# Shadow Prices Handbook

## Valuation and weighting of emissions and environmental impacts

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# Preface

We are pleased to present this Shadow Prices Handbook, which sets out at length the methodology employed for our shadow price calculations and the use of these prices to derive weighting factors for individual environmental themes. This means the Handbook can also serve as a useful scientific background document, in which all relevant factors, methodological choices and assumptions are explicitly cited.

The document also presents two sets of shadow prices and weighting factors. These data can be used in a numerous types of economic and environmental analysis, provided it is borne in mind that they are average values for the Netherlands and that local conditions may vary. The cited prices are valid for the year 2008. With respect to government policy, the reference situation is that of around September 2009.

# Contents

# Summary

## Introduction

***What are shadow prices?***
Shadow prices are constructed prices for goods or production factors that are not traded in markets. Environmental quality is one example. By using these shadow prices, the environment can nonetheless be included in various kinds of economic analysis. Shadow prices then provide an indication of the value of a particular good – in this case environmental quality – to society.

In the context of the present study, shadow prices can be regarded as estimates of the value of environmental goods. These estimates can be used in a variety of applications, including cost-benefit analyses and investment decisions, or as a weighting method for comparing the relative severity of different environmental impacts.

***Aim and nature of this study***
It was in 2002 that CE Delft last elaborated a set of shadow prices and these are now seriously outdated. This study has several aims: to develop a new set of shadow prices that can be used for valuing and weighting environmental impacts; to explain the use of shadow prices; and to provide a user's manual giving guidance on what kind of shadow price should be used in what situation.

As a unique addition, the present study has combined the most recent developments in the environmental science of characterisation factors and the economics of valuation into a methodologically consistent framework. In this way a contribution has been made to both the literature on valuation of externalities and the environmental science literature concerned with elaborating weighting factors.

## Use of shadow prices
Valuation and weighting are two key elements of the use of shadow prices.

**Valuation** is used in analysing the wider social consequences of investment decisions. With the aid of shadow prices, environmental impacts can be taken on board along with financial considerations and compared with one another, as in Social Cost-Benefit Analyses (SCBAs), for example. Here the aim is to use shadow prices to obtain an as comprehensive assessment as possible of all the impacts attending the (investment) decision. With valuation, use is generally made of the shadow prices of individual emissions.

**Weighting**, in contrast, is used mainly in environmental impact analyses in which the identified impacts are compared. Weighting of environmental impacts is sometimes carried out as a final step in Life Cycle Analysis (LCA) to condense the results into a single, uniform figure. Financial valuation is frequently employed as a weighting method in LCAs and in practical calculation tools like the 'Envirometer' (for small and medium-sized businesses) and GreenCalc (for comparing the environmental performance of buildings). With weighting, use is generally made of the shadow prices of environmental themes.

In this study, shadow prices have been elaborated both for individual emissions and for environmental themes, with the linkages between individual emissions and themes being established using characterisation factors. Characterisation factors provide an indication of the relative importance of a pollutant in terms of its contribution to a particular environmental impact.

## Quantitative results on shadow prices of emissions

Table 1 summarises some of the most frequently used shadow prices calculated in the present study. They are all expressed in €/kg emission from Dutch territory in 2008 (i.e. in 2008 prices). In this study, shadow prices have been calculated using the abatement and damage cost methods (as explained in Section 4 of this summary). In most cases the latter method has been adopted.

**Table 1 Shadwo prices of emissions on Dutch territory in 2008 according to two calculation methods ($€_{2008}$/kg pollutant)**

| Pollutant | Abatement costs | Damage costs |
|---|---:|---:|
| $CO_2$ | 0.0250 | 0.0250* |
| CFC-11 | 149 | 159 |
| $NO_x$ | 8.72 | 10.6 |
| $SO_2$ | 5.00 | 15.4 |
| $NH_3$ | 11.7 | 27.8 |
| NMVOC | 5.00 | 2.54 |
| $PO_4$ | 11 | 1.80 |
| P to water | 10.9 | 1.78 |
| N to water | 7.00 | NA |
| $PM_{10}$ | 2.30 (50.0)** | 41.0 |
| $PM_{2.5}$ | 2.30 (50.0)** | 64.8 |
| Dioxins | 92.00E06 | 5.09E07 |
| As (arsenic) | 466 | 811 |
| Cd (cadmium) | 4700 | 127 |
| Cr (chromium) | 36,900 | 33.5 |
| Ni (nickel) | 1,800 | 5.37 |
| Pb (lead) | 225 | 408 |

Notes: These figures are averages. Future impacts of these emissions (on environmental policy or on endpoints) have been included in these values and, where relevant, discounted to the year of emission using a 2.5% discount rate with no risk premium.

* Damage costs based on abatement costs.

** For $PM_{10}$ and $PM_{2.5}$ the precise policy context is as yet unclear, implying an estimated shadow prices of either € 2.30 or € 50.

These shadow prices are average values for emissions from an average emission source at an average location in the Netherlands. They include the risk-free discounted future impacts of the emission in 2008 (using discount rates without risk adjustments). In the case of damage costs, impacts on populations outside the Netherlands have been assigned the same value as in the case of resident Dutch populations. Financial transfers such as subsidies and taxes are *not* included in the shadow prices.

On the basis of this, shadow prices have been calculated not only for these pollutants, but also for several environmental policy themes like noise, land use and final waste. Table 2 reports the values adopted for these environmental themes.

Table 2 **Shadow prices for noise, land use and final waste on Dutch territory in 2008 according to two calculation methods (€$_{2008}$/unit)**

| Theme | Unit | Abatement costs | Damage costs |
|---|---|---|---|
| Final waste* | kg | 0.18 | NB |
| Land use** | M2 | NB | 0.612 |
| Noise*** | dB | | |
| >50 dB | above threshold | 70 | 12.7 |
| >70 dB | | 70 | 82.6 |

Notes: These figures are averages.

* Although final waste is not one of the impact categories used in Life Cycle Assessment (LCA), for the Dutch government it does constitute a separate policy theme involving dedicated policy efforts and a shadow price for it was also calculated in the previous study (CE, 2002a).

** Valuation is highly dependent on the type of land used; see Section 5.5.2.

*** The figure reported for noise is based on road traffic. Shadow prices for rail traffic are lower, those for air traffic generally higher.

## User guidelines on shadow prices

### *Choice between damage costs and abatement costs*

As can be seen from Table 1 and Table 2 above, in this study we have calculated shadow prices according to two methods.

The first method proceeds from the costs that need to be incurred to secure the environmental policy targets in question. This is known as the *abatement cost method* and was used to calculate the shadow prices published by CE Delft in 2002. In this method the shadow price is calculated as the cost of the most expensive technique required to meet government targets. From an economic perspective, the abatement costs are thus equal to the Pigovian charge[1] that would have to be paid to achieve the set targets.

In the second approach, the *damage cost method*, environmental quality is valued on the basis of the estimated damage occurring as a result of emissions and other changes in natural capital. The damage cost method proceeds from people's willingness to pay not to damage the environment and is commonly used by economists for assigning a value to externalities.

The question now is which type of shadow prices are to be used in which contexts. The general rule is that if a project leads to changes in environmental quality, damage costs should be used, while if it leads to changes in the efforts required to secure environmental targets, abatement costs are preferable.

In practice, this results in damage costs being used in most cases. Only for those environmental themes for which the government has agreed absolute targets (which are also binding) will a project imply no change in environmental quality, merely a change in the efforts required to secure the environmental targets concerned. Thus, in a situation in which all power generation comes under the EU ETS, a policy measure making low-energy lamps compulsory in public buildings and areas will lead to no net additional $CO_2$ savings on a national scale. The value of these emission reductions equals their emission trading price, in turn equal to the marginal costs of the most expensive measure to meet the targets. Likewise, the avoided $NO_x$ will not in

1 A tax levied on a non-market activity that generates negative externalities.

fact lead to any real reduction in $NO_x$ emissions, but to reduced efforts to secure internationally agreed $NO_x$ emission targets.
The Dutch government has set absolute caps on emissions of $CO_2$, $SO_2$, $NO_x$, NMVOC and $NH_3$. In the case of these emissions, the environmental impacts of a project are valued using abatement costs, *unless it is a project that influences the targets themselves* (e.g. an SCBA on the need for more stringent climate policy).

### *Use in non-average situations*
The shadow prices calculated in this report are averages for the Netherlands, based on emissions in 2008. Consequently, these figures cannot be used for non-average situations and in such cases the following recommendations can be given:

*Different emission sources*
In the case of abatement costs there is no problem. With damage costs, the value of above all $PM_{2.5}$ (and to a lesser degree NMVOC) depends very much on the magnitude of the emissions. The value to be assigned to emissions from specific sources, such as traffic, may therefore deviate substantially from the figures presented here. In Annex C, Section C.3.9, a table provides values that can be used for the damage costs of transport emissions.

*Different population densities*
The main reason that emission location is important is because damage estimates depend on population density and the ecosystems present in the region. In this study the average for the Netherlands has been taken (398 people/$km^2$), with no attempt made to make damage estimates for regions with higher or lower population densities. Within the Netherlands, though, there is no problem using abatement costs for more sparsely or densely populated areas.

*Different countries and regions*
The abatement costs may not be used for other countries or regions, because policies there will differ from Dutch policies. The damage costs can be used up to a certain point. In Section 5.6.3 we provide estimates for the EU-27. Although adjustments could feasibly be made for individual countries in the EU-27, this has not been done in the present study. Adjustments for non-EU countries is possible up to a point. An option for correcting the values calculated here for the difference in income level between the EU-27 and non-EU countries is provided in Section 5.6.4. This yields a very rough estimate, because other countries also deviate from the EU-27 average in terms of air currents and population densities, which will also be of influence on damage estimates.

*The future*
Generally speaking, estimated damages increase as people become wealthier. This means that for emissions occurring in the future current damage estimates need to be corrected for an additional value that arises because people have a positive income elasticity for environmental quality. The value to be assigned to $CO_2$ emissions increases even more with time according to the damage estimates and, if increasingly stringent climate policy is assumed, for the abatement costs too. Section 5.6.1 provides information on which value should be adopted for $CO_2$ emissions.

### *Shadow price 'shelf life'*

The shadow prices are valid for the situation in 2008, but can be used without any problem for a number of years to come. In the *short term* the shadow prices adopted here can be corrected for inflation by using the consumer price indices for the Eurozone. Because the damage costs are based on the willingness to pay for environmental quality and this WTP rises with income, the damage costs should also be corrected for changes in income levels. In this study we have assumed an income elasticity of 0.85 with respect to environmental quality.

For the *longer term*, users of these shadow prices should themselves make a reasoned judgment on the 'shelf life' of the shadow prices developed here.

The presented abatement costs remain valid until:

- New policy targets are agreed for the emissions in question. This is particularly likely to occur in response to new developments in international climate policy. If the EU decides to go for a 30% reduction target in 2020, then the shadow price based on *prevention costs* rises to € 0.05/kg $CO_2$. This also holds for *damage cost* estimates up to 2020, since these are based on prevention cost figures (see Section 5.5.2 and Annex C.2.4).
- Technological breakthroughs occur.
- Drastic changes in resource prices occur.

The damage costs change if:

- Different values are elaborated for the pollutants used here or for the values assigned to the underlying factors, such as the valuation of life expectancy.
- A different methodology is developed for, say, valuation of option values and risk assessment, with consequences for the valuation of environmental goods.

In general, it can be said that the values are more durable in the case of the damage costs, because the underlying variables (such as dose-response functions, pollutant dispersion and valuation of endpoints) will change little over time. With damage costs, though, the science involved is continually advancing and new studies, or a new methodology, may therefore yield new insights.

### Weighting

In environmental impact analyses use is made of characterisation factors. Characterisation factors are numbers indicating how much a given pollutant (e.g. 1 kg of $CO_2$) contributes to a particular environmental impact (e.g. climate change). Since $CO_2$, $CH_4$ and CFCs all contribute to the reinforced greenhouse effect, the mutual relationship between them can then be established using characterisation factors.

Following characterisation, an environmental impact analysis (like a Life Cycle Assessment, LCA) yields scores for the various environmental themes of climate change, acidification, eutrophication and so on. To condense the scores on these themes into a single figure, use can then be made of weighting. Weighting indicates the relative importance of the environmental theme – the relative significance of climate change compared with acidification, for example.

Shadow prices can be used as weighting factors. They then indicate the relative magnitude of the costs of the various environmental impacts. With weighting factors based on abatement costs, these are the costs of meeting government targets; with weighting factors based on damage costs, the costs of the damages resulting from the emissions.

In the present report, two sets of weighting factors have been developed: for abatement costs and damage costs. Table 3 reports the most important of these.

**Table 3 Two sets of weighting factors for emissions in the Netherlands in 2008 (€$_{2008}$/kg-equivalent)**

| Impact category | Abatement costs | Damage costs |
|---|---|---|
| | (set 1b)* | (set 2)* |
| Climate change ($CO_2$-eq.) | 0.0250 | 0.0250 |
| Ozone depletion (CFC-11-eq.) | 30.0 | 39.1 |
| Acidification ($SO_2$-eq.) | 0.594 | 0.638 |
| Photo-oxidant formation (NMVOC-eq.) | 5.00 | 0.585 |
| PM formation ($PM_{10}$-eq.) | 50.0 | 51.5 |
| Eutrophication, fresh water (P from STP-eq.) | 10.9 | 1.78[1] |
| Eutrophication, marine water/land (N-eq.)[2] | 7.00 | 12.5 |
| Human toxicity (1,4-DB-eq.)[3] | NA | 0.0206 |
| Ionising radiation (U235-eq.) | NA | 0.0425 |
| Land use ($m^2$ per year) | NA | 0.612 |
| Abiotic resource depletion | NA | 0 |

Notes:

* For more information on the types of weighting sets, see Chapter 6.

1 Based on value estimates based on ReCiPe endpoints.

2 For weighting set 1b: marine eutrophication; for weighting set 2: terrestrial eutrophication.

3 For weighting set 1b: human toxicity would be specified in terms of kg $PM_{10}$-eq. To avoid double counting, this impact is taken into account via the weighting factor of PM formation.

These weighting factors can be used to weight the results of an LCA, say, and aggregate them to a single, uniform figure.

Because the set of weighting factors based on damage costs is more extensive than that based on abatement costs, the obvious course is to use the set based on damage costs for weighting purposes. The greatest differences are in the weighting factors for human toxicity: based on abatement costs, these are a factor 100 higher.

To the theme of abiotic resource depletion we have assigned a shadow price of zero. In properly functioning markets, future scarcity will be reflected in prices and there will be no externalities. The issue of whether abiotic resource depletion is unfair to future generations is entirely contingent on the question of what happens with the profits generated by resource extraction.

Given the weighting factors for the various environmental themes, abatement and damage costs can be derived for other individual environmental pollutants falling under those themes as long as characterisation factors are available. In Annex J of this report we provide a list of values for over 400 pollutants.

## Methodological underpinning

This report details the methodological underpinning for the shadow prices calculated. In the context of the present study we have not sent out questionnaires, computed the costs of technologies or developed any other quantitative research methods, but have based ourselves on existing sources. In this summary we provide a brief synopsis of the main premises underlying the abatement cost method, the damage cost method and the characterisation of environmental impacts (required to arrive at weighting factors).

> The terminology employes below is technical and intended for analysts experienced in the valuation and weighting of environmental impacts. In this summary we make no attempt to explain this terminology (this is done in the main report and the various annexes), but are concerned solely to provide basic justification for the methodology adopted.

### *Characterisation factors*

In this study we have made use of the characterisation factors calculated in the ReCiPe project. Over the past five years this project has endeavoured to achieve consistency in the characterisation factors at the endpoint and midpoint levels. This has resulted in a consistent set of characterisation factors in which the relationship between the midpoint and endpoint level has been delineated using a consistent methodology.

Although the ReCiPe project was completed in mid-2009, there may still be a few minor adjustments (errata) to the characterisation factors calculated. For our calculations of shadow prices we have taken the characterisation factors as they stood on 1-11-2009. Any revisions subsequent to this date have *NOT* been included in calculating the weighting factors or damage costs presented in the current report.

Characterisation factors are associated with different perspectives, among them the time scale over which impacts are included. For the set of characterisation factors we have used the hierarchist perspective, which can be regarded as the average of the time scales distinguished in the ReCiPe project.

### *Abatement costs*

Abatement costs are given by the intersection of the marginal cost function and the policy target in the year in which that target is to be secured. If the policy in question makes use of charges, though, the abatement costs are equal to the highest charge.

Table 4 reports the policy targets adopted in the present study.

Table 4 **Background scenarios and policy targets adopted for the various pollutants**

| Pollutant/impact | Scenario, autonomous trends | Policy target |
|---|---|---|
| $CO_2$ | GE scenario | 20% emission reduction in 2020 |
| CFC-11 | - | Waste disposal fee as per Decree on Disposal of White and Brown Goods |
| $NO_x$ | GE scenario | Anticipated National Emissions Ceiling (NEC) target for 2020 (186 kt) |
| $SO_2$ | GE scenario | Anticipated NEC target for 2020 (35 kt) |
| $NH_3$ | GE scenario | Anticipated NEC target for 2020 (119 kt) |
| NMVOC | GE scenario | Anticipated NEC target for 2020 (143 kt) |
| $PO_4$ | - | Administrative fine under Fertiliser Act |
| N | - | Administrative fine under Fertiliser Act |
| $PM_{10}$ and $PM_{2,5}$ | - | EU Directives regarding concentrations<br>Cost effectiveness criterion in National Emission Guidelines for Air |
| Final waste | - | Cost effectiveness criterion in draft National Waste Management Plan |
| dB rail >55 | - | Noise control policy (Euro per dB-dwelling) |
| dB road >50 | | |
| dB aircr. >45 | | |

For the costs of the abatement techniques and technologies for pollutants with fixed policy targets ($CO_2$, $NO_x$, $SO_2$, $NH_3$, NMVOC) we have based ourselves on relevant studies and the ECN/MNP databases.

One of the problems in the case of abatement costs is how to allocate the 'joint costs' of techniques and technologies that reduce emissions of more than just one pollutant. This allocation has been effectuated using an iterative procedure. Joint cost allocation introduces an element of arbitrariness into cost estimates.

The abatement costs have been converted to weighting factors using the ReCiPe mid-point characterisation factors. Using these weighting factors, cost estimates were then made for pollutants for which no immediate policy targets have been set (but which fall under the same environmental theme). The fact that certain pollutants (like $NO_x$) fall under several environmental themes introduces another arbitrary element into the weighting factors and cost estimates of those pollutants for which we have not used marginal costs for calculating abatement costs.

### *Damage costs*
In this study the damage costs have been calculating using the Impact Pathway Approach, which traces the release of emissions via a causal chain from dispersion-dose-response through to physical impacts. These physical impacts were then monetised. For most pollutants and themes we have used the Impact Pathway Approaches established in the NEEDS project (an ExternE-related European study on the external costs of energy use, completed in 2008). Via NEEDS we were able to establish physical impacts on the following endpoints:
- Human health (morbidity and mortality).
- Ecosystems (biodiversity).
- Agricultural crops.
- Materials and buildings.

For environmental impacts not covered by NEEDS, such as ozone depletion and freshwater eutrophication, we have used the Impact Pathway Approach developed in ReCiPe for endpoint characterisation factors. In this context we could only establish physical impacts for human health and ecosystems, which could, in some cases, be extended by means of supplementary studies.

All physical impacts have been assigned monetary values in accordance with the values provided in NEEDS:
- Premature death (chronic and acute mortality) is valued in accordance with the NEEDS 2008 recommendation using a VOLY of € 40,000 for chronic mortality and € 60,000 for acute mortality (both in prices of 2000).
- Illness (morbidity) is valued using the values provided by the NEEDS (2008) project.
- Changes in biodiversity are valued via Kuik et al., 2008.
- Crop impacts are valued at market prices.
- Impacts on buildings and building materials are valued at repair costs.

Table 5 provides a synopsis of the approaches adopted for the individual themes and pollutants.

**Table 5 Approaches adopted for the various pollutants classified by environmental theme**

| Environmental theme | Pollutants estimated directly | Approach | Estimated endpoints | Missing endpoints |
|---|---|---|---|---|
| Climate change | $CO_2$ | Literature analysis | HH, ES, CR, BLD | |
| Ozone depletion | - | ReCiPe + literature | HH, ES, CR | |
| PM formation | $PM_{10}$, $PM_{2.5}$, $NO_x$, $SO_2$, $NH_3$ | NEEDS | HH | |
| Photo-oxidant formation | NMVOC, $NO_x$, $SO_2$ | NEEDS | HH, ES, CR | |
| Acidification | $NO_x$, $NH_3$, $SO_2$ | NEEDS | ES, CR, BLD | |
| Eutrophication, fresh water | - | ReCiPe | ES | |
| Eutrophication, soil | $NO_x$, $NH_3$, $SO_2$ | NEEDS | ES, CR | |
| Human toxicity | Cd, As, Ni, Pb, Hg, Cr, formaldehyde, dioxins | NEEDS | HH | |
| Ionising radiation | Cesium, iodine, hydrogen (tritium), carbon, krypton, radon, thorium, uranium | NEEDS | HH, ES | CR |
| Noise | dB | Literature | HH | |
| Land use | - | ReCiPe | ES | CR** |

Notes:

* HH = human health; ES = Ecosystems; BLD = buildings; CR = crops.

** Land use also has an impact on crops, as prices of land will rise. This effect is probably a pecuniary externality, which has therefore not been included in the present study.

In a number of cases, our estimates proceed from different premises from those adopted in NEEDS or ReCiPe and are consequently not *directly* comparable.

Damage costs have been converted to weighting factors using the ReCiPe midpoint characterisation factors. Using these weighting factors, cost estimates were then made for pollutants for which no damage estimates were available. In doing so, we established the relationship between these pollutants at the endpoint level.

# List of abbreviation

All the abbreviations and acronyms used in this report are explained the first time they are used. For ease of reference they are here listed alphabetically.

| | Meaning |
|---|---|
| AGF | Age Group Functions |
| AOT 40 value | Accumulated Ozone Concentration above a Threshold of 40 ppbV |
| BAT | Best Available Technique(s) |
| Bq | Becquerel |
| BREF | Bat Reference Document |
| CAFÉ-CBA | Cost-benefit Analyses for Clean Air for Europe (EU research programme) |
| CASES | Cost Assessment for Sustainable Energy Systems |
| CBA | Cost-benefit analyses |
| CCS | Carbon capture and storage |
| CDM | Clean Development Mechanism |
| CE method | Choice Experiment, a type of stated preferences research |
| CEA | Cost-Effectiveness Analysis |
| CFC | Chlorofluorocarbon |
| COI | Cost of Illness |
| CPI | Consumer Price Index |
| CRF | Concentration-Response Function |
| CV | Compensation Variation |
| CVM | Contingent Valuation Method |
| DALY | Disability Adjusted Life Year |
| dB | Decibel |
| DPSIR | Driving Forces-Pressures-States-Impacts-Responses |
| DW | Disability Weight |
| ECN | (the Netherlands) Energy Research Centre |
| EDP | Ecosystem Damage Potential |
| EESC | Effective Equivalent of Stratospheric Chlorine |
| EIA | Environmental Impact Assessment |
| EMEP | European Monitoring and Evaluation Programme |
| FDA | American Food and Drug Administration |
| FES | Netherlands Economic Structure Enhancing Fund |
| GDP | Gross Domestic Product |
| GE scenario | Global Economy scenario |
| GHG | Greenhouse gases |
| GWP | Global Warming Potential |
| HEATCO | Developing Harmonised European Approaches for Transport Costing and Project Assessment (EU research programme) |
| HICP | Harmonised Indes of Consumer Prices |
| HP | Hedonic pricing |
| HTP | Human Toxicity Potential |
| HPM | Hedonic Pricing Method |
| HUI | Health Utility Index |
| HWM | Hedonic Wages Method |
| IAM | Integrated Assessment Model |
| IIASA | Institute for Applied Systems Analysis |
| IMPACT | Internationalisation Measures and Policies for All external Costs of Transport (EU research programme) |
| IPA | Impact Pathway Approach |

| | Meaning |
|---|---|
| IPPC directive | (the EU's) Integrated Pollution and Prevention Control Directive |
| LAP | (the Netherlands') National Waste Management Plan |
| LCA | Life Cycle Assessment |
| LCIA | Life Cycle Impact Assessment |
| LIME model | Life Cycle Impact Assessment method based on Endpoint modelling |
| LRS | Lower respiratory symptoms |
| MACC | Marginal Abatement Cost Curve |
| MNP | (the Netherlands') Environmental Assessment Agency (prior to April 2008) |
| MPC | Maximum Permissible Concentration |
| MPR | Maximum Permissible Risk |
| MRAD | Minor Restricted Activity Days |
| MTR level | Maximum Tolerated Risk level |
| NEC | National Emission Ceilings |
| NEEDS | New Energy Externalities Developments for Sustainability (EU research programme) |
| NeR | (the Netherlands') National Emission Guidelines for Air |
| NHM | Northern Hemispheric Modeling |
| NMVOC | Non-Methane Volatile Organic Compunds |
| NOGEPA | Netherlands Oil and Gas Exploration and Production Association |
| NPV | Net Present Value |
| NVKL | Dutch Association of Refrigeration and Air Conditioning Companies |
| ODS | Ozone-depleting substance(s) |
| OEI Guidelines | Dutch guidelines for calculating the economic impacts of infrastructure projects |
| PBL | (the Netherlands') Environmental Assessment Agency (after April 2008) |
| PBq | PetaBequerel |
| PDF | Potentially Disappeared Fraction of species |
| PPP | Purchasing power parity |
| PRTP | Pure rate of time preference |
| QALY | Quality Adjusted Life Year |
| RAD | Restricted Activity Days |
| REACH | (the EU's) Regulation, Evaluation and Authorisation of CHemicals directive |
| RGF | Risk Group Functions |
| RP | Revealed preference |
| SCBA | Social cost-benefit analysis |
| SCC | Social Cost of Carbon |
| SIA | Secondary Inorganic aerosols |
| SNAP | Sectoral classification based on emission inventories |
| SP | Stated preference |
| SRM | Source-receptor matrices |
| TC | Travel cost |
| TCM | Travel Cost Method |
| TSP | Total suspended matter |
| VEDP | Value of Ecological Damage Potential |
| VOC | Volatile Organic Carbon |
| VOLY | Value of Life Year |
| VPF | Value of Prevented Fatality |
| VROM | (the Netherlands') Ministry of Housing, Spatial Planning and the Environment |
| VSL | Value of Statistical Life |
| WHO | Word Health Organisation |
| WLD | Work loss days |
| WTA | Willingness to Accept |
| WTP | Willingness to Pay |

| | Meaning |
|---|---|
| YLD | Years Lived with Disability |
| YOLL | Years of Life Lost |

# Glossary of terms

This study draws on knowledge from both the environmental and economic sciences. As certain terms from this dual sphere will not be familiar to all the users of this handbook, there follows a glossary, with abbreviations as appropriate. To keep explanations short, we have sometimes written '(etc.)' to indicate that a term like 'pollutants' or 'emissions' should also be understood as including a broader range of environmental impacts such as land use.

| | |
|---|---|
| Abatement costs | All the costs that society must incur to secure the government's environmental targets; these generally take the form of investment costs. |
| Annuity method | Depreciation method in which the annual burden of interest and repayment is constant throughout the entire depreciation period. |
| Characterisation factor | Number indicating the contribution of a standard quantity of a pollutant (etc.) to a given environmental impact. The bigger the characterisation factor, the greater the contribution. |
| Compensating variation (CV) | A measure of the change in a person's individual welfare relative to an original (or reference) situation due to implementation of a given project. The CV is the maximum sum that a person benefiting from the change can forfeit without becoming worse off than before the project (their willingness to pay); it is also the minimum sum the losing party needs in order not to suffer a loss of welfare if the project does go ahead (their willingness to accept). The CV is a measure of the consumer surplus. (See also *Equivalent variation*.) |
| Consumer surplus (CS) | (An approximation of) the difference between the maximum sum that someone (the consumer) is willing to pay for a good or service and the sum that is actually paid. |
| Cost-benefit analysis (CBA) (also: social CBA, SCBA) | An analysis method for establishing the monetary value of all the benefits and disbenefits experienced by all parties in a (national) society as a result of a given project being implemented, supplemented by (preferably quantitative) information on impacts that cannot be satisfactorily expressed in monetary terms. |
| Cost effectiveness | The ratio between the costs and actual impacts of a given government policy. Cost effectiveness can be defined from the perspective of government, end users or society as a whole. |
| Cost effectiveness analysis (CEA) | An analysis method for assessing which of a number of project alternatives or variants can secure the ('unidimensional') project objective at lowest cost or, in other words, with which alternative or variant the best result can be achieved in terms of the project objective within a given financial budget. |
| Damage costs | All the damage experienced by individuals as a result of environmental pollution (etc.). |
| Damage cost indicators | Indicators constructed on the basis of direct linkage between emissions (etc.) and associated damage costs. Damage cost indicators provide general information on dispersal and dose-response relationships. |

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| Default values | In the context of this study, central estimates of monetary values that can be applied by users in situations where is no full knowledge of all the relevant conditions. |
| Direct impact | The welfare impact of an environmental policy on the target groups obliged to implement it (see also *Indirect effect*). |
| Disability Adjusted Life Year (DALY) | The number of years of healthy life lost by a population as a result of illness. |
| Discount rate | Interest rate used for calculating the present value of a sum of money that will be paid or received some time in the future. (See also *Social discount rate*.) |
| Dispersion model | A model that uses climatological and other data to establish a relationship betweeen emissions and immissions (i.e. pollutant concentrations, also referred to as *dose*). It allows immissions to be calculated from emission factors (or indicators) determined from emission measurements. |
| Dose-effect relationship | In toxicology, the principle that states that the likelihood of a deleterious effect occurring rises as more of the toxic substance is added. |
| Dose-response relationship | See *Dose-effect relationship*. |
| Economies of scale | The reduction in average production costs occurring in the longer term when the scale of production is increased (as a result of spillover effects, for example, or more efficient use of production facilities). Economies of scale are one of the main reasons that 'natural monopolies' exist, i.e. economic activities that can be most efficiently implemented by just one or several producers. |
| EcoSense | A model developed at the University of Stuttgart for pan-European modelling of the relationship between pollution (etc.) and damages. The model comes in two variants: normal and light. A simplified version of the model is available on Internet[2]. |
| Endpoint | The level where environmental impacts ultimately occur, with a distinction generally made between impacts on health, biodiversity and ecosystems, buildings and production. |
| Environmental theme | Themes identified in Dutch environmental policy, for example, climate change, acidification, eutrophication, desiccation. |
| Equivalence factor | see C*haracterisation factor*. |
| Ex ante | Beforehand. |
| Ex post | After completion. |
| Externality | An unintended change in the welfare of third parties for which no compensation is received. In more formal terms: An externality arises if the utility or production functions of an economic agent (the one 'subjected to' the externality) include a real variable, the value of which is affected by the behaviour of a different economic agent (the one 'imposing' the externality) who does not include it in their decision-making. |
| ExternE | An acronym for Externalities of Energy, a European research programme that quantifies the impact of pollutants (etc.) on human health and the environment by tracking them from source through to impact. |

[2] See http://ecosenseweb.ier.uni-stuttgart.de/index.html.

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| Impact-Pathway Approach (IPA) | A method developed within ExternE to assess the impact of specific pollutants on various endpoints in monetary terms. The main steps include assessment of emissions, dispersion, establishing dose-response functions and monetisation of impacts. |
| Income elasticity | The income elasticity of demand is a measure of the change in demand for a particular good resulting from a change in income. |
| Indirect impact (also: Derived impact) | Any impact of an environmental policy not classed as a direct impact (see *Direct impact*). |
| Internal impact | Any impact of an environmental policy for which, via transactions and market trading, a market price arises reflecting the value assigned to it by individual actors. |
| Life Cycle Assessment (LCA) | An analysis method for quantifying the sum total of environmental impacts of a given product or service over its entire lifetime, from raw materials extraction via the usage phase to final waste disposal. It is sometimes referred to as a 'cradle to grave' environmental analysis. |
| Midpoint | Impact categories whereby the analysis takes place at the level of environmental themes. |
| NEEDS | The last phase of the ExternE series of projects, focusing on external costs of energy production. The values of damage costs developed within the NEEDS project using IPA and EcoSense modelling have been used extensively throughout this report. |
| Net Present Value (NPV) | A profitability or decision criterion used in cost-benefit analysis. The sum obtained by deducting the present value of the projected costs of a given investment from the present value of the projected returns. In a CBA the NPV is calculated using the social discount rate; if the NPV is positive, project implementation is deemed economically viable. |
| Normalisation | Method in which each impact score is expressed as a relative share in the aggregate impact score of all the interventions associated with the entire economic system of a particular region. |
| OEEI | Acronym for a Dutch research programme on the economic impacts of infrastructure (initiated by the Ministries of Transport, Public Works & Water Management and Economic Affairs). It resulted in the OEI guidelines. |
| Pareto-optimality | An economic situation is said to be Pareto-optimal if resource use and economic production are allocated in such a way that any other allocation that leads to additional benefits to any other member of society is at the expense of somebody else's welfare. |
| Project | An investment or series of mutually related investments equivalent to (or accompanied by) a government intervention in the market. To avoid a situation in which, with a series of investments, profits from one element mask losses from another, this definition needs tightening. A project can then be defined as the minimum set of mutually related investments that is anticipated to be technically feasible and economically viable. |

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| ReCiPe | A Dutch research programme developing an integrated method of life cycle impact assessment that combines midpoint and endpoint approaches by devising a uniform set of characterisation factors. |
| Revealed preference (also: Revealed behaviour) | An estimate of demand based exclusively on actual observation of how consumers respond to changes in prices and/or income. (See also *Stated preference*.) |
| Shadow price | A value given to a good or service for which there is no price formation in a market where supply meets demand. In principle, the shadow price equals the increase in welfare resulting from one additional unit of the good or service in question. The shadow price is often used in situations in which supply and demand for the unpriced good would be in reasonable equilibrium. |
| Social cost-benefit analysis (SCBA) | See *Cost-benefit analysis*. |
| Social discount rate | The discount rate used in CBA to calculate the present value of a project's social costs and benefits. It differs from the interest rate used for discounting private investments. The Dutch government currently prescribes a social discount rate of 2.5% (in real terms) in a risk-free environment. In practice a risk premium can be added to account for non diversified risks. |
| Stated preference (also: stated behaviour) | A method used to estimate demand for a good or service based on consumers' responses when asked how they would respond in a hypothetical situation to a change in prices and/or income. (See also *Revealed preference*.) |
| Weighting | A method in which the impact scores for the various impact categories are each assigned a weight before being summed to yield a single final score. |
| Willingness to Accept (WTA) | The minimum sum that a person is willing to accept in exchange for forfeiting a good or service or for accepting a disbenefit (in the form of damage or nuisance, for example). |
| Willingness to Pay (WTP) | The maximum sum that a person is willing to pay to have access to a good or service or avoid a disbenefit (in the form of damage or nuisance, for example). |
| Years of life lost (YOLL) | A measure of premature death in a population. |
| Value of Life Year (VOLY) | A value assigned to one year of life in full health, which can be revealed directly using SP methods. VOLY can be seen as valuation of YOLLs or DALYs. |

# 1 Introduction

## 1.1 Introduction

Shadow prices are constructed prices for goods or production factors that are not traded in actual markets. In economic analysis use is generally made of market prices. For some goods, among which environmental ones, no such prices are available, however, because no markets exist for them. To enable the inclusion of environmental goods in economic analyses, use is made of so-called shadow prices. These prices then provide an indication of the value of a particular good – in this case the environment – to society.

Shadow prices are implicit prices: the price of environmental quality cannot be determined directly in the marketplace and so needs to be calculated. In general terms, there are two ways to assign shadow prices to environmental quality. The first proceeds from the costs that need to be incurred to secure environmental policy targets and is known as the abatement cost method. The second assigns value to environmental quality based on the estimated damage occurring as a result of emissions and other changes in natural capital and is known as the damage cost method. Both methods have their own specific areas of application and should not be seen as necessarily competing with one another. Among users of shadow prices, however, there is considerable confusion as to which set of prices should be used under what circumstances (CE, 2007b).

