58E-01
Acrolein	1.42E+04
Acrylamide	8.95E+02
Acrylic acid	8.53E+02
Actinides, radioactive, unspecified	0.00E+00
Alcohol, diacetone	2.60E+00
Aldehydes, unspecified	1.96E+02
Allyl chloride	2.32E+01
Ammonia	1.16E+01
Aniline	8.90E+00
Aniline, p-chloro-, hydrochloride	8.12E+01
Anthracene	8.19E-01
Antimony	1.54E+04
Arsenic	1.13E+05
Barium	1.55E+03
Benzaldehyde	1.02E+00
Benzenamine, 2-methoxy-5-nitro-	6.69E+00
Benzene	4.97E+00
Benzene, (epoxyethyl)-	5.20E+00
Benzene, 1,2,3-trimethyl-	1.07E+01
Benzene, 1,2,4-trichloro-	1.49E+01
Benzene, 1,2,4-trimethyl-	1.08E+01
Benzene, 1,2-dichloro-	2.69E+00
Benzene, 1,3,5-trimethyl-	1.17E+01
Benzene, 1,3-dinitro-	9.57E+02
Benzene, 1,4-dichloro-	2.30E+00
Benzene, 1-propyl-	5.35E+00
Benzene, 3,5-dimethylethyl-	1.12E+01
Benzene, ethyl-	6.34E+00
Benzene, hexachloro-	1.32E+03
Benzene, pentachloro-	3.40E+02
Benzidine	2.21E+02
Benzidine, 3,3'-dichloro-	1.38E+02
Benzo(a)pyrene	1.23E+02
Benzotrichloride	1.46E+03
Benzyl chloride	9.34E+00
Beryllium	8.17E+05
Biphenyl, 4-amino-	8.30E+01
Bis(2-chloro-1-methylethyl)ether	1.61E+01
Bis(2-chloroethyl)ether	4.72E+01
Bromoform	2.92E+01
Butadiene	1.87E+01
Butadiene, hexachloro-	4.05E+01
Butanal	6.70E+00
Butane	2.98E+00
Butane, 1,1,1,3,3-pentafluoro-, HFC-365mfc	1.99E+01
Butane, 2,2-dimethyl-	2.04E+00
Butane, 2,3-dimethyl-	4.57E+00



	€/kg
Butane, nonafluoroethoxy, HFE-569sf2	1.48E+00
Butane, perfluoro-	2.22E+02
Butane, perfluorocyclo-, PFC-318	2.58E+02
Butanol, 2-methyl-1-	4.13E+00
Butanol, 2-methyl-2-	1.93E+00
Butanol, 3-methyl-2-	3.43E+00
Butene, 1,4-dichloro-2- (trans)	1.98E+02
Butyl acetate	2.27E+00
C.I. disperse yellow 3	1.87E+00
Cadmium	8.00E+04
Caprolactam	1.62E+00
Carbamic acid, ethyl ester	2.46E+00
Carbon dioxide	2.50E-02
Carbon dioxide, fossil	2.50E-02
Carbon dioxide, land transformation	2.50E-02
Carbon disulfide	1.09E+02
Carbon monoxide, biogenic	2.28E-01
Carbon monoxide, fossil	2.28E-01
Carbon-14	0.00E+00
Catechol	9.52E-01
Cesium-134	0.00E+00
Cesium-137	0.00E+00
Chlorendic acid	1.80E+01
Chlorine	4.81E+02
Chloroform	7.25E+01
Chloromethyl methyl ether	5.57E+01
Chromium	7.13E-01
Chromium VI	1.27E+04
Cobalt	9.91E+03
Cobalt-58	0.00E+00
Cobalt-60	0.00E+00
Copper	3.80E+01
Cumene	4.47E+00
Cyanide	1.63E+01
Cyclohexane	2.45E+00
Cyclohexanol	4.38E+00
Cyclohexanone	2.53E+00
Cyclohexene, 4-vinyl-	1.60E+00
Cyclohexylamine	3.27E+00
Cyclopentadiene, hexachloro-	3.43E+03
Decabromodiphenyl oxide	1.88E+03
Decane	3.25E+00
Diethyl ether	3.76E+00
Diethyl ketone	3.50E+00
Diglycidyl resorcinol ether	1.41E+01
Dihydrosafrole	9.61E-01
Diisopropyl ether	3.36E+00
Dimethyl carbonate	2.11E-01
Dimethyl ether	1.62E+00
Dimethylcarbanyl chloride	4.97E+02
Dinitrogen monoxide	7.45E+00
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	2.24E+08
Dipropylthiocarbamic acid S-ethyl ester	3.15E+01
Dodecane	3.02E+00