Shadow prices are frequently used (by both government and industry) in studies as well as in practical applications. Roughly speaking, three types of application can be distinguished:
1. **Cost-benefit analyses and investment decisions**: environmental impacts play a major role in many economic decisions. In the case of a new road, for example, it is not only the road's cost-effectiveness that needs considering but also the unintended side-effects, including those relating to the environment. Assigning shadow prices to these environmental impacts means they are expressed in similar units to the financial and economic data, permitting a better underpinned decision on the desirability and practicalities of the investment.
2. **Weighting**: in environmental analyses like Life Cycle Assessment or Environmental Impact Assessment, where the various environmental impacts need to be weighted in some way, shadow prices can be used to this end. For example, a company can use shadow prices to assess whether coffee packaging made of an aluminium-based laminate performs better environmentally than plastic packaging. Government, for its part, can use shadow prices to determine whether it is environmentally sounder to recycle paper or incinerate it and recover the energy content.
3. **Benchmarking and indicators**: a company, organisation or country can use shadow prices to compare its environmental performance with that of other companies, organisations or countries. To do this, all relevant environmental impacts are weighted using shadow prices, possibly going on to compare these with financial items, as with the 'Envirometer', for example (CE, 1998).

## 1.2 Project background

In the 1990s CE Delft was commissioned by Thermphos, Stimular and the Dutch Ministry of Housing, Spatial Planning and the Environment (VROM) to develop a set of shadow prices. These prices, based on abatement costs, were employed as a key element in the 'Envirometer' developed by Stimular, as a management tool at Thermphos and in a range of social cost-benefit analyses (SCBA) for government (see, for example, CPB, 2005).

These shadow prices are now outdated, however, for two reasons. In the first place, the prices drawn up in 2002 are only valid for emissions occurring up to the year 2010. New policy targets (on acidification and $CO_2$, for example) mean the 2002 figures need to be reviewed. Secondly, the 'old' shadow prices are based on abatement costs, while financial valuation of environmental goods is being based increasingly on damage costs.

Against this background, over the past year Thermphos, Stimular and VROM have each individually asked CE Delft whether an update of the old set of shadow prices is feasible. Because of the considerable overlap between the different parties' inquiries it was decided, as in 2002, to instigate a joint project, to be jointly funded by the three parties. In the current project two new sets of shadow prices have been developed: one based on abatement costs, the other on damage costs.

In addition, all three parties expressed a need for a handbook to be written in which the potential use of shadow prices in policy-making and commercial activities is explained and a clear exposition given of when shadow prices according to abatement costs should be used, and when these should be based on damage costs.

## 1.3 Project objective

The aim of this project is threefold:
1. To develop two sets of shadow prices, one based on damage costs, the other on abatement costs, valid for the situation in 2008. These shadow prices can be used as default values[3] in cost-benefit analyses and other types of economic analysis.
2. To elaborate these sets of shadow prices into weighting factors that can be used in Life Cycle Assessment and other types of environmental analysis.
3. To draw up a report discussing the use of shadow prices, presenting recommendations on the choice between abatement costs and damage costs, and providing technical background on how we arrived at our particular sets of shadow prices.

To achieve these aims meant combining environmental and economic knowledge on weighting and valuation of emissions and environmental goods. Environmental weighting, which is common practice in Life Cycle Assessment, and monetary valuation of environmental quality have until now been two largely distinct areas of research. Combining them has several advantages. On the one hand, this allows that a broader range of emissions can be valued than

[3] By 'default values' we mean that from the perspective of the user of this Handbook, the values reported here can be viewed as a central estimate of the value to be applied in situations where there is no full knowledge of relevant conditions. However, it should be noted that from the point of view of the process of generating them, these shadow prices do not take any 'default values', as they are created on the basis of modelling and depend on a set of assumptions.

is standard practice today. By utilising the results of economic valuation studies, on the other hand, environmental scientists can be provided with a more coherent framework for weighting the outcomes of environmental impact analyses.

The present, extensive document reports on our methods and seeks to provide maximum transparency as regards the procedures we have adopted. In this way we hope to provide due insight into the chosen methodology as well as help those using shadow prices decide whether or not they can be appropriately used for the particular purpose they have in mind.

## 1.4 Relationship with other studies

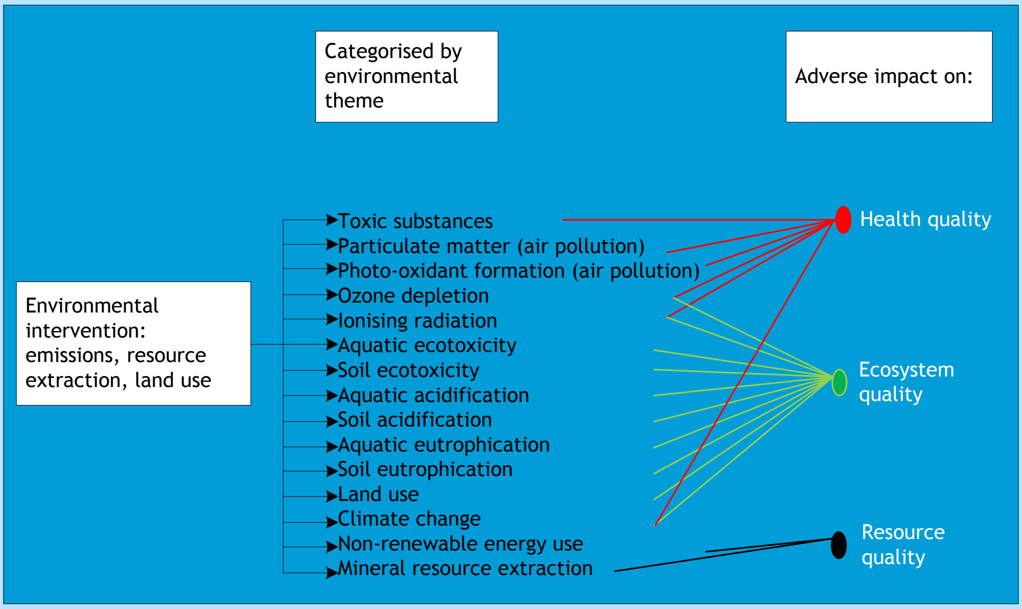
This study combines recent research findings from three 'tendencies', with the aim of arriving at as consistent as possible estimates of monetary values of environmental quality which can be used in both financial/economic analyses (such as SCBA) and as a weighting method in environmental analyses (such as LCA or EIA). The three tendencies can be characterised as follows:
a Development of characterisation factors and weighting sets for use in Life Cycle Assessment (LCA), as in the ReCiPe project.
b Financial valuation of environmental quality, as developed in Europe within the ExternE series of studies (NEEDS/CASES/MethodEx), the IMPACT transport manual and the (ongoing) EXIOPOL project.
c Manuals concerned with cost-benefit analysis, as in the Netherlands with the 'OEI Guidelines', the later supplement on 'valuation of nature', and the 'SCBA Guidelines for Environmental Policy'.

Further to a), since the 1990s major environmental research efforts have been expended on rendering a range of environmental pollutants amenable to comparison. This is achieved by using characterisation factors, normalisation and weighting. Characterisation factors are numbers that indicate how much of a standard quantity of a given pollutant contributes to a particular environmental impact. The bigger the characterisation factor, the greater the contribution. With respect to the environmental impact 'climate change', for example, methane has a higher characterisation factor than carbon dioxide. This means a kilogram of methane causes more global warming than a kilo of carbon dioxide (see also Box 1). Characterisation is sometimes supplemented by normalisation and/or weighting. In the case of normalisation, each impact is expressed in terms of its relative share in the aggregate impact score of all the interventions associated with the entire economic system in a particular region. This makes it easier to grasp the relative importance of each of the impacts. In the case of weighting, the scores for each of the various impact categories are weighted and then summed to yield a single, final score. This makes it easier to compare two product alternatives, for example.

In the Netherlands, characterisation factors have been developed at the so-called midpoint level by CML, Leiden University's Institute of Environmental Sciences (Guinée et al., 2002), and for the endpoint level by PRé Consultants (PRé, 2000). In the recently completed ReCiPe project (Goedkoop et al., 2009) the premises of the two methods were brought into line with one another and a uniform set of characterisation factors was developed for both the midpoint and endpoint level (cf. Section 2.4). This set of factors has been used in the present project. Shadow prices can then be seen as one of the weighting methods for rendering results at midpoint and endpoint level amenable to comparison and expressed in one and the same (monetary) unit. In addition, the endpoint valuations elaborated in the ReCiPe project have been used to determine the damage costs associated with some environmental themes.

**Box 1: Impact assessment and characterisation in LCA**

In the context of LCAs, assessing the environmental impacts of a given product is referred to as LCIA (Life Cycle Impact Assessment). In the central element of impact assessment, characterisation, all environmental interventions are multiplied by their associated characterisation factors. For each intervention, this provides a picture of its contribution to one or more impact categories. For each impact category, these contributions can then be summed, yielding an aggregate impact score for each. The list of impact scores gives a picture of the product's environmental impacts and is often referred to as its 'environmental profile'.




Source: Adopted from RIVM (2009a).

Further to b), since 1991 the ExternE series of projects has been engaged in estimating the external costs of energy production. To date, this network has involved over 50 research teams from more than 20 countries. The NEEDS (New Energy Externalities Developments for Sustainability) project, a research programme for the European Commission lasting from 2004 to 2008, is the most recent research undertaken by this network. Despite difficulties and uncertainties, ExternE has become a well-recognised source on externalities estimation, for both methodology and results.[4]

A related European-funded research project was CASES (Costs Assessment for Sustainable Energy Systems[5]), which was concluded in 2008. The main goal of CASES was to compile coherent and detailed estimates of both the external and internal costs of energy production for the EU and several non-EU countries under different energy scenarios until 2030.

4 For more information see http://www.externe.info.

5 For more information, see: http://www.feem-project.net/cases/.

For determining shadow prices in the present project we have used an Excel spreadsheet tool developed in the NEEDS/CASES framework by the University of Stuttgart. This tool uses input from the ecological-economic model EcoSense.

A third major project, implemented in part by the same institutes involved in the NEEDS and CASES projects, was MethodEx.[6] This project (2004-2006) focused on the external costs of pollution from sectors other than power generation, such as industry, agriculture and waste disposal. In the present report we make no direct reference to the MethodEx results because the work performed within that project relates largely to the same sources investigated in NEEDS and the Excel tool developed in the MethodEx project is less comprehensive and up-to-date than that developed in NEEDS.

For the transport sector, a specific estimate of external costs has been carried out under a project known as IMPACT (Internalisation Measures and Policies for All external Costs of Transport; CE, 2008b). IMPACT shows how to estimate the external costs of transport, how these can be used for pricing policy in EU member states and what the impact of such policy is likely to be. In the present study we have used results from the IMPACT study on various occasions, mainly for differentiating damage costs according to various locational circumstances.

A major 6$^{th}$ framework programme called EXIOPOL is presently investigating external cost estimates that can be used to supplement the NEEDS/CASES framework.[7] As this programme has not yet been concluded, the preliminary research results should be handled with some caution. We have used information from this project mainly to refine our own findings and estimates.[8]

Further to c), in the Netherlands there has been considerable focus in recent years on harmonising the premises and procedures used in conducting social cost-benefit analyses. This followed the completion of the Betuwe rail route, a project for which various SCBAs yielded conflicting results. The harmonisation efforts led to the adoption, in 2000, of the so-called 'OEI Guidelines' for calculating the economic impacts of infrastructure projects (CPB, 2000). In 2004 a separate supplement was issued for impacts on nature ('nature-inclusive SCBAs'; Ruijgrok et al., 2004). Because these impacts are often hard to estimate individually, in 2006 a separate handbook was issued with specific indicators for valuing nature (Ruijgrok et al., 2006). In 2007 the 'SCBA Guidelines for Environmental Policy' were published (CE, 2007b). The present report, which can in some ways be regarded as a handbook of standard estimates for environmental valuation, thus constitutes a practical extension of the cited SCBA Guidelines.

6 For more information, see: http://www.methodex.org/.

7 For more information, see: http://www.feem-project.net/exiopol/index.php.

8 In the future, information from EXIOPOL might be used to estimate damage costs for eutrophication, for example.

## 1.5 Scope

In this study we present sets of shadow prices and weighting factors that can be used in a range of economic and environmental analyses. These shadow prices are average values for emissions originating from unknown sources in the Netherlands in 2008. Within this project, guidelines have been developed on which of the two sets of shadow prices or weighting factors should be employed in such analyses, depending on the issue addressed by the practitioner. We thereby distinguish the following types of analysis tool: external cost estimates and cost-benefit analyses, life cycle assessments and benchmarking exercises. What this document does not purport to be is a user manual on how these tools are themselves to be used. There is consequently no coverage of the issues typically encountered in these analyses (such as 'system definition', sensitivity analysis, distributional effects, allocation issues and so on). For cost-benefit analyses the (Dutch) reader is referred to the 'OEI Guidelines' (CPB, 2000) and the 'SCBA Guidelines for Environmental Policy' (CE, 2007b). For Life Cycle Assessment we refer the reader to Guinée et al. (2002).

Neither is this study a textbook on the valuation of environmental goods or weighting of environmental impacts. The objective of this study was to arrive at two sets of shadow prices and weighting factors suitable for practical use. The underlying estimates have been made by CE Delft on the basis of the best available scientific understanding. In doing so, we have oriented ourselves around what can be currently seen as the *mainstream* in the sciences of valuation, characterisation and weighting – with a slight preference for that which is most *recent*. This means that while *other* valuation and weighting methods will be mentioned here (and due references provided), we shall only be entering into very limited discussion as to whether the method adopted by ourselves is superior. Given the vast literature on valuation and weighting, it would indeed be unfeasible to provide a synopsis of all the methods currently in use. It is therefore up to those using the shadow prices or weighting factors elaborated in this handbook to decide whether the figures presented here are preferable to those cited in other publications.

Unless otherwise specified, all the shadow prices in this document are expressed in terms of €/kg emission. These prices have been calculated as averages for the Netherlands. In all cases, decisions on whether these averages can be used in a tool like CBA or LCA must be made by practitioners. As the justification for such a decision will always be governed by the specific issue for which the shadow prices are being employed, the question of whether or not use of national data is justified cannot be answered by us within the present study. Local circumstances such as population density, existing pollution levels and locally implemented standards may mean the data presented here are not always appropriate for use at the local level (municipality or province, for example). Neither can additional impacts in other countries, including developing nations, be determined using these shadow prices. Finally, the correct use of shadow prices also depends very much on the source of the pollution involved: transport emissions, for instance, cause far more damage to human health than the average emission because they occur at breathing level. Using these average values to calculate the damage associated with transport emissions will therefore certainly yield an underestimate. Because we consider these issues important in the context of shadow prices, in this handbook we indicate how the figures presented here can be converted if national averages are deemed unsatisfactory. The actual conversion step is beyond the scope of the present study, however.

In all cases the shadow prices and weighting factors presented here are expressed (ultimately) as ‘central values’. We are fully aware that this may imply an accuracy that is unjustified. The shadow prices have themselves been calculated on the basis of a multitude of uncertain factors. The formal treatment of uncertainty in this study (cf. Section 5.7) shows that the variance is extremely great – so great, in fact, that use of shadow prices is not to be recommended at first sight. This holds not only for the prices developed in the present study, but also for other valuation or weighting methods for environmental goods (although these do not generally encompass any formal treatment of uncertainty). Yet it is a choice between the devil and the deep blue sea, as the saying goes: either one opts not to use shadow prices, with the upshot that financial data cannot be compared with environmental impacts and the latter cannot be compared and weighted with respect to one another, or one does use them, in the full recognition that the results are anything but certain. This choice will depend partly on what purpose the shadow prices are being used for and what degree of certainty one requires of the results. Sensitivity analyses may sometimes be a useful way of getting to grips with uncertainties.

## 1.6 Report structure

This report sets out the scientific underpinning for the calculation and use of shadow prices in applied economic and environmental studies. This means the main intended readership of this handbook are analysts and researchers who use shadow prices in cost-benefit and allied analyses or for weighting environmental impacts.

Besides the scientific justification provided here, other practical manuals on application of shadow prices have also been written within the organisations that have funded the present study.

This scientific background report is structured as follows. Chapter 2 outlines the conceptual framework of this study and establishes the relationship between valuation and weighting. Chapter 3 is concerned with calculation of the shadow prices, with two approaches being distinguished: shadow prices according to abatement costs and according to damage costs. In Chapters 4 and 5 these two sets of shadow prices are then calculated. Chapter 6 goes into the use of shadow prices for weighting purposes and elaborates three different weighting sets for shadow prices based on midpoints. Chapter 7 discusses the practical use of shadow prices and presents recommendations on which set of prices should be used in which type of application.

In several lengthy annexes we present detailed information on how we arrived at our shadow prices. As this information was so extensive and would have had serious repercussions for the report’s readability, we opted to include this information in separate annexes.

The document is also published in a Dutch-language version.

## 1.7 Methodology and process

### 1.7.1 Methodology

In calculating both damage costs and abatement costs use was made solely of existing models and studies. This means, for example, that no new survey-type valuation studies were conducted. In the case of abatement costs, besides earlier studies use was also made of an extensive dataset employed in a SCBA carried out in the Netherlands on acidification ceilings (CE, 2008a), which was based on the dataset developed in the so-called ‘Options Document on Energy and Emissions 2010/2020’ (ECN/MNP, 2006). For damage costs, besides literature studies use was made of the Excel tool developed in the NEEDS framework. For each of the shadow prices, the methodology adopted is described in detail in the annexes.

### 1.7.2 Shadow price units

The shadow prices presented in this report relate to emissions of environmental pollutants from the Netherlands’ territory in the year 2008. In each case these prices are expressed in €/kg emission, at 2008 price levels (often shortened to $€_{2008}$).

The shadow prices are average values for the Netherlands and may vary with local circumstances and the nature of emissions generation (high stacks versus tailpipes, etc.). This variance is cited in the report where relevant and in some cases elaborated with reference to an illustrative table. However, no general methodology has been elaborated for converting these prices to shadow prices keyed to individual sources and emission locations.

Impacts on residents of countries other than the Netherlands have been valued the same as for Dutch residents. Emissions have an impact not only now but in the future, too. In some cases these future impacts have already been implicitly included in the valuation.[9] In cases where future impacts have been calculated, a 2.5% discount rate was used with no ‘risk premium’.

### 1.7.3 Rounding up of shadow prices

The shadow prices calculated in this study have been only minimally rounded up, to a floating comma with three decimal places.[10] This degree of precision means the values presented in this report appear to have greater certainty than is in fact the case. However, because these shadow prices will be used in, say, cost-benefit analyses in which our values will often be multiplied by a factor of a million or more, we felt it would be better to leave it to the practitioner to decide on how the data should ultimately be rounded. In the final table of costs and benefits in a SCBA, the practitioner should therefore decide to what level he or she wishes to round off the monetary values obtained. To our mind, it is more in line with the nature of shadow prices for us not to prescribe a particular degree of rounding deemed by us to be ‘responsible’.

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9 One example in this context is the question, via surveys, how much it is worth to people if they were given an additional six months to live at the end of their life. This question already assumes a certain implicit discounting, because people are obliged to give their present value.

10 This means the set of numbers 1260; 126; 12.6; 1.26; 0.126; 0.0126 all have the same precision.

### 1.7.4 Project supervision

The project was supervised by representatives of the commissioning parties:
- Dirk den Ottelander and Jacquelien Wijkhuijs (Thermphos).
- Rutger Pol (Dutch Ministry of Housing, Spatial Planning and the Environment (VROM).
- Adriaan van Engelen and Marc Herberigs (Stimular Foundation).

Earlier versions of this study have been reviewed by a Committee of Experts comprising the following:
- Dr. Rob Aalbers (Netherlands Bureau for Economic Policy Analysis, CPB).
- Drs. Luke Brander (Institute of Environmental Studies, IVM, Free University of Amsterdam).
- Dr. Reinout Heijungs (Institute of Environmental Sciences, CML, University of Leiden).
- Dr. Arjan Ruijs (Netherlands Environmental Assessment Agency, PBL).

We are very grateful to both the group of supervisors and the Committee of Experts for their efforts. It goes without saying that we ourselves bear sole responsibility for the results presented here.

### 1.7.5 Expertise

It proved unfeasible to derive all the information we needed from the literature. In exploring and elaborating the numerous issues encompassed in the present study we have made grateful use of information from several (international) experts in this field, often via email. In the framework of this study, questions on particular subtopics were put to the following experts:
- Dr. P. Preiss (Institute of Energy Economics and the Rational Use of Energy IER, Stuttgart).
- Dr. Simone Tilmes (National Center for Atmospheric Research, Boulder, USA).
- Prof. Ståle Navrud (Agricultural University of Norway).
- Drs. Lauran van Oers (Institute of Environmental Sciences, CML, University of Leiden).
- Dr. Ir. Onno Kuik (Institute of Environmental Studies, IVM, Free University of Amsterdam).

We thank them for their willingness to answer our questions and engage in discussions on our specific premises. Again, it goes without saying that they bear no responsibility for the results presented here.

# 2 Conceptual framework

## 2.1 Introduction

In this chapter we set out the conceptual framework adopted in the present study. In doing so, we establish a relationship between the economic and environmental analysis frameworks that have been employed for calculating the shadow prices and the use of these prices in Life Cycle Assessment, Social Cost-Benefit Analysis and similar types of analysis. In Section 2.2 we first explain how shadow prices are employed for the dual purpose of monetary valuation and weighting. In Section 2.3 we then set out the theoretical foundations of shadow price calculation. The economic theory behind the use of shadow prices is discussed and the two methods employed for calculating such prices – based on abatement costs and damage costs – are introduced. Section 2.4 turns to the environmental side of the issue, dealing with the characterisation and weighting of emissions and environmental impacts. It will be argued that shadow prices provide an effective means of assigning relative weights to these impacts. In Section 2.5 we examine the relationship between weighting and valuation and explain the philosophy underlying the present handbook.

Although this chapter sets out the conceptual framework of the project, it does *not* deal with the methods used for assigning a value to environmental goods in anything but superficial terms. The methods adopted for monetary valuation are treated in Chapter 3.

## 2.2 Use of shadow prices

In this report shadow prices are understood to be hypothetical prices for scarce environmental goods[11]. They are currently used as a tool in a wide range of decision-making processes, particularly by government and industry.

Shadow prices can be assigned to emissions, pollution, environmental impacts and environmental quality. In this study they are calculated for emissions. In doing so, however, we shall in some cases need to establish the relationship between emissions, pollution, environmental impacts and environmental quality. In the economic introduction (Section 2.3) we shall frequently refer to environmental quality as the inverse of pollution. While this is standard economic terminology, however, it is not an entirely accurate reflection of how the two relate.

11 'Hypothetical' in the sense that these prices are assumed to occur if markets for environmental goods and services were in place.

In practice, shadow prices can provide input to the decision-making process in two ways[12]:

- In analysing the social effects of an investment decision, shadow prices can be used to take environmental impacts on board alongside financial considerations by assigning them a monetary value. Such might be the case in a Social Cost-Benefit Analysis (SCBA), for example. Here the principal aim is ***valuation,*** with shadow prices providing a means of comparing environmental impacts with other monetised items in order to obtain an as comprehensive assessment as possible of all the impacts attending the (investment) decision.
- In environmental analyses, shadow prices can be used to assign a relative weight to each of the environmental impacts identified. This is the case with Life Cycle Assessments (LCAs), Environmental Impact Assessments (EIAs) and benchmarking exercises, for instance. Here the prime aim is environmental ***weighting,*** with shadow prices providing a means of comparing disparate environmental impacts.

**Valuation** of environmental impacts via shadow prices takes place, in principle, in every SCBA in which externalities are assigned a monetary value. In the Netherlands, for example, shadow prices have been used in recent years in SCBAs on offshore wind power (CPB, 2005) and on the impact of the European REACH Directive (Witmond et al., 2004). One problem in this context, though, is that the various SCBAs differ in *how* the environmental impacts are monetised. One of the prime aims of the present project has therefore been to elaborate guidelines on how financial valuation should preferably be effectuated.

**Weighting** of environmental impacts is sometimes carried out as the final step in an LCA to condense the results into a single figure. Various weighting methods have been proposed in the literature, grounded among other things in the concept of ‘distance to target’ (VROM, 1993) and the use of expert panels (Huppes et al., 2007). Nne of these weighting methods is designed precisely for monetising impacts, though (cf. CE, 2002b or Steen, 1999). Monetary valuation is often used as a weighting method in LCAs as well as in practical environmental calculation tools like the ‘Envirometer’ (for small and medium-sized businesses) and GreenCalc (for comparing the environmental performance of buildings).

## 2.3 Welfare-theoretical principles relevant to valuation

Valuing environmental quality means expressing the value of environmental quality to society in monetary terms. Because in many cases the value of environmental quality cannot be obtained directly (via market prices, for example), it must be obtained through calculation.

The research tradition of monetising environmental impacts dates back to the 1930s, when citizens of the United States sought compensation via the courts for the sulphur dioxide emissions of a Canadian mining company (Read, 1963). In the Netherlands, environmental impacts were first monetised in the academic world in the late-1960s, when scientists put a concrete price on noise nuisance (cf. IVM, 1972). Since then, valuation has become part and parcel of environmental economics studies and major efforts have been

12 This distinction relates to the use made of shadow prices and not so much to their substance, for valuation is in principle also a form of weighting, as prices are essentially an indication of the relative social utility of one particular good compared with that of another.

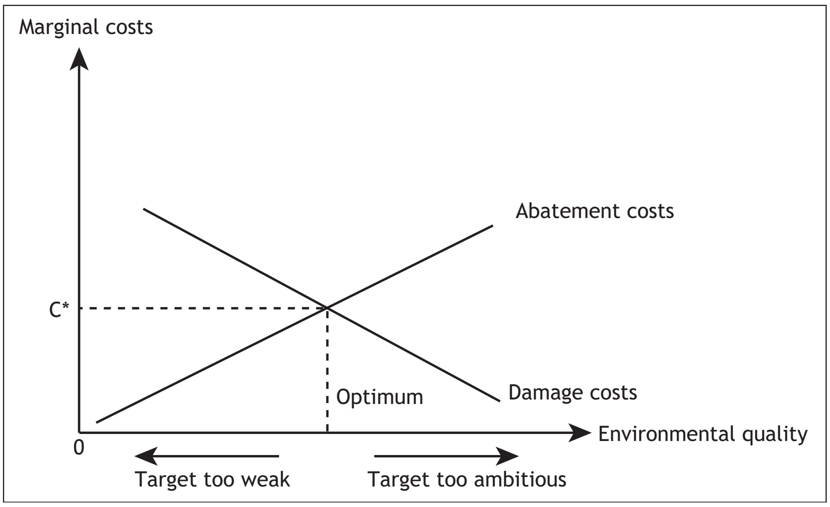
expended on both methodological development and practical valuation studies (Hoevenagel and De Bruyn, 2008).

### 2.3.1 Equilibrium pricing

From an economic perspective, most environmental services are not provided by market mechanisms. We cannot go to the supermarket to buy clean air, biodiversity or protection from environmental hazards. Such services are nonetheless scarce, since their availability is limited and affected by our processes of consumption and production (Hueting, 1980). In economic terms one speaks of the existence of ‘negative externalities’: side-effects of production and consumption that have an adverse effect on the welfare of others without monetary compensation being paid for this welfare loss.[13]

It is instructive to imagine for a moment that a market for environmental services did exist. How much clean air would we then buy? Standard economic theory tells us that society would end up at the point where the benefits of one additional unit of clean air equals the cost of one additional unit of pollution reduction. In other words, the moment pollution abatement would become costlier than the value attached to clean air, we have reached an ‘optimal’ level of pollution (referred to by economists as Pareto-optimality). The associated marginal costs are termed the *equilibrium price* for this particular environmental impact category. This price is an indication of the value attached by society to the impact in question. In Figure 1 this point is marked C*. It is the point at which the marginal costs of abatement equal the marginal costs of damages due to pollution.

**Figure 1 Optimal level of pollution and associated equilibrium shadow price according to economic theory**




Note: Environmental quality is to be interpreted here as the inverse of pollution.

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13 In formal economic language, externalities arise when: (1) certain agents’ utility or production functions include real (not pecuniary) variables whose values are chosen by others who do not take the corresponding welfare effects into account; and (2) the party imposing the negative (positive) external effect does not pay (receive) compensation to (from) the victims (from the beneficiaries) equal to their loss (gain) in welfare (Baumol and Oates, 1988). Other restrictive conditions are sometimes added to the definition of externalities (Mishan 1981, p. 393), such as (3) the activity imposing the externality is not forbidden by law, and (iv) the loss of welfare to the victim is not the result of emotions (such as jealousy or hate) that are considered morally questionable.

Note that as environmental quality improves, marginal abatement costs increase, reflecting the general tendency that at the beginning of the pollution reduction process less costly technologies can be used. Also, over time abatement technologies become more efficient because of economies of scale and technological advancement (learning curves). The marginal damage costs in Figure 1 can be seen as a proxy for the marginal benefits to society of improvements to environmental quality and show a downward trend, reflecting the law of diminishing marginal utility deriving from such improvements.

It is not hard to see that this 'optimal' level of pollution and the associated equilibrium price will not be the same for all pollutants. This is due in the first place to the cost curves for pollution abatement differing widely from one impact category to another. It is far cheaper, for example, to achieve a 20% cut in the $SO_2$ emissions driving acidification, among other things, than to do the same for the $CO_2$ emissions driving climate change, which requires far costlier abatement technologies. The second reason is that society attaches a different value to different environmental impacts. As global warming is likely to have a far greater impact on the economy and on human health, reducing $CO_2$ emissions will be of far more value to society than cutting $SO_2$ emissions. The consequence of these (hypothetical) lines of reasoning would be that society puts a higher value on additional $CO_2$ emissions than on additional $SO_2$ emissions.[14]

### 2.3.2 Shadow prices

Equilibrium prices indicate the true economic value of pollution *if all externalities are internalised*. Although such prices can in principle be developed and used to assign a monetary value to emissions, this is not generally undertaken. The main reason is that such prices only report the external costs of a particular project to society *if* the current pollution level is 'optimal'. The more the current situation deviates from that optimum, however, the more in error such estimates of additional costs will be. In most cases current environmental quality is not located at the optimum because of a lack of (effective) environmental policies.

Figure 2 depicts a situation in which current environmental quality is at A, under current environmental policies, with marginal abatement costs of Ca. The current level of environmental quality (interpreted here as the inverse of pollution) is thus below the optimal level of O. Hence, the marginal damage costs associated with the current situation are given by Cd. Because not all externalities are internalised, in this case one can derive *shadow* prices. A shadow price can be defined as the infinitesimal change along an objective function e.g. arising from an infinitesimal change in a constraint.[15]

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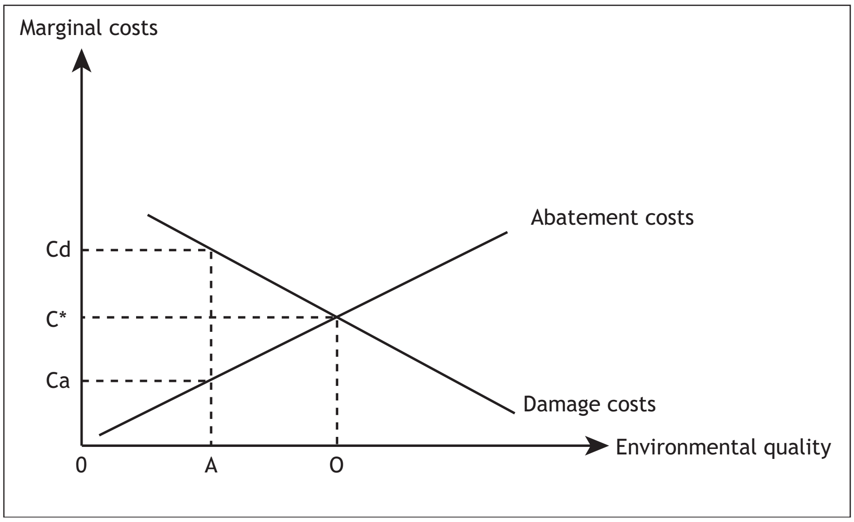
14 We note here that in reality the optimum is hard to locate owing to dynamic development of the curves involved. See also the discussion in Chapter 7.

15 Formally, the shadow price is the value of the Lagrange multiplier at the optimal solution, which means that it is the infinitesimal change in the objective function arising from an infinitesimal change in the constraint. If there are no constraints, the shadow price is equal to the price at the optimum.

Figure 2 Prices of externalities under current political efforts




Note: Environmental quality is to be interpreted here as the inverse of pollution.

Thus, at the present level of pollution (point A), two shadow prices can be obtained:

a A shadow price for the damage cost function, equal to the infinitely small increase (decrease) in damages due to an infinitely small decline (increase) in environmental quality. This shadow price, given by Cd, expresses the so-called marginal damage costs.
b A shadow price for the abatement cost function, equal to the infinitely small increase (decrease) in abatement costs due to an infinitely small increase (decline) in environmental quality. This shadow price, given by Ca, expresses the so-called marginal abatement costs.[16]

Both shadow prices thus yield a value for the (marginal) change in the condition of the environment to society. In the situation represented in Figure 2, the abatement cost approach gives the marginal cost to society of policy efforts to *maintain* environmental quality A, while the damage cost approach gives the marginal cost to society of small *deviations from* environmental quality A.

The difference between these two approaches needs to be properly understood and is illustrated by the following example (see also Chapter 7). Suppose the government has set an $NO_x$ emission target of 250 kt for 2010, which it seeks to secure by rolling out policies including standards and $NO_x$ emissions trading. The transport ministry then launches a road-building programme and the attendant Environmental Impact Assessment indicates this will lead to 5 kt extra $NO_x$ emissions in 2010, clearly jeopardising the government’s emission target. Additional policy efforts are hence required which, at the margin, will cost an amount equal to Ca per unit $NO_x$. In this case the shadow price of $NO_x$ is equal to Ca, as the total level of environmental quality will be unaffected by the existence of the national policy plan to achieve the goal in 2010.

[16] The method used for calculating this type of shadow price is known by several names: the abatement cost approach, the avoidance cost approach or the target-consistent pricing approach.

Now suppose the government announces that the 250 kt target no longer applies and is to be revoked. In that case the additional costs to society will be determined by the damage costs. The additional 5 kt of emissions will, at the margin, cause damages equal to Cd. In this case, therefore, the costs to society should be taken equal to the damage costs.

**The general rule is therefore: if a project leads to changes in environmental quality, these changes should be valued according to shadow prices based on damage costs. If a project leads to changes in environmental policy-induced abatement efforts, these changes should be valued according to shadow prices based on abatement costs.** In practice, of course, the situations may be rather more complicated than outlined here. In Chapter 7 the question of which shadow prices should be used under what circumstances will be elaborated in greater detail. It is only in a hypothetical, ideal situation in which policy targets reflect the socially optimal level of pollution that the two approaches to shadow price estimation yield the same outcome.

## 2.4 Characterisation and weighting

There are known to be over 10,000 pollutants with a potentially deleterious impact on our environment. In analysing the impact of these pollutants and other environmental interventions, environmental scientists have long sought to condense the huge mass of data their analyses generate into a single indicator. This has been effectuated in a two-step approach: characterisation and subsequent weighting of the results.

Characterisation is a process whereby indices known as characterisation factors are used to indicate how much a standard quantity of a given pollutant contributes to a particular environmental impact. The higher the characterisation factor, the greater the contribution. For the impact category 'climate change, for example, the gas methane has a higher characterisation factor than carbon dioxide. This is equivalent to saying that a kilo of methane causes more global warming than a kilo of carbon dioxide.

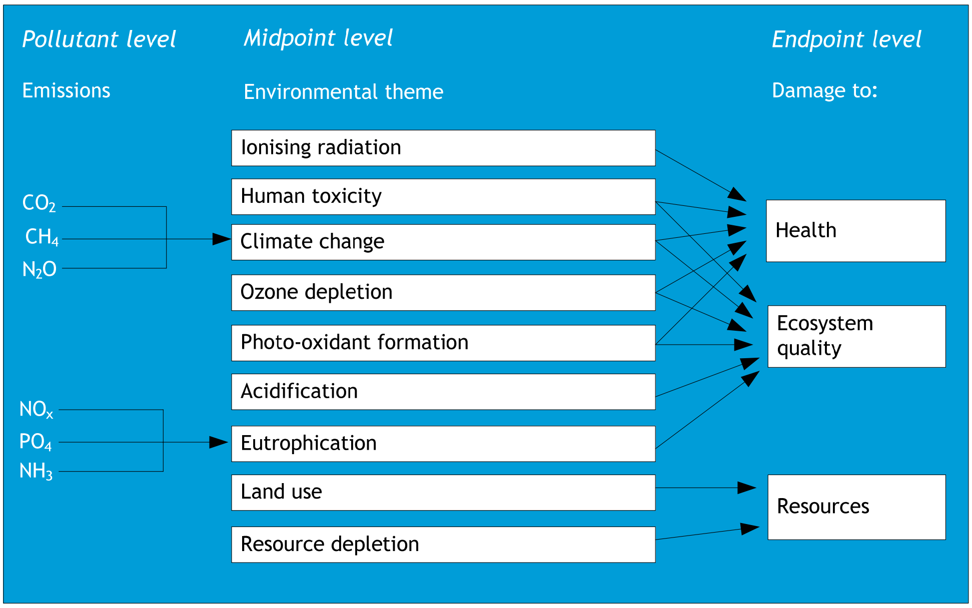
Although these characterisation factors can be used to group emissions into a number of discrete environmental impact categories like 'climate change', 'acidification' or 'human health impacts' (an issue returned to below), these categories are still not amenable to comparison. To provide a reproducible means of comparing these disparate impacts, they are often weighted relative to one another. As mentioned, various methods have been proposed in the literature to this end, based among other things on the notion of 'distance to target' (VROM, 1993) or use of expert panels (NOGEPA, 1999). Shadow prices provide another means of weighting environmental impacts. These prices put a figure on the value of emissions relative to one another and to other goods used and traded in society. In assigning a *value* to emissions, as in SCBA, it is generally the value of the emissions relative to other financial items that is of principal interest. When emissions are *weighted*, however, it is the relationships among those various emissions that are of prime concern. The weighting factors can then be regarded as the relative socio-economic importance attached to the various environmental impacts. These issues will now be discussed in more detail.

### 2.4.1 Pollutant level, midpoint level and endpoint level

Pollutants are by-products of economic activities that are emitted to the atmosphere, water and soil. Examples include $CO_2$ and $SO_2$ as well as particles of arsenic or benzene. Many of these substances have a more or less similar effect: $NO_x$, $SO_2$ and $NH_3$ all contribute to acidification, for example. Acidification can be considered a relevant environmental theme expressing changes in the state of the natural environment. In similar fashion, environmental scientists distinguish a total of 10 to 20 relevant indicators that together characterise the state of the environment. Besides 'acidification', these include 'climate change', 'freshwater eutrophication' and 'photo-oxidant formation', for example. These are referred to as the *midpoint* impacts of emissions. These changes in the state of the environment are important because they go on to have an *ultimate impact*, on human health or biodiversity, for example. These latter impacts are known as *endpoint* impacts.

In environmental science three related levels are thus distinguished: the pollutant level, the midpoint level and the endpoint level. Figure 3 shows the relationship between them.

**Figure 3 Relationships between pollutant level, midpoint level and endpoint level**




While midpoints correspond to environmental themes, endpoints refer to the changes in human welfare resulting from environmental pollution. In reality, there are many more midpoint and endpoint categories feasible than those used in the present study (cf. Section 5.2.2). Some of these are listed in Box 2.

**Box 2: Midpoint and endpoint categories**

Possible midpoint impact categories include (according to Guinée et al., 2002):
- Depletion of abiotic resources.
- Impacts of land use.
- Climate change.
- Stratospheric ozone depletion.
- Human toxicity.
- Ecotoxicity (freshwater aquatic, marine aquatic, terrestrial).
- Photo-oxidant formation.
- Acidification.
- Eutrophication.
- Odour.
- Noise.
- Waste heat.
- Casualties.
- Depletion of biotic resources.
- Desiccation.

In the literature there is no umambiguous list of possible endpoints (cf. Freeman, 1993; PRé, 2000). Broadly speaking, the following endpoints can be distinguished:
- Impacts on human health (premature death and illness).
- Other human impacts (e.g. odour, visibility, visual aesthetics).
- Impacts on natural resources (e.g. crops, fisheries) and recreational activities (e.g. the productive functions of ecosystems).
- Impacts on ecosystems resilience (including biodiversity).
- Impacts on materials and buildings.

The contribution of each pollutant to the midpoint and endpoint levels is determined in environmental science using characterisation factors, which are now discussed.

### 2.4.2 Characterisation factors

Characterisation factors are used in tools like LCAs and EIAs to express the relationships between pollutant, midpoint and endpoint level.

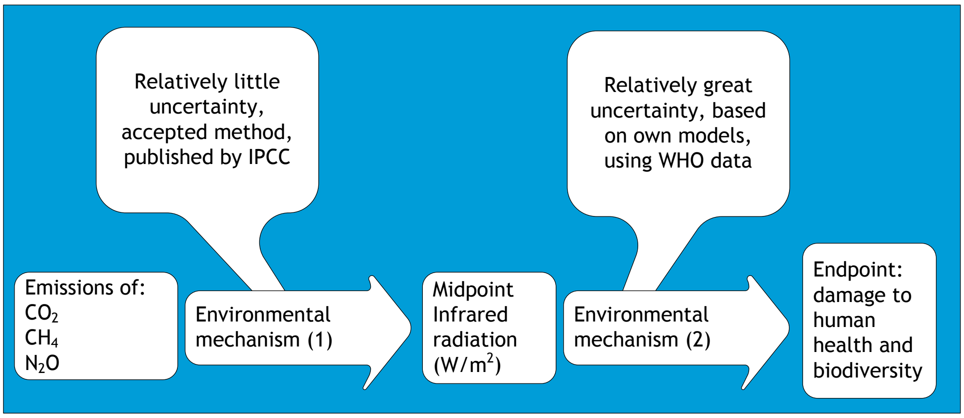
An LCA comprises of a sequence of phases, one of the first of which is to inventory all the pollutant emissions, consumed resources and immaterial impacts like land use. These impacts are often organised into (midpoint) impact categories like climate change. Midpoint characterisation factors aggregate individual emissions (e.g. kg $CO_2$, kg $CH_4$) to these integrated environmental themes. In the case of climate change, for instance, the indicator is infrared radiation (in $W/m^2$), while the characterisation factor is Global Warming Potential (GWP in kg $CO_2$-eq. per kg). Endpoint characterisation factors translate these environmental themes into impacts on human health, biodiversity and cultural and natural heritage. The endpoint impact categories for climate change, for example, are damage to human health and to biodiversity, which have as their respective indicators DALYs (Disability Adjusted Life Years) and PDF (Potentially Disappeared Fraction of species) (cf. Chapter 5 and Annex H).

In general terms, then, there exist two sets of characterisation factors:
- **Midpoint** characterisation factors, as defined in the LCA Handbook (Guinée et al., 2002). In this case the impacts of an environmental intervention at the level of emissions are aggregated to so-called environmental policy themes based on their potential contribution to these themes.
- **Endpoint** characterisation factors, as defined in Ecoindicator 99 (PRé, 2000). In this case the impacts are aggregated around the ultimate 'receptors' (human health, biodiversity and depletion of natural resources).

Until recently there was little harmonisation between these two sets of characterisation factors. The use of different premises for determining the midpoint and endpoint characterisation factors meant an LCA practitioner could potentially have a certain influence over the results depending on the chosen type of characterisation. In technical terms: there were no conversion factors for moving from midpoint to endpoint.

The ReCiPe project, started in 2001, seeks to bring an end to this situation and develop consistent sets of characterisation factors at midpoint and endpoint level, with a view to integrating the midpoint and endpoint methods. Figure 4 depicts the relationship between midpoint and endpoint for climate change. Extra greenhouse gas emissions lead to extra absorption of infrared radiation and therefore contribute to global warming. To represent the impact of different greenhouse gases on global warming, use is made of midpoint characterisation factors like Global Warming Potentials (GWPs, expressed in kg $CO_2$-eq./kg) to characterise the amount of infrared radiation absorbed by each of the gases in question and their residence time in the atmosphere. Environmental models can then be used to determine the ensuing damage to health and biodiversity per kg $CO_2$-eq. This means ReCiPe provides not only midpoint and endpoint characterisation factors but also specifies the relationship between them.

Figure 4 **Relationship between midpoints and endpoints in ReCiPe. Example of climate change, linking to human health and ecosystem damage**




Source: Adapted from Goedkoop et al., 2009.

In ReCiPe, endpoint characterisation factors have been determined for the following endpoints:
– Human health.
– Ecosystems (biodiversity).
– Availability of abiotic resources (cf. Section 3.5.5).

Endpoints relating to capital goods and agricultural productivity were not within the scope of ReCiPe.

### 2.4.3 Weighting

The environmental impacts at midpoint or endpoint level do not lead to an unequivocal result of the environmental analysis. For years, aggregation to a single score has been a source of major controversy in LCA circles and beyond. Aggregation is bound to involve a certain degree of subjectivity, with any decision on the weighting factors being based on a qualitative consideration of the results of the LCA. This is why some people hold it would be better for weighting to be dispensed with altogether. Others hold that this kind of weighting takes place anyhow whenever a decision is made, whether it be explicitly or implicitly, and that it would therefore be better to establish a single, formalised procedure, because this would at least render the criteria transparent.

Normally speaking, weighting takes place on the basis of either midpoint impacts or endpoint impacts, with the methods employed in the two cases not essentially differing. In broad terms, two options can be distinguished (Warringa, 2003)[17]:

A) Weighting based on expert panels or questionnaires. In this case a group of respondents are asked their opinion on the relative importance of a series of environmental impacts. Expert panels endeavour to aggregate the scores for the impacts on the various environmental themes on the basis of expert opinion. The advantage of this approach is that experts are well-informed about the potential effects of environmental problems. Questionnaires can also be used for getting non-experts to assess the relative weight of the endpoints, particularly in the case of endpoint weighting. The advantage of this approach is that at endpoint level a weighting can be found that is closer to people's general preferences. Examples of weighting factors using expert panels are the NOGEPA weighting factors at midpoint level (NOGEPA, 1996; Huppes et al., 2007) and the weighting proposed in Eco-indicator 99 at endpoint level (PRé, 2000).
B) Weighting based on monetisation. In this case weighting is carried out on the basis of the estimated social costs of the environmental impacts. Here a distinction can be made between weighting according to abatement costs (Vogtlander and Bijma, 2000; CE, 2002b), whereby the costs that need to be incurred to secure government targets are the determining factor, or weighting according to damage costs at the endpoint level (cf. Steen, 1999).

Shadow prices can be used for the latter form of weighting, based on monetisation. The weighting factors can then be seen as representing the relative socio-economic importance attached to the respective environmental impacts.

[17] Besides these two options, weighting can also be performed using the 'distance to target' method (VROM, 1993), although in many cases this does not solve the weighting problem entirely. For this reason this method is not considered further here.

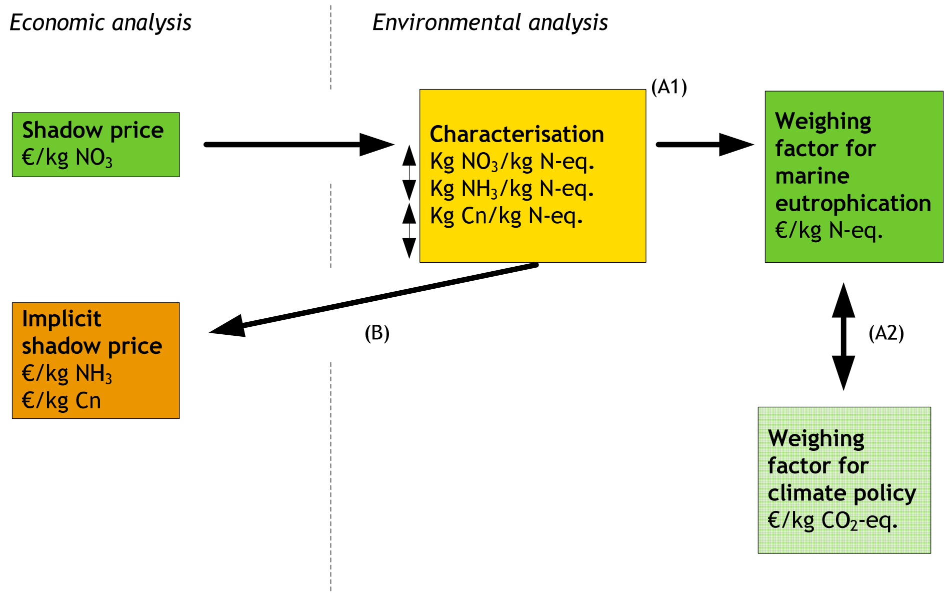
## 2.5 Relationship between characterisation, valuation and weighting

It was stated above that economic valuation can be used in environmental analyses where weighting is important. Conversely, though, economic valuation can also make use of insights developed in environmental science. This is particularly true in the context of characterisation factors, as these factors establish a fixed relationship between pollutant level and environmental impact:

– *Valuation* of an individual pollutant can be extended to *valuation* of all pollutants having an allied environmental impact through judicious use of characterisation factors.
– Characterisation factors determine the *weighting* factors to be used for individual environmental themes.

Figure 5 explains how this works.

**Figure 5 Interrelationship between environmental and economic analysis in this handbook**




Assume, for example, that we have calculated a shadow price for $NO_3$, using abatement costs or damage costs. $NO_3$ is a key factor in the eutrophication of coastal waters. There are also other pollutants involved, though, including $NH_3$, Cn and N. In various studies on characterisation factors, values are reported for the relative contributions of each of these pollutants to the theme of eutrophication of marine waters. By multiplying the calculated shadow price by the characterisation factor a price is obtained (in €/kg N-eq., see **(A1)**) that can be used as a weighting factor and compared with other factors, for climate policy, for example (see **(A2)**). These weighting factors represent the mutual relationship between the environmental impacts in terms of their seriousness when these are approximated via economic analysis.[18]

[18] For damage costs this holds because damage is related to environmental impact. In the case of abatement costs this is strictly only true if government policy is addressing the environmental impact. This would mean that policy targets reflect the true degree of damage that the various pollutants cause. See also Section 6.6.

On the other hand, weighting factors can also be used for assigning a value to those pollutants for which the economic analysis failed to find a value, as in the case of the impact of $NH_3$ and Cn on coastal water eutrophication. Using the characterisation factors, an implicit valuation of $NH_3$ en Cn emissions can then in principle be carried out for their impact on this environmental theme (see **(B)**). It should be remarked that both $NH_3$ and Cn have impacts on several themes.

Consequently, characterisation factors play a pivotal role in this study. One of the problems, however, is that the characterisation factors need to be compatible with the method used for economic valuation. Only if the characterisation factors and the calculated shadow prices are conceptually compatible can we adopt the route outlined here. The use of characterisation factors in the present study will be discussed in Chapter 6.

## 2.6 Conclusions

Shadow prices are implicit prices for environmental quality that can be used in a range of economic and environmental tools. Valuation and weighting are two key elements of the use of shadow prices. In this Chapter we have identified two sets of such prices: one based on the supply function for environmental quality (abatement costs), the other on the demand function for environmental quality (damage costs).

Coupling shadow prices to environmental impacts enables the latter to be given a monetary value. Environmental impacts can be valued at either midpoint level (e.g. climate change, acidification, eutrophication) or endpoint level (e.g. human health, biodiversity). In the literature there are various different sets of characterisation factors available. In the present study we have generally based ourselves on the characterisation factors from the ReCiPe project. These factors have been used on the one hand to elaborate a set of weighting factors for environmental impacts at midpoint level and on the other to extend the set of shadow prices to yield a more complete set.

# 3 Methodological framework

## 3.1 Introduction

In the previous chapter we set out the conceptual framework of the present study. We now take a closer look at the theory behind calculation of shadow prices for individual pollutants. This chapter thus provides a general introduction to shadow price calculation. In doing so, we examine how the supply and demand functions for environmental quality can best be approached, with the abatement cost method yielding an estimate of the supply of environmental quality and the damage cost method an estimate of demand for it.

In Section 3.2 we first consider the calculation of abatement costs, where the marginal costs of securing policy targets is the issue of concern. In Section 3.3 the value to be assigned to demand for environmental quality is then determined by examining the damage occurring as a result of changes in that quality. Valuation of environmental goods will always be a controversial issue and in Section 3.4 we therefore examine a number of ethical concerns about the practice. Section 3.5, finally, provides a synopsis of the premises and assumptions we have used in determining the shadow prices presented in Chapters 4 and 5.

For readers familiar with the extensive literature on the subject of financial valuation, Sections 3.2 and 3.3 of this chapter will perhaps not contain that much new information and may even be deemed superficial or incomplete. The aim of this study, however, is not to provide a textbook on valuation, but to elaborate a set of practical shadow prices for use in economic and environmental tools. The reason for including these sections nonetheless stems from the idea that a minimum notion of what valuation precisely entails is required if one is to embark on the use of shadow prices.

## 3.2 Shadow prices according to abatement costs

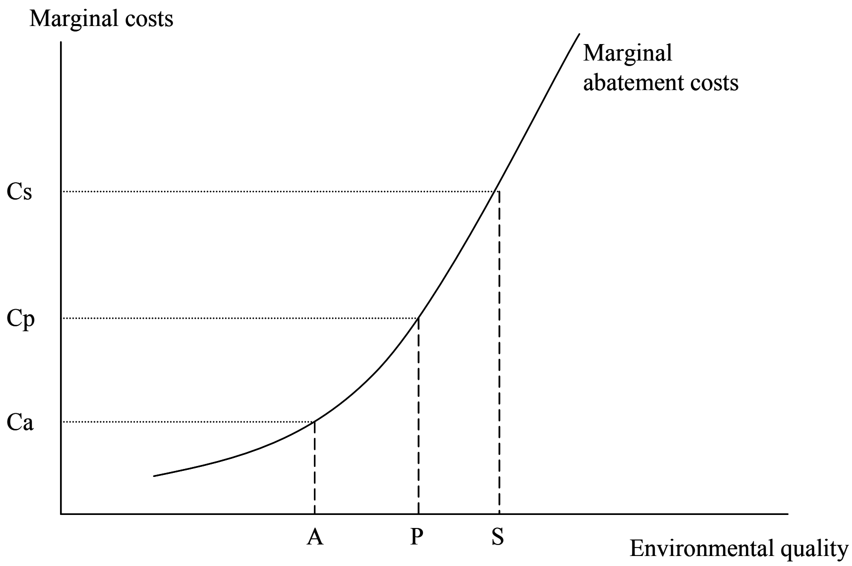
The abatement cost function provides a proxy for the supply of environmental quality. It determines how much it would cost to supply an additional level of environmental quality. Shadow prices for abatement costs are determined using the abatement cost curve, as outlined in Section 2.3. In this curve, ideally all costs associated with reducing environmental pressure should be included. Hence, the complete version of the curve includes the costs of environmental (i.e. abatement) technologies, the (opportunity) costs of reduced output, the costs of environmental regulation (e.g. administrative costs) and the indirect costs associated with the higher cost of output due to application of the abatement measures.

In practice, however, only a subset of these costs is included, viz. the costs of the technological or organisational measures implemented to reduce emissions.[19] This makes it impossible to carry out a direct comparison with the damage costs in order to determine the 'optimal' level of pollution (cf. Sections 4.2.1 and 7.3.2).

### 3.2.1 Types of shadow prices using abatement costs

In principle one can distinguish three types of shadow prices based on abatement costs, proceeding from three different starting points. First of all, in Figure 6 the current level of environmental quality is set at A. In addition, policy-makers may seek to reduce emissions to level P by means of environmental policy interventions. Finally, scientists, for their part, might have determined that the sustainability threshold of this pollutant is equivalent to S, the point at which the pollution no longer exceeds the natural capacity of ecosystems to absorb such substances.

**Figure 6 Three shadow prices according to the abatement cost approach**




Note: Environmental quality is to be interpreted here as the inverse of pollution.

The corresponding three shadow prices are now:
Ca: the shadow price of maintaining current environmental quality.
Cp: the shadow price of the policy target.
Cs: the shadow price of the sustainability threshold.

In practice, only Cp is used as a shadow price. This is because Ca clearly embodies a normative point of view, for nowhere is it laid down that the current level of environmental quality must be maintained. Since the aim of most environmental policy programmes is to reduce emissions, for most environmental themes Ca is in fact not in line with society's preferences.[20] Cs, on the other hand, also embodies a normative point of view, but a more

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[19] In some cases (e.g. the cost curves underlying the so-called 'Options Document' (ECN/MNP, 2006)), the costs of reduced output are also included (the 'shrinkage scenario').

[20] One exception would be for environmental themes characterised by conservationist arguments, such as land use. It can be assumed that society is wiling to conserve certain landscapes, implying acceptance of a stand-still principle in which further deterioriation is deemed unacceptable.

compelling one than Ca, for it has been argued by numerous economists that remaining within ecosystem limits is a precondition for any economic system to provide sustained human welfare (see, for example, Daly and Cobb, 1989; Hueting and De Boer, 2004). This echoes earlier statements to the effect that economics is ultimately a teleological science that should seek the most cost-effective means of achieving given ends (Robbins, 1935; Hennipman, 1978). Remaining within sustainability constraints can then be regarded as a given end. In practice things prove rather more complicated, though. For example, a sustainability threshold for global warming is often set at a temperature rise of 2 degrees Celsius. In itself, however, even this is an arbitrary choice, since even with this threshold irreversible man-made changes to ecosystems are inevitable (Lynas, 2008). In identifying this 2°C threshold, certain implicit value judgments on acceptability and costs are clearly already being made. It can be argued, moreover, that a strict sustainability criterion conflicts with other worthy goals like poverty alleviation, sanitation programmes or infant mortality reduction, since allocation of funds to achieve strict sustainability prevents alternative allocation to the latter goals. Adopting the strict sustainability criterion then assumes the superiority of that criterion to other normative principles, which is itself a normative position (at least, as long as one holds that humanity itself is not endangered if we fail to respect these strict sustainability constraints).

We have no wish or ambition to discuss here at length the question whether nature imposes an absolute constraint on our economic activities and whether that constraint is currently being overstretched (but see for a discussion De Bruyn, 2000, Chapter 2). We simply note that it is policy targets that are most frequently used for determining shadow prices according to abatement costs. This approach was originally proposed by CE Delft and developed in the form of a set of shadow prices that has been widely used in the Netherlands (CE, 2002a). In this approach the shadow prices are obtained with reference to the marginal costs of meeting the policy target in question.

### 3.2.2 Techniques to reveal abatement costs

The construction of shadow prices based on abatement costs for use in valuation and weighting involves two steps:

1. Analysing the policy goals and determining the marginal costs of achieving these goals.
2. Establishing a mechanism for allocating 'joint costs'.

#### Analysing policy goals and determining marginal costs

There are two type of policy goal that can be used in the abatement cost method:

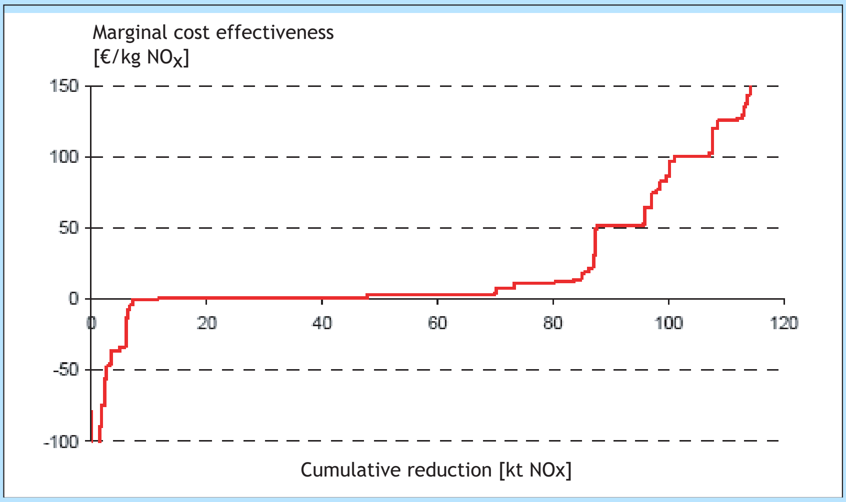
a Policy goals for which no national targets are set, but which use economic instruments to regulate pollution. Examples include Dutch surface water policy, whereby there is a charge on effluent discharges that is calculated per 'pollution unit'. In this case, then, the marginal costs of the policy target are equivalent to the maximum charge levied on the effluent discharge. If there is an emission trading scheme in place, the marginal costs may be equal to the price of emission allowances. If industry or consumers prefer to pay fines rather than implement abatement measures, these charges may provide another estimate of the shadow price.

b Policy goals for which national targets have been set, but which do not (exclusively) use economic instruments to regulate pollution. Examples here include emission targets for $CO_2$, $NO_x$, $SO_2$, NMVOC, etc. In this case the marginal costs must be estimated with reference to the so-called marginal abatement cost curve (MACC: see Box 3).

**Box 3: The marginal abatement cost curve**

The basis of the abatement cost approach is the marginal cost curve for pollution reduction, also known as the abatement cost curve or the marginal abatement cost curve (MACC). Since the mid-1990s such cost curves have frequently been determined for numerous pollutants across a wide range of countries, regions, municipalities and firms. As an example, Figure 7 shows the MACC for the reduction of $NO_x$ in 2020 according to the 'Options Document on Energy and Emissions' (ECN/MNP, 2006). The curve is derived from the costs associated with a broad range of potential abatement methods. As some of these techniques involve a discrete choice (apply or not), the MACC is not a smooth curve. As can be seen, there is a small set of measures that are actually profitable, as indicated by the negative costs. These are generally measures relating to reducing inputs of energy and materials.

**Figure 7 $NO_x$ emissions abatement costs**




Source: ECN/MNP, 2006

The cost curve does not itself lead to a shadow price unless a target is set that intersects this curve somewhere. In the literature, political targets are generally used, i.e. targets laid down in national environmental policy plans. The $NO_x$ costs in Figure 7 may serve as an illustration. In 2005 current $NO_x$ emissions as high as 350 kt/y were reported. International targets for the year 2010 are 260 kt, however, and the policy target is therefore to reduce emissions by 90 kt over the next 5 years. From Figure 7 we see that the marginal costs associated with this policy goal are about € 50/kg $NO_x$. This can consequently be regarded as a preliminary shadow price for $NO_x$ pollution. It is preliminary, because it ignores any synergy with measures to reduce other pollutants, such as $CO_2$. As $CO_2$ measures also reduce $NO_x$, the total policy goal is in fact less than 90 kt. Calculations by CE Delft based on the cited Options Document indicate that the costs of meeting the $NO_x$ target may fall from € 50 towards € 8/kg $NO_x$ if all synergistic reductions are taken into account. Hence, it is very important to make due allowance for any such synergy with policies on other pollutants.

Allocating 'joint costs'

One of the problems involved in determining shadow prices is how to allocate measures that reduce emissions of more than one pollutant across these various pollutants. Generally speaking, there are two ways to tackle this problem:

1. Economic optimisation.
2. An iterative procedure.

*Economic optimisation* proceeds from the perspective of an array of measures being implemented to cut emissions of specific pollutants, for each of which a policy goal has been developed. By minimising the aggregate costs involved in reducing all these pollutants, a set of least-cost techniques is identified. The costs of these techniques can subsequently be divided over the pollutants according to their contribution to the overall least costs. However, this contribution must still be assessed with reference to something like distance-to-target or endpoint characterisation factors. Hence, the economic optimisation principle does not entirely solve the problem of joint cost allocation.

Alternatively, an *iterative procedure* can be used to allocate costs to reductions of the respective emissions. In the first iteration, to establish the abatement costs associated with each of the pollutants, the other emissions can be valued at either zero, at some earlier estimate of the abatement costs or at a shadow price derived in another manner. In each subsequent iteration, estimates are improved by taking due account of other pollutants by valuing them using the cost estimate of the previous iteration. Although this method is intuitively appealing, it runs the risk of adopting a suboptimal set of estimates for the total reduction of costs for the economy (path dependency in the iteration procedure).

In conclusion, there is no one scientifically valid method for allocating joint costs. Any method adopted for this purpose will inevitably introduce an element of arbitrariness into estimates of abatement costs.

### 3.2.3 Abatement costs and characterisation

As explained in Section 2.5, in this study we use characterisation factors for two purposes:

a To estimate weighting factors.
b To estimate shadow prices for pollutants for which no costs are available.

The key question, though, is whether midpoint or endpoint characterisation factors should be used for calculating abatement costs. This question can best be answered by examining how government environmental policy is shaped: after all, abatement costs provide an estimate of the cost of securing the policy targets in question.

Government environmental policy is very clearly oriented more towards midpoints (environmental themes) than endpoints (health, crop damage and so on). Climate policy is an example involving emissions not only of $CO_2$ but also of other greenhouse gases. Acidification, eutrophication, noise nuisance and waste policy are all separate environmental policy themes. Here, then, it makes sense to calculate abatement costs using midpoint characterisation factors. This issue will be discussed further in Chapter 6.

## 3.3 Shadow prices according to damage costs

The damage cost approach attempts to estimate the demand function for environmental quality. This demand is driven by the ability of people to pay for that quality. In other words: how much of their income would they be willing to sacrifice to obtain an additional unit of environmental quality. This is commonly referred to as the Willingness to Pay (WTP). Another approach is to consider the extent to which people are willing to accept environmental damage. This is known as the Willingness to Accept (WTA). The concepts of WTP and WTA are thus both defined in terms of individuals' preferences.[21]

### 3.3.1 Methods

There are several methods available for estimating WTP, falling broadly into two categories: stated preference and revealed preference methods. Stated preference methods use questionnaires to assess people's WTP for maintaining or improving environmental quality. For many environmental themes this is rather tricky, as most people have no clear understanding of what environmental quality actually implies for their lives. Interviews containing questions like 'How much would you pay to reduce $SO_2$ emissions by 1 ktonne?' will not yield any meaningful results, simply because 1 kt of $SO_2$ emissions remains an abstract artifact. Questionnaires therefore need to be carefully designed so respondents can refer to concrete phenomena that are comprehensible to them. This implies that WTP is generally estimated at the endpoint level and refers to concrete environmental impacts such as those on human health, ecosystem resilience or crops, fisheries and biodiversity.

Among stated preference methods the most popular is the Contingent Valuation Method (CVM), whereby survey respondents are asked directly about their WTP for a certain good, described carefully in the study scenario. For example, respondents may be asked for their WTP for preserving a certain ecosystem endangered by global warming. Another option is to ask a question about the Willingness to Accept (WTA) loss of that ecosystem; this approach is considered to yield less credible results, however (see Box 4).[22]

21 These preferences should be comprehensive, stable and coherent. 'Comprehensive' is taken to mean that individuals must be able to make meaningful preference comparisons between the specific costs or benefits under consideration and the standard of measurement (normally money). 'Stable' means that preferences should not vary arbitrarily over time and that different theoretically valid methods of eliciting a person's preferences should yield comparable results. 'Coherent' means that for a given person the preferences elicited should be internally consistent when viewed in the light of some acceptable theory of preference (Bateman et al., 2002).

22 A variant of CVM is the Choice Experiment (CE) method, in which the respondent is given a set of alternatives and asked to choose the most preferable. The WTP for certain attributes (the risk of mortality, for example) is then revealed using econometric analysis.

**Box 4. The WTP-WTA discrepancy and voluntary vs. involuntary risks**

Despite theoretical assumptions that WTP and WTA values should be roughly equivalent for small income losses, there is a large body of empirical evidence that WTA values are typically two or more times higher than parallel estimates of WTP. There seem to be several reasons for this phenomenon, including: (1) psychological aversion to loss: losses matter more to people than do commensurate gains, and the reduction of losses is worth more than foregone gains; (2) income constraints: WTP is constrained by an individual's income while WTA is not; (3) ethics/legitimacy: in the context of valuation of human or species health or safety, some people may deem it unethical to reduce levels of health or safety in exchange for money.

It is noteworthy that in CVM studies the concept of WTP is more often used, with some researchers explicitly or implicitly rejecting the use of WTA as incredible. However, while WTP is definitely more appropriate than WTA in certain cases (e.g. for determining the merit of a proposed habitat improvement programme), it is questionable whether WTP is an appropriate measure for establishing, say, compensation to a coastal population after an oil spill. For resource losses, the concept of WTA seems more correct than WTP. The reluctance of analysts to use WTA for environmental losses means activities with negative environmental impacts will be unduly encouraged, because the real value of the associated losses will be underestimated (Brown and Gregory, 1999).

Another factor influencing the value revealed during valuation studies is the nature of the risk. There seems to be an additional premium attached to involuntary risk. According to some studies, the WTP for avoiding road accidents (where the actions are perceived to be voluntary in the sense that the risk is under an individual's control) is lower than the WTP for avoiding train accidents (ExternE, 2005). Hence, for valuing damages that occur in a situation of involuntary risk, either a certain premium or a lower discount rate might be used.

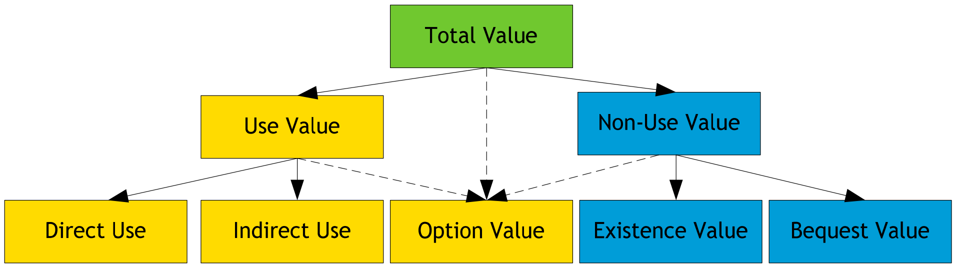
The damage costs reported in the present study often refer to health risks associated with air pollution, which are generally involuntary. Because valuation of human health damage costs in NEEDS is based largely on the VOLY obtained in a survey where the scenario described (involuntary) exposure to air pollution, there is no need for additional adjustment or adding a premium for involuntariness, as this should already be included in the estimates due to the nature of data collection.

An alternative approach uses revealed preference methods, whereby the value for environmental quality can in some cases be measured by using other markets as proxies for the non-existing market for environmental quality. If house prices are lower in polluted areas than in cleaner ones, an implicit price for environmental quality is provided by property price differentials. However, as house prices are codetermined by many other factors besides environmental quality, it may be hard to tease out the specific influence of the latter.

### 3.3.2 Types of values relevant in WTP

Stated and revealed preferences tend to reveal only part of the total value of environmental services. Total economic value can be divided into two broad components: use value and non-use value (Figure 8). Market prices normally reveal the direct use value. Indirect use value relates to the value ascribed to specific functions of certain ecosystems, such as erosion control and flood control or nutrient entrapment. In addition, natural resources may be valued simply for their potential availability in the future. The position of this so-called option value in the overall scheme is controversial: there is no consensus among environmental economists as to whether this is a subcategory of use or of non-use value.

Figure 8 Total economic value and its components




Source: Dziegielewska et al. 2007.

The category of non-use value captures those elements of value that are unrelated to any current, future or potential uses. Existence value reflects benefits related to satisfaction that a given good exists. People may be willing to pay a certain amount to preserve endangered species even if they know they will never visit their habitats or use them in any way. Bequest value refers to benefits from ensuring that certain goods will be preserved for future generations.

The problem with valuation is that use values are easier to establish than non-use values (even in CVM), as respondents have more direct knowledge about the costs of particular forms of use. Bequest values tend to be more ethical in nature and thus harder for respondents to estimate. Revealed preference methods can only establish use values, while stated preference methods can reveal total economic value, i.e. including non-use value and option value.

### 3.3.3 Estimating damage costs

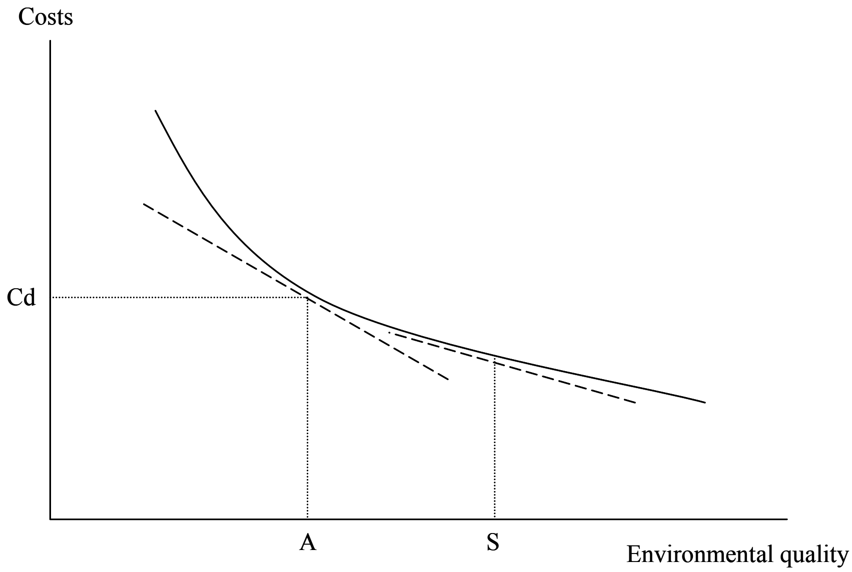
As stated above, one of the problems with estimating WTP for environmental quality is that people have little knowledge about the relationship between emissions and the aspects of value that might be threatened by the ensuing pollution. Respondents cannot therefore simply be asked for their WTP for reducing certain emissions. For this reason WTP is often calculated by investigating the *damages* due to pollution, with these damages geared to the endpoints developed in environmental science.