	€/kg
Epichlorohydrin	2.94E+02
Ethane	1.04E+00
Ethane, 1,1,1,2-tetrachloro-	2.53E+01
Ethane, 1,1,1,2-tetrafluoro-, HFC-134a	3.61E+01
Ethane, 1,1,1,2-tetrafluoro-2-bromo-, Halon 2401	7.50E+00
Ethane, 1,1,1-trichloro-, HCFC-140	3.73E+00
Ethane, 1,1,1-trifluoro-, HFC-143a	1.12E+02
Ethane, 1,1,1-trifluoro-2,2-chlorobromo-, Halon 2311	4.20E+00
Ethane, 1,1,2,2-tetrachloro-	1.94E+01
Ethane, 1,1,2,2-tetrafluoro-, HFC-134	2.75E+01
Ethane, 1,1,2-trichloro-	4.50E+02
Ethane, 1,1,2-trichloro-1,2,2-trifluoro-, CFC-113	1.91E+02
Ethane, 1,1,2-trifluoro-, HFC-143	8.83E+00
Ethane, 1,1-dichloro-1-fluoro-, HCFC-141b	2.37E+01
Ethane, 1,1-difluoro-, HFC-152a	3.13E+00
Ethane, 1,2-dibromo-	3.38E+02
Ethane, 1,2-dibromotetrafluoro-, Halon 2402	2.21E+02
Ethane, 1,2-dichloro-	7.52E+01
Ethane, 1,2-dichloro-1,1,2,2-tetrafluoro-, CFC-114	2.78E+02
Ethane, 1,2-difluoro-, HFC-152	1.33E+00
Ethane, 1-chloro-1,1-difluoro-, HCFC-142b	5.99E+01
Ethane, 1-chloro-2,2,2-trifluoro-(difluoromethoxy)-, HCFC-235da2	8.75E+00
Ethane, 2,2-dichloro-1,1,1-trifluoro-, HCFC-123	3.39E+00
Ethane, 2-chloro-1,1,1,2-tetrafluoro-, HCFC-124	1.58E+01
Ethane, 2-chloro-1,1,1-trifluoro-, HCFC-133a	3.47E+01
Ethane, chloro-	5.50E-01
Ethane, chloropentafluoro-, CFC-115	1.97E+02
Ethane, fluoro-, HFC-161	3.00E-01
Ethane, hexafluoro-, HFC-116	3.05E+02
Ethane, pentachloro-	2.58E+01
Ethane, pentafluoro-, HFC-125	8.75E+01
Ethanol	3.40E+00
Ethanol, 2-butoxy-	4.08E+00
Ethanol, 2-methoxy-	2.60E+00
Ethene	8.45E+00
Ethene, chloro-	3.54E+01
Ethene, dichloro- (cis)	3.78E+00
Ethene, dichloro- (trans)	3.31E+00
Ethene, tetrachloro-	1.76E+02
Ethene, trichloro-	3.16E+00
Ether, 1,1,1-trifluoromethyl methyl-, HFE-143a	1.89E+01
Ether, 1,1,2,2-Tetrafluoroethyl 2,2,2-trifluoroethyl-, HFE-347mcc3	1.44E+01
Ether, 1,1,2,2-Tetrafluoroethyl 2,2,2-trifluoroethyl-, HFE-347mcf2	9.35E+00
Ether, 1,1,2,2-Tetrafluoroethyl methyl-, HFE-254cb2	8.98E+00
Ether, 1,1,2,3,3,3-Hexafluoropropyl methyl-, HFE-356mec3	2.53E+00
Ether, 1,1,2,3,3,3-Hexafluoropropyl methyl-, HFE-356pcc3	2.75E+00
Ether, 1,1,2,3,3,3-Hexafluoropropyl methyl-, HFE-356pcf2	6.63E+00
Ether, 1,1,2,3,3,3-Hexafluoropropyl methyl-, HFE-356pcf3	1.26E+01
Ether, 1,2,2-trifluoroethyl trifluoromethyl-, HFE-236ea2	2.47E+01
Ether, 1,2,2-trifluoroethyl trifluoromethyl-, HFE-236fa	1.22E+01
Ether, 2,2,3,3,3-Pentafluoropropyl methyl-, HFE-365mcf3	2.75E-01



	€/kg
Ether, di(difluoromethyl), HFE-134	1.58E+02
Ether, difluoromethyl 2,2,2-trifluoroethyl-, HFE-245cb2	1.77E+01
Ether, difluoromethyl 2,2,2-trifluoroethyl-, HFE-245fa1	7.15E+00
Ether, difluoromethyl 2,2,2-trifluoroethyl-, HFE-245fa2	1.65E+01
Ether, ethyl 1,1,2,2-tetrafluoroethyl-, HFE-374pc2	1.39E+01
Ether, pentafluoromethyl-, HFE-125	3.73E+02
Ethyl acetate	1.77E+00
Ethyl acrylate	1.88E+00
Ethylene glycol	3.15E+00
Ethylene glycol monoethyl ether	3.26E+00
Ethylene oxide	2.32E+01
Ethyne	7.20E-01
Fertiliser, nitrogen	5.11E-01
Fluoranthene	4.35E+00
Formaldehyde	2.67E+02
Formic acid	2.71E-01
Heptane	4.17E+00
Hexane	7.11E+00
Hexane, 2-methyl-	3.47E+00
Hexane, 3-methyl-	3.08E+00
Hexane, perfluoro-	2.33E+02
HFE-227EA	3.85E+01
HFE-236ca12 (HG-10)	7.00E+01
HFE-263fb2	2.75E-01
HFE-329mcc2	2.30E+01
HFE-338mcf2	1.38E+01
HFE-338pcc13 (HG-01)	3.75E+01
HFE-347pcf2	1.45E+01
HFE-43-10pccc124 (H-Galden1040x)	4.68E+01
Hydrazine, methyl-	1.23E+01
Hydrocarbons, aliphatic, alkanes, cyclic	2.40E+00
Hydrocarbons, aromatic	5.04E+00
Hydrocarbons, chlorinated	1.20E+02
Hydrogen fluoride	6.12E+02
Hydrogen-3, Tritium	0.00E+00
Hydroquinone	6.53E+00
Iodine-129	0.00E+00
Iodine-131	0.00E+00
Iodine-133	0.00E+00
Isobutane	2.60E+00
Isobutene	5.30E+00
Isobutyraldehyde	4.34E+00
Isopentane	3.42E+00
Isoprene	9.20E+00
Isopropyl acetate	1.78E+00
Krypton-85	0.00E+00
Lead	3.54E+04
Lead-210	0.00E+00
Maleic anhydride	9.34E+00
Manganese	7.36E+03
manure N	5.53E-01
m-Cresol	2.15E+00
Mercury	1.18E+06
Methacrylic acid, methyl ester	1.74E+01