One of the first major studies to explore this route was grounded in the ExternE framework, initiated in the 1990s. In this case pollution impacts are translated as far as possible into their ultimate impacts on valuable goods: human health, capital goods and ecosystem services. By valuing pollution-related damages in these three categories, a proxy can be derived for the Willingness to Pay (WTP) for an additional unit of environmental quality. It should be noted that although damages to human health are not explicitly valued via the marketplace, in both environmental economics and health sciences there is a vast body of literature on the valuation of human health. By linking environmental valuation to this research data a far more reliable estimate can be made of the value of environmental quality. This is the approach that has been adopted in the present study (cf. Chapter 5).

### 3.3.4 Non-linearity, marginal and average damage costs

The WTP depends on the current level of environmental quality. However, the damage function itself is essentially non-linear owing to the existence of thresholds and irreversibility in ecosystem behaviour.[23] In addition, environmental quality can be considered a normal economic good which is subject to the law of diminishing marginal utility. People in densely forested countries tend to value forests lower than those living in countries where virtually all forests have been logged. At the margin, however, the damage costs will approach linearity. By investigating the damages arising at the present level of pollution (in Figure 9, level A), one can arrive at an estimate of the damage costs (Cd). It should be noted that virtually all the studies on the subject have assumed damage costs to be linear. In other words, they estimate the environmental damage (generally at the present level of pollution) and then divide the costs by this level of pollution in order to arrive at the damage costs per unit pollution. In this way the marginal damage costs become identical to the average damage costs. While this may be true for marginal changes, it should be noted that for larger, systemic changes this value cannot be used to estimate the value of environmental quality. This is illustrated in Figure 9, depicting the marginal damage function. If pollution were to drop from the current level of A to S, for example, damages might be overestimated using the marginal damage estimate from point A.

**Figure 9** **Damage cost estimation at the level of current pollution (A) under non-linear marginal damage cost functions**




In this document the marginal damages occurring at the present level of pollution are referred to as the shadow price.[24]

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[23] For the Netherlands a sea level rise of 20 cm will have negligible effects on the economy, but a 4-metre rise will have a tremendous impact as the country may become largely submerged.

[24] Strictly speaking, this is not a shadow price, because no constraint is formulated, but rather a valuation of the marginal environmental damage. In line with common parlance, however, we shall refer to this valuation as a shadow price.

### 3.3.5 Damage cost estimation and characterisation

Because damage cost estimation is very much congruent with the endpoint approach, the most obvious course of action is to use endpoint characterisation factors in the damage cost approach. In many ways the two approaches are in fact very similar, for in calculating both damage costs and endpoint characterisation factors use is made of environmental and health models to estimate the relationship between emissions and endpoints. While the endpoint approach describes the physical impacts in terms of a set of physical indicators, however, the damage cost approach goes one step further and actually assigns a monetary value to them.

In principle, then, a damage estimate for emissions can also be obtained through direct valuation of the endpoints, with the relationship between these endpoints and individual emissions being established using characterisation factors (cf. Section 2.5). This will be elaborated in Chapter 5; which approach is preferable given data availability is an issue discussed in Chapter 6.

## 3.4 Ethical perspectives on valuation

Assigning an economic value to impacts on the environment and human health may raise moral objections. Below we discuss the most important objections to economic valuation *as such*, thereby ignoring the various other moral dilemmas concerning determination of the actual values themselves, such as whether values for personal risk reduction may be used for risks imposed on others, or the intergenerational weighting of costs and benefits (the choice of social discount rate).

### Economic valuation denies nature's intrinsic value

To have an intrinsic value implies being a goal-in-itself, independent of any instrumental or use value to anyone or anything else. Irrespective of the existence of such value, however, economic valuation does not imply any denial thereof. Economic valuation only facilitates and rationalises choices between alternatives for which scarce resources must be used (time, money). If money is spent on alternative A, it cannot be spent on alternative B. In this weighting of choices, the acknowledgement of intrinsic value may very well be taken into account. After all, even the most devoted conservationists must decide on which environmental projects to spend their money on and how much to spend on feeding themselves. When deciding how much of our budget to devote to development aid, we do not deny the intrinsic value of people living in the developing world. Economists look at people's willingness to pay for various goods and objectives and use this to deduce economic values for them. Of course, people may disagree about other people's preferences and (moral) values, and thus their willingness to pay. However, economists are merely observing what occurs in society.

### The economic value of nature is infinite, since its existence is a prerequisite for (human) life. As such, nature may not be weighted against other goods

Although the economic value of nature *as such* may be infinite (or extremely high: Constanza et al., 1997), it is not so 'at the margin'. Choices as to whether or not to prevent a minor environmental intervention do not put humanity or nature as such in jeopardy. Economic valuation is always concerned with valuation of these kinds of marginal (i.e. relatively small) interventions. When we say the price of bread is one Euro, we merely mean this is the price of one additional loaf. The multiplication of all the food in the world by its price gives a meaningless number, though, since we cannot live without food.

### The values of goods like good health and material consumption goods are incommensurable

The objection here is that the value of good health, for example, cannot be expressed in the same metric as the value of things like cars and televisions (for a discussion of the issue, see e.g. Alfred, 2006). This issue has essentially already been discussed. First, economic valuation takes place at the margin. The discussion is not generally whether or not to save a particular life, but how to reduce health risks by a certain percentage. People show themselves both competent and willing to compare the use of resources for health risk reduction and more material needs. Second, even in the case of specific health impacts it should be noted that resources can always be used to avert alternative health impacts and that choices thus always have to be made.

### It is immoral to put a price on human life

Although it may seem 'cold-hearted' to rationalise decisions regarding human life, there is no reason to see this as *immoral*. There is no moral imperative to save lives at all costs (for example at the expense of one's own life). Two kinds of choices should be distinguished, however. First, there are situations in which specific lives are at stake, as with miners trapped in a collapsed mine. In such cases there seems to be no limit to the expenses incurred to save lives. Even in such cases, though, there are still trade-offs to be made: resources used to save these particular lives cannot be used simultaneously to save other victims. Economic valuation is not about putting a price tag on such specific lives, however, but about the value of *statistical lives*. The policy choices for which economic valuation is employed relate to marginal changes in the risks people face. For example, if a certain risk is reduced from fifteen in a million to fourteen in a million for a population of one million, then one *statistical life* is saved. Economists observe that such comparative assessments of risks and possible gains are regularly made in everyday life, as in choices between modes of transportation. From such social choices economists can deduce values for statistical lives. So while life as such may be priceless, safety in the sense of statistical risk reduction is not. The same reasoning holds for the protection of nature (see e.g. Power, 1996). Furthermore, it should be noted that in policy-making one cannot abstain for moral reasons from any valuation of statistical lives at all; if one abstains from *explicit* economic valuation, this only means that valuations are made implicitly and presumably inconsistently. Since financial resources are limited, all policy-making (including that undertaken in the past) involves making choices (CE, 2002b).

### Economic valuation is useless in policy-making

Some critics object to economic valuation by saying that by focusing on the preferences of individuals, only self-interest is taken into account. They argue that issues like environmental protection should be judged on the basis of public interest, that is, what is best for society as a whole. Whether or not the public interest is the same as the sum of individual self-interests is a controversial issue in political philosophy that remains unresolved. We can only emphasise that the WTP-based shadow prices that can be used in cost-benefit analysis are not a substitute for a political process; they merely provide information on people's preferences, e.g. how much people are willing to pay for a certain change in environmental quality.

## 3.5 Basic premises of shadow price calculation

In this section we discuss the basic premises adopted in developing our sets of shadow prices and weighting factors. Chapters 5 and 6 deal with the specific premises underlying calculations of values and weighting factors for, respectively, abatement and damage costs and propose concrete values for these costs and the associated weighting factors.

### 3.5.1 Temporal and geographical scope of emissions

Shadow prices have been calculated for emissions originating within the Netherlands' territorial borders (excluding the Antilles) for the year 2008[25].

In the case of abatement costs this means that shadow prices do not (necessarily) cover all the emissions of residents of the Netherlands, some of which occur abroad.[26] This is because the vast majority of policy targets relate to emissions occurring on Dutch territory.

In the case of damage costs there is a need to make some kind of assumption regarding the damage occurring *outside* Dutch territory. This damage can be estimated using dispersion models and dose-effect relationships (cf. Chapter 5). In calculating shadow prices we have taken as our point of departure that the (unit) *value* assigned to that damage is the same as for the Dutch situation. In other words: the damage inflicted on people in the Netherlands is valued precisely the same as the damage inflicted on those living in, say, Germany or Poland. This assumption will be discussed in Chapter 5.

### 3.5.2 Average values for the Netherlands

The shadow prices provided in this report are average values for the Netherlands. Depending on the particular context, actual shadow prices may deviate from this average. In the case of abatement costs this may hold if additional local policies are in place to keep air quality within statutory limits, for example. In the case of damage costs, actual values will depend on issues like population density and the location and precise nature of the emissions concerned. As an illustration: ground-level emissions of transport particulates are generally more damaging than particulates from industrial stacks. In Chapter 7 we take a closer look at the potential use of our set of shadow prices for situations deviating from the Dutch average.

### 3.5.3 Prices

The shadow prices presented in this handbook (in Euros) are at 2008 price levels. The prices cited in some of the literature and models consulted are in 2000 or 2005 prices. In such cases we have adjusted them to 2008 levels using the consumer price index of the European Central Bank for the Euro zone (for data and justification, see Annex A).

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[25] Some estimates of abatement costs are based on data for 2007, which were assumed to remain unchanged for the year 2008.

[26] An indicator like GDP is applicable to residents of the Netherlands.

Shadow prices can be viewed as prices for emissions exclusive of financial transfers like taxes and subsidies. In the case of damage costs this implies that the shadow costs are not necessarily equal to the external costs. After all, some of the damage induced has already been internalised by way of charges or participation in an emissions trading scheme. The issue of interpreting shadow prices in terms of external costs is discussed in Chapter 7.

### 3.5.4 Characterisation factors

The choice of characterisation factors was one of the most complex issues in the present study. In the first place it is important to obtain some kind of harmonisation between the midpoint and endpoint factors in order to facilitate calculations between the two levels, which may be important for developing weighting factors and determining a larger set of shadow prices than is feasible on the basis of data analysis. Secondly, this study presents shadow prices for the Netherlands, implying that characterisation factors should be drawn up as far as possible for the Dutch situation. This is particularly relevant for endpoint factors, where the precise impact of emissions is co-determined by country-specific variables like weather and population density.

Given the set of characterisation factors available in the ReCiPe project, we opted to align ourselves with these, because in that project the relationship between midpoints and endpoints has been built into a consistent framework. There is a downside, though, as these characterisation factors are not keyed specifically to the Dutch situation.

### 3.5.5 The category ‘abiotic resource depletion’

Many LCA studies include the damage category ‘abiotic resource depletion’ to cover the risk that humanity will run out of resources for use by future generations. In this study no damage costs have been estimated for this category.

ReCiPe (Goedkoop et al., 2009) included the endpoint category ‘availability of resources’. They chose to look at the geological distribution of mineral and fossil resources and assess how the use of these resources causes marginal changes in efforts to extract future resources.

However, in conventional economics the depletion of resources is not considered a real or technical externality, but rather a pecuniary externality. Pecuniary externalities operate through prices. For example, if person A buys masses of cheese, the price of cheese will rise, which negatively affects person B who wishes to buy this cheese. This is part of the efficient operation of markets, though, and is not therefore deemed an externality. Pecuniary externalities consequently do not affect the efficiency of economic systems, although they do affect the distribution of well-being. Hotelling’s theory states that the depletion of non-renewable resources is included in the prices of these resources and that the costs of depletion have therefore been internalised.

Opponents of this view cite the burden placed on future generations by current resource use. Similarly, several ecological economists consider over-consumption of non-renewable resources to be unfair to future generations. They argue that putting a price on irreplaceable natural resources is like auctioning the Mona Lisa to a very small group: the price would be too low, since other parties, including people living in the future, cannot bid. A second argument is made by ExternE. Under the assumption that current interest rates are higher than the social preference rate that should be used for social issues, external costs can indeed be attached to the depletion of abiotic resources. A third argument, provided by Cleveland (1991), is that the extraction of non-renewable resources involves negative externalities like pollution of rivers and soils. Because of the steadily declining grades of remaining ores, moreover, such externalities are expected to increase in the future. Hence, extraction of such resources now (with attendant consumption of a portion of the limited stocks) is held to result in higher future external costs.

These views all depend on the idea that it is the stocks themselves that are limiting future human welfare and that wealth is created solely via resource markets. However, both propositions can be questioned. If resource rents are invested in, say, extraction technology, future generations may also benefit from technological advance. Studies on historical unit mineral prices show declining rather than rising trends, providing empirical evidence that scarcity is becoming less and less of a constraint on economic activity (Barnett and Morse, 1963). Simon (1981, p.46) has remarked that 'as economists or as consumers, we are interested in the particular services that resources yield, not in the resources themselves'. The value of these services can be represented by the price of the resources and these show declining long-term trends, both in extraction costs and in price. Hence, future generations may in fact well become better off through our rates of resource extraction.

In sum, although the depletion of abiotic resources might be considered unfair to future generations, in the case of well-functioning markets there will be no external costs involved. We have therefore assumed a zero figure for the damage costs of abiotic resource depletion. The abatement costs for this impact category have likewise been taken to be zero, because there are currently no policies addressing the finiteness of mineral resources.

# 4 Shadow prices according to abatement costs

## 4.1 Introduction

This chapter is concerned with the abatement costs calculated in the present study. First, in Section 4.2, the calculation procedure is described. Next, in Section 4.3, the resultant abatement costs are presented. In Section 4.4 we discuss the applicability of these costs to other regions and countries and for longer time horizons. The issue of uncertainty is also discussed. Precise calculation of the abatement costs for each theme is set out at length in Annex B.

## 4.2 The methodology used in this study

### 4.2.1 General valuation methodology

Shadow prices according to abatement costs are calculated with reference to the marginal costs of securing standing environmental policy targets. The shadow prices are determined, in other words, by the costs of the *most expensive* of the set of *least-cost* measures required to achieve the targets.

That the shadow price by definition indicates the *minimum* social costs of additional emission reduction required to meet the policy target means that policy inefficiencies are ignored. After all, the costs of the environmental policy will generally prove higher in actual practice, because the government does not allocate the available 'emissions space' across the target groups in a cost-optimum manner, for example, basing itself instead to an extent on arguments of equity, competitiveness or other such considerations. Firms may also opt not to pay pollution charges, preferring instead to implement costlier waste prevention measures because of marketing considerations. These are not implicit costs of environmental policy, however, but implicit costs of other kinds of policy like market, industry or income policy. For this reason, in our view these costs should not be allocated to the abatement costs.[27]

As argued in Chapter 3, national abatement costs can be calculated with reference to:

- The economic instruments rolled out for policy targets not involving a national emissions cap.
- National policy targets pursued via some form of environmental policy package.

[27] This is obviously debatable. One member of the Committee of Experts considered this an arbitrary assumption. In this context we recognise that environmental policy *influences* industry policy, for example, and that corrective measures are therefore required. Whether these should be allocated to industry policy or environmental policy depends on which policy target is judged to be primary, and which subsidiary. Because we are here concerned with elaborating shadow prices *for the environment*, in the present study we take environmental policy as primary, but without wishing to imply any inherent priority of environmental policy over other policy areas.

In the second of these approaches, abatement costs are determined from the marginal costs of the technical and organisational measures employed to secure the stated targets. One problem here is that such measures often lead to simultaneous cuts in several different pollutants. Allocation of these joint costs will always be a somewhat arbitrary affair (cf. Section 3.2.2). The estimates presented in this study are based on background studies by ECN and PBL, with priority being given to climate policy (cf. Annex B). In addition, the results have been compared with the cost curves drawn up by interdepartmental working groups as a follow-up to the Options Document (ECN/MNP, 2006). In allocating joint costs we have assumed top priority for climate policy, as stated, with other policy targets only being set after finalisation of that policy. Given the ongoing debates in Europe about future policy on acidification, this would seem a reasonable reflection of policy practice.

### 4.2.2 Priority pollutants and targets

We have estimated the abatement costs for eleven priority pollutants, each related to specific environmental theme(s) (see Section 4.3.2). In the case of climate change, for example, the abatement costs for $CO_2$ have been calculated. Any costs associated with the other greenhouse gases are thus not included – under the assumption that these are *lower* than in the case of $CO_2$ emissions, so that the costs of reducing the latter indeed represent the marginal costs of securing the political targets.

Most policy targets are keyed to a particular date in the future like 2010 or 2020 and the required emission cuts are hence subject to future trends in emissions growth. To determine the marginal costs associated with a given target, scenarios must therefore be used that estimate future emissions in the absence of policy measures. Although such 'autonomous' emissions paths are widely available, due care must be taken that no policy plans have already been included in these scenarios.

The abatement costs for pollutants for which a national emission cap is in place (e.g. $CO_2$, $NO_x$, $SO_2$, $NH_3$ and NMVOC) have been calculated using the cost curves in the Options Document (ECN/MNP, 2006).[28] These curves include options for emission reduction in several economic sectors. To what extent the measures underlying the cost curves are applicable depends largely on estimated economic developments. For example, if the aluminum industry is anticipated to reduce its output by 2020 for reasons of competitiveness, the cost effectiveness of abatement measures in this particular industry is likely to decrease. We have therefore also relied partly on the estimates of marginal costs provided in the cited ECN/MNP literature and used these to make an educated guess of the total costs involved and the subdivision of these costs over the various pollutants.[29]

The political targets and background scenarios used in our calculations are listed in Table 6. These targets are justified in Annex B, along with the underlying calculations.

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28 Including later revisions of these curves elaborated by interdepartemental working groups.

29 In our own estimates we allocated any joint costs based on an iterative procedure in which priority was given to climate change policies over other policy measures.

Table 6 Background scenarios and policy targets adopted for the various pollutants

| Pollutant/ impact | Scenario, autonomous trends | Policy target |
|---|---|---|
| $CO_2$ | GE scenario* | 20% emission reduction in 2020 |
| CFC-11 | - | Waste disposal fee as per Decree on Disposal of White and Broen Goods |
| $NO_x$ | GE scenario | Anticipated National Emissions Ceiling (NEC) target for 2020 (186 kt) |
| $SO_2$ | GE scenario | Anticipated NEC target for 2020 (35 kt) |
| $NH_3$ | GE scenario | Anticipated NEC target for 2020 (119 kt) |
| NMVOC | GE scenario | Anticipated NEC target for 2020 (143 kt) |
| $PO_4$ | - | Administrative fine for exceeding phosphate 'usage norm' as per Fertiliser Act |
| N | - | Administrative fine for exceeding nitrogen 'usage norm' as per Fertiliser Act |
| $PM_{10}$ and $PM_{2,5}$ | - | EU directives regarding concentrations<br>Cost effectiveness criterion in National Emission Guidelines for Air |
| Final waste | - | Cost effectiveness criterion in National Waste Management Plan: 150% of landfill charges |
| dB rail >55 | - | Noise control policy (Euro per dB-dwelling) |
| dB road >50 | | |
| dB aircr. >45 | | |

* Global Economy (GE) is a frequently used scenario that relies heavily on market forces and international trade liberalisation. It includes relatively high estimates of future economic growth, but also anticipates large-scale environmental problems.

## 4.3 The set of shadow prices and weighting factors based on abatement costs

### 4.3.1 Methodological description, per environmental theme

Proceeding from the government policies cited in Table 6, for each of the pollutants a shadow price was estimated, making use of a wide range of literature sources. The respective procedures are set out and justified in Annex B. Below, we report the main elements for each environmental theme and the corresponding pollutants.

#### Climate change

In 2007 the government's climate change programme 'Clean and Efficient' (Werkprogramma Schoon en Zuinig; VROM, 2007a) was reviewed by ECN, supported by MNP. To make dependence on European policy explicit, two scenarios were thereby used: 'EU high' and 'EU low'. In the first, the emission reduction target is 30% and the $CO_2$ price € 50/t (2007 prices) in 2020. In the second these figures are 20% and € 20/t. ECN consequently assumes that domestic emissions will indeed be reduced to a cost figure of € 20 and € 50 per $tCO_2$, respectively. As a central value for emissions occurring in the year 2008 we have taken a value of € 25/$tCO_2$, based on the projections of the NEEDS project (NEEDS, 2008). If the more stringent target of 30% emissions reduction is adopted, the shadow price should be adjusted to € 50/$tCO_2$.

### Ozone depletion

To a very large extent, emissions of ozone-depleting substances are now prohibited altogether. In the case of exemptions, we have derived an approximate figure for shadow prices from the costs of processing under the collection scheme in place for old stocks of CFCs and halons. The waste disposal fee for 'standard' large white goods is € 5, compared with € 17 for fridges and freezers. This difference of € 12 per fridge/freezer can be allocated exclusively to the CFC emissions avoided: around 0.4 kg (H)CFC per unit. As CFC-11 is the main focus of the (H)CFC recovery scheme, we have adopted a shadow price of € 30/kg CFC-11.

### Acidification

The shadow prices for the individual pollutants $NO_x$, $SO_2$ and $NH_3$ , which all have an impact on acidification, are based on the anticipated NEC targets for 2020 and marginal cost figures from various sources (ECN/MNP, 2006 and IIASA data). These pollutants impact on other environmental themes, too. In our own calculations, joint costs were broken down by introducing the assumption of top priority being given to climate policy and all climate-related policy measures being allocated to the theme of climate change.

### Photo-oxidant formation

For NMVOC a National Emission Ceiling of 143 kt is anticipated for 2020. The measures set out in the Options Document will not be sufficient to achieve this target and additional policy will be required. MNP has estimated the cost effectiveness of these additional measures to be around € 5/kg NMVOC (MNP, 2007a).[30] In addition, the Netherlands Emission Guidelines for Air (NeR) cites an indicative reference value of € 4.60/kg VOC emission for the cost effectiveness of abatement measures (SenterNovem, 2009).[31] We therefore estimate marginal abatement costs of around € 5/kg NMVOC. This is substantially higher than the figure of € 0.90/kg NMVOC cited in the 2002 update (CE, 2002a). The difference stems from the NEC target for 2020 (143 kt), which is more stringent than that in the Fourth National Environmental Policy Programme, 2010 (163 kt).

### Eutrophication

With its production of manure and use of artificial fertilisers, it is the agricultural sector that is mainly responsible for eutrophication. The costs that need to be incurred by this sector are therefore a good measure of the shadow price of eutrophication. The Fertiliser Act (Meststoffenwet) lays down administrative fines for exceeding the phosphate and nitrogen 'usage norms'. Based on these charges, we have set the shadow prices for phosphate ($PO_4$) and nitrogen (N) at € 11/kg and € 7/kg, respectively. These values are higher than those in the 2002 update (CE, 2002a), because the charges have since been increased.

### PM formation

Although there are no *emission* targets in place for $PM_{10}$ and $PM_{2.5}$, European air quality standards were incorporated into Dutch legislation in 2007 in the form of limit values for airborne *concentrations*. In the NeR (SenterNovem, 2009) an outdated figure is cited as cost effectiveness threshold, with no obligation to implement measures costing over € 2.30/kg avoided particulates. Although a variety of costlier measures to reduce $PM_{10}$ emissions have been

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30 Based on Figure 3.1 (p.35): a reduction of approx. 20 kt NMVOC for approx. € 100 million.

31 NB: Although these values are given in the most recent Guidelines (SenterNovem, 2009, with respect to Section 2.11 unchanged since December 2006), they date from the mid-1990s.

reviewed in several studies, it is hard to relate them to specific concentration targets. We have therefore based ourselves on the NeR figure[32]. In addition, $NO_x$ and $SO_2$ also play a part in $PM_{10}$ formation. If $PM_{10}$ were to be assigned a (far) higher value than € 2.30/kg, $NO_x$ and $SO_2$ would have a share in $PM_{10}$ formation that is irreconcilable with the shadow prices for these pollutants based on their respective targets and marginal abatement costs. As an indication, though, in the tables we report an alternative figure of € 50/kg for abatement costs. This value is based on the estimated cost on the cost effectiveness of measures available to industry. The observed costs of emission reduction in transport exceed € 50/kg, while abatement costs in agriculture appear to be somewhat lower (cf. Annex B.4.2).

### Human toxicity
Although there are no national emission caps in force for toxic chemicals, maximum permissible concentrations (MPC) have been set. Shadow prices for such pollutants were therefore indexed to $PM_{10}$ using MPC ratios, taking only the figure of € 2.30/kg $PM_{10}$ just cited.

### Final waste
Final waste is not an impact category used in Life Cycle Assessment. However, because it constitutes a separate policy theme for the Dutch government, involving dedicated policies, in the previous update (CE, 2002a) a shadow price for final waste was reported. In the present study an update for this value is also provided. In the treatment of the weighting factors in Chapter 6 the theme of final waste is not discussed, though.

Since 1 January, 1996 there has been a ban on land filling waste that can be incinerated or recycled. The material ultimately remaining after the various potential processing steps, which is not amenable to further incineration or recycled, is termed ‘final waste’. For this waste the latest draft National Waste Management Plan (Landelijk afvalbeheerplan; VROM, 2008a) lays down ‘useful application’ as a minimum standard, unless the additional costs exceed 150% of the landfill tariff, which in 2007 was around € 120/t (including the associated environmental tax). From this we derive a figure of € 180/t for the marginal abatement costs for final waste.

### Noise
Noise nuisance is caused by industry, road, rail and air traffic and various other sources. To prevent and control it, the Noise Nuisance Act (Wet geluidshinder) has been in place since the late 1970s. It provides a legal basis for noise policy, by setting maximum allowed noise levels for roadsides and industrial areas. The Ministry of Transport, 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 the threshold of € 3,000 per dB-dwelling reduction, the measure is deemed efficient. We have taken this threshold level as the shadow price.

[32] We anticipate the national $PM_{2.5}$ target being secured. $PM_{10}$ targets are supposed to be more binding.

It would be preferable, though, to have the shadow price depend on the degree of nuisance and the type of noise involved. Since people experience the noise of various traffic modalities differently[33], we recommend applying the following thresholds:
– 50 dB for road traffic noise.
– 55 dB for railway noise. And
– 45 dB for aircraft noise.

### 4.3.2 Results per pollutant (valuation)

Table 7 summarises the abatement costs calculated for each of the priority pollutants. It also shows the environmental themes associated with each of these pollutants.

Table 7 **Abatement costs per priority pollutant (€$_{2008}$/kg; central value in brackets)**

| Pollutant | Shadow price (*central value)* | Theme |
|---|---|---|
| $CO_2$ | 0.02-0.05 *(0.025)* | Climate change |
| CFC-11 | 30 | Ozone depletion |
| $NO_x$ | 5-10 *(9)* | Acidification, photo-oxidant formation (smog), PM formation |
| $SO_2$ | 5-10 *(5)* | Acidification, photo-oxidant formation, PM formation |
| $NH_3$ | 4 | Acidification, PM formation |
| NMVOC | 5 | Photo-oxidant formation |
| $PO_4$ | 11 | Freshwater eutrophication (eutrophication) |
| N | 7 | Marine eutrophication (eutrophication) |
| $PM_{10}$* | 2.30-50 | Human toxicity |
| Final waste | 0.18 | Final waste |
| dB rail >55 | 3,000 | Noise (€ per dB-dwelling) |
| dB road >50 | | |
| dB aircraft >45 | | |

* For $PM_{10}$ the precise figure is as yet unclear. Based on existing (outdated) policy, the valuation is € 2.30/kg. New policy is currently under development, though, which may well lead to an increase in abatement costs to € 50/kg. See Annex B.4.

The central values have been calculated on the basis of characterisation factors reflecting the relationship between the pollutants (cf. Annex D) and the analysis in Annex B.

### 4.3.3 Extending the number of values at pollutant level

The set of abatement costs presented above is limited to eleven pollutants. The list can be extended, though, by using the characterisation factors employed in environmental impact studies (cf. Sections 2.4 and 2.5). By taking the environmental relationship between the pollutants as the point of departure, there also arises an *implicit* valuation for pollutants associated with the same environmental impacts but for which no shadow price has been calculated. In LCAs and other types of environmental analysis the climate change impact of 1 kg $CH_4$ is taken equal to that of 25 kg $CO_2$, for instance (based on Goedkoop *et al.*, 2009), which means the shadow price for $CH_4$ works out at € 0.625/kg if the central value of € 0.025/kg is taken for $CO_2$.

[33] At a given noise level people tend to experience the greatest nuisance from aircraft noise and least from railway noise (see Annex B).

Using the ReCiPe midpoint characterisation factors a more extensive list of *implicit* abatement costs can then be calculated.

Behind this extended list of implicit abatement costs lies the assumption that government policy is 'efficiently' designed in both economic and environmental terms. In the above example this means the government, in drawing up its policies, sets the marginal costs of reducing the contribution of $CH_4$ emissions to climate change equal to the marginal costs of reducing $CO_2$ emissions. In practice this is obviously not (necessarily) the case, though. There are strong indications, for example, that policy on other (non-$CO_2$) greenhouse gases is currently *de facto* cheaper than policy addressing $CO_2$ (CE, 2005). For this reason, the actual abatement costs for $CH_4$ should be *lower* than the implicit abatement costs. In the present study no attempt has been made to calculate the actual abatement costs for each pollutant.

In Chapter 6 there is a more extensive discussion of characterisation and the problems associated with allocating emissions across the various themes. Table 8 reviews the pollutants most frequently encountered in environmental impact analyses. Annex J provides a full list of implicit shadow prices for all the 400 pollutants considered.

**Table 8 Shadow prices of emissions in the Netherlands in 2008 based on abatement costs (€$_{2008}$/kg pollutant)**

| Pollutant | Total | Climate change | Ozone depletion | Acidification | Photo-ox. formation | Eutrophication | PM formation | Human tox., air | Human tox., water |
|---|---|---|---|---|---|---|---|---|---|
| $CO_2$ | 0.0250 | 0.0250 | | | | | | | |
| $CH_4$ | 0.6250 | 0.6250 | | | | | | | |
| $N_20$ | 7.45 | 7.45 | | | | | | | |
| CFC-11 | 149 | 119 | 30.0 | | | | | | |
| CFC-12 | 303 | 273 | 30.0 | | | | | | |
| CFC-113 | 183 | 153 | 30.0 | | | | | | |
| CFC-114 | 278 | 250 | 28.2 | | | | | | |
| CFC-115 | 197 | 184 | 13.2 | | | | | | |
| HCFC-22 | 46.8 | 45.3 | 1.50 | | | | | | |
| $NO_x$ | 8.72 | | | 2.32 | 5.00 | 0.896 | 0.506 | | |
| $SO_2$ | 5.00 | | | 4.13 | 0.406 | | 0.460 | | |
| $NH_3$ | 11.7 | | | 10.1 | | 0.784 | 0.736 | | |
| NMVOC | 5.00 | | | | 5.00 | | | | |
| $PO_4$ | 11.0 | | | | | 11.0 | | | |
| P to water | 10.9 | | | | | 10.9 | | | |
| P to soil (fertiliser) | 0.577 | | | | | 0.577 | | | |
| P to soil (manure) | 0.545 | | | | | 0.545 | | | |
| $NO_3$ to water | 7.14 | | | | | 7.14 | | | |
| $NO_3$ to air | 0.896 | | | | | 0.896 | | | |
| N to water | 7.00 | | | | | 7.00 | | | |
| N to soil/air (manure) | 0.553 | | | | | 0.553 | | | |
| N to soil/air (fertiliser) | 0.511 | | | | | 0.511 | | | |

| Pollutant | Total | Climate change | Ozone depletion | Acidification | Photo-ox. formation | Eutrophication | PM formation | Human tox., air | Human tox., water |
|---|---|---|---|---|---|---|---|---|---|
| $PM_{10}$ * | 2.30 (50) | | | | | | 2.3 (50) | | |
| $PM_{2.5}$* | 2.30 (50) | | | | | | 2.3 (50) | | |
| CO | 0.009 | | | | | | | 0.009 | |
| Benzo(a)pyrene | 92,000 | | | | | | | 92,000 | |
| Dioxins | 92,000,000 | | | | | | | 92,000,000 | |
| As (arsenic) | 466 | | | | | | | 184 | 282 |
| Cd (cadmium) | 4,700 | | | | | | | 184 | 4,520 |
| Co (cobalt) | 3,370 | | | | | | | 460 | 2,910 |
| Cr (chromium) | 36,900 | | | | | | | 36,800 | 108 |
| Hg (mercury) | 8,140 | | | | | | | 613 | 7530 |
| Ni (nickel) | 1800 | | | | | | | 368 | 1,430 |
| Pb (lead) | 225 | | | | | | | 184 | 41.0 |
| Zn (zinc) | 227 | | | | | | | 0.920 | 226 |
| Fluoride | 1,840 | | | | | | | 1,840 | 0.21 |
| Final waste | 0.180 | | | | | | | | |
| Noise (dB-dwelling over threshold**) | 3,000 | | | | | | | | |

* For $PM_{10}$ the precise figure is as yet unclear. Based on existing (outdated) policy, the valuation is € 2.30/kg. New policy is currently under development, though, which may well lead to an increase in abatement costs to € 50/kg. See Annex B.4. As the policy targets for $PM_{10}$ appear to be more stringent than for $PM_{2.5}$ the figure for $PM_{10}$ has been taken as determining.

** Threshold levels are 50 dB for road traffic noise, 55 dB for railway noise and 45 dB for aircraft noise.

As can be seen in Table 4, for a number of pollutants the total figure is given by the sum of values for several different environmental themes. This is particularly the case with the acidifying pollutants $NO_x$, $SO_2$ and $NH_3$, which also contribute to photo-oxidant formation, eutrophication and PM formation. The first issue to be tackled here is the problem of *joint costs*, discussed in Sections 3.2.2 and 4.2.1. The allocation of joint costs will always be a somewhat arbitrary affair. The estimates used in the present study are based on background studies by ECN and PBL, which give top priority to climate policy (cf. Annex B). The next problem is how the marginal costs of individual pollutants are to be allocated across the environmental themes. On this point a pragmatic approach has been adopted in which the marginal abatement costs calculated for the individual priority pollutants were allocated across the various themes in such a way that the sum total of the individual components in turn also generates realistic results, once uncertainty margins have been allowed for. The shadow price for the priority pollutant $NO_x$, for example, is between € 5/kg and € 10/kg. Allocating € 10 to the theme of acidification is implausible, though, because via the characterisation factors this would lead to an unrealistic value of € 18/kg for $SO_2$. This would be irreconcilable with the abatement costs for $SO_2$, which are also in the range € 5-10/kg. The method is explained at greater length in Annex D.

## 4.4 Temporal and spatial variation

### 4.4.1 Abatement costs for other countries or local circumstances

The shadow price of an emission is given by the intersection of the marginal abatement cost curve (MACC) and the policy target, often depicted as a vertical line. Policy targets are determined in the political process on the basis of societal preferences in the country concerned. Such preferences may obviously vary. Not only may there be fundamental differences in perceptions of risk. Additionally, different policy targets may be set as a result of differences in available funding, owing to differences in per capita income, for instance. Finally, abatement costs as well as direct damage costs may also vary from region to region because of differences in local circumstances such as population density or landscape type. This means that shadow prices calculated according to the abatement cost method are only valid in the region to which the policy targets apply.

Within the Netherlands, too, the same considerations hold for a number of themes where local limits are in force for certain pollutants. This means any increase in ambient levels of these pollutants arising through development projects must be locally compensated. To the extent that the costs of local measures are higher than the national average, or the limits represent a more stringent target than national policy, the abatement costs will be an underestimate of the actual costs incurred at the local level.

### 4.4.2 Temporal changes in abatement costs

In this study, marginal abatement costs have been calculated for the year 2008.[34] The shadow price of a number of emissions is given by the intersection of the MACC and the policy target concerned, both of which may change over time.