	€/kg
Methane	6.25E-01
Methane, biogenic	6.76E-01
Methane, bromo-, Halon 1001	8.13E+03
Methane, bromochlorodifluoro-, Halon 1211	2.27E+02
Methane, bromodifluoro-, Halon 1201	5.21E+01
Methane, bromotrifluoro-, Halon 1301	5.39E+02
Methane, chlorodifluoro-, HCFC-22	4.69E+01
Methane, chlorofluoro-, HCFC-31	4.23E+01
Methane, chlorotrifluoro-, CFC-13	3.60E+02
Methane, dibromo-	3.85E-02
Methane, dibromodifluoro-, Halon 1202	3.90E+01
Methane, dichloro-, HCC-30	5.42E+01
Methane, dichlorodifluoro-, CFC-12	1.62E+03
Methane, dichlorofluoro-, HCFC-21	3.78E+00
Methane, difluoro-, HFC-32	1.69E+01
Methane, dimethoxy-	1.39E+00
Methane, fluoro-, HFC-41	2.30E+00
Methane, fossil	6.76E-01
Methane, iodotrifluoro-	1.00E-02
Methane, monochloro-, R-40	1.36E+01
Methane, tetrachloro-, CFC-10	1.39E+04
Methane, tetrafluoro-, CFC-14	1.85E+02
Methane, trichlorofluoro-, CFC-11	3.02E+02
Methane, trifluoro-, HFC-23	3.70E+02
Methanol	1.75E+00
Methyl ethyl ketone	3.15E+00
Methyl formate	2.28E-01
Molybdenum	3.54E+03
Molybdenum trioxide	2.05E+02
m-Phenylenediamine	7.48E+00
m-Xylene	9.35E+00
Naphthalene	1.88E+01
Nickel	1.01E+03
Nitrate	8.96E-01
Nitrotriacetic acid	6.60E-02
Nitrobenzene	2.88E+02
Nitrogen fluoride	4.30E+02
Nitrogen oxides	8.72E+00
Nitroglycerin	1.60E+00
N-Methylolacrylamide	1.24E+01
NM VOC, non-methane volatile organic compounds, unspecified origin	5.00E+00
N-Nitrosodiethylamine	1.46E+04
N-Nitrosodiphenylamine	1.50E+00
N-Nitrosodipropylamine	7.77E+03
N-Nitrosopiperidine	3.57E+01
Noble gases, radioactive, unspecified	0.00E+00
Nonane	3.50E+00
o-Cresol	2.71E+00
Octane	3.83E+00
o-Phenylenediamine dihydrochloride	2.74E-01
o-Toluidine hydrochloride	1.67E+00
o-Xylene	8.90E+00
PAH, polycyclic aromatic hydrocarbons	4.72E+01



	€/kg
Particulates, < 10 um	2.30E+00
Particulates, < 10 um (mobile)	2.30E+00
Particulates, < 10 um (stationary)	2.30E+00
Particulates, < 2.5 um	2.30E+00
Particulates, > 2.5 um, and < 10um	2.30E+00
p-Cresidine	5.29E-01
Pentanal	6.45E+00
Pentane	3.34E+00
Pentane, 2,3-dihydroperfluoro-, HFC-4310mee	4.10E+01
Pentane, 3-methyl-	4.05E+00
Pentane, perfluoro-	2.29E+02
PFC-9-1-18	1.88E+02
PFPME	2.58E+02
Phenol	2.13E+00
Phenol, 2,4,5-trichloro-	5.80E+00
Phenol, 2,4,6-trichloro-	1.77E+00
Phenol, 2,4-dichloro-	8.17E+02
Phenol, 2,4-dimethyl-	6.33E+00
Phenol, 2,4-dinitro-	1.50E+02
Phenol, pentachloro-	1.00E+02
Phosphorus	4.32E+04
Phthalate, dimethyl tere-	5.38E+00
Plutonium-238	0.00E+00
Plutonium-alpha	0.00E+00
Polonium-210	0.00E+00
Polychlorinated biphenyls	8.30E+01
Propanal	6.75E+00
Propane	1.49E+00
Propane sultone	1.18E+02
Propane, 1,1,1,2,2,3-hexafluoro-, HFC-236cb	3.35E+01
Propane, 1,1,1,2,3,3,3-heptafluoro-, HFC-227ea	8.05E+01
Propane, 1,1,1,2,3,3-hexafluoro-, HFC-236ea	3.43E+01
Propane, 1,1,1,3,3,3-hexafluoro-, HCFC-236fa	2.45E+02
Propane, 1,1,2,2,3-pentafluoro-, HFC-245ca	1.73E+01
Propane, 1,1,3,3-tetrafluoro-, HFC-245fa	2.58E+01
Propane, 1,2-dibromo-3-chloro-	2.46E+03
Propane, 1,2-dichloro-	6.14E+02
Propane, 1,3-dichloro-1,1,2,2,3-pentafluoro-, HCFC-225cb	1.58E+01
Propane, 2,2-dimethyl-	1.46E+00
Propane, 2-nitro-	1.89E+01
Propane, 3,3-dichloro-1,1,1,2,2-pentafluoro-, HCFC-225ca	3.65E+00
Propane, perfluoro-	2.21E+02
Propargyl alcohol	4.49E+01
Propene	9.50E+00
Propene, 1,3-dichloro-	2.78E+03
Propene, 1-chloro-1-	8.58E+01
Propionic acid	1.27E+00
Propylene glycol	3.86E+00
Propylene glycol methyl ether	3.00E+00
Propylene glycol t-butyl ether	3.91E+00
Propylene oxide	9.75E+01
p-Xylene	8.55E+00
Pyrene	7.73E+00
Pyridine	2.44E+02



	€/kg
Radium-226	0.00E+00
Radon-222	0.00E+00
Safrole	2.58E-01
s-Butyl acetate	2.33E+00
Selenium	1.88E+04
Silver	8.92E+04
Sodium azide	7.25E+01
Styrene	8.47E+00
Sulfur dioxide	5.00E+00
Sulfur hexafluoride	5.70E+02
Sulfur oxides	5.00E+00
Sulphur, trifluoromethyl pentafluoride	4.43E+02
t-Butyl acetate	4.48E-01
t-Butyl alcohol	8.95E-01
t-Butyl ethyl ether	2.06E+00
t-Butyl methyl ether	2.19E+00
Thallium	1.29E+04
Thioacetamide	1.87E+01
Thiourea	1.55E+00
Thorium-230	0.00E+00
Tin	1.17E+02
Toluene	7.29E+00
Toluene diisocyanate	2.99E+04
Toluene, 2,4,6-trinitro-	1.14E+03
Toluene, 2,4-diamine	3.63E+01
Toluene, 2,4-dinitro-	3.31E+02
Toluene, 2,6-dinitro-	3.38E+03
Toluene, 2-chloro-	1.64E+01
Toluene, 2-ethyl-	7.60E+00
Toluene, 3,5-diethyl-	1.10E+01
Toluene, 3-ethyl-	8.60E+00
Toluene, 4-ethyl-	7.65E+00
Toluene, dinitro-	6.42E+01
Triethyl amine	1.81E+00
Undecane	3.25E+00
Uranium	0.00E+00
Uranium alpha	0.00E+00
Uranium-234	0.00E+00
Uranium-235	0.00E+00
Uranium-238	0.00E+00
Vanadium	7.96E+03
Vinyl acetate	7.84E+00
Xenon-133	0.00E+00
Xylene	2.37E+00
Zinc	1.13E+03



Table 54 Abatement costs for emissions to soil

	€/kg
2,4,5-T	3.93E+00
2,4-D	8.81E-01
Abamectin	1.03E+02
Acephate	2.17E+02
Acifluorfen	1.10E+00
Aldicarb	6.07E+02
Alpha-cypermethrin	3.89E+00
Aluminum phosphide	7.73E+01
Amitraz	1.13E+01
Anthracene	1.15E-01
Antimony	6.97E+02
Arsenic	6.26E+02
Asulam	6.83E-03
Atrazine	5.18E+00
Azinphos-methyl	1.65E+00
Barium	3.80E+02
Bendiocarb	7.75E+00
Benomyl	2.83E-03
Bentazone	5.75E+00
Beryllium	7.43E+01
Bifenthrin	1.11E+01
Bitertanol	2.16E+00
Botran	2.74E+00
Bromoxynil	1.07E+00
Buprofezin	1.12E+01
Butylate	3.98E-01
Cadmium	1.27E+03
Captan	1.67E-03
Carbaryl	6.88E-01
Carbendazim	5.84E-01
Carbofuran	2.71E+01
Carboxin	3.93E-01
Chlorfenvinphos	7.38E+02
Chlorimuron-ethyl	2.25E-01
Chlormequat	1.75E+00
Chlorothalonil	8.99E+00
Chlorpropham	5.31E+00
Chlorpyrifos	3.01E+01
Chlorsulfuron	1.40E+00
Chromium	3.22E-04
Chromium VI	1.90E-01
Clofentezine	4.03E+00
Cobalt	8.63E-19
Copper	2.53E-01
Cyanazine	3.47E+00
Cyfluthrin	7.52E+00
Cypermethrin	3.15E+03
Cyromazine	3.13E+02
Daminozide	8.60E-02
Deltamethrin	4.00E-01
Diazinon	1.15E+02
Dicamba	1.25E+00
Dichlorvos	1.44E+01