In assessing the 'shelf life' of the shadow prices, due allowance should be made for four developments:

a *Economic trends may pan out differently*
First of all, the MACC might change due to a change in emissions. If emissions rise due to growth of economic output, for example, the marginal costs of securing policy targets will likewise rise. If emissions fall, as in the present credit crisis, expectations are that it may become cheaper to meet emissions targets if the crisis leads to a structural downturn in economic growth.

b *Technological advance*
Because of economies of scale and learning curves, over time abatement technologies become more efficient and/or cheaper. If policy targets are formulated for years in the relatively distant future, costs should be corrected for such technological advance. Virtually all MACCs use *ex-ante* cost information. Some studies have pointed to the divergence between *ex-post* and *ex-ante* cost-effectiveness analysis (CEA). *Ex-ante* CEA tends to overestimate the costs to a certain extent. For cases where both types of CEA were available, the international literature reports a factor 2 to 5 difference between the two (IVM, 2006; Harrington, 2000; Burtraw, 1996; SEI, 1999). The main reasons are that learning effects and scale effects tend to be underestimated *ex ante* and that some cost studies are conducted for strategic reasons, for example to obstruct more stringent environmental policies. In many such cases, simpler means were subsequently found to meet targets that were not examined during the studies. In the US, for example, $SO_2$ regulation has resulted mainly in

34 Any values from 2007 have been taken to remain unchanged in 2008.

cleaner coal being purchased, rather than in implementation of the numerous technical measures identified in earlier cost estimates.

c *Trends in energy prices*
Energy prices are sometimes a major factor in the projected cost of abatement measures. High energy prices mean the MACC bends down, because energy savings lead to lower costs. This only applies to the shadow price of $CO_2$, though.

d *Revision of policy targets*
Over time, policy targets may be revised as society's preferences change, due for example to greater concern about a particular environmental problem or new scientific understanding regarding the toxicity or hazard of particular emissions. Changes in the MACC may also lead to policy targets being revised.

In summary, marginal abatement costs will change over time. This is above all problematical if the derived shadow prices are used for valuation purposes. Depending on how each of the four cited factors plays out, the assigned values would need to be revised. For weighting this may be a less relevant issue, though, because there will be less change in the mutual relationship between the themes due to their being influenced to some extent by the same factors. In actual policy practice there are feedback mechanisms, moreover. If the costs of emissions reduction prove higher than projected, policy targets can be adjusted downwards, and vice versa. Because of these feedback mechanisms, the marginal abatement costs are generally more robust than one would expect on the basis of one single component element like technological advance.

### 4.4.3 Uncertainty

Given the above, shadow prices are characterised by a certain degree of uncertainty, particularly if one seeks to project such prices into the future. The factors determining these prices are themselves to a certain extent unpredictable, as in the cases of political, international willingness to lay down tighter (or laxer) targets, or technological advance. It is important to note that in this respect shadow prices differ little from market goods. The prices of raw materials like oil are equally dependent on political, economic and technological developments. This uncertainty obviously increases the further one seeks to peer into the future.

# 5 Shadow prices according to damage costs

## 5.1 Introduction

Shadow prices for environmental quality can, alternatively, be determined using data on the damages resulting from pollution. This chapter presents the shadow prices obtained in this way. Damage costs have been estimated directly from the endpoints. All damage costs presented in the subsequent sections and in Annex C are expressed as the value of the damages due to emissions of the specific substances in the Netherlands (or the EU-27, as indicated) discounted to the year 2008 and in 2008 prices.[35] The damage costs include all measurable negative effects that can be attributed to environmental pollution. These may or may not be similar to the concept of external costs, an issue that is addressed not here, but in Chapter 7 (Section 7.3.1).[36]

These negative effects include direct impacts only, even though there may also be indirect impacts. Productivity losses in agriculture, for example, may lead to starvation. However, these indirect effects are not included, *except* in the case of greenhouse gas emissions, where Integrated Assessment Models (cf. Annex C.2.1) generally do include such indirect impacts.

The structure of this section is as follows. First, in Section 5.2, the general methodology is outlined and various methodological choices are introduced. Next, in Section 5.3, the Impact Pathway Approach is presented and in Section 5.4 an alternative implicit valuation is described using the endpoint characterisation factors from the ReCiPe study. Section 5.5 summarises our estimates of damage costs for a set of specific pollutants in the Netherlands. Extensive descriptions of how we arrived at each cost figure are provided in Annex C. In Section 5.6, temporal and spatial variation of the estimates are discussed and estimates of damage costs for the EU-27 given. Section 5.7 deals with uncertainty and in Section 5.8, finally, the figures obtained in this study are compared with those of other studies.

The reader should note that this chapter deals only with the methodology adopted in this study for estimating damage costs. Issues such as whether the resultant cost figures adequately reflect the Willingness to Pay (WTP), the categorisation of values of environmental goods, estimation methods and ethical considerations have already been addressed in Chapter 3 (Sections 3.3 and 3.4, in particular). The use of these damage costs in cost-benefit analysis and other applications is also not addressed in this chapter, but will be discussed in Chapter 7.

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35 Additionally, valuation of non-EU impacts of emissions have been assumed the same as EU impacts. Discounting is sometimes done explicitly and sometimes implicitly. An example of implicit discounting is when the discounting can already be included in the value of WTP. In the NEEDS project, WTP was derived from the CVM question, which was formulated more or less as follows: How much would you be willing to pay for extending your life by 6 (and in the other versions, by 3) months? As people are assumed to express their present value for this, an implicit discounting is already included in the value of their life expectancy.

36 Only in the case of land use changes do we refer to external costs, because these estimates have been derived on this basis and are not necessarily related to physical damages.

## 5.2 The methodology used in this study

Within the damage cost method, three approaches can be distinguished for establishing the relationship between pollution and the value of the damages to which it gives rise:
- Direct investigations using revealed or stated preferences.
- Estimation of impacts at the level of endpoints, using the Impact Pathway Approach.
- Estimation of impacts at the level of endpoints, using endpoint characterisation factors.

As we have used all these approaches in the present study, they will now each be described.

### 5.2.1 Approach 1: Direct investigations using revealed or stated preferences

For certain environmental problems, the relationship between environmental disturbance and monetary value can be directly investigated. This is especially true of tangible assets affected by local environmental problems. People can be asked to value a specific protected area, for example. Alternatively, property prices or recreational travel times can be used as a 'revealed preference' for assigning a value to the protected area. Most estimates for the environmental theme of noise are obtained using this method. By observing the differential in property prices between noisy and quiet areas an implicit value can be derived for various types of noise.

For more general environmental themes, as with the impact of specific pollutants on various endpoints, this approach is more problematical, however. We cannot ask people how they would value a kg of $SO_2$ as they do not clearly understand the relationship between $SO_2$ and their demand for environmental quality. In such cases it is necessary to define certain functions relating the impact of specific pollutants to specific endpoints. Direct investigations on valuation of endpoints (as in Approach 1) can be then used as inputs for Approach 2.

### 5.2.2 Approach 2: Estimation of endpoints using the Impact Pathway Approach

Damage costs can also be estimated for so-called endpoints: the ultimate impact of emissions on entities valued by human society (cf. Section 2.4.1). In this study four such endpoints have been distinguished[37]:

a Impacts on human health (premature death and illness).
b Impacts on productivity of ecosystems.
c Impacts on materials and buildings.
d Impacts on ecosystems resilience (including biodiversity).

One of the problems with valuing impacts on endpoints is that information is generally lacking on the precise impact of a given project or policy plan on such endpoints. In most cases, however, an environmental impact assessment (EIA) will provide information on the projected change in emissions due to project implementation. The question now is how to translate this information into impacts on endpoints and subsequently into monetary estimates.

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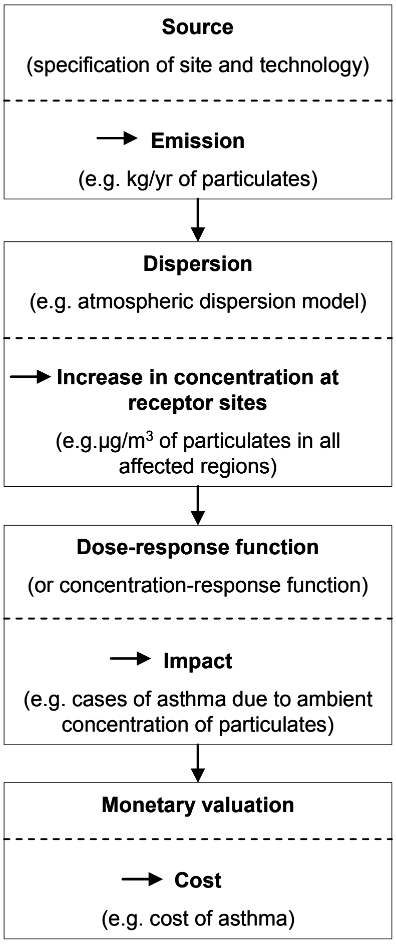
[37] This effectively implies that the endpoints odour, visibility, cultural heritage and visual aesthetics have been ignored. In Annex C some of these endpoints will be described and estimates for these from the literature will be given. However, these impacts are not included in the shadow prices presented in this report.

To assess damage costs per unit of specific pollutants in monetary terms, an analysis method has been developed that is known as the Impact Pathway Approach (NEEDS, 2008a; see Figure 10).

The Impact Pathway Approach (IPA) has been used in several international research projects initiated by the European Commission, starting with the original ExternE study implemented in mid-1990s. Recent updates to the ExternE series include the NEEDS project. Another EC-funded project using the IPA is CASES. These projects have been designed to develop methodology and provide estimates of the externalities of energy conversion and transportation. The ExternE methodology aims to cover all relevant (i.e. non-negligible) externalities identified through the impact-pathway approach.

**Figure 10 The Impact Pathway Approach**




Source: Based on NEEDS, 2008a.

The various steps are now described.

### *Step 1: Source-Emissions*

This step identifies, within a geographical grid, all relevant emission sources. In the EcoSense model used in the final stages of the ExternE project, the emissions were taken from the EMEP (European Monitoring and Evaluation Programme) database with a spatial resolution of approximately 50 x 50 $km^2$.

### *Step 2: Dispersion-Receptor sites*
This step translates emissions into concentrations at specific, geographically diversified receptor points (sometimes called immissions). For classical air pollutants, dispersion and chemical transformation in Europe have been modeled using the EMEP/MSC-West Eulerian model, which also includes meteorological data. Source-receptor matrices have been derived which allowed a change in concentration or deposition to be attributed to each unit of emission and for each of the EMEP grid cells across Europe. Model runs have been performed for a 15% reduction of each airborne pollutant. Within the model, meteorological conditions are averaged across four representative meteorological years. For emissions in the years 2000-2014, dispersion results reflect the estimated background emissions in 2010. For other future years, the estimated background emissions modeled for 2020 were used. It should be noted that the chemical reactions and interactions are fairly complex. For example, a reduction of $NO_x$ emissions leaves more background $NH_3$ for reaction with background $SO_2$ than without $NO_x$ reduction. The reaction of additional free $NH_3$ with $SO_2$ increases the concentration of sulphates at certain locations (NEEDS, 2008).

### *Step 3: Dose-response functions and impacts*
This step establishes the relationship between pollution concentration and physical impacts at the endpoint level. With the aid of a so-called concentration-response function and with reference to the size of the exposed population, physical impacts have been calculated for each grid cell. Population density data were taken from SEDAC (2006).

Three types of physical impacts are described:
- Mortality: the risk of premature death due to reception of the pollutant. A distinction can be made here between acute mortality (immediate death) and chronic mortality (death occurring after a certain period of exposure to a given pollutant). Acute mortality may be the result of photo-oxidant formation (smog), for example, while chronic mortality is typically associated with emissions of particles (primary and secondary). For classical air pollutants, reduced life expectancy (YOLL, years of life lost) was found to be the most important endpoint.
- Morbidity: the risk of developing a disease due to reception of the pollutant. The following effects have been evaluated and factored in to our final calculations: restricted activity days, work loss days, hospital admissions and medication use.
- Potentially disappearing species: a measure of how pollutants impact on ecosystems and biodiversity.

The physical endpoints are described in more detail in Annex H.

For impacts on materials and productivity changes in environmental services (e.g. fisheries, forests, crop losses), no physical impact is normally given, with estimates being directly transferred in monetary terms.

### *Step 4: Monetary valuation*
The final step is monetary valuation. Impacts on productivity changes are revealed directly via market prices. Impacts on materials are revealed by examining restoration costs. Impacts on human health and ecosystems cannot be directly observed via the market, however. These have therefore been estimated using various methods.

The monetary values recommended in ExternE for YOLL were derived from questionnaires. In the NEEDS project, VOLY was valued directly using CVM (i.e. a stated preferences method), asking people about their WTP for 3 or 6 months' longer life due to air quality improvement. The monetary values for diseases proposed by the economic expert group have been derived on the basis of informal meta-analysis and the most recent robust estimates (ExternE, 2005). Finally, impacts on ecosystems have been estimated using the results of a meta-analysis of studies related to valuation of biodiversity changes by Kuik et al. (2008). Valuation of these ecosystem impacts will be described in Section 5.3.3.

*Discussion of Impact Pathway Approach*

It should be noted that the full Impact Pathway Approach can be used only for those impacts for which it is possible to determine specific units of environmental impact, such as emission of specific pollutants in kilograms, and dose-response functions related to these units. The best example of an endpoint that can be modelled using the IPA is the impact of pollution on human health. If, according to epidemiological tests, an increased concentration of a specific pollutant leads to a certain increase of the number of cases of a certain disease (and if this disease shortens average human life expectancy by a given number of years), using medical statistics we can arrive at a number of years lost due to a disease which can be expressed in YOLLs or DALYs and then evaluated in monetary terms. However, devising dose-response models for endpoints like visual aesthetics or recreational value would be very hard. Although we can establish a relationship between the source of damage and a receptor (e.g. the shorter the distance to the source of visual intrusion, the higher the damage in terms of visual disturbance or loss of recreational amenities), we would lack a common unit for valuation.

For those endpoints where defining a unit of environmental impact is problematical and where valuation depends primarily on perception related to specific, local phenomena, a common practice in valuation is to use empirical studies of greatest relevance for the case being analysed (i.e. use Approach 1).

### 5.2.3 Approach 3: Estimation of endpoints using characterisation factors

For certain pollutants the IPA yields a reasonable estimate of damages at the endpoint level and is able to translate this back to the release of a certain pollutant to the atmosphere. However, the scope of the impact-pathway approach is limited to classical pollutants only. For other environmental themes, therefore, an alternative approach has been adopted in which we directly value the ReCiPe endpoints. This approach is probably less accurate, as ReCiPe uses only two endpoints (human health and biodiversity), albeit that these can be considered the most important ones.

However, due to differences in modelling approaches with respect to geographical scale, time and grid cell size, there are important differences between the modelling of emission-receptor pathways in NEEDS and ReCiPe. In Annex G these differences are highlighted. Owing to the nature of the underlying modelling results, we believe the NEEDS approach will yield more credible results for economic damage estimation in the present project (see also Annex G).[38] Consequently, Approach 2 is preferable to Approach 3. However, in cases where data for the former are unavailable we shall revert to the latter.

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[38] The ReCiPe approach yields results that have not been discounted, for example. Especially when impacts occur in the more distant future, there will be differences between the NEEDS and ReCiPe approaches.

## 5.3 Valuation in Approach 2 (Impact Pathway Approach)

The Impact Pathway Approach puts a value on damages per unit emission of specific pollutants. The relationship between pollutant load and monetary value is sometimes established directly and sometimes by way of physical indicators. The following impacts have been determined using the IPA:
- Premature death (chronic and acute mortality); see Section 5.3.1.
- Illness (morbidity); see Section 5.3.2.
- Impacts on biodiversity; see Section 5.3.3.
- Impacts on crops; see Section 5.3.4.
- Impacts on buildings; see Section 5.3.5.
- Other impacts; see Section 5.3.6.

In the IPA we have to use certain indicators for estimating each of these impact categories. By and large, we have adopted the values from the NEEDS study for this purpose. For a proper understanding of the procedure, a brief explanation of physical indicators is in order. The two indicators that are relevant here are Years of Life Lost (YOLL) and Potentially Disappeared Fraction (PDF) of species (cf. Box 5 and Annex H).

**Box 5. Physical endpoint indicators**

The two indicators most commonly used in the Impact Pathway Approach are YOLL and PDF.

YOLL (Years of life lost) is a widely used indicator of premature death in a given population. Ozone, photo-oxidant formation (smog), toxic substances and ionising radiation may all result in premature death. YOLL, an indicator developed in the health sciences, corresponds to the number of deaths multiplied by the standard life expectancy at the age at which death occurs.

The PDF (potentially disappeared fraction) of selected species is an indicator of biodiversity loss. For a given land use type, a certain 'standard' 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. In the NEEDS project the PDF approach was used for assessing the ecosystem impacts of acidification and eutrophication. The EcoSense model employed in that project for the modelling and valuation of different pollution impacts used PDF-values per unit deposition from a study by Kuik et al. (2008).

Annex H provides more information on YOLL and PDF as well as several other physical indicators.

Impact of non-classical pollutants within NEEDS has been assessed using the concept of DALYs. For a discussion of ambiguities between monetary evaluation of Years of Life Lost (YOLL) and Disability Adjusted Life Years (DALYs), see Section 5.4.2.

### 5.3.1 Valuating YOLL (premature deaths) through VOLY

Value of Life Year (VOLY) is a relatively new concept, for which the first empirical surveys were conducted in the 1990s. Until recently it has attracted little attention, and reliable estimates from surveys asking explicitly about valuation of life expectancy gain are scarce.

In the NEEDS project, VOLY was valued directly by means of CVM (i.e. a stated preference method), asking people about their WTP for a 3 or 6 months' longer life due to air quality improvement. This contrasts with the Value of Statistical Life (VSL) method, in which revealed preferences dominate the empirical basis (via wage differentials between high-risk and low-risk jobs, for example).

In NEEDS (2008c) it is argued that, at least in the context of air pollution, valuation of mortality using VOLY is superior to valuation with VSL for several reasons, including the following:
- Air pollution cannot be identified as the primary cause of an individual death, only as a contributing cause.
- VSL fails to take into account that the loss of life expectancy per death is very much less for air pollution-related deaths (around six months) than for the typical accidents (30-40 years) on which VSL calculations are based. In other words, air pollution impacts mainly on death at end-of-life, while accidents are often in mid-life.

The first international study using CVM to obtain a direct estimate of VOLY was conducted within the NewExt phase of the ExternE project series[39] between 2001 and 2003. The questionnaire was developed according to a format previously used in the USA and Canada and described in Krupnick et al. (2002) that focused on air pollution reduction benefits. It was used to conduct surveys in the UK, Italy and France. The valuation question was phrased in terms of reducing the risk of dying during the coming ten years. The hypothetical risk reduction would be achieved thanks to a medication which would have to be bought through annual payments. Based on the results of this survey, the NewExt team recommended a VOLY of € 50,000 (NEEDS, 2008c).

Based on the experience of NewExt, the decision was taken to develop a new questionnaire for the NEEDS[40] project, part of the ExternE series. The survey was implemented in 9 countries (UK, France, Poland, Czech Republic, Hungary, Germany, Switzerland, Spain, Denmark) on representative samples of the population in one major city in each country. The respondents answered valuation questions about implementing air pollution reduction policies that would result in an increase in life expectancy of (1) six months and (2) three months. Based on these empirical results, the NEEDS team recommended a mean VOLY of € 40,000 for the EU-25. As VOLY is highly correlated with income, the mean value calculated for the group of New Member States is lower than average (€ 33,000), and the mean value calculated from the sample for the EU-15 plus Switzerland is higher than average (€ 41,000) (NEEDS, 2008c).

[39] ExternE (External costs of Energy) is a series of research projects initiated by the European Commission aimed at estimating socio-environmental damages related to energy conversion. For more information, see Section 1.4.

[40] New Energy Externalities Developments for Sustainability, a European Commission research project implemented during the period 2004-2008.

**Box 6: The relationship between VOLY and VSL**

One of the methods used for comparing the costs and benefits of public policy measures which effectively save human life employs the concept of Value of Statistical Life (VSL). The standard approach is to place a monetary value on the life-saving benefits of specific (actual or hypothetical) measures by estimating Willingness to Pay for mortality risk reduction. As the specific people whose lives are saved by regulations cannot be identified, the concept of VSL refers to 'statistical lives'.

The economic value of a statistical life has been the subject of empirical studies for over 30 years now and the concept is part of generally accepted economic methodology. Although VOLY is considered a different concept from VSL, there is a certain correlation between the two. In theory, at least, VSL can be seen as a discounted sum of annual VOLYs. In its simplest form the relationship between VSL and VOLY is:

$$\text{VSL} = \text{VOLY}\left\{\frac{1}{(1+\delta)} + \frac{1}{(1+\delta)^2} + \ldots + \frac{1}{(1+\delta)^{L_{acc}}}\right\}$$

where $\delta$ is the discount rate and $L_{acc}$ the average life expectancy loss due to accidents, since VSL studies are based on accidental deaths. Such calculations typically yield a figure of between 20 and 40 for the ratio VSL/VOLY (NEEDS, 2008c). There are grounds for scepticism about the validity of this type of conversion, however. For example, a small advance in the time of death resulting from air pollution, usually in old age, may be viewed as very different from violent death in an accident, resulting in a major loss of life expectancy. Direct valuation studies (using CVM or related techniques) can therefore be deemed a more reliable method of eliciting the VOLY.

Within the NEEDS project two different values of Years of Life Lost were applied for mortality: (1) € 40,000 for chronic mortality (YOLLchronic), reflecting years of life lost due to chronic exposure that only becomes apparent after several years of exposure, and (2) € 60,000 for acute mortality (YOLLacute), for effects occurring in the same year as exposure.[41] The higher value for acute mortality is not based on direct studies on preferences but on discounting the average time span between dose and impact in the case of chronic mortality. Due to the CVM set-up, the value of € 40,000 includes implicit discounting by the surveyed.[42] For infant mortality NEEDS took a value of € 3 million, reflecting the notion that, according to several studies, the value of a statistical life for infants is perceived as higher than that for adults.

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41 It should be noted that these monetary values are adopted uniformly across the entire area covered within the NEEDS project. No differences in GDP per capita values are taken into account in valuing human mortality and morbidity, i.e. the same VOLY of € 40,000 is used for all the countries of the EU-15. This may be surprising; one might expect, for example, that the values for human mortality and morbidity based on stated preferences studies would be higher for the Netherlands than the average figure for the EU, as GDP per capita in the Netherlands is higher than the EU average. However the damages resulting from emissions in the Netherlands diffuse over a wider area and affect receptors beyond the country's borders (for some substances, like $NH_3$, this dispersion effect is far less pronounced than for others, like NMVOC). As the level of wealth in some of the countries affected by Dutch emissions is lower than in the Netherlands, underestimating the values in the Netherlands by applying the average VOLY is at least to some extent counterbalanced by overestimation in other areas. It may also be noted that for designing policies for the entire EU, using average EU values is politically justified (common European values and policies).

42 Respondents were asked about their WTP for 3 or 6 months' longer life due to air quality improvement. Since this happens at the end of their lives, people tend to implicitly discount these sums.

Within NEEDS, the VSL for infants is assumed to be twice that for adults.[43]

### 5.3.2 Estimating the costs of illness

The category of Years of Life Lost includes only the effects of diseases related to reduction of life expectancy. In the NEEDS project other effects were accounted for separately and include the following categories: new cases of chronic bronchitis, medication use, restricted activity and work loss days, and costs of hospital admissions. These effects were evaluated based on the scientific literature. For some effects, like the value of new cases of chronic bronchitis, the estimates are based on CVM studies (for this specific endpoint respondents were asked for their WTP to avoid chronic bronchitis), while for others the costs were derived from figures for the standard cost of a visit to the doctor or stay in hospital; for work loss days, an average European daily productivity loss value was used (ExternE, 2005).

Table 9 lists the monetary values of the health impact endpoints adopted in the NEEDS project (in Euros of 2000). Table 9 also reports which of the NEEDS damage estimates are based on direct financial costs and which on non-financial welfare losses estimated using a variety of survey techniques.

**Table 9 Monetary values of the health impact endpoints adopted in the NEEDS project (€$_{2000}$) and methods applied for arriving at these values**

| Endpoint | Unit | Value (€$_{2000}$) | Method* |
|---|---|---|---|
| Life expectancy reduction (YOLLchronic) | Year | 40,000 | SP |
| Increased mortality risk YOLLacute | Year | 60,000 | SP |
| New cases of chronic bronchitis | Case | 200,000 | SP |
| Medication use/bronchodilator use | Per 1 use | 1 | RP |
| Lower respiratory symptoms, cough days | Day | 38 | SP |
| Restricted activity days | Day | 130 | RP |
| Work loss days | Day | 295 | RP |
| Minor restricted activity days | Day | 38 | RP |
| Respiratory hospital admissions | Case | 2,000 | FC |
| Cardiac hospital admissions | Case | 2,000 | FC |
| Value of prevented fatality (VPF) | Case | 1,500,000 | SP |
| Mortality infants (2 times VPF) | Case | 3,000,000 | SP |

* SP – stated preferences; RP – revealed preferences; FC – financial costs.

Source: NEEDS, 2008 and NEEDS, 2008a.

It should be noted that the same values are used for all receptors (i.e. people). In the EcoSense model used in NEEDS, the receptor domain covers the whole of Europe. Additionally, for the health impacts of classical pollutants a Northern Hemispheric Model has been used, with the impact of emissions from Europe also being estimated for areas outside Europe. For these impacts, too, a simplified approach of using the same CRF functions and valuation of impacts has been adopted.

[43] It can be noted that in earlier stages of the ExternE series a VSL of € 1 million was used and a VOLY of € 50,000.

### 5.3.3 Estimating the monetary value of PDFs

Monetary valuation of ecosystem and biodiversity effects is far less developed than valuation of a statistical life. Two general approaches can be distinguished:

1. Approaches based on restoration costs.
2. Stated preference methods (particularly the Contingent Valuation Method), which allow direct assessment of WTP and so are more suitable for calculating external costs.

In the EcoSense model used in the NEEDS project, monetary evaluation of PDF was initially based on the minimum restoration costs of improving land use type from one with a lower number of different species to one with a higher number, i.e. under the assumption of restoration being performed to increase biodiversity. The methodology for estimating and evaluating biodiversity loss due to land use change is described in detail in Ott et al. (2006). The habitat restoration selected as a reference value was least-cost restoration of integrated arable land in Germany into organic arable land, involving a biodiversity increase of at least 20%. As this type of land conversion is common in all the countries considered, it was selected as the minimal marginal cost of improving biodiversity per PDF and $m^2$ (NEEDS, 2008a). NEEDS (2006) provides more detailed information on restoration costs for different land use categories in Europe calculated per PDF and $m^2$.

Later, however, the evaluation was updated according to the average WTP taken from Kuik et al. (2008). The value of PDF used for Europe now amounts to € 0.47/PDF/$m^2$ rather than the € 0.45/PDF/$m^2$ based on the study by Ott et al. (2006). The small difference between these values boosts confidence in the value used in the EcoSense model (NEEDS, 2008a). This approach has been developed further in the CASES project.[44] In that project PDF was defined in terms of Ecosystem Damage Potential (EDP), which is practically identical to PDF. On the basis of 24 studies on ecosystem valuation, the average value per EDP per hectare per year was calculated as equalling € 4,706 (median value: € 604). Values in other than euro currencies were adjusted using purchasing power parity exchange rate factors and converted to 2004 price levels using GDP deflators. EDP was found to be insensitive to income levels (or GDP per capita). A positive relationship was found between population density and biodiversity value, which is logical because if more people live in the vicinity of an area with high biodiversity, there will be more people that value that biodiversity. In this approach, the PDF/EDP value covers a broad range of value categories that can be attached to biodiversity, including both use value (e.g. recreational) and non-use value (existence, intrinsic value). In the cited study, diminishing returns to scale were found, so that with increasing size of ecosystems the value per hectare declines. In addition, as biodiversity change increases, the values per unit of biodiversity diminish (CASES, 2008).

[44] Cost Assessment of Sustainable Energy Systems, a project of the European Commission focusing on the total costs of energy production ($6^{th}$ Framework Programme).

Based on the meta-analysis of the international literature on the external costs of land use change (as stated, 24 different studies), a willingness-to-pay function for biodiversity change was estimated, with the following result:

$$\ln(VEDP) = 8.740 + 0.441 \ln(PD) + 1.070\ FOR - 0.023\ RIV + 0.485\ COA - 2.010\ dEDP - 0.312 \ln(AREA)$$

where:

VEDP = Value of Ecological Damage Potential (EDP is basically the same as PDF but measured per hectare).
PD = population density.
FOR = dummy variable for forest ecosystems.
RIV = dummy variable for river ecosystems.
COA = dummy variable for coastal ecosystems.
dEDP = change in EDP.
AREA = size of ecosystem in hectares.

For more details on this method of calculating values for biodiversity change, the reader is referred to CASES (2008) and studies referenced therein.

In the NEEDS estimates the average value of € 0.47 per $m^2$ was taken, with adjustment for the share of natural soil in each modelling grid cell and country-dependent sensitivity of soil (NEEDS, 2008a).

### 5.3.4 Impact on crops

Impact on crops belongs to the category of ecosystem services and is related specifically to agriculture. Within the NEEDS project the effects of $SO_2$ and ozone were modelled using concentration-response functions. Changes in crop yields dependent on $SO_2$ concentration were calculated for wheat, barley, potato, sugar beet and oats. For ozone, the relative yield change was calculated for rice, tobacco, sugar beets, potato, sunflower and wheat.

Another approach adopted within the ExternE studies aimed to investigate the costs of mitigating certain impacts on crops. Two effects were assessed: acidification of agricultural soils and fertilisation effects due to nitrogen deposition. For acidification, an upper-bound estimate of the amount of lime required to balance atmospheric acid inputs on agricultural soils across Europe was estimated. As for fertilisation effects, because deposition of oxidised nitrogen is beneficial to crops, the reduction in fertiliser needs was calculated.

In NEEDS monetary valuation of crop changes is based on the price per tonne of specific crops, with a number of reference sources being used for this purpose (NEEDS, 2008a).

### 5.3.5 Impact on building materials

Impact on building materials has been modelled using the impact-pathway approach adopted in the ExternE project series. The impacts of air pollutants on buildings include loss of mechanical strength, leakage, and failure of protective coatings due to materials degradation. For several materials, dose-response functions were established. In a two-step approach, these concentration-response functions (CRF) 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 of critical degradation, i.e. the point where replacement or maintenance is deemed necessary.

In the EcoSense model used in NEEDS, CRF were implemented for the following materials: limestone, sandstone, natural stone, mortar, rendering, zinc and galvanised steel, paint on steel, paint on galvanised steel, and carbonate paint.

Monetary values for impacts on building materials in maintenance costs per $m^2$ are based on expert estimates and range from € 33 for mortar and rendering to € 299 for limestone, natural stone and sandstone (in 2000 prices) (NEEDS, 2008a).

### 5.3.6 Other impacts

Other impacts include those on recreational values, odour, visibility and aesthethics. These impacts have not been taken into account in the present study for two reasons. First, because estimates of these impacts are highly uncertain and have only been established in a small number of studies that followed a less well-scrutinised methodology. Second, based on these studies, it could be argued that the total impact of these effects taken together is probably small compared to the other impacts that have been estimated. These values should be included only if they are believed to be important under specific circumstances. References and estimation procedures for these kinds of impacts are included in Annex C.3.7.

## 5.4 Valuation in Approach 3 (implicit valuation based on ReCiPe endpoints)

Another route for establishing shadow prices according to damage costs would be to use the characterisation factors and establish monetary values for the ReCiPe endpoints directly (cf. Goedkoop et al., 2009). This delivers a far more straightforward route than Approach 2. Because of the nature of the ReCiPe modelling assumptions, however, this route is less specifically oriented towards damage estimation (cf. Annex G).

### 5.4.1 Physical endpoints in ReCiPe

In ReCiPe three physical endpoints are distinguished:
- Human health, estimated in Disability Adjusted Life Years (DALY).
- Ecosystem biodiversity, estimated in PDF (or species/$m^2$).
- Abiotic resource availability, estimated in €.

For the reasons outlined in Section 3.5.5, in this study we have not included resource availability as an endpoint category, as it does not constitute an economic problem.

PDFs have already been outlined in Box 5 at the beginning of Section 5.3.
Box 7 below introduces the concept of DALY. Further information on DALY and its components can be found in Annex H.

**Box 7: The concept of Disability Adjusted Life Years (DALY)**

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.

$$DALY = YOLL + YLD$$

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).

In short, one DALY represents the loss of one year of equivalent full health (WHO, 2008).

### 5.4.2 Monetary valuation of DALY

In this Section we discuss how to arrive at a monetary value for a statistical year of healthy life. One such indicator is VOLY (value of life year). One option is therefore to use the estimate from the NEEDS project, i.e. a mean VOLY for EU-25 of € 40,000 for chronic diseases.

One downside of this method is that VOLY in principle only reflects a value of YOLL, so the value of YLD may not be properly included in VOLY. This depends on whether the disability weights included in YLD are representative of the direct estimation of the WTP to avoid illness. Whether using VOLY as a monetary estimate of DALY results in a significant under- (or over-) estimate of a sum of mortality- and morbidity-related effects for specific environmental themes depends on two things:

- The most important endpoint for the pollutants. For $PM_{2.5}$, for example, YOLL represents about 67% of the total damage value per unit emission. In this case, because the vast majority of the impact is attributed to changes in mortality, using VOLY to value one DALY seems justified. However, if the pollutants result in more YLD-effects than YOLL-effects, using VOLY to value one DALY could give a biased result.
- The difference between direct valuation of the damage costs of diseases (using stated or revealed preference methods) and YLD valued at € 40,000. Below, we estimate this difference for a specific disease.

For morbidity with classical pollutants, NEEDS uses other literature sources to estimate the costs of medical treatment, costs related to days of work lost, etc. For example, lower respiratory symptoms are valued at € 38 per day, or € 13,880 per year. According to the WHO (2008), the disability weight for lower respiratory infections (episodes) is 0.279. If we wished to compare the two approaches for this case, we could calculate 0.279 * 40,000 = 11,160. In this case, then, valuing morbidity effects using YLD and its monetary value of € 40,000 would result in an underestimate, because with direct estimation we obtain a higher value. However, one cannot adjust YLD based on this one example and for other YLDs no relationship could be established between the valuation of the incidence of diseases from NEEDS.

We therefore suggest adopting a value of € 40,000 (in 2000 prices) or € 55,021 (in 2008 prices) for one DALY, thereby following the approach taken in the NEEDS project. For certain environmental themes resulting in more YLD-effects than YOLL-effects, this may result in slight under- or overestimation.

### 5.4.3 Monetary valuation of PDF

In the ReCiPe project, damage to ecosystem diversity is measured as the loss of species occurring in a certain area in one year. Characterisation factors were developed for terrestrial areas, but also for freshwater and marine areas. Here, we shall use only the characterisation factors for terrestrial areas (since the monetary valuation in the NEEDS project can only be applied to terrestrial areas).

The loss of species during a year is simply the PDF per $m^2$ per year multiplied by the average species density per $m^2$. The terrestrial species density is estimated at 1.38E-08 per $m^2$ (Goedkoop et al., 2009). The monetary values that have been used are derived from NEEDS and shown in Table 10.

**Table 10 Monetary valuation (€$_{2008}$)**

| | Euro |
|---|---|
| PDF per $m^2$ per year | 0.55 |
| Specie per year | 4.0E7 |

## 5.5 The set of shadow prices based on damage costs

In this section we present our estimated shadow prices according to damage costs for various pollutants, thereby using two weighting schemes. In the first scheme we use our estimates for the individual pollutants and aggregate these to environmental themes at the midpoint level. The second weighting scheme is based on a direct estimate of ReCiPe endpoints for all environmental themes, similar to Approach 3, outlined above.

### 5.5.1 Approach adopted

In Section 5.2 three different approaches for estimating damage costs were presented. Table 11 reports which method has been used for each of the environmental themes considered.

Table 11 **Approaches adopted for the various individual pollutants, classified by environmental theme**

| Environmental theme | Pollutants directly estimated | Approach* | Estimated endpoints** | Ignored endpoints |
|---|---|---|---|---|
| Climate change | $CO_2$ | Approach 2^ | HH; ES, crops, build. | |
| Ozone depletion | - | Approach 3^^ | HH; ES, crops | |
| PM formation | $PM_{10}$^^^, $PM_{2.5}$, $NO_x$, $SO_2$, $NH_3$ | Approach 2 | HH | |
| Photo-oxidant formation | NMVOC, $NO_x$, $SO_2$ | Approach 2 | HH, ES, crops | |
| Acidification | $NO_x$, $NH_3$, $SO_2$ | Approach 2 | ES, crops, build. | |
| Eutrophication of fresh water | - | Approach 3 | ES | |
| Eutrophication of soils | $NO_x$, $NH_3$, $SO_2$ | Approach 2 | ESs, crops | |
| Human toxicity | Cd, As, Ni, Pb, Hg, Cr, formaldehyde, dioxins | Approach 2 | HH | |
| Ionising radiation | cesium, iodine, hydrogen (tritium), carbon, krypton, radon, thorium, uranium | Approach 2 | HH, ES | Crops |
| Noise | dB | Approach 1 and 2 | HH | |
| Land use | - | Approach 3 | ES | Crops*** |

Notes:

* Approach 1 = direct estimation of damages; Approach 2 = Direct estimation of endpoints using NEEDS data; Approach 3 = Implicit estimation of ReCiPe endpoints.

** HH = human health; ES = Ecosystems; build. = buildings.

*** Land use also has an impact on crops, as prices of land will rise. Since this effect is most likely a pecunairy externality, it has been ignored in this study.

^ For climate change, the method is not exactly the same as the Impact Pathway Approach for other pollutants, as emission sources and dispersion are not modelled (because of the nature of CO2 impacts, which are independent of emissions location).

^^ For ozone depletion we discounted the direct estimate of ReCiPe endpoints, as these are reported without any discounting (for procedures, see Annex C.6). To the ReCiPe estimates we added an estimate for crop losses.

^^^ The value for $PM_{10}$ is a weighted average of the values for PMco and $PM_{2.5}$, both of which are included in $PM_{10}$.

### 5.5.2 Results

The full results of the quantitative analysis are presented in Annex C. Here we provide a brief summary.

Cost estimates for **climate change** have been derived from a variety of studies on damage costs using Integrated Assessment Models (IAM). They were subsequently compared with the values obtained using the abatement cost approach, and a set of values for a period 2010-2050 is recommended based on the two approaches. For the current decade until 2020, the damage costs are based on abatement costs according to the arguments outlined in Annex C.

Valuation of impacts of so-called **classical pollutants** ($SO_2$, $NO_x$, PM, $NH_3$ and NMVOC) has been based on the results of the Impact Pathway Approach (IPA) adopted in the NEEDS project, where $PM_{10}$ consists of $PM_{2,5}$ (a share of 61%) and PMco (39%)[45]. The impacts of these substances have been assessed for human health, agricultural crops, buildings and materials. Based on the concentration-response functions (CRFs) and monetary values developed in the course of the ExternE series, a set of values of damage costs per tonne of pollutant has been calculated after modelling a 15% emission reduction for each pollutant in different regions using the EcoSense model. The following endpoints were included: mortality (both chronic and acute, valued using a YOLL of € 40,000 (for chronic mortality), morbidity (diseases and symptoms), crop yields, buildings and materials, and ecosystems. For valuation, we largely adopted the values from the NEEDS project (for detailed elaboration of the values used in the damage cost estimates, see Section 5.3).

For **toxic pollutants**, i.e. heavy metals, formaldehyde and dioxins, damage costs have been assessed for human health only. The proposed values are based on the results of the NEEDS project, which summarises the outcomes of a variety of other studies. Monetary valuation of morbidity and mortality was performed by multiplying the YOLL and YLD attributed to various diseases caused by these pollutants by a VOLY of € 40,000.

Valuation of the impact of **ozone-depleting substances** was not a topic included in the NEEDS project. Given the further lack of such estimates in the literature, we opted to base our estimates of human health impacts on the ReCiPe methodology. The effect of a change in UV-B radiation on human health was calculated using the AMOUR model (van Dijk et al., 2008). The resulting damage factor is expressed in DALYs per unit change in Effective Equivalent of Stratospheric Chlorine (EESC). These have been translated into the characterisation factors in DALYs per tonne CFC-11-equivalent for each group of ozone-depleting substances. In the case of human health effects, monetary valuation was obtained by using a standard value for a VOLY. For capital effects, the endpoint damage for different crops has been multiplied by the estimated producer costs, based on Hayashi et al. (2006).

Assessment of **ionising radiation** has been based on the results of the NEEDS project and includes only human health-related valuation. The fate and exposure factors used in NEEDS were calculated on the basis of the UNSCEAR (1993, 2000) methodology. As a result of radiation absorption, health impacts may manifest themselves in the form of fatal and non-fatal cancers and hereditary defects. Valuation of these impacts was based on the number of DALYs per cancer. For fatal cancers, the resulting YOLL was multiplied by the VOLY of € 40,000 and the cost of illness was added. For non-fatal cancers, the cost of illness was used. For valuation of hereditary effects, a standard value of statistical life (VSL) equal to € 1.5 million per case was taken.

Valuation of **noise** effects draws on the results of the HEATCO project funded by the EC and focusing on transport externalities. The guidelines developed within HEATCO take annoyance and health impacts to be 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 valuations. Quantifiable health costs were added to these values to derive a total value for noise.

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45 These percentages have been determined on the basis of current $PM_{2.5}$ and $PM_{10}$ emissions in the Netherlands; see Annex C 3.1.

For **land use**, the NEEDS approach to valuing ecosystems has been applied. The NEEDS approach uses Potentially Disappeared Fraction (PDF), the inverse of relative species abundance. 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. For valuation of land use, the average value of PDF from the study of Kuik et al. (2008) was applied. This is an average global value (mainly for Europe and North America) and will therefore not reflect very specific local conditions in the Netherlands. The impacts of land-use changes on PDF have been taken from the ReCiPe project (Goedkoop et al., 2009), which also gives averages for Europe. We therefore interpret these values for land use changes as values for the EU-27 (and not the Netherlands). However, they could be used in the Dutch context as a first approximation.

Table 12 gives the results at the level of each pollutant for the damage costs based on the information in Annex C. The damage costs relate to the costs of emissions in 2008 in the Netherlands, expressed in € per kg pollutant in price levels of 2008 (for a discussion on the geographical scale of the estimates, see Section 3.5.1).

**Table 12 Shadow prices of emissions in the Netherlands in 2008 based on damage costs (€$_{2008}$/kg pollutant)**

| | Human health | Ecosystems | Crops and buildings | Total |
|---|---|---|---|---|
| $CO_2$** | 0.00487 | 0.0201 | | 0.0250 |
| $CH_4$ | 0.122 | 0.503 | | 0.625 |
| $N_2O$ | 1.45 | 6.00 | | 7.45 |
| CFC-11 | 62.4 | 95.6 | 1.25 | 159 |
| CFC-12 | 103 | 219 | 1.31 | 324 |
| CFC-113 | 67.8 | 123 | 1.34 | 193 |
| CFC-114 | 84.3 | 201 | 1.25 | 287 |
| CFC-115 | 52.6 | 148 | 0.75 | 202 |
| HCFC-22 | 12.8 | 36.4 | 0.05 | 49.2 |
| Halon-1211 | 350 | 38.0 | 3.17 | 391 |
| $NO_x$ | 9.27 | 1.48 | -0.167 | 10.6 |
| $SO_2$ | 14.5 | 0.453 | 0.430 | 15.4 |
| $NH_3$ | 23.1 | 4.86 | -0.200 | 27.8 |
| NMVOC | 2.16 | -0.0873 | 0.462 | 2.54 |
| P (manure) | | 0.0890 | | 0.0890 |
| P (fertilizer) | | 0.0947 | | 0.0947 |
| P (from STP) | | 1.78 | | 1.78 |
| $PM_{10}$ (particulates) | 41.0 | | | 41.0 |
| $PM_{2.5}$ (particulates) | 64.8 | | | 64.8 |
| Dioxins | 5.09E07 | | | 5.09E07 |
| As (arsenic) | 811 | | | 811 |
| Cd (cadmium) | 127 | | | 127 |
| Cr (chromium) | 33.5 | | | 33.5 |
| Ni (nickel) | 5.37 | | | 5.37 |
| Pb (lead) | 408 | | | 408 |

| | Human health | Ecosystems | Crops and buildings | Total |
|---|---|---|---|---|
| $CH_{20}$ | 0.275 | | | 0.275 |
| Uranium-235 (air)* | 1.16E09 | | | 1.16E09 |
| Uranium-235 (water)* | 1.27E08 | | | 1.27E08 |

* Radiation emissions are measured in Bequerel. The damage costs reported here are expressed in € per petabecquerel (PBq).

** Valuation for climate change was not originally broken down into damage to human health and to ecosystems. To do so, we used the ReCiPe endpoint factors for health damage (in DALYs) and ecosystem damage (in PDFs) per kg $CO_2$. These damages were each valued directly, from which the ratio between health and ecosystem damage was determined.

In addition to these substances, Table 13 gives the damage costs of noise from road, rail and aircraft. Up to 70 dB the damage costs consist only of annoyance costs (€ 10/€ 15 per dB above threshold). Above 70 dB, the damage costs include annoyance costs and costs related to health effects occurring above these noise levels.[46]

**Table 13 Shadow prices for noise exposure in the Netherlands, based on damage costs (€$_{2008}$ per dB above threshold, per year per person exposed)**

| $L_{den}$ dB(A) | Road | Rail | Aircraft |
|---|---|---|---|
| 50-70 | 12.71 | 12.71 | 19.06 |
| > 70 | 20.33 | 20.33 | 27.96 |
| *> 70 (add for health effects)* | *62.27* | *62.27* | *60.99* |
| **Threshold** | **50 dB** | **55 dB** | **50 dB** |

Source: Own calculations based on HEATCO, 2006.

Table 14 reports the damage costs for several land use types. As outlined above, because of the nature of the underlying studies only European values can be given. They may serve as a first approximation for values in the Netherlands.

46 For example, the damage costs of 75 dB road noise equals: (€ 12.71 x 20 + € 20.33 x 5 + € 62.27) = € 418 year per person exposed.

Table 14 **Values of external costs for occupation of a certain area of land for different land use types (€$_{2008}$, EU averages)**

| Land use type | External costs (€ per m$^2$ per year) |
|---|---|
| Monoculture crops/weeds | € 0.77 |
| Intensive crops/weeds | € 0.74 |
| Extensive crops/weeds | € 0.71 |
| Monoculture fertile grassland | € 0.63 |
| Intensive fertile grassland | € 0.51 |
| Extensive fertile grassland | € 0.38 |
| Monoculture infertile grassland | € 0.47 |
| Extensive infertile grassland | € 0.24 |
| Monoculture tall grassland/herb | € 0.75 |
| Intensive tall grassland/herb | € 0.58 |
| Extensive tall grassland/herb | € 0.42 |
| Monoculture broadleaf, mixed forest and woodland | € 0.35 |
| Extensive broadleaf, mixed and yew LOW woodland | - |
| Broad-leaved plantation | € 0.45 |
| Coniferous plantations | € 0.50 |
| Mixed plantations | € 0.61 |
| Continuous urban | € 0.78 |
| Vineyards | € 0.48 |

Source: Own calculations based on Goedkoop et al. 2009 and Kuik et al. 2008.

## 5.6 Temporal and spatial variation

### 5.6.1 Dynamic developments

The financial value assigned to emissions via monetisation of the resultant damage may, for a variety of reasons, change over the course of time.

In the first place there may be alterations in the physical damage itself as time progresses. If there is a rise in population density, for example, the aggregate impact of emissions will also rise. The same holds for economic growth: here, too, this will mean more Euros of capital being exposed to potential damage. Finally, dose-effect relationships may not be linear: if emissions grow, the marginal damage due to an additional unit emission may be greater than that associated with earlier emissions. On the other hand, developments may also lead to a reduction in future physical damage, through adaptation or structural economic change, say. One example for the Netherlands is provided by agriculture, which is more vulnerable than most economic sectors to climate change, while its relative share in GDP is declining.

It is not only the physical damage that may change over time, but also financial valuation thereof. Concerns about an unpolluted environment generally increase in proportion to income, reflected in a greater Willingness to Pay (WTP). It is therefore to be anticipated that as prosperity continues to grow in the future, the financial value assigned to emissions will also rise. In the literature the debate is currently focused on the issue of whether the environment is a ‘luxury good’ (with an income elasticity greater than 1) or a normal economic good (with an income elasticity between 0 and 1). A number of empirical studies have sought to establish the income elasticity of WTP for environmental quality, with the more recent of them making use of stated preferences. Most of these studies claim income elasticities of a little less than 1 (e.g. Kristrom and Riera, 1996; Ready et al., 2002; Hökby and Söderqvist, 2003). They covered various European countries and a range of environmental impacts. One study on valuation of air quality in a district of China arrives at a different conclusion. Given the estimated income elasticities derived, air quality there is considered a luxury good (Wang and Mullahy, 2006). A plausible explanation for this might be that certain environmental problems are more urgent in China than in European regions. At the same time the degree of income elasticity appears to depend on income level, too. Following the lead of the NEEDS project, we propose to here adopt a figure of 0.85 for income elasticity.[47] Assuming 2% economic growth, this leads to a value increase of 1.7% (see Annex A.3).

These rules apply to all pollutants. In estimating the external costs associated with $CO_2$ emissions there is one further complication, relating to the non-linear modelling of these emissions. If worldwide emissions are not reduced, an additional unit of $CO_2$ emission may have a greater impact in the future than now owing to steadily rising atmospheric $CO_2$ levels. Table 15 presents the estimates of values for $CO_2$ in different time horizons based on informal meta-analysis of various studies as presented in the IMPACT Handbook (CE, 2008b).

**Table 15 Recommended values for the external costs of climate change (in €/t$CO_2$) changing over time, expressed as single values for a central estimate, lower and upper values**

| | Central values (€/tonne $CO_2$) | | |
|---|---|---|---|
| Year of emissions | 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.

### 5.6.2 Regional variation

Thus far, the shadow prices reported represent averages for the Netherlands for emissions in 2008. However, these averages may not be truly representative of local circumstances. Especially for pollutants contributing to acidification, PM formation, eutrophication and toxicity, local circumstances may deviate from these national averages. This is due to local variations in climatic conditions (winds), population density and quality of soils.

If local circumstances are particularly relevant for a specific use of shadow prices, we would advocate a separate model run with EcoSense and adjustment of the modelling results according to the assumptions made in the present study. These adjustments should at least cover price levels and monetary estimates of biodiversity loss. Another possibility is to use a value transfer procedure (cf. Annex E). This might be applied if there are data available on differences between the factors that influence the value estimate for a given region and the average country-specific factors.

47 This value is an expert judgment based on informal meta-analysis of a variety of studies related to this subject. It is worth noting that income elasticity varies across environmental services/impacts. For any given case, therefore, a range of elasticity values can be taken from the scientific literature. In the most simplified approach and if no elasticity data are available, an income elasticity of 1 can be used. In such cases this should be clearly indicated, though, as such an approach may lead to overestimation.

### 5.6.3 Damage costs for the EU-27

The NEEDS/CASES/ExternE projects, on which we based some of the estimates presented in Chapter 4, have established estimates for the EU-27 as a whole as well as specific estimates for each of these countries. CRF functions and monetary valuation of endpoints are the same for the whole receptor domain, which includes the whole of Europe and adjacent areas. The ultimate impact of emissions from Europe on the receptors located outside Europe has been modeled via North-Hemispheric modeling.

Table 16 reports damage estimates for the EU-27, given by the emission-weighted averages of the results for each individual country.

**Table 16 Shadow prices of emissions in the EU-27 in 2008 based on damage costs (€$_{2008}$/kg pollutant)**

| | Human health | Ecosystems | Crops and buildings | Total |
|---|---|---|---|---|
| $CO_2$ | 0.00487 | 0.0201 | | 0.0250 |
| $CH_4$ | 0.122 | 0.503 | | 0.625 |
| $N_20$ | 1.45 | 6.00 | | 7.45 |
| CFC-11 | 62.4 | 95.6 | 1.25 | 159 |
| CFC-12 | 103 | 219 | 1.31 | 324 |
| CFC-113 | 67.8 | 123 | 1.34 | 192 |
| CFC-114 | 84.3 | 201 | 1.25 | 287 |
| CFC-115 | 52.6 | 148 | 0.75 | 202 |
| HCFC-22 | 12.8 | 36.4 | 0.05 | 49.2 |
| Halon-1211 | 350 | 38.0 | 3.17 | 391 |
| $NO_x$ | 7.87 | 1.30 | 0.480 | 9.64 |
| $SO_2$ | 8.73 | 0.254 | 0.263 | 9.25 |
| $NH_3$ | 13.0 | 4.69 | -0.220 | 17.5 |
| NMVOC | 1.29 | -0.097 | 0.227 | 1.42 |
| P (manure) | | 0.0890 | | 0.0890 |
| P (fertiliser) | | 0.0947 | | 0.0947 |
| P (from STP) | | 1.78 | | 1.78 |
| $PM_{10}$ (particulates) | 22.6 | | | 22.6 |
| $PM_{2.5}$ (particulates | 33.8 | | | 33.8 |
| Dioxins | 5.09E07 | | | 5.09E07 |
| As (arsenic) | 728 | | | 728 |
| Cd (cadmium) | 115 | | | 115 |
| Cr (chromium) | 18.2 | | | 18.2 |
| Ni (nickel) | 3.16 | | | 3.16 |
| Pb (lead) | 383 | | | 383 |
| $CH_{20}$ | 0.275 | | | 0.275 |
| Uranium-235 (air)[48] | 1.16E09 | | | 1.16E09 |
| Uranium-235 (water) | 1.27E08 | | | 1.27E08 |

The estimates for the EU-27 should, of course, be used if the subject of valuation or weighting is at the level of the EU-27. The choice of damage value (country-specific or EU-average) therefore depends on the type of project or policy being evaluated. In designing new European regulations, for example, it makes more sense to use EU-27 values, while for projects of national scope country-specific values would be more appropriate.

48 Note that radiation emissions are measured in Bequerel. The damage costs reported here are expressed in € per petabecquerel (PBq).

Note that for $CO_2$, because damages are calculated using Integrated Assessment Models for global damages, in most cases no additional adjustment is needed. In the Fund model used in NEEDS to estimate the damage costs associated with greenhouse gas (GHG) emissions, the values of damages are implicitly adjusted for income level because VSL is evaluated in monetary terms using a constant (200) times per capita income. Income elasticity is assumed to be unity. For projects with a global/European scope we would therefore recommend simply using these values. Different recommendations may apply, however, to the estimates of GHG shadow prices based on damage costs for projects with a national scope. In such cases, the value transfer procedures described in the following section may apply.

### 5.6.4 Damage estimates for non-EU countries

Damage valuation should ideally be based on high-quality, primary valuation studies. Such studies are not always available, however, which means that in efforts to provide reliable estimates of environmental damages for a given region or country, researchers and policy-makers often have to refer to primary valuation studies carried out for other regions and 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 can be regarded as the final step of the Impact Pathway Approach, required in cases where relevant primary valuation studies are lacking.

When transferring damage estimates to other (non-European) countries, several decisions and assumptions need to be made:

- A choice must be made between using country-specific or EU-average damage values. In most cases the latter are probably more appropriate, unless there is sufficient evidence that such factors as background pollutant concentrations and receptor density are very similar in the country under consideration to those in a particular European country for which specific values are available.
- By using either EU-27-average or country-specific values and adjusting only for economic differences (income level) we assume that all other factors (such as background pollutant concentrations, receptor density, meteorological conditions) are more or less the same. In certain specific contexts, however, some of these factors may deviate significantly from the EU average. For human health impacts, for instance, receptor density is expressed in terms of population density; if there is a significant difference in population density between the European average and the region concerned, this can be corrected using a factor given by the ratio between the population densities of the respective regions. The same holds for local impacts. These may differ from the country-specific figures or EU-27 averages provided in this handbook. In cases when sufficient and reliable information is available on some of the factors influencing damage estimates, values can be adjusted accordingly.
- For values based on stated preferences surveys we assume that the preferences of the population in the location to which we transfer the values are more or less the same as in Europe. Typically, differences in values obtained in different countries occur not only because of differences in income but also owing to other factors like age structure, religion, political regime and so on. In a simplified approach, we will not be able to control for these differences.

Bearing in mind the above limitations and recognising that, regardless of the valuation method adopted, the values obtained in surveys are highly dependent on individual income, in the simplest approach we propose adjusting the damage values according to the ratio of income levels of the relevant populations. The most popular and reliable statistic approximating the level of individual income is per capita GDP. In order to have a real measure of income, we therefore propose using per capita GDP at Purchasing Power Parity (PPP). Additionally, it is advised to use an income elasticity factor.

The adjusted WTP estimate at the project/policy site, $WTP_p$, can then be calculated as follows:

$$WTP_p = WTP_E \, (Y_p \,/\, Y_E)^{\beta} \qquad (1)$$

where $WTP_E$ is the original value estimate from our handbook, $Y_E$ and $Y_p$ are per capita GDP at PPP levels for the EU or a selected European country and project/policy site, respectively, and $\beta$ is the income elasticity of demand for the environmental good in question (based on NEEDS, 2007a).[49] Within the NEEDS project, it was assumed that $\beta$ equals 0.85.

For more information on alternative methods of value transfer and for further discussion on the topic, the reader is referred to Annex E. The exact method of value transfer may depend on project scope. In projects of international or global scope, there may be ethical objections to using a different value for a statistical human life (expressed in VSL or VOLY, say) depending on location. In CBAs, country-specific values should be applied within a given country. This implies using a much lower VSL in India, say, than in a European country. In CBAs on EU Directives affecting all European member states, with very different income/GDP per capita, however, an average EU value should be used. In global CBAs it can be argued that for ethical reasons global average values should be used.[50]

Our recommended approach in such cases is to perform a sensitivity analysis using different valuation methods, assess whether the results differ significantly and, if so, identify implications for the given policy or project.

49 It is assumed that income elasticity comes from a double logarithmic model, where ß is a regression coefficient for a relationship between WTP and income, with both variables transformed using a natural logarithm. Such models are convenient to use because ß fits the definition of income elasticity: it shows the percentage growth in WTP resulting from 1% growth in income. Such a model assumes constant elasticity of income for all income levels.

50 Personal communication with Ståle Navrud, 16.04.2009.

## 5.7 Uncertainty

The monetary values of damages per unit of the specific pollutants presented in this handbook have been estimated using a variety of assumptions and models. Each step of the analysis involves a certain degree of uncertainty,[51] which accumulates and effectively increases with each further step (e.g. the uncertainty at the level of CRF is aggravated with additional uncertainty surrounding monetisation of effects identified through CRF). In the sections below, we first outline some of the most important sources of uncertainty in the damage estimates and subsequently provide a more formal description of how this uncertainty can be roughly calibrated for the IPA.

### 5.7.1 Major sources of uncertainty in valuation of health damages

Health damages are the single largest contributor to the total damages associated with most pollutants. This section therefore considers the uncertainties surrounding valuation of mortality and morbidity effects.

In the ExternE series of projects the impact of classical pollutants on human health was assessed by summing the impacts of specific substances on several endpoints such as mortality (in some cases assessed separately for adults and infants) and morbidity. In the NEEDS project special surveys were conducted to reveal direct WTP for a longer and healthier life due to improved air quality. The values obtained in CVM studies and the resulting values per tonne of pollutants are only rough estimates. Among the factors contributing to ambiguity are the following:

1. It is impossible to precisely calibrate the dose-response relationship described in the survey scenario using the modelling applied in the EcoSense model. In the survey scenario, a hypothetical reduction in air pollution would result in extending an individual life by 3 or 6 months (the respondents had to assign a monetary value to both changes in life expectancy). In the EcoSense model a 15% reduction in pollutant emissions is modelled. It is to be queried whether these two elements match. It should be noted, though, that the ExternE experts made every effort to design the entire estimation methodology as plausibly as possible.
2. It is impossible to know exactly what lies behind the values obtained from the surveys. In the CVM surveys carried out in NEEDS it was stressed that the respondents were asked to value an increase in their own life expectancy only, combined with improved quality of life due to improved air quality. However, from responses to several additional questions at the end of the questionnaire it was evident that some of the respondents' motives included broader values such as bequest value (i.e. willingness to leave a cleaner environment for future generations).

In the alternative approach using DALYs (as used in the NEEDS project for non-classical pollutants) the situation is even more ambiguous. The most questionable element is the disability weight used to obtain the YLD measure. Disability weights have been assigned to specific diseases by health care experts and are not based on carefully designed CVM studies like the one carried out in NEEDS to calculate the value of YOLL. The main advantage of disability weights is their usefulness for comparative purposes: if one disease has been assigned a lower disability weight than another, it means that living one year with the former is preferable to living one year with the latter. In situations where there was no better information on the costs associated with

[51] It is useful to distinguish risks from uncertainty. Risk relates to a situation where we have at least some idea of the probability of a given effect occurring. Often, though, we have no idea of the probability at all. This is true uncertainty. For example, we cannot (yet) assign probabilities to particular effects of climate change.

certain endpoints, disability weights have been used (via YLD) to translate the qualitative impact of diseases into longevity. If, based on studies using other valuation methods, disability weights proved unreliable, then monetary valuation based on DALYs would also be rendered more questionable (especially for those diseases for which YLD constitutes a large share of the DALY measure).

When a person dies prematurely from a disease due to (air)pollution, there is obviously a sense of loss and grief among family and friends. These values have not been covered by the valuation approach adopted in NEEDS, however. With regard to values such as 'enjoyment of life' and 'happiness', the best answer we can give is that these are probably partly covered, but it is impossible to say to what extent.

### 5.7.2 Formal treatment of uncertainty

Because of the complexity of calculating damage costs within IPA (several stages of estimation, each with its own uncertainties), assessing overall uncertainty for final damage estimates is by no means straightforward. The methodology used within the NEEDS project is based on assessing the geometric standard deviations ($\sigma_g$) of the damage cost estimates under the assumption that these are distributed lognormally. For classical pollutants, NEEDS (2008b) found a value of approximately 3 for $\sigma_g$. According to the characteristics of lognormal distribution this means that, for classical pollutants, there is a 68% probability of the true value lying within an interval given by the central value[52] divided by three and the central value multiplied by three.

For greenhouse gases, the analysis of uncertainty in NEEDS is based on Tol (2005), who has reviewed numerous damage cost analyses. He gathered over 100 estimates of marginal damage costs and used them to define a probability density function. The function proves to be strongly right-skewed, with a median of \$ 3.8/t$CO_2$, a mean of \$ 25.4/t$CO_2$ and a 95% confidence level of \$ 95/t$CO_2$. NEEDS (2008b) conclude that, excluding negative costs, the distribution is not too different from a lognormal with its tail of high estimates of low probability. According to their calculations, the standard deviation for these estimates is 5. We may therefore conclude that 68% of the estimates fall within an interval between the central value divided by five and that value multiplied by five. For the mean estimate from Tol (2005) of \$ 25.4 this would imply the following confidence interval: [5, 127].

Under the same assumption of lognormality of damage distribution, NEEDS (2008b) calculated geometric means and standard deviations for trace pollutants. They estimated a $\sigma_g$ of 4 for As, Cd, Cr-VI, Hg, Ni and P and of 5 for dioxins. Hence, it can be stated that for As, Cd, Cr-VI, Hg, Ni and P the true values lie with a 68% probability within an interval between the central value divided by four and that value multiplied by four, while for dioxins the true values lie with a 68% probability within an interval between the central value divided by five and that value multiplied by five.

[52] 'Central value' refers to the values reported in the relevant tables in this handbook for the damage costs for the various specific pollutants.

### 5.7.3 Practical approach to uncertainty in SCBA

This formal treatment of uncertainty does not probably provide any meaningful leverage in practical SCBA, given the very large ranges involved (a factor of 1/3 to 3 between the upper and lower bounds within 68% probability bounds). While we are certainly in favour of communications regarding uncertainty being as transparent as possible, including an upper and lower bound with the estimates would have an all too predictable impact on every SCBA: at the upper bound environmental impacts will dominate all other costs and benefits, while at the lower bound environmental impacts will be negligible.

Although the choice of how to deal with uncertainty is ultimately up to the practitioners conducting the SCBA, we can imagine them working with the central values presented in this handbook and using a sensitivity analysis to present the consequences if the true economic value of the environment were to be double, say. For the lower bound we could imagine the abatement costs being taken. These may constitute an underestimate of the damage under the assumption that government policies are generally too loose, given the actual ecological damage of pollutants. In addition, abatement costs may be less uncertain (but with these costs there is no formal treatment of uncertainty available, however).

The question of whether abatement costs or damage costs should be used in SCBA to value external effects will be dealt with in Chapter 7.

## 5.8 Comparison with estimates from other studies

Within Europe, there are two main ‘families’ of external cost estimates. The first derives from the ExternE framework and has been used in the present report. In addition to the NEEDS estimates used here, estimates from projects like CASES, MethodEx, NewExt, HEATCO also use the ExternE methodology. The alternative approach is constituted by the studies undertaken in the Cost-benefit Analysis for Clean Air for Europe (CAFÉ CBA)(AEA, 2005).[53] The values per tonne of emissions from NEEDS and CAFÉ CBA are not always close, especially when it comes to country-specific values, even though the set of CRF functions applied is very similar. Differences in values for specific countries between the NEEDS project and CAFÉ CBA are due primarily to the fact that the NEEDS estimates refer to the damages due to emissions *released by* a given country, while those in CAFÉ CBA refer to emissions *occurring in* a given country. Average EU values should be more or less similar, however. As an illustration, Table 17 reports the EU-average estimates cited in CAFÉ CBA and NEEDS for several classical pollutants. Below, we go some way to explaining the differences between them. The NEEDS values do not include estimates of damages to biodiversity and those derived from Northern-Hemispheric Modelling, since these were not included in the CAFÉ CBA estimates.[54]

53 Projects like TREMOVE have also used this methodology (http://www.tremove.org/).

54 This of course implies that the values here do not correspond to the values in Section 5.6.3.

Table 17 **Comparison of illustrative values from the NEEDS and CAFÉ CBA projects, EU averages (€/kg, adjusted to $€_{2008}$), excluding impacts on biodiversity and impacts due to dispersion through the Northern Hemisphere**

| Pollutant | NEEDS | CAFÉ-CBA* |
|---|---|---|
| NMVOC | 1.03 | 1.20 |
| $NO_x$ | 8.17 | 5.29 |
| $PM_{2.5}$ | 33.6 | 31.2 |
| $SO_2$ | 8.61 | 6.73 |

Sources: CE, 2008b; CASES, 2008; own calculations.

* Values from CAFÉ CBA are estimated for emissions occurring in the year 2010.

Possible sources of differences between these values include the following:

- In CAFÉ CBA a different source-receptor matrix (SRM) is used, with no division into sub-regions and only one set of meteorological conditions, while in NEEDS the modelling matrix is more detailed.
- In CAFÉ CBA one scenario related to background emissions is used, while in NEEDS two different scenarios are used (one for the years 2000-2014 and one thereafter).
- CAFÉ CBA values only cover health costs and ozone-related crop losses, while NEEDS estimates cover other crop impacts, too.
- CAFÉ CBA uses a different value for VOLY than NEEDS. In the CAFÉ CBA study a range of estimates is given, reflecting a different basis for calculating mortality and morbidity impacts. In Table 17 we reproduce the values based on a median figure of € 52,000 for VOLY in the CAFÉ-CBA (the VOLY used later in NEEDS was slightly lower, € 40,000). It may be noted, however, that the values from the NEEDS project are higher than the CAFÉ CBA values, so this cannot be the reason for the differences.

# 6 Weighting factors

## 6.1 Introduction

In this chapter the sets of damage and abatement costs elaborated in Chapters 4 and 5 are used to devise three sets of weighting factors for use in LCA and other types of environmental impact analysis.

The most important step in developing weighting factors is to ensure consistent application of a coherent set of characterisation factors to the shadow prices.

## 6.2 Methodology

In this project we have developed three sets of weighting factors based on valuation of individual pollutants (see Chapter 2):
1. A set based on abatement costs characterised at midpoint level.
2. A set based on economic damage costs at endpoint level characterised at midpoint level.
3. A set based on direct valuation of health and biodiversity damage at endpoint level.

The use of characterisation factors is unavoidable, because shadow prices are available for a limited number of pollutants only. Characterisation factors give an indication of a pollutant's relative contribution to a given environmental impact. If we had shadow prices for all the 1,000 pollutants covered in an LCA, there would be no need to employ characterisation factors, because the individual shadow prices would themselves embody characterisation. We would then be able to use the shadow prices to determine the relative importance of pollutants A and B. If we only dispose over a shadow price for pollutant A, though, there is no option but to use characterisation factors.

In moving from valuation to sets of weighting factors, two practical problems arise:
1. Multiple impacts: the fact that many pollutants have impacts on multiple environmental themes, across which the shadow price therefore needs to be allocated.
2. Implicit characterisation: the fact that the damage estimates for multiple pollutants within a given theme already bring with them an implicit characterisation, which may deviate from the midpoint characterisation provided by ReCiPe. How are such differences to be addressed?

These problems are relevant for the first two sets of weighting factors only. In each case we have sought the best possible strategy for dealing with them. The respective methodologies will now be described.

### 6.3 Weighting factors according to abatement costs

#### 6.3.1 Introduction

For weighting based on abatement costs, we proceed from a single priority pollutant per environmental theme. This is often the pollutant taken as equivalence factor in the midpoint characterisation factors. For acidification, for example, $SO_2$ is the reference pollutant, which means that every other pollutant having an impact on this environmental theme is defined in terms of kg $SO_2$. In principle, it is this priority pollutant that determines the valuation. This pollutant may have an impact on multiple environmental themes. In that case a split across the various themes is achieved by examining the value of pollutants having an impact on one environmental theme only.

In order to then extend the valuation per priority pollutant to all possible pollutants, use has been made of ReCiPe midpoint characterisation factors[55].

One problem with abatement costs is how to allocate the costs of measures that reduce emissions of more than one pollutant across these various pollutants. The weighting factors ultimately elaborated will depend on how these so-called ‘joint costs’ are treated (cf. Sections 3.2.2 and 4.2.1).

#### 6.3.2 Results, by impact category, for use in weighting

The pollutants have been translated to impact categories, analagously to the procedure outlined in Section 4.2. Because margins are extremely complicated and indeed undesirable in the context of weighting, for those pollutants for which the value has been calculated within certain margins a central value has been included. For $CO_2$ this is € 0.0250/kg, equivalent to the estimates in the NEEDS project (cf. Annex B). For $SO_2$ and $NO_x$ we analysed the joint costs with climate policy and assessed what value would lead to a consistent estimate for both pollutants, given that both relate to multiple themes (see below). It emerged that a central value of € 5/kg for $SO_2$ and € 9/kg for $NO_x$ yielded a consistent estimate.

Table 18 reports the values for the respective environmental themes. Two variant sets of weighting factors are provided, calculated using different values for PM formation:

- Set 1a consists of weighting factors based on a PM value of € 2.30/kg. This figure is based on SenterNovem (2009) and forms the starting point of the analysis.
- Set 1b contains weighting factors if a PM value of € 50/kg is assumed. This alternative value is based on a rough estimate of the cost effectiveness of the measures required to meet future policy targets (cf. Annex B.4.2).

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55 An exception was made for human toxicity. For various pollutants (e.g. cadmium and mercury) the government has set maximum tolerable risk levels (MTRs). In calculating the abatement costs for human toxicity we have therefore based ourselves on these MTRs rather than on midpoint characterisation factors. For weighting, however, human toxicity is excluded to avoid double counting of impacts (both PM formation and human toxicity are expressed in terms of $PM_{10}$-eq.).

Table 18 **Weighting factors for emissions in the Netherlands in 2008 by environmental theme (midpoint, €$_{2008}$/kg equivalent)**

| Impact category | Equivalence factor | Abatement costs 1a (€/kg) | Abatement costs 1b (€/kg) |
|---|---|---|---|
| Climate change | $CO_2$ eq. | 0.0250 | 0.0250 |
| Ozone depletion | CFC-11-eq. | 30.0 | 30.0 |
| Acidification | $SO_2$-eq. | 4.13 | 0.594 |
| Photo-oxidant formation | NMVOC-eq. | 5.00 | 5.00 |
| Eutrophication, fresh water | P-eq. | 10.9 | 10.9 |
| Eutrophication, marine water | N-eq. (to water) | 7.00 | 7.00 |
| PM formation | $PM_{10}$-eq. | 2.30 | 50.0 |
| Human toxicity | $PM_{10}$-eq. | NA | NA |
| Noise (€ per dB-dwelling) | dB rail >55 | 3,000 | 3,000 |
| | dB road >50 | 3,000 | 3,000 |
| | dB aircr. >45 | 3,000 | 3,000 |

Note: [3] To avoid double counting, no weighting factor is attached to human toxicity. Physical impacts of $PM_{10}$-eq. weighted with the factor for PM formation.

As can be seen in Table 18, an alternative value for PM formation has consequences for the theme of acidification, too. This is due to the problem, of relevance for several themes, that the priority pollutant impacts on more than one theme. In such cases the pollutant's contribution to the individual theme has been calculated on the basis of its relative input to other themes. In the case of acidification, $SO_2$ has been taken as the determinant because all the other acidifying pollutants in ReCiPe are expressed in kg $SO_2$-eq. However, $SO_2$ is a pollutant with impacts on three themes. The price of € 5/kg $SO_2$ (set 1a)[56] therefore had to be broken down into contributions to the themes of acidification, photo-oxidant formation and PM formation, as shown in Table 19.