	€/kg
Dicofol	3.61E+01
Difenzoquat	9.52E-01
Diflubenzuron	1.01E+00
Dimethipin	1.16E+01
Dimethoate	2.39E+00
Dinoseb	3.34E+02
Diphenamid	5.01E-01
Dipropylthiocarbamic acid S-ethyl ester	2.04E+00
Diquat dibromide	1.01E+02
Disulfoton	4.69E+01
Dithianon	2.67E+00
Diuron	4.32E+00
Dodine	1.06E-01
Endosulfan	3.52E+00
Endothall	1.04E-01
Ethephon	1.13E+01
Ethion	9.15E+00
Ethoprop	2.55E+03
Fenamiphos	2.02E+01
Fenarimol	3.63E+01
Fenbuconazole	4.67E+00
Fenbutatin oxide	1.03E+03
Fenitrothion	7.27E+00
Fenpropathrin	1.70E+00
Fenpropimorph	4.35E+01
Fentin acetate	3.86E+01
Fentin hydroxide	2.44E+01
Fenvalerate	4.72E+00
Ferbam	9.94E+01
Fertiliser, nitrogen	5.11E-01
fertilizer P	5.77E-01
Fluometuron	5.61E-01
Flusilazole	1.38E+01
Flutolanil	3.36E-01
Folpet	3.11E+00
Fomesafen	1.15E+01
Fonofos	1.21E+01
Fosetyl-aluminium	8.65E-02
Glufosinate ammonium	1.09E+01
Glyphosate	1.70E-03
Hexaconazole	2.05E+01
Hexazinone	4.26E-01
Hexythiazox	1.69E+00
Imazalil	1.44E+00
Imazaquin	6.10E-02
Iprodion	9.52E-02
Isofenphos	4.26E+01
Isoxaben	1.14E+00
Kresoxim-methyl	8.05E-01
Lactofen	1.12E+02
Lead	6.90E+00
Lindane	2.35E+02
Linuron	2.69E+01
Malathion	1.92E-02



	€/kg
Maleic hydrazide	3.11E-01
Mancozeb	1.93E-03
Maneb	9.29E-02
Manganese	4.37E+00
manure N	5.53E-01
manure P	5.45E-01
MCPA	8.79E+00
MCPB	3.04E-01
m-Cresol	1.25E-01
Mepiquat chloride	6.51E-01
Mercury	2.12E+04
Metalaxil	2.04E+00
Methamidophos	1.77E+01
Methidathion	5.38E+00
Methiocarb	6.51E+00
Methomyl	1.77E+01
Methoxychlor	7.06E-01
Metiram	9.41E+00
Metolachlor	1.60E+00
Metribuzin	1.65E+00
Metsulfuron-methyl	1.56E-01
Mevinfos	7.48E-01
Molinate	1.96E+01
Molybdenum	1.07E+02
Myclobutanil	5.98E-01
Naled	4.12E+01
Napropamide	3.36E-01
Nickel	2.81E+00
Norflurazon	6.49E-01
Oryzalin	7.02E-01
Oxamyl	6.33E-01
Oxydiazon	2.55E+01
Oxyfluorfen	1.11E+02
Paclobutrazol	8.28E-01
Parathion	2.67E+00
Parathion, methyl	1.91E+01
Pendimethalin	2.44E+00
Phenmedipham	2.17E-03
Phorate	8.95E+00
Phosmet	7.94E-01
Picloram	1.17E+00
Pirimiphos methyl	6.16E+00
Prochloraz	1.28E+01
Procymidone	1.04E+00
Profenofos	6.81E+00
Prometryn	8.99E-01
Pronamide	4.88E+00
Propachlor	4.53E-01
Propamocarb	8.95E-01
Propanil	1.04E+00
Propargite	2.42E+00
Propene, 1,3-dichloro-	7.59E+01
Propham	7.50E-02
Propiconazole	6.28E+00