Table 19 **Breakdown of $SO_2$ value by theme (set 1a)**

| Pollutant | Characterisation factor | Theme price (€/kg) | Price for $SO_2$ (€/kg) |
|---|---|---|---|
| $SO_2$ total: | | | 5.00 |
| – impact on photo-oxidant formation | 0.0811 kg NMVOC-eq. | 5.00 | 0.406 |
| – impact on PM formation | 0.2 kg $PM_{10}$-eq. | 2.30 | 0.46 |
| – impact on acidification | 1 kg $SO_2$-eq. | 4.13 | ← 4.13 |

The $SO_2$ prices for photo-oxidant formation and PM formation have been calculated by multiplying the respective ReCiPe midpoint characterisation factor representing the physical impact of $SO_2$ for the theme in question by the so-called 'theme price'. The $SO_2$ value for the theme of acidification is then given by the total figure minus these two prices.

[56] A higher value for PM formation in set 1b corresponds with a higher price for $SO_2$ (slightly above the upper bound of € 10/kg in Table 7). See also Section 4.3.3.

## 6.4 Weighting factors according to damage costs

In the damage cost approach, two kinds of weighting factors are available:
- Weighting factors based on the damage estimates from Chapter 5 using ReCiPe midpoints.
- Weighting factors based on implicit damage estimates based on ReCiPe endpoint factors.

### 6.4.1 Results for weighting with shadow prices of damage costs

This approach uses the estimates from Chapter 5 to arrive at estimated weighting factors. However, as outlined in detail in Annex D, the procedure for aggregating these estimates into a weighting scheme is far from unambiguous. This is due mainly to certain pollutants impacting on multiple environmental themes.

In 'Approach A' the monetary valuations (damage costs) of different pollutants were directly estimated from the results of the NEEDS project. For pollutants like $NO_x$ and $SO_2$ with multiple environmental impacts, we need a way to split damage costs across themes. We shall explain this approach with reference to the example shown in Figure 11, below.

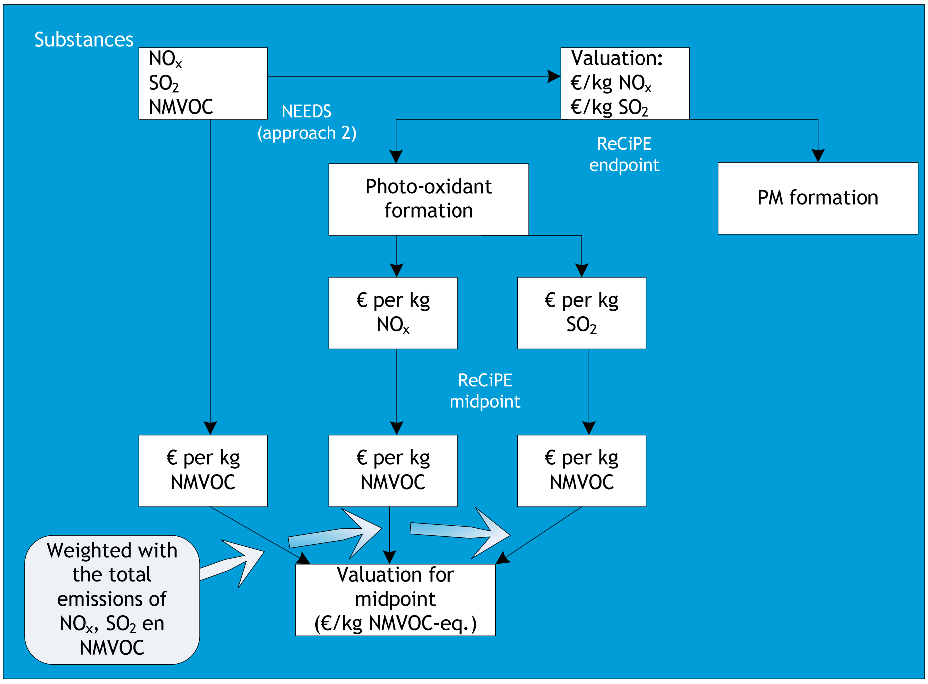
$NO_x$ and $SO_2$ emissions have an impact on two environmental themes: photo-oxidant formation and PM formation. From the NEEDS project, damage costs for $NO_x$ and $SO_2$ can be taken, but these damage costs need to be split over these two themes. This has been done using ReCiPe endpoint characterisation factors.

Now, though, we have three shadow prices for the theme photo-oxidant formation:
- the value derived from the impact that $SO_2$ has on the theme;
- the value derived from the impact that $NO_x$ has on the theme;
- the value for the pollutant NMVOC, taken directly from NEEDS.

In order to obtain just one value we first use ReCiPe midpoint characterisation factors to convert the damage costs per kg $NO_x$ and $SO_2$ into damage costs per kg NMVOC, which are different for each pollutant. We then weight the three damage costs using total Dutch emissions of $NO_x$, $SO_2$ and NMVOC (in the year 2008). We now have a shadow price for photo-oxidant formation (in €/kg NMVOC-eq.).

This approach was also applied to other pollutants and themes.

Figure 11 Schematic example of Approach A




Using this approach, which is described in more detail in Annex D, we established the values reported in Table 20 for the respective impact categories.

Table 20 **Weighting factors for emissions in the Netherlands in 2008 by environmental theme ($€_{2008}$/kg-equivalent)**

| Impact category | Damage costs ($€_{2008}$/kg) | | | |
|---|---|---|---|---|
| | Human health | Ecosystems | Cap & land | Total |
| Climate change ($CO_2$-eq.) | 0.00487 | 0.0201 | 0.0201 | 0.0250 |
| Ozone depletion (CFC-11-eq.) | 37.9[1] | | 1.25 | 39.1 |
| Acidification ($SO_2$-eq.) | | 0.453 | 0.185 | 0.638 |
| Photo-oxidant formation (NMVOC-eq.) | 0.585 | | | 0.585 |
| PM formation ($PM_{10}$-eq.) | 51.5 | | | 51.5 |
| Eutrophication, freshwater (P from STP-eq.) | | 1.78 | | 1.78 |
| Eutrophication, terrstrial (N-eq.) | | 14.3 | -2.02 | 12.5 |
| Human toxicity (1.4 DB-eq.) | 0.0206 | | | 0.0206 |
| Ionising radiation (U235-eq.) | 0.0425 | | | 0.0425 |
| Land use ($m^2$ per year) | | 0.612 | | 0.612 |

[1] Based on the implicit valuation of ReCiPe endpoints.

For the damage costs of land use we used a weighted average of the values in Table 8 in Chapter 5, taking into account the land use types that exist in the Netherlands (see Annex C 9). For the damage costs of noise we were unable to derive a single value: see Table 8 in Chapter 5.

### 6.4.2 Weighting factors based on implicit valuation of ReCiPe endpoints

Finally, we established a second method for weighting the various environmental themes, employing direct valuations of the ReCiPe endpoints. As outlined in Section 5.4, for this purpose we used a specific, adjusted, valuation of DALYs and PDFs using the uplift factor of 1.7% (see Annex A.3). Table 21 reports the weighting schemes and the split between human health and ecosystems using a direct valuation of ReCiPe endpoints.

Table 21 **Weighting factors for emissions in the EU in 2008, using direct valuation of ReCiPe endpoints (€$_{2008}$/kg-equivalent)**

| Impact category | Human health | Ecosystems | Total |
|---|---|---|---|
| Climate change ($CO_2$-eq.) | 0.0770 | 0.318 | 0.395 |
| Ozone depletion (CFC-11-eq.) | 96.8 | | 96.8 |
| Acidification ($SO_2$-eq.) | | 0.233 | 0.233 |
| Photo-oxidant formation (NMVOC-eq.) | 0.00215 | | 0.00215 |
| PM formation ($PM_{10}$-eq.) | 14.3 | | 14.3 |
| Eutrophication, freshwater (P from STP-eq.) | | 1.78 | 1.78 |
| Eutrophication, terrestrial (N-eq.) | NA | NA | NA |
| Human toxicity (1.4 DB-eq.) | 0.0386 | | 0.0386 |
| Ionising radiation, Uranium-235 to air (U235-eq.) | | 0.000902 | 0.000902 |
| Land use ($m^2$ per year) | | 0.612 | 0.612 |

As can be seen, a number of noteworthy differences occur with endpoint valuation using ReCiPe midpoint factors, especially in the sphere of climate change. In the next section the reasons for these differences are identified.

## 6.5 Comparison of the sets of weighting factors

A concise summary of the various sets of weighting factors is provided in Table 22. The differences between them will now be discussed in greater detail.

Table 22 **Three sets of weighting factors for emissions in the Netherlands in 2008 (€$_{2008}$/kg-equivalent)**

| Impact category | Weighting set 1 | | Weighting set 2 | Weighting set 3 |
|---|---|---|---|---|
| | 1a | 1b | | |
| Climate change ($CO_2$-eq.) | 0.0250 | 0.0250 | 0.0250 | 0.395 |
| Ozone depletion (CFC-11-eq.) | 30.0 | 30.0 | 39.1 | 96.8 |
| Acidification ($SO_2$-eq.) | 4.13 | 0.594 | 0.638 | 0.233 |
| Photo-oxidant formation (NMVOC-eq.) | 5.00 | 5.00 | 0.585 | 0.00215 |
| PM formation ($PM_{10}$-eq.) | 2.30 | 50.0 | 51.5 | 14.3 |
| Eutrophication, freshwater (P from STP-eq.) | 10.9 | 10.9 | 1.78[1] | 1.78 |
| Eutrophication, marine/terr. (N-eq.)[2] | 7.00 | 7.00 | 12.5 | NA |
| Human toxicity (1.4-DB-eq.)[3] | NA | NA | 0.0206 | 0.0386 |
| Ionising radiation (U235-eq.) | NA | NA | 0.0425 | 0.000902 |
| Land use ($m^2$ per year) | NA | NA | 0.612 | 0.612 |

1 Based on value estimates based on ReCiPe endpoints.
2 For weighting set 1: marine eutrophication; for weighting set 2: terrestrial eutrophication.
3 For weighting set 1: human toxicity has been specified in terms of kg $PM_{10}$-eq. To avoid double counting, this impact is taken into account via the weighting factor for PM formation.

In comparing the three sets of weighting factors several things are immediately apparent:

– For climate change, weighting set 3 (ReCiPe) yields impacts fifteen times greater than sets 1 and 2. The reason for this is that in weighting set 3 damages occurring in the future are not discounted; if this is done, the results are identical to the results obtained with weighting sets 1 and 2.

The same holds for ozone depletion: in this case weighting set 3 yields impacts two to three times greater than sets 1 and 2, because of future damage not being discounted in ReCiPe.[57]

- For the theme of PM formation, the abatement costs with weighting set 1a are substantially lower than the damage costs. The main reason for this is that in calculating the costs we proceeded from the still valid (but highly outdated) regulations embodied in the NeR. If a reliable alternative cost estimate can be made based on the new policy being drawn up for $PM_{2.5}$ (or $PM_{10}$) after 2010, the attendant abatement costs will have to be radically readjusted and the split between $PM_{10}$ formation and acidification will also change. This is reflected in the results of weighting set 1b, which have been calculated with an indicative estimate of € 50/kg $PM_{10}$.
- For the themes of acidification and photo-oxidant formation, the abatement costs (weighting set 1), which are grounded in government policy, are generally higher than the damage costs (weighting sets 2 and 3)[58]. This may be due mainly to the chosen allocation of joint costs. The damage cost method shows that the main impacts of $SO_2$ are to be ascribed to secondary particles, which therefore fall under the theme of PM formation. In the abatement cost approach (weighting set 1a), the main impact of $SO_2$ is on the theme of acidification itself, however, because the value put on the abatement costs for $PM_{2.5}$ is as low as it is because there is as yet no government policy from which these might be reliably derived.
- For the themes of acidification, photo-oxidant formation and PM formation, weighting set 2 yields damage costs that are greater than those obtained with weighting set 3. These differences may be due to differences in geographical orientation: weighting set 2 calculates damages for the Netherlands, while set 3 is concerned with Europe as a whole (with the Netherlands having a greater population density). In addition, for acidification and photo-oxidant formation weighting set 3 makes no allowance for damages to buildings and crops.
- For human toxicity, the damage costs with weighting set 3 are twice as high as with set 2. This may be due to the fact that future damages are not discounted in Approach 3, so that damage estimates are higher.

The precise differences between the environmental modelling and calculation of physical impacts in NEEDS and ReCiPe are discussed in Annex G.

## 6.6 Use of weighting factors for shadow prices for 400 environment pollutants

Based on the weighting factors in sets 1 and 2 and the ReCiPe midpoint characterisation factors, as a final step an extensive list of implicit damage and abatement estimates can now be drawn up. This is done by using the environmental relationship between pollutants contributing to the same theme. This yields an implicit valuation for both the abatement and damage costs. This valuation took place under the following assumptions:

- For the abatement costs, it was assumed that government policy is economically efficient and designed in accordance with the environmental relationship between pollutants (cf. treatment of this issue in Section 4.3.3). As this is certainly not the case, due caution should be exercised when using these implicit abatement costs for the purposes of valuation. In this context we would note that, for reasons set out in Chapter 7, for these

57 Note that with weighing set 2 we have based ozone depletion on ReCiPe endpoints, but in doing so *have* discounted future damages.

58 An exception is the value for acidification in weighting set 1b compared to that value in set 2.

400 pollutants there seems to be only limited scope for using abatement costs for valuation purposes. Their use for weighting is still feasible, though.

- For the damage costs (of emissions to air), weighting set 2 and ReCiPe midpoint characterisation factors were used. A linear relationship was assumed between the contribution of the individual pollutant to the theme (as with acidification, say) and the resultant damage at the endpoint level. NB. For the damage costs of emissions to soil and water, an alternative route was needed. Proxy values for those emissions were determined based on direct valuation of ReCiPe endpoints.

Full lists of implicit damage costs and abatement costs for 400 pollutants are provided separately in Annex J of this report.

# 7 Use of shadow prices

## 7.1 Introduction

This Chapter deals with the practical use of shadow prices and provides guidance on whether abatement costs or damage costs should be used, making a broad distinction between their use for valuation purposes (for estimating external costs and in cost-benefit analyses) and as weighting factors (in LCA and other types of environmental impact analysis). Differences in their use by government and industry are also discussed.

First of all, in Section 7.2, we present the shadow prices calculated according to the abatement cost and damage cost methods. The two sets of prices are compared and the robustness of the estimates is discussed, along with the potential for revision as time progresses. We also discuss the 'shelf life' of the figures: at what point will these sets of shadow prices need to be revised?

Next, in Section 7.3, we consider the use of shadow prices as a valuation method in the context of welfare analysis. This is where external cost estimates and use in cost-benefit analyses are discussed. We elaborate user guidelines and explain the differences between shadow prices based on abatement costs and damage costs with reference to several examples. In Section 7.4 we discuss their use as weighting factors. In doing so, we show that the three weighting sets presented in Section 6.5 may in practice lead to different recommendations and argue under what circumstances which set is preferable. Finally, in Section 7.5, we discuss the use of shadow prices in industry.

## 7.2 General use of shadow prices

### 7.2.1 Use as default values for the average Dutch situation (2008)

The shadow prices proposed in this Handbook will often be used as *default values* for valuation and weighting. They provide an indication of the average value to be assigned to emissions occurring within the Netherlands.

For their use as default values, it is important to be aware how these shadow prices are constructed (cf. Section 1.7.2 and footnote 2 in Section 1.3): they are average prices for pollutant emissions originating on Dutch territory in 2008.[59] The future impacts of these emissions (whether on environmental policy or on endpoints) have been factored into the valuation and, where relevant, discounted to the year of emission.[60] Impacts on populations outside the Netherlands have been assigned the same value as in the case of the resident Dutch population.[61]

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59 They are averages because the type and location of the emission can sometimes have a major impact on the precise value.

60 In a number of cases the VOLY of € 40,000 has not been discounted, because there is already implicit discounting in the CVM (cf. Chapter 5). In the remaining cases, calculations have been based on a risk-free discount rate of 2.5% (i.e with no risk premium), offset by an income elasticity of 0.85 for goods relating to environmental quality and 2% economic growth up to 2030 and 1% subsequently. Cf. Annex A.

61 Because of differences in climate and demography (population density and structure), the impacts may well differ from impacts on residents of the Netherlands, though. The CRFs

The shadow prices are spot estimates relative to the situation in 2008. They can be used for valuing projects that lead to marginal changes. However, if the project leads to non-marginal changes (a halving of emissions, say) these estimates cannot be used. In such cases, targets alter in the abatement cost approach, while background concentrations change in the damage costs approach. In terms of Figure 2 in Section 2.3.2, the entire area under the curve then needs to be estimated rather than working with a spot estimate.

Unless otherwise stated, all the shadow prices in this report are expressed as €/kg emissions (at 2008 price levels). Table 23 lists the damage costs and abatement costs for the most relevant pollutants.

**Table 23 Shadow prices of emissions in the Netherlands in 2008, based on abatement and damage costs (€$_{2008}$/kg)**

| Pollutant | Abatement costs | Damage costs |
|---|---|---|
| $CO_2$ | 0.0250 | 0.0250* |
| $CH_4$ | 0.625 | 0.625 |
| $N_20$ | 7.45 | 7.45 |
| CFC-11 | 149 | 159 |
| CFC-12 | 303 | 324 |
| CFC-113 | 183 | 193 |
| CFC-114 | 278 | 287 |
| CFC-115 | 197 | 202 |
| HCFC-22 | 46.8 | 49.2 |
| $NO_x$ | 8.72 | 10.6 |
| $SO_2$ | 5.00 | 15.4 |
| $NH_3$ | 11.7 | 27.8 |
| NMVOC | 5.00 | 2.54 |
| $PO_4$ | 11 | 1.80 |
| P to water | 10.9 | 1.78 |
| P to soil (fertiliser) | 0.577 | 0.0947 |
| P to soil (manure) | 0.545 | 0.0890 |
| N to water | 7.00 | NA** |
| $PM_{10}$ | 2.30 (50.0)*** | 41.0 |
| $PM_{2.5}$ | 2.30 (50.0)*** | 64.8 |
| Dioxins | 92.00E06 | 5.09E07 |
| As (arsenic) | 466 | 811 |
| Cd (cadmium) | 4,700 | 127 |
| Cr (chromium) | 36,900 | 33.5 |
| Ni (nickel) | 1,800 | 5.37 |
| Pb (lead) | 225 | 408 |

* Based on abatement cost figures (see Annex C.2.4).

** Because there is no damage estimate for nitrogen in NEEDS or ReCiPe, which means the set of damage costs is exclusive of the theme of marine eutrophication.

*** The precise value for $PM_{10}$ and $PM_{2.5}$ is currently unclear. The value of € 2.30/kg is based on outdated policy. New policy is currently under development, though, which may cause the abatement costs to increase to € 50/kg. Cf. Annex B.4.

Comparing the damage costs and abatement costs, we see that for certain heavy metals (cadmium, chromium and nickel) the latter are higher. This also holds for the eutrophying pollutant phosphorus and for NMVOC. In most cases, though, the damage costs exceed the abatement costs. This holds for PM, for

(Concentration Response Functions) have not been calculated country-by-country, however, but for Europe as a whole.

which the damage costs are almost € 65/kg, while the abatement costs are scarcely more than € 2/kg. With $SO_2$, $NO_x$ and $NH_3$, too, the damage costs are substantially higher than the abatement costs. It is important to stress, though, that a comparison between damage and abatement costs does not equate to a rough-and-ready SCBA of environmental policy: these sets of damage and abatement costs are merely a subset of the total costs and benefits associated with such policy (cf. Sections 7.3.1 and 7.3.2). It can therefore not be decided purely on the basis of a comparison between abatement and damage costs whether a particular policy should be tightened or relaxed.

### 7.2.2 Use in future years

The shadow prices apply to the situation in 2008, but can certainly be used for a number of years into the future. With time, though, a certain amount of revision will be unavoidable. We distinguish two kinds of adjustment:

a Adjustment of price level and year.
b Fundamental adjustments.

As time progresses, adjustments may be called for because the prices in a given SCBA have been calculated for the year 2010, for example. The shadow prices presented here should then be adjusted to the new price level by correcting for inflation. In addition, the damage costs require an additional correction in view of the positive income elasticity that exists for environmental quality. This is done by multiplying the growth (or shrinkage) of income relative to the 2008 level by an income elasticity of 0.85 (cf. Annex A). These are non-fundamental adjustments, because they embody no change in the basic system of shadow price calculation.

In the case of $CO_2$ emissions, an important issue is that impacts become increasingly severe as the atmospheric concentration of greenhouse gases rises. The value to be assigned to $CO_2$ emissions consequently depends on expectations regarding future trends in that concentration. Our own figures for future $CO_2$ emissions are reported in Section 5.6.1.

Fundamental adjustments will be required, though, in cases involving changes to the system variables underpinning calculation of the abatement and damage costs. Abatement costs will change if policy targets are revised, if economic trends prove to diverge from projections (at the time of writing, economies are shrinking everywhere) or if there are major technological breakthroughs (or the prices of technologies change because of drastic changes in raw material prices, say).[62] The most likely driver will be new developments in international climate policy, though. If the EU goes for a 30% reduction target in 2020, the shadow price for *both abatement costs and damage costs* rises to € 0.05/kg$CO_2$.[63] Another potential driver is international policy on air pollution: if the NEC targets for 2020 are far more, or less, stringent than the Dutch targets estimated in Annex B (based on current proposals: $NO_x$ 186 kt, $SO_2$ 35 kt, $NH_3$ 119 kt, NMVOC 143 kt), the abatement costs will need to be adjusted accordingly. And European policy on $PM_{2.5}$ is currently being elaborated that will lead to substantially higher abatement costs than the figures cited here (cf. MNP, 2007b).

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62 With respect to negative economic growth: if there are several years of shrinkage and policy targets are not adjusted, it will become cheaper to secure the envisaged emissions cuts and the abatement costs will consequently be lower than projected.

63 The reason why the damage costs also need to be increased is that these costs, for the short term, have been calculated using the abatement costs (cf. Annex C).

In the case of damage costs, the values have more 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 with respect to (a) use of a discount rate for calculating impacts occurring in the distant future; (b) valuation of a statistical human life in the context of environmental pollution; (c) valuation of option values and bequest values. In addition, those environmental themes for which a value has not yet been calculated using the Impact Pathway Approach (e.g. eutrophication) will in the future also become amenable to this route (according to the task description of the EXIOPOL project; cf. Section 1.4).

In conclusion, then, analysts making use of these shadow prices should exercise their own good judgment in assessing whether these prices are still valid for the purpose at hand. With abatement costs, the main issue is to examine whether there have been new policy developments since September 2009; with damage costs, whether the methodology adopted in this study is still to be deemed valid. Regardless of developments on these fronts, we would recommend updating the current set of shadow prices once every 5 to 8 years based on the most recent estimates.

### 7.2.3 Application to Dutch regions and other countries

The two sets of shadow prices have in principle been developed for the Netherlands and cannot therefore simply be transposed without further ado to the regional (provincial) level or to other countries. This is particularly relevant for SCBAs. If an SCBA is carried out specifically at the provincial level, the shadow prices presented here may not simply be used as they stand. With the abatement costs it will be necessary to assess the extent to which provincial government has its own additional policy over and above standing national policy, as when air quality standards are being exceeded, for example. With damage costs, it will need to be assessed how far the provincial situation deviates from the national average, particularly with respect to population density.

As discussed in Section 4.4.1, abatement costs cannot simply be transposed to countries other than the Netherlands. For damage costs, in this project we have proposed a benefit-transfer methodology (cf. Section 5.6.4 and Annex E) that can be used to obtain a very rough estimate of the damage costs arising in other countries.

### 7.2.4 Application to specific emission sources

To an extent, shadow prices also depend on the type of emission source. This holds only for certain specific categories of damage costs, though. With the damage costs of emissions contributing to local environmental impacts – in particular $PM_{2.5}$ and its precursors ($SO_2$, $NO_x$, $NH_3$) – the height at which the emissions take place is also extremely relevant. This is because the resultant health impacts occur at breathing level and emissions at this level in densely populated areas (as with vehicle emissions) will be more damaging than those originating higher up (via stacks, for example).

These problems will have greatest relevance in the case of transport emissions being valued according to damage costs. In such cases use can be made of the set of damage costs developed in the IMPACT project specifically for this purpose (CE, 2007a). In Annex C 3.9 of the present handbook we reproduce the values most frequently used and compare these with the NEEDS estimate used here.

### 7.2.5 Uncertainty

Both sets of shadow prices are characterised by a degree of uncertainty. In the case of abatement costs this is due first of all to interpretations of government policy. At the present time the set can be deemed fairly reliable, because the political targets in question have all been established relatively recently (with the exception of $CO_2$ emissions in 2020). As the years pass, though, their reliability will gradually diminish, because of trends in emissions and changes in policy challenges (i.e. the portion of emissions that still needs reducing). This may be either in a positive or negative sense, for example in the case of lower than projected emissions due to an economic downturn. There may also be any number of revisions to standing policy. On top of this, there is a certain amount of uncertainty regarding the actual costs of the technical measures used to meet policy targets, which are often estimated *ex ante*. Finally, there are costs that are incurred across multiple pollutants and environmental themes. Allocation of these costs to the respective environmental impacts is essentially an arbitrary affair on which there is no scientific consensus on a standard allocation method.

Damage costs are inherently more uncertain than abatement costs. This is due primarily to the wide variation in published estimates of dose-effect relationships and monetary valuation of impacts. Data on these are often adopted from the international literature and corrected for location-specific conditions.

## 7.3 Use in cost-benefit analysis and external cost estimates

One specific use of shadow prices is for putting a monetary value on individual environmental impacts, as with (social) cost-benefit analyses and estimation of external costs. In this section we examine such uses and provide a reasoned exposition of whether abatement costs or damage costs should be used in this context. First of all, we examine the issue of whether the shadow prices calculated in the present study are equivalent to external costs. Next, we discuss the general rule for use of abatement costs in cost-benefit analyses and external cost estimation. Finally, with reference to four illustrative examples we elaborate on the choices to be made in concrete situations.

### 7.3.1 External cost estimates

In economics, external costs, or externalities, are a familiar concept. Environmental impacts are a typical example of an externality: an impact on the welfare of others not taken into account in the decision-making process of the party imposing the impact.[64]

[64] In simple, everyday terms; for a more exact definition see the footnote in Section 2.3.1. In this section we are at any rate not referring to pecuniary externalities.

In principle, externalities are best represented by damage costs, for these are a measure of the change in welfare occurring as a result of environmental pollution. These damage costs are not (necessarily) equivalent to the external costs, however, for two reasons:

1. The damage is not always equal to the value assigned to environmental quality.
2. Some of the damage may already be internalised in government policy.

These will now be discussed in turn.

### The damage is not always equal to the value assigned to environmental quality

The damage estimates are based on the notion that Willingness to Pay corresponds to the actual damage to health, buildings, ecosystems and agricultural crops. In calculations of damage, use has been made of a discount rate (cf. Annex A). While use of such a rate is in itself justified for this purpose (after all, the discount rate represents the opportunity costs of the claim on capital), its value may not necessarily be in line with actual human preferences in this area. Based on notions of stewardship, responsible citizenship or moral imperatives vis-à-vis future generations, people may uphold the moral principle that it is socially undesirable to inflict damage on others – even when such behaviour is not regulated under current institutional arrangements.[65] Ideally speaking, these damage estimates should therefore be extended with a set of moral values. This is beyond the scope of the present study, however. We would recommend, though, that if the abatement costs are *higher* than the damage costs, analysts might argue for taking the abatement costs as a proxy for damages when estimating externalities. This is the line of reasoning adopted by ourselves (and others) in estimates of the short-term damage associated with climate change, for example, where the abatement costs exceed the damage estimates reported in the literature (cf. Annex C).[66]

### Some of the damage may already be internalised in government policy

Under standing government policy, some portion of the damage may already be internalised. For example, a levy on effluent discharges to surface waters means the external costs no longer equal the damage, some of which is now factored into the decisions of the party causing the pollution. In this case the external costs equal the difference between the damages and the charge paid.

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[65] On ethical principles in the context of intergenerational discounting in the climate debate, cf. Davidson (2008).

[66] If the abatement costs exceed the (short-term) damage costs, this may be an indication there are moral values at stake that are not monetised in the damage cost method. It might equally well be argued, though, that government policy is suboptimal (and in fact too stringent). Consequently, no general rule can be given, and the analyst should in such cases make a reasoned decision as to whether abatement costs or damage costs should be adopted.

**Box 8. Distributional effects and internalised risks**

This study reports, among other things, the damage costs associated with the health risks of air pollution. Most of these risks are involuntary. It should be noted, though, that in some cases people elect voluntarily to live in more polluted or noisier areas or properties, provided they receive due compensation in the form of lower property prices (this relationship can be revealed using the Hedonic Pricing method; cf. Annex E). Lower property prices due to pollution mean that home-owners selling their property make less of a profit, so effectively the costs of pollution are not internalised but the effect is merely distributional: there is a transfer of welfare from property sellers to property buyers.

Besides, moving into or out of certain areas, people may internalise certain environmental risks, especially those related to natural disasters and accidents, in two ways: (1) making up-front expenditures to avoid or mitigate losses; (2) purchasing insurance. Regardless of these actions, though, the value of the external costs remains the same, as these costs are not being paid by the party causing the pollution.

In conclusion, shadow prices (based on damage costs) are not always synonymous with estimated external costs. Before using them for the latter purpose, analysts should therefore closely examine the relationship between current government policy and the extent to which non-damage categories might be expressed in people's preferences.

### 7.3.2 Determining the optimum

According to standard economic thinking, a comparison of abatement costs and damage costs should provide information on the extent to which a particular environmental policy (or set of policies) is at the economic optimum or whether it in fact needs tightening or relaxing. This is what is suggested by Figure 1 in Section 2.3.1. Although this is indeed theoretically the case, the two sets of costs cannot simply be compared for this purpose, however, for three reasons:

- Some of the damage costs have already been internalised (see previous section), so it would be more appropriate to compare the remaining external costs with the abatement costs.
- The abatement costs comprise only the costs of technical measures (and in some cases those of output reduction), thus ignoring administrative charges, government failure and so on.
- The shadow prices are exclusive of any indirect (knock-on) costs or benefits of environmental policy.

This means that abatement and damage costs cannot be compared as a means of carrying out a kind of rough-and-ready SCBA. Similarly, there is little point in concluding that policy needs tightening or relaxing purely because of a difference between damage and abatement costs. The most that is indicated by any major difference is that policy may not be optimal.

### 7.3.3 Choice between damage and abatement costs in CBA

In Chapter 2 a general rule was introduced for the use of abatement or damage costs: **if a project leads to changes in environmental quality, damage costs should be used; if it leads to changes in efforts to secure environmental targets, abatement costs should be used.**

The idea behind this rule is simple (and was in fact already stated implicitly in the 'OEI Guidelines'). Every new development (e.g. road-building) leads to additional emissions, and these should be valued using damage costs. For a limited number of environmental themes, though, the government has set absolute targets. These targets determine the environmental quality ultimately achieved and are not (normally) revised because a new road is built. If the targets are to be secured, construction of the road will mean additional measures having to be taken in order to achieve the desired environmental quality. The value of the emissions in question is thus given by the costs of the additional measures that need to be taken to secure the targets. The costs of these measures are precisely the abatement costs.

It is important to note that this rule only holds in areas of environmental policy where *absolute* targets have been set and where those targets are binding.[67] The Dutch government's current targets vis-à-vis the various environmental themes are described in Annex B. At present there are absolute emission caps in place for the following pollutants only: $CO_2$, $SO_2$, $NO_x$, NMVOC and $NH_3$. These targets are specific to the Netherlands. If the SCBA also has the Netherlands as geographical scope, for these emissions the environmental impacts ensuing from a project should be valued according to abatement costs, *unless it is a project that impacts on the targets themselves*.

In those cases where the government levies charges in the absence of absolute targets, this reasoning no longer applies. A charge does not set an absolute cap on the emissions in question, but internalises some fraction of the environmental damage. That damage should therefore in principle be valued by determining the *difference* between the damage costs and the abatement costs, as this represents the external impact that is not yet internalised. Given that abatement costs are based on the *marginal* charge, in such cases it may make sense to value the emissions using damage costs and then deduct from this the charges paid (which do not always equal the marginal charge) to arrive at a figure for the external costs. We would note in this context that the analyst should take good care not to subsequently include the charge as a direct effect (in a SCBA a charge counts only as a distributional effect).

Finally, we would note that in a precisely designed SCBA the general rule comes to the fore of its own accord. Consider a SCBA being conducted for a new motorway, with an increase in $NO_x$ emissions of 2 kt being projected. First the 'zero alternative' is considered, with an assessment being made of how emissions would develop if the motorway were not built, under the assumption of standing $NO_x$ emission targets being secured. The 'project variant' then comprises the road, and the extra $NO_x$ emissions are in the first place monetised using damage costs. At the same time, the analysts are aware that these additional emissions in fact mean that additional policy must be rolled out if international commitments are to be met. The project variant is therefore taken to encompass the policy measures that needs to be implemented over and above standing policy to meet international $NO_x$ commitments. The value of these additional measures are, roughly speaking, equal to the abatement costs. On the benefit side, though, these extra

[67] If the targets are not binding, the abatement costs are zero.

measures will mean a 2 kt decrease in $NO_x$ emissions. In summary, then, it can be said that the damage resulting from extra $NO_x$ emissions can therefore also be valued directly from the abatement costs. Because SCBA practitioners will in many cases not have detailed knowledge of the ins and outs of environmental policies, the above rule provides a simple and handy means for properly monetising environmental impacts.

### 7.3.4 Examples of use

Below we provide a few examples explaining the use of shadow prices in practical cost-benefit analysis, restricting ourselves in each case to environmental impacts. These examples are illustrative rather than exhaustive, and intended solely to elucidate the choice between shadow prices based on abatement and damage costs. Unless otherwise specified, the scale of analysis is the Netherlands, with a 30 year time horizon.

A. *SCBA on road pricing.* In this case there are impacts on emissions of $CO_2$, $NO_x$, NMVOC and $PM_{2.5}$. For the first three of these there are absolute targets in place. For the period up to 2020 these emissions are valued using the abatement costs. Thereafter, they are valued using the damage costs, making due allowance for the positive income elasticities for environmental quality and the rising damage costs of $CO_2$. In the case of $PM_{2.5}$ there are no absolute targets in place, nor any pollution charges. Here, then, valuation is performed according to damage costs for the entire period, again with due allowance made for income elasticities.

B. *SCBA on acidification ceilings for 2020.* In this SCBA the European Commission's current proposals for $SO_2$, $NO_x$, NMVOC and $NH_3$ emissions are subjected to a SCBA. In this case it is the targets themselves that are under scrutiny and the environmental impacts are therefore valued using damage costs. In this SCBA there are consequently no fixed 'targets'.

C. *SCBA on waste incineration versus landfill.* The aim of this SCBA is to assess whether the land filling of waste leads to greater economic welfare than the current practice of incineration with renewable energy recovery. Compared with the latter route, landfill results, *inter alia*, in greater emissions of $CH_4$ (from the landfill site) and $CO_2$ (in the absence of energy generation) and more land use. Emissions of $CH_4$ and $CO_2$ are regulated via Dutch climate policy and should therefore be valued according to abatement costs, while land use should be valued using damage costs. An additional complication arises, though, because the energy generated in municipal waste incinerators helps secure renewable energy targets. If the Netherlands stops producing renewable energy via these incinerators, alternative sources of such energy would have to be exploited to secure the targets agreed at the European level. In this case, then, the $CO_2$ emissions should be valued according to the marginal costs of securing the renewable energy targets (not provided in this report).

D. *SCBA on building a new business estate in a Dutch municipality.* A new business estate attracts more traffic, the emissions of which need to be monetised. In this case the scale of the SCBA is no longer the Netherlands but a municipality, district or province and the national targets are no longer binding: at the local level, additional emissions do not immediately lead to extra damage. The fact that this increase in local emissions implies a need for additional emission cuts in other regions has no consequences for the scale of this SCBA. In this case, therefore, all the environmental impacts should be valued according to damage costs. It should be noted, though, that the damage costs provided in this report are *average* values for the Netherlands as a whole and that in the cited SCBA it would be better to calculate the local damage costs directly, using the EcoSense model, for example.

These examples demonstrate that analysts should always carefully consider whether a SCBA should be elaborated using abatement or damage costs. For a substantial number of environmental impacts it will suffice to work with the latter. For environmental impacts having an effect on absolute targets corresponding with the scale of the SCBA, however, impacts should be valued using abatement costs.