	€/kg
Propoxur	4.09E+00
Pyrazophos	1.22E+01
Pyriproxyfen	3.47E-01
Quizalofop ethyl ester	6.79E+00
Resmethrin	1.42E+00
Rotenone	1.56E+01
Selenium	1.75E+04
Sethoxydim	2.58E-01
Silver	2.46E+03
Simazine	2.85E+01
Tebufenozide	2.24E+00
Tebuthiuron	6.26E-01
Teflubenzuron	4.03E+01
Terbacil	1.21E+00
Terbufos	1.15E+03
Terbutryn	1.01E+02
Thallium	3.89E+02
Thiabendazole	9.50E-01
Thifensulfuron-methyl	3.96E+00
Thiobencarb	8.67E+00
Thiodicarb	3.01E+00
Thiophanat-methyl	1.86E-01
Thiram	1.60E+00
Tin	4.58E-02
Tolclophos-methyl	8.33E+00
Tralomethrin	2.97E+01
Triadimefon	2.90E+00
Triadimenol	8.49E-01
Tri-allate	6.23E+00
Triasulfuron	1.01E+01
Triazofos	7.71E+02
Trichlorfon	3.73E-01
Trifluralin	1.94E+01
Triforine	1.32E+01
Vanadium	9.96E+01
Vinclozolin	6.21E+00
Zinc	1.52E+01
Zineb	3.24E-01



Table 55 Abatement costs for emissions to water

	€/kg
1,4-Dioxane	3.43E-01
1-Butanol	4.05E-02
2-Benzothiazolethiol	3.15E-01
2-Butenal	3.06E+00
Acenaphthene	1.96E-01
Acetamide	8.76E-02
Acetone	2.01E-02
Acetonitrile	1.10E-01
Acetophenone	3.77E-02
Acrylamide	1.12E+01
Acrylic acid	3.34E-02
Acrylonitrile	1.21E+01
Allyl chloride	1.35E+00
Ammonium. ion	7.00E+00
Aniline	1.48E+00
Aniline. p-chloro-. hydrochloride	6.12E+00
Anthracene	7.73E-03
Antimony	1.29E+03
Antimony-124	0.00E+00
Arsenic. ion	3.36E+04
Barium	9.27E+02
Benzaldehyde	5.82E-02
Benzenamine. 2-methoxy-5-nitro-	1.04E+00
Benzene	8.26E-01
Benzene. 1.2.4-trichloro-	1.05E+01
Benzene. 1.2-dichloro-	1.28E+00
Benzene. 1.3-dinitro-	6.62E+01
Benzene. 1.4-dichloro-	1.37E+00
Benzene. chloro-	1.80E+00
Benzene. ethyl-	5.04E-02
Benzyl chloride	1.09E+00
Beryllium	4.90E+01
Bis(2-chloro-1-methylethyl)ether	2.78E+00
Bis(2-chloroethyl)ether	1.85E+00
Bis(chloromethyl)ether	2.09E+04
Bromate	1.47E+01
C.I. direct blue 218	2.71E-02
C.I. disperse yellow 3	3.61E+00
Cadmium. ion	2.78E+02
Caprolactam	3.11E-02
Carbamic acid. ethyl ester	1.27E-01
Carboxylic acids. unspecified	1.91E-02
Catechol	1.67E-01
Cesium-134	0.00E+00
Cesium-137	0.00E+00
Chlorine	3.91E+02
Chloroform	4.58E+01
Chromium VI	7.08E-01
Chromium. ion	1.21E-03
Cobalt	4.30E-19
Cobalt-58	0.00E+00
Cobalt-60	0.00E+00
Cumene	6.42E-02



	€/kg
Cupferron	5.59E+00
Cyanide	8.64E+00
Cyclohexane	4.76E-02
Cyclohexene. 4-vinyl-	4.16E+00
Cyclohexylamine	2.78E-01
Decabromodiphenyl oxide	3.34E-03
Dipropylthiocarbamic acid S-ethyl ester	4.90E+00
Ethane. 1.1.2.2-tetrachloro-	6.35E+00
Ethane. 1.2-dibromo-	1.47E+02
Ethane. 1.2-dichloro-	3.93E+01
Ethane. chloro-	2.18E-01
Ethane. hexachloro-	2.71E+03
Ethanol. 2-butoxy-	1.82E-01
Ethanol. 2-methoxy-	3.57E-01
Ethene. chloro-	1.07E+01
Ethene. dichloro- (trans)	1.09E+01
Ethene. tetrachloro-	1.09E+02
Ethene. trichloro-	1.26E-01
Ethyl acrylate	1.15E-01
Ethylene glycol	1.58E-02
Ethylene oxide	4.28E+00
Fluoranthene	8.60E+00
Formaldehyde	1.23E+01
Hydrazine	2.85E+01
Hydrocarbons. aromatic	3.36E+01
Hydrogen-3. Tritium	0.00E+00
Hydroquinone	2.48E-02
Iodine-129	0.00E+00
Iodine-131	0.00E+00
Lead	5.01E+02
Maleic anhydride	1.52E-05
Manganese	1.58E+03
Manganese-54	0.00E+00
m-Cresol	2.05E-01
Mercury	5.68E+04
Methacrylic acid. methyl ester	5.45E-01
Methane. dichloro-. HCC-30	2.64E+01
Methane. tetrachloro-. CFC-10	1.33E+04
Methanol	1.24E-02
Methyl ethyl ketone	6.30E-03
Molybdenum	2.92E+03
Molybdenum trioxide	2.55E+01
m-Phenylenediamine	1.20E+00
Naphthalene	4.76E+00
Nickel. ion	1.57E+01
Nitrate	7.14E+00
Nitrilotriacetic acid	4.32E-03
Nitrite	6.90E+00
Nitrobenzene	4.12E+01
Nitrogen	7.00E+00
Nitrogen. organic bound	7.00E+00
Nitroglycerin	2.03E-01
N-Methylolacrylamide	9.34E-01
N-Nitrosodiphenylamine	4.85E+00



	€/kg
o-Cresol	9.34E-02
PAH. polycyclic aromatic hydrocarbons	4.55E+00
p-Cresidine	5.43E-01
Phenol	2.60E-02
Phenol. 2.4.5-trichloro-	8.99E-01
Phenol. 2.4.6-trichloro-	3.47E-01
Phenol. 2.4-dichloro-	1.92E+00
Phenol. 2.4-dimethyl-	1.11E+00
Phenol. 2.4-dinitro-	5.84E+00
Phosphate	1.10E+01
Phosphorus	1.09E+01
Phthalate. butyl-benzyl-	6.65E-02
Phthalate. dibutyl-	4.30E-01
Phthalate. dioctyl-	1.52E+01
Polychlorinated biphenyls	2.55E+02
Propane. 1.2-dichloro-	2.71E+02
Propane. 2-nitro-	4.62E+00
Propene. 1.3-dichloro-	2.32E+01
Propylene oxide	1.05E+01
Pyrene	3.98E-01
Pyridine	3.61E+00
Radium-226	0.00E+00
Selenium	2.39E+04
Silver. ion	1.11E+03
Silver-110	0.00E+00
Sodium azide	8.69E-01
Styrene	9.11E-01
t-Butyl alcohol	1.27E+00
t-Butyl methyl ether	1.22E-01
Thallium	4.46E+03
Thiourea	1.60E-01
Tin. ion	2.85E-01
Toluene	1.61E-01
Toluene diisocyanate	2.81E+04
Toluene. 2.4.6-trinitro-	9.38E-02
Toluene. 2.4-dinitro-	1.26E+00
Toluene. 2.6-dinitro-	6.19E+00
Toluene. 2-chloro-	3.91E+00
Toluene. dinitro-	1.24E+01
Tribufos	4.32E+03
Tributyltin compounds	7.94E+02
Uranium alpha	0.00E+00
Uranium-234	0.00E+00
Uranium-235	0.00E+00
Uranium-238	0.00E+00
Vanadium. ion	8.37E+02
Xylene	1.73E-01
Zinc. ion	8.19E+01