## 7.4 Use in LCA and weighting environmental impacts

In environmental analyses the various environmental impacts identified can be weighted using shadow prices. In many cases these will be Life Cycle Assessments (LCAs) inventorying a product's environmental impacts during the production, consumption and possibly waste phase. In such cases weighting is required if the comparison of products or of phases in a production chain yields different results for different impact categories.

Weighting factors, described in Chapter 6, can be used to weight impacts on different environmental themes so that ultimately a single aggregate score can be given. In this study three sets of weighting factors have been developed:
1. A weighting set with abatement costs characterised at midpoint level.
2. A weighting set with endpoint economic damage costs characterised at midpoint level.
3. A weighting set with direct valuation of health and biodiversity damage at endpoint level.

In the following section, use of these weighting factors is illustrated with reference to an example.

### 7.4.1 Use in weighting environmental impacts

For a better understanding of how the various weighting factors affect the results of an LCA, we compare the production chains of one kilo of beverage carton (liquid packaging board) and one kilo of HDPE, two materials used for packaging beverages like milk. The various environmental impacts are then weighted in three ways to yield a single environmental impact score (in Euros).

The results are shown in Figure 12, Figure 13 and Figure 14. As can be seen, the different weighting sets yield divergent results. Surprisingly, liquid packaging board creates the least environmental impact when weighted according to abatement costs (set 1a) and direct valuation at endpoint (set 3), while it is HDPE that scores best when weighting is based on damage costs at midpoint level (set 2)

Figure 12 Environmental impact scores per kg liquid packaging board and HDPE (in Euros) with weighting based on abatement costs (set 1a)

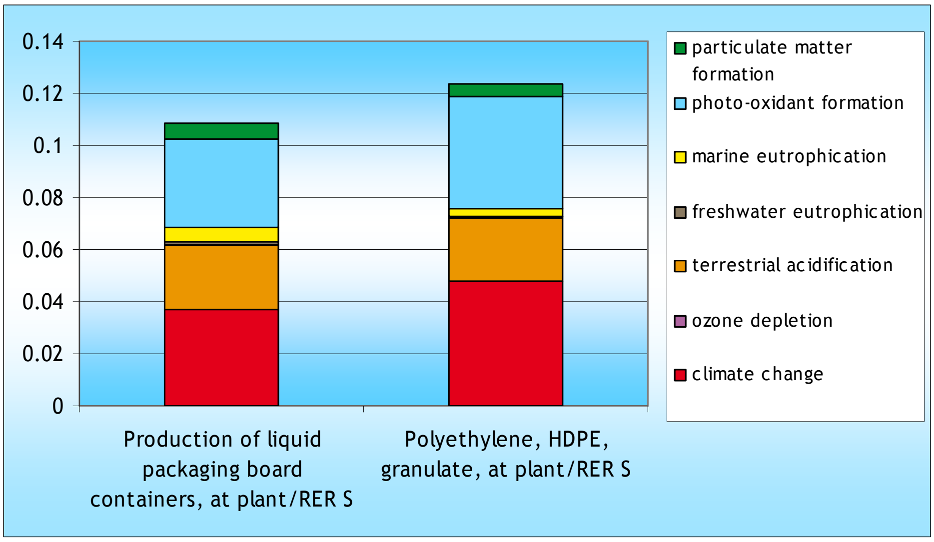



Figure 13 Environmental impact scores per kg liquid packaging board and HDPE (in Euros) with weighting based on economic damage costs at midpoint level (set 2)

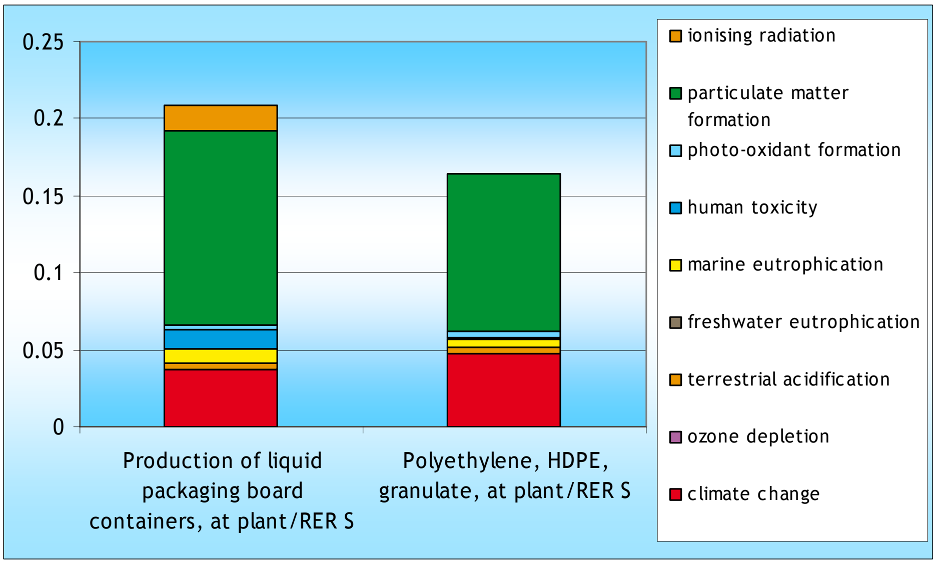
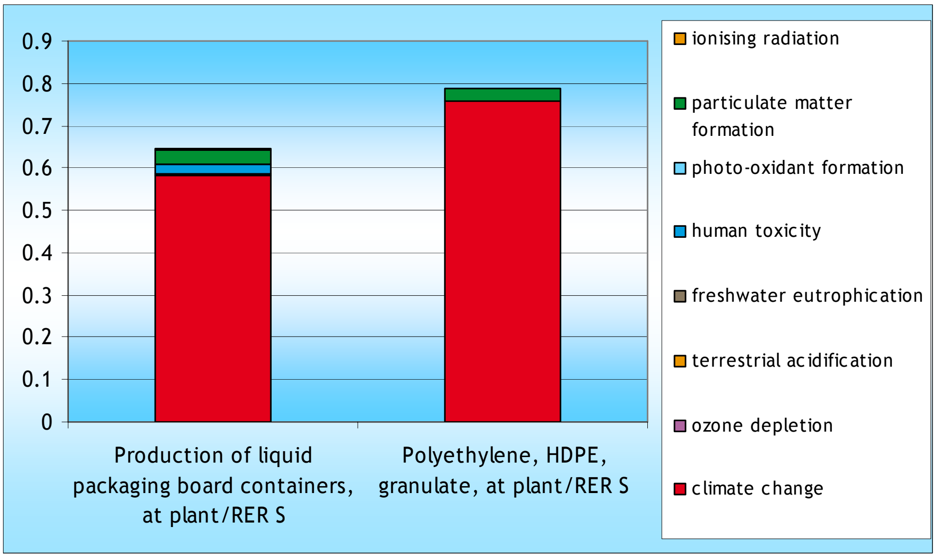


Figure 14 **Environmental impact scores per kg liquid packaging board and HDPE (in Euros) with weighting based on direct damage valuation at endpoint level (set 3)**[68]




The difference in results between the first two sets of weighting factors is due to the relatively large contribution of the theme 'PM formation' in set 2 compared with set 1a. This is because the abatement costs for this theme in set 1a are far lower (€ 2.30/kg) than the damage costs (€ 65/kg), as is therefore the weighting factor, too.

The substantial difference between the two weighting variants based on damage costs is due to a methodological discrepancy between the underlying data models. In ReCiPe (set 3) there is no discounting of damages, which means that major damage occurring in the longer term is assigned the same weight as damage occurring at the present time. As a result, climate impacts become especially important, because these relate particularly to the longer term. NEEDS (set 2) does employ a discount rate (of 3% up to 2030 and 2% thereafter) and with this set it is therefore the short-term health impacts of particulates that make a major contribution. In this way the two sets provide complementary information; if just one set is to be used, set 2 merits preference over set 3, because the discount rate used is generally accepted, although there is room for ethical discussion on the exact figure to be taken for this purpose.

In all likelihood it will have to be decided on a case-to-case basis whether the set of weighting factors based on abatement costs (set 1) or on damage costs (set 2) is preferable. Generally speaking, it can be said that abatement costs are more precise and link up well with calculations of the cost effectiveness of policy measures. Damage costs, on the other hand, are concerned more with impacts on social welfare. Which set of policy measures is preferable therefore depends on the envisaged aim of the weighting process.

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68 In this practical case study, land use has not been included because the associated weighting factor gives an average value for the Netherlands, while the land use impacts associated with production of liquid packaging board and HDPE occur abroad (in Scandinavia). For this environmental theme a different weighting factor should therefore be calculated, based on locally present 'nature types'.

## 7.5 Use of shadow prices in industry

In the past, industry, government at various levels, researchers and NGOs have all used shadow prices to improve their understanding of environmental impacts, weight such impacts and assign them a financial value.[69] The question of whether abatement costs or damage costs are to be used will depend partly on the purpose for which shadow prices are being used. This is determined above all by the particular perspective being adopted. In this Section we consider the perspective that might apply for business and industry.

If a *broad welfare perspective* is adopted, then the aim is to assign a monetary value to the welfare impact of environmental impacts. The underlying assumption here is that the activity or project needs to be assessed from a broader societal perspective. In that case, a combination of damage costs and abatement costs should be employed, following the rules set out in Section 7.4 for monetising emissions. For weighting these emissions, it is best to use a weighting set constructed around *damage costs* (cf. Section 7.5). In social cost-benefit analyses the broad welfare perspective is adopted as standard practice.

If a so-called *bounded rationality* perspective is adopted, on the other hand, then the premise is accepted that a company or policy target group should operate within the bounds of the law. In this perspective a company seeks to contribute to the social values expressed in the political forum. In such cases valuation and weighting of environmental impacts should be performed on the basis of *abatement costs*.

Below we first identify some of the specific user objectives common among policy target groups. In Section 7.3 we have already considered the perspectives adopted by researchers and government analysts conducting cost-benefit analyses.

### 7.5.1 In-house corporate communications

Many companies make internal policy decisions that have an impact on nature and the environment and, increasingly, they are giving due consideration to such impacts. Here, shadow prices can contribute to the decision-making process in two ways:

- In financial analyses or investment decisions, environmental impacts can be included along with financial considerations because they have been monetised using shadow prices. This means they can play a role in the decision-making process.
- In environmental analyses (LCA, EIA), the various environmental impacts identified can be weighted using shadow prices, facilitating decisions as to what is better for the company or the environment.

The question of whether abatement or damage costs should be used for the purpose at hand will depend very much on the background and preferences of the company and the faith placed in the reliability of the results.

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69 The shadow prices published by CE Delft in 2002 (CE, 2002a) have been used in a number of social cost-benefit analyses, including an SCBA on an offshore wind farm (CPB, 2005) and on funding from the Netherlands Economic Structure Enhancing Fund (FES). Within Dutch industry, shadow prices are used mainly for measuring environmental performance, as with the so-called Envirometer or GreenCalc. Since 2002 CE Delft has carried out environmental analyses (both LCAs and EIAs of investments) using its 2002 shadow prices for Campina, Electrabel, Amsterdam's municipal Waste/Energy Company, Nederlandse Aardoliemaatschappij, Thermphos, the Netherlands Association of Paper Nappy Producers, Bouwend Nederland, South Holland provincial executive, Waterbedrijf Europoort and Coca Cola.

What damage costs primarily express is the impact on social welfare in the Netherlands, regardless of any government policy that may be in place. Damage costs are inherently more uncertain, though. They can be used if a company is concerned about the impact of its activities on local residents, for example, and feels called to adopt a 'frontrunner' role as an organisation with responsibilities to society going beyond what current government policy prescribes. By using damage costs a company is taking the implicit stance that the government is not going far enough with its environmental policy and should be doing more, even if that would cost the company more.

What abatement costs primarily express is the impact on the company itself in relation to its operating environment, particularly the government. Abatement costs can be used if the company wants an assessment of the impacts of its operations on nature and the environment, but in doing so does not aspire to going beyond adhering to government targets in the areas in question. Abatement costs are used by companies wishing to shoulder their part of the burden in securing politically agreed environmental policy targets, even if that entails additional expenditure by the company.

### 7.5.2 External corporate communications

Shadow prices are used by industry, in particular, for the purpose of external communications on their environmental performance, frequently for the following reasons:

- Justification of a corporate decision to the competent authority. The environmental impact of the policy decision is communicated to the local, provincial and/or national government.
- Image improvement. Particularly when a company is profiling its 'corporate social responsibility', it may inform the wider public about the overall impact of its entire business, via an environmental annual report, say. In this way it can provide insight into trends in its environmental performance over time.
- Benchmarking. If the overall environmental performance of various different companies is known, benchmarking can be carried out to assess how well a given company scores compared with its competitors.

In all these cases, a summary assessment can in principle be performed with the aim of expressing the overall environmental impacts of a company decision, or a company as such, in a single indicator. The various different impacts can then be weighted using (relative) shadow prices.

In practice, though, shadow prices are less frequently used for this purpose. To the best of our knowledge, none of the Netherlands' provincial authorities use abatement costs or damage costs to assess companies' environmental performance, for example. There is mention in the NeR of the abatement cost method, which can be used for integrated assessment environmental impacts. However, in the European Intergrated Pollution Prevention and Control (IPPC) directive no method is proposed for this purpose. A short series of phone calls indicated that this is not likely to change in the future, either, one of the reasons being the environmental movement's scepticism about a 'trade-off' of emissions via an economic calculus, the fear being that this will mean a watering-down of the IPPC directive.

As far as we know, Thermphos and the Japanese company Osakagas are the only two companies that use shadow prices in their environmental annual report.

### 7.5.3 User guides for individual companies

For Stimular and Thermphos, two organisations that have been instrumental in realising this handbook, specific user guides have been prepared geared to the organisation's particular mission.

# 8 References

**ABT, 2000**
Out of Sight : The Science and Economics of Visibility Impairment, Executive Summary
Boston (MA) : ABT Associates, Inc., 2000

**AEA, 2005**
Methodology for the Cost-Benefit analysis for CAFE
Volume 2: Health Impact Assessment
Didcot : AEA, Technology Environment (AEA), 2005
www.cafe-cba.org, accessed August 2009

**Alfred, 2006**
J. Alfred
Incommensurability and monetary valuation
In : Land Economics 82 (2): 141-161

**Arnesen and Nord, 1999**
T. Arnesen and E. Nord
The value of DALY life: problems with ethics and validity of disability adjusted life years
In : BMJ, no. 319 (1999); p. 1423-1425

**Arrow, 2007**
K.J. Arrow
Global Climate Change : A Challenge to Policy
In : The Economists' Voice, Vol. 4, Issue. 3 (2007); Article 2

**Barnett and Morse, 1963**
H.J. Barnett and Ch. Morse
Scarcity and Growth : The Economics of Natural Resource Availability. 3rd reprint 1973
Baltimore : John Hopkins University Press, 1963

**Bateman et al., 2002**
I.J. Bateman, R.T. Carson, B. Day, M. Hanemann, N. Hanley,T. Hett, M. Jones-Lee, G. Loomes, S. Mourato, E. Ozdemiroglu, D. Pearce, R. Sugden and J. Swanson
Economic Valuation with Stated Preference Techniques. A Manual
Cheltenham (UK) : Edward Elgar, 2002

**Baumol and Oates, 1988**
W.J. Baumol and W.E. Oates
The Theory of Environmental Policy, 2nd edition
New York : Cambridge University Press, 1988

**Brown and Gregory, 1999**
T.C. Brown and R. Gregory
Why the WTA-WTP disparity matters
In : Ecological Economics, no. 28 (1999); p.323-335

**De Bruyn, 2000**
S.M. de Bruyn
Economic growth and the Environment : an empirical analysis
Boston : Kluwer Academic Publishers, 2000

**Burtraw, 1996**
D. Burtraw
The $SO_2$ emissions trading program : cost savings without allowance trades
In : Contemporary Economic Policy vol. 14 (1996) no. 2; p. 79-94

**Bush et al., 1972**
J. Bush, S. Fanshel and M. Chen
Analysis of a Tuberculin Testing Program Using a Health Status Index
In : Socio-economic Planning Sciences 6: 49-68

**Capros and Mantzos, 2000**
P. Capros, L. Mantzos
Kyoto and technology at the European Union : costs of emission reduction under flexibility mechanisms and technology progress
In : Int. J. Global Energy Issues, 14 pp. 169-183, 2000

**CASES, 2008**
External Costs database (Euro per ton values)
Deliverable D.2.2, Cost Assessment of Sustainable Energy Systems (CASES) project
http://www.feem-project.net/cases/downloads_presentation.php
accessed 18 April 2009

**CE, 1998**
G.C. Bergsma en G. de Weerd
De Van Melle MKB-Milieubarometer
Delft : CE Delft, 1998

**CE, 2002a**
M.D. Davidson
Update schaduwprijzen, financiële waardering van milieuemissies op basis van Nederlandse overheidsdoelen
Delft : CE Delft, 2002

**CE, 2002b**
M.D. (Marc) Davidson, R.C.N. (Ron) Wit
Weging van emissiereducties : Is explicitering en uniformering zinvol voor de Rijksoverheid?
Delft : CE Delft, 2002

**CE, 2005**
S.M. (Sander) de Bruyn, M.J. (Martijn) Blom, R.C.N. (Ron) Wit,
H.J. (Harry) Croezen, G.E.A. (Geert) Warringa, B.E. (Bettina) Kampman, met assistentie van P. (Piet) Boonekamp (ECN)
Evaluatie doelmatigheid binnenlands klimaatbeleid Kosten en effecten, 1999-2004
Delft : CE Delft, 2005

**CE, 2007a**
M. Maibach, C. Schreyer, D. Sutter (INFRAS) ; H.P. van Essen, B.H. Boon, R. Smokers, A. Schroten (CE Delft) ; C. Doll (Fraunhofer Gesellschaft – ISI) ; B. Pawlowska, M. Bak (University of Gdansk)
Internalisation Measures and Policies for All external Costs of Transport (IMPACT), Deliverable 1 : Draft Handbook on estimation of external costs in the transport sector, revised version
Delft : CE Delft, 2007

**CE, 2007b**
S.M. (Sander) de Bruyn, M.J. (Martijn) Blom, A. (Arno) Schroten, M. (Machiel) Mulder
SCBA Guidelines for Environemntal Policy
Delft : CE Delft, 2007

**CE, 2008a**
E.P. (Ewout) Dönszelmann, S.M. (Sander) de Bruyn, M.H. (Marisa) Korteland, F. (Femke) de Jong, M.N. (Maartje) Sevenster (allen CE Delft) ; M. (Michiel) Briene, M. (Manfred) Wienhoven, J. (Jaap) Bovens (allen Ecorys)
Maatschappelijke effecten vermindering luchtverontreiniging: MKBA van mogelijke NEC-plafonds
Delft : CE Delft, 2008

**CE, 2008b**
M. Maibach, C. Schreyer, D. Sutter (INFRAS) ; H.P. van Essen, B.H. Boon, R. Smokers, A. Schroten (CE Delft) ; C. Doll (Fraunhofer Gesellschaft – ISI) ; B. Pawlowska, M. Bak (University of Gdansk)
Handbook on estimation of external costs in the transport sector. Internalisation Measures and Policies for All external Costs of Transport (IMPACT), Version 1.1
Delft : CE Delft, 2008

**CE, 2008c**
B. Kampman, L. de Boer and M. Otten
Kosten en effecten van beleidsmaatregelen
Delft : CE Delft, 2008

**Cleveland, 1991**
C.J. Cleveland
Natural Resource Scarcity and Economic Growth Revisited : Economic and Biophysical Perspectives
In : Ecological Economics : The Science and Management of Sustainability / R. Constanza (ed). New York : Columbia University Press, New York, p. 289-317

**Commissie Integraal Waterbeheer, 1998**
Werkgroep III
Schadelijkheid versus maatstaf, notitie
S.l. : Coordinatiecommissie Uitvoering Wet Verontreniging Oppervlaktewater (CUWVO), 1998

**Constanza et al., 1997**
R. Constanza
The value of the world's ecosystem services and natural capital
In : Nature, vol. 387, no. 6630 (1997); p. 253-260

**CPB, 2000**
C.J.J. Eijgenraam, C.C. Koopmans, P.J.G. Tang and A.C.P. Verster
Evaluatie van infrastructuurprojecten : leidraad voor Kosten-batenanalyse
Onderzoeksprogramma Economische Effecten Infrastructuur
Den Haag : Centraal Planbureau (CPB), 2000

**CPB, 2003**
R.A. de Mooij and P. Tang
Four Futures of Europe
Den Haag : Centraal Planbureau (CPB), 2003

**CPB, 2005**
Annemiek Verrips (CPB), Hage de Vries (ECN), Ad Seebregts (ECN) en Mark Lijesen
Windenergie op de Noordzee : een maatschappelijke kosten-batenanalyse
Den Haag : Centraal Planbureau (CPB), 2005

**Daly and Cobb, 1989**
H.E. Daly and J.C. Cobb
For the common good
Boston : Beacon Press, 1989

**Davidson, 2006**
M.D. Davidson
A social discount rate for climate damage to future generations based on regulatory law
In : Climate Change, vol. 76, no.1-2, (2006); p. 55-72

**DLR, 2006**
Wolfram Krewitt, Barbara Schlomann
Externe Kosten der Stromerzeugung aus erneuerbaren Energien im Vergleich zur Stromerzeugung aus fossilen Energieträgern
Stuttgart/Karlsruhe : Deutsches Zentrum für Luft- und Raumfahrt (DLR)/ Fraunhofer Institut für System- und Innovationsforschung (ISI), 2006

**Dziegielewska et al., 2007**
D. Dziegielewska, T. Tietenberg, N. Niggol Seo (eds.)
Total Economic Value : topic in Encyclopedia of Earth
http://www.eoearth.org/article/Total_economic_value
accessed on 20 September 2009

**EC, 2007**
Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions - Limiting global climate change to 2 degrees Celsius - The way ahead for 2020 and beyond (COM/2007/0002 final)
Brussels : Commission of the European Communities, 2007

**ECN, 2005**
Dril, A.W.N van en H.E. Elzenga
Referentieramingen 2005-2020, ECN-MNP, ECN-rapport C-05-018, RIVM-rapport 773001031
Petten : ECN, 2005
(http://www.rivm.nl/bibliotheek/rapporten/773001031.pdf)

**ECN, 2006**
B.W. Daniëls, J.C.M. Farla
Optiedocument energie en emissies : 2010-2020
Petten : Energieonderzoek Centrum Nederland (ECN), 2006

**ECN/MNP, 2006**
B.W. Daniëls, J.C.M. Farla
Optiedocument energie en emissies 2010/2020
Petten ; Bilthoven : Energieonderzoek Centrum (ECN) ; Milieu- en Natuurplanbureau (MNP), 2006

**ECN, 2009**
P. Kroon, W. Wetzels
Onderbouwing actualisatie BEES B. : Kosten en effecten van de voorgenomen wijziging van het besluit emissie-eisen stookinstallaties
Petten : Energieonderzoek Centrum Nederland (ECN), 2009

**ESPREME, 2007**
Alison Searl
ESPREME, Estimation of willingness-to-pay to reduce risks of exposure to heavy metals and cost-benefit analysis for reducing heavy metals occurrence in Europe
Workpackage 6 – D05a, Exposure response functions for HM impacts on human health
http://espreme.ier.uni-stuttgart.de/
accessed 13 August 2009

**ExternE, 2005**
P. Bickel and R. Friedrich
Externalities of Energy, Methodology (ExternE), 2005 update
Luxembourg : European Commission, 2005
http://www.externe.info/brussels/methup05a.pdf
accessed 20 August 2009

**Fantke, 2008**
P. Fantke
Appendix: Parameterisation of the environmental fate and exposure models of WATSON
In : Bachmann, T.M., Fantke, P., Jolliet, O., Margni, M. and Rosenbaum, R.
NEEDS Project, FP6
Rs1b_D2.1 : Methodological improvements for the assessment of external costs due to indirect human exposure through ingestion and due to further substances so far unaddressed

**Freeman, 1993**
A.M. Freeman III
The Measurement of Environmental and Resource Values : Theories and Methods
Washington DC : Resource for the Future, 1993

**Goedkoop et al., 2009**
M.J. Goedkoop, R. Heijungs, M. Huijbregts, A. De Schryver, J. Struijs and R. van Zelm
ReCiPe 2008 : A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level, First edition Report I: Characterisation
http://www.lcia-ReCiPe.net
accessed 6 January 2009

**Guinée et al., 2002**
Jeroen B. Guinée (eindred.), Marieke Gorrée, Reinout Heijungs, Gjalt Huppes, René Kleijn, Arjan de Koning, Lauran van Oers, Anneke Wegener Sleeswijk, Sangwon Suh, Helias A. Udo de Haes, Hans de Bruijn, Robbert van Duin, Mark A.J. Huijbregts
Handbook on life cycle assessment : operational guide to the ISO Standards
Dordrecht (et al.) : Kluwer Academic Publishers, 2002

**Hanley et al., 2001**
N. Hanley, J.F. Shogren and B. White
Introduction to Environmental Economics
New York : Oxford University Press Inc., 2001

**Harrington, 2000**
W. Harrington, R.D. Morgenstern, P. Nelson
On the accuracy of regulatory cost estimates
In : Journal op Policy Analysis and Management vol. 19, (2000) no. 2; p. 297-322

**Hayashi et al., 2006**
K. Hayashi, A. Nakagawa, N. Itsubo and A. Inaba
Expanded Damage Function of Stratospheric Ozone Depletion to Cover Major Endpoints Regarding Life Cycle Impact Assessment
In : The International Journal of Life Cycle Assessment , vol.11, no.3 (2006); p. 150–161

**HEATCO, 2006**
P. Bickel (et al.)
Developing Harmonised European approaches for Transport Costing and Project Assessment (HEATCO)
Deliverable D5 : Proposal for Haromised Guidelines
Stuttgart : IER, Germany, 2006

**Hennipman, 1977**
P. Hennipman, Edited by H. van den Doel and A. Heertje
Welvaartstheorie en economische politiek
Alphen a/d Rijn : Samson, 1977

**Hökby en Söderqvist, 2003**
S. Hökby, T. Söderqvist
Elasticities of Demand and Willingness to Pay for Environmental Services in Sweden
In : Environmental and Resource Economics, Vol. 26 , no. 3 (2003); p. 361-383

**Hoevenagel and De Bruyn, 2008**
R. Hoevenagel en S.M. de Bruyn
Nog weinig waardering voor milieuwaardering
In : F. Oosterhuis, et al., ‘Aan schaarste geen gebrek’, p. 31-42

**Hueting, 1980**
R. Hueting
New scarcity and economic growth
Amsterdam/Brussel : Agon Elsevier, 1974 (Dutch edition)
Amsterdam : North-Holland Publishing Company, 1980 (English edition)

**Hueting and De Boer, 2004**
R. Hueting, B. de Boer
Sustainable national income and multiple indicators for sustainable development
In : Measuring sustainable development, integrated economic, environmental and social frameworks, p. 39-52
Paris : OECD, 2004

**Huppes et al., 2007**
G. Huppes, M.D. Davidson, J. Kuyper, L. van Oers, H.A. de Haes and G. Warringa
Eco-efficient environmental policy in oil and gas production in The Netherlands
In : Ecological Economics, vol. 61, no. 1 (2007); p. 43-51

**IIASA, 2006**
M. Amann, W. Asman, I. Bertok, J. Cofala, C. Heyes, Z. Klimont, M. Posch, W. Schöpp and F. Wagner
Emission control scenarios that meet the environmental objectives of the Thematic Stratgey on Air Pollution
In : NEC Scenario Analysis report no. 2, IIASA, 2006

**IIASA, 2007**
Cost-effective Emission Reductions to meet the Environmental Targets of the Thematic Strategy on Air Pollution under Different Greenhouse Gas Constraints
NEC Scenario Analysis Report No. 5
Laxenburg : IIASA, 2007

**IIASA, 2008**
National Emission Ceilings for 2020 based on the 2008 Climate & Energy Package
NEC Scenario Analysis Report No. 6
Laxenburg : IIASA, 2008

**INFRAS/IWW, 2004**
C. Schreyer, M. Maibach, W. Rothengatter, C. Doll, C. Schneider, D. Schmedding (INFRAS; IWW)
External costs of transport : update study
Karlsruhe/Zürich/Paris : the International Union of Railways (UIC), 2004

**IVM, 1972**
H.M.A. Jansen, en J.B. Opschoor
De invloed van geluidshinder op de prijzen van woningen
Amsterdam : VU, Instituut voor Milieuvraagstukken (IVM), 1972

**IVM, 2006**
F. Oosterhuis (ed.), et al.
Ex-post estimate of costs to business of EU environmental legislation : final report
Amsterdam : VU, Institute for Environmental Studies (IVM), 2006

**JRC, 2007**
P. Russ, T. Wiesenthal, D. Regemorter and J.C. Císcar
Global Climate Policy Scenarios for 2030 and Beyond : Analysis of Greenhouse Gas Emission Reduction Pathway Scenarios with the POLES and GEM-E3 models
Luxembourg : Office for Official Publications of the European Communities, 2007
http://ec.europa.eu/dgs/jrc/downloads/jrc_reference_report_2007_11_climatepolicy.pdf
accessed 20 August 2009

**Kuik et al., 2008**
O. Kuik, L. Brander, N. Nikitina, S. Navrud, K. Magnussen and E.H. Fall
Report on the monetary valuation of energy related impacts on land use changes, acidification, eutrophication, visual intrusion and climate change
Deliverable D3.2 CASES project
http://www.feem-project.net/cases/documents/deliverables/D_03_2%20non%20human%20ext%20cost%20final.pdf
accessed October 2009

**Kristrom and Riera, 1996**
B. Kristrom and P. Riera
Is the income elasticity of environmental improvements less than one?
In : Environmental and Resource Economics, Vol.7, No.1 (1996); p. 45-55

**Krupnick et al., 2002**
A. Krupnick, A. Alberini, M. Cropper, N. Simon, B. O'Brien, R. Goeree and M. Heintzelman
Age, Health and the Willingness to Pay for Mortality Risk Reductions : A Contingent Valuation Survey of Ontario Residents
In : Journal of Risk and Uncertainty, vol. 24, no. 2 (2002); p. 161-186

**Lopez and Murray, 1998**
A.D. Lopez and C.J.L. Murray
The Global Burden of Disease, 1990-2020
In : Nature Medicine, Vol. 4, no. 11 (1998); p. 1241-1243

**Lynas, 2008**
M. Lynas
Zes graden : Onze toekomst op een warmere planeet
Utrecht : Uitgeverij Jan van Arkel, 2008

**Mauskopf and Moreles, 2000**
J. Muskopf and R.A. Morales
Measuring the Pain : Suffering and Functional Disability Associated with Foodborne Illness, proceeding from a conference 'Valuing the Health Benefits of Food Safety', University of Maryland, 14-15 September 2000

**MethodEx, 2006**
Methods and data on environmental and health externalities: harmonising and sharing of operational estimates
BeTa-MethodEx look-up tables for damage costs per tonne of emission for regional and global air pollutants, including trace metals and some organics
http://www.methodex.org/news.htm
Accessed July 2009

**Miedema and Oudshoorn, 2001**
H.M.E. Miedema and C.G. Oudshoorn
Annoyance from transportation noise: relationships with exposure metrics DNL and DENL and their confidence intervals
In : Environmental Health Perspectives, vol. 109, no. 4 (2001); p. 409-416

**Mishan, 1981**
E.J. Mishan
Introduction to normative economics
New York : Oxford University Press, 1981

**MNP, 2005**
Beoordeling van het Prinsjesdagpakket Aanpak Luchtkwaliteit 2005
Bilthoven : Milieu en Natuur Planbureau (MNP), 2005

**MNP, 2007a**
W.L.M. Smeets, W.F. Blom, A. Hoen, B.A. Jimmink, R.B.A.Koelemeijer, J.A.H.W. Peters, R. Thomas, W.J. de Vries
Kosteneffectiviteit van aanvullende maatregelen voor een schonere lucht
Bilthoven : Milieu- en Natuur Planbureau (MNP), 2007

**MNP, 2007b**
J. Matthijsen and H. ten Brink
$PM_{2,5}$ in the Netherlands : Consequences of the new European air quality standards
Bilthoven : Milieu en Natuur Planbureau (MNP), 2007

**MNP, 2008**
G.J.M. Velders, J.J.M. Aben, W.F. Blom, J.D. van Dam, H.E.Elzenga, G.P. Geilenkirchen, P. Hamming, A. Hoen, B.A. Jimmink, R.B.A. Koelemeijer, J. Matthijsen, C.J. Peek, C.B.W. Schilderman, O.C. van der Sluis, W.J. de Vries
Concentratiekaarten voor grootschalige luchtverontreiniging in Nederland : Rapportage 2008
Bilthoven : Milieu- en Natuurplanbureau (MNP), 2008

**Murray, 1994**
C.J.L. Murray
Quantifying the Burden of Disease: the Technical Basis for Disability-Adjusted Life Years
In : Bulletin World Health Organization, vol. 7, no. 3 (1994); p. 429–445

**Murray, 1996**
C.J.L. Murray, Rethinking DALYs
In : The Global Burden of Disease/ C.J.L. Murray and A.D. Lopez (ed.)
Cambridge, MA. : Harvard University Press, 1996

**NEEDS, 2006**
W. Ott, M. Baur, Y. Kaufmann, R. Frischknecht and R. Steiner
Assessment of Biodiversity Losses
NEEDS deliverable D.4.2.- RS 1b/WP4,
Priority 6.1: Sustainable Energy Systems and, more specifically
Sub-priority 6.1.3.2.5: Socio-economic tools and concepts for energy strategy
http://www.needs-project.org/docs/results/RS1b/RS1b_D4.2.pdf

**NEEDS, 2007a**
S. Navrud
Value Transfer Techniques and Expected Uncertainties
NEEDS deliverable No 2.1 – RS 3a
Priority 6.1: Sustainable Energy Systems and, more specifically,
Sub-priority 6.1.3.2.5: Socio-economic tools and concepts for energy strategy.
http://www.needs-project.org/2009/Deliverables/Needs_Rs3a_D2.1.doc

**NEEDS, 2007b**
R. Torfs, F. Hurley and B. Miller
Final Report on casual links between pollutants and health impacts – A set of concentration-response functions
NEEDS deliverable Rs1b_D3.7
Priority 6.1:Sustainable Energy Systems and, more specifically
Sub-priority 6.1.3.2.5: Socio-economic tools and concepts for energy strategy
http://www.needs-project.org/docs/results/RS1b/NEEDS_Rs1b_D3.7.pdf

**NEEDS, 2007c**
D. Anthoff
Report on marginal external damage costs inventory of greenhouse gas emissions
NEEDS deliverable 5.4-RS1b
Priority 6.1: Sustainable Energy Systems and, more specifically
Sub-priority 6.1.3.2.5: Socio-economic tools and concepts for energy strategy
http://www.needs-project.org/2009/Deliverables/RS1b%20D5.4-5.5.pdf

**NEEDS, 2008a**
P. Preiss, R. Friedrich and V. Klotz
Report on the procedure and data to generate averaged/aggregated data
NEEDS deliverable No. 1.1 – RS 3a
Priority 6.1: Sustainable Energy Systems and, more specifically
Sub-priority 6.1.3.2.5: Socio-economic tools and concepts for energy strategy
http://www.needs-project.org/2009/Deliverables/RS3a%20D1.1.zip

**NEEDS, 2008b**
J. Spadaro, and A. Rabl
Final Report on the Uncertainty on the Transfer/Generalization of ExternE Results (revised in March 2009)
NEEDS deliverable D 3.2
Priority 6.1: Sustainable Energy Systems and, more specifically
Sub-priority 6.1.3.2.5: Socio-economic tools and concepts for energy strategy
http://www.needs-project.org/2009/Deliverables/RS3a%20D3.1.zip

**NEEDS, 2008c**
B. Desaigues, D. Ami, A. Bartczak, M. Braun-Kohlova,S. Chilton, V. Farreras, A. Hunt, M. Hutchison,C. Jeanrenaud, P. Kaderjak, V. Maca,O. Markiewicz, H. Metcalf, S. Navrud,J.S. Nielsen,R. Ortiz, S. Pellegrini, A. Rabl, R. Riera, M. Scasny, M-E. Stoeckel, R. Szanto and J. Urban
Final report on the monetary valuation of mortality and morbidity risks from air pollution
NEEDS deliverable no. 6.7
Priority 6.1: Sustainable Energy Systems and, more specifically,
Sub-priority 6.1.3.2.5: Socio-economic tools and concepts for energy strategy
http://www.needs-project.org/docs/results/RS1b/NEEDS_RS1b_D6.7.pdf

**NOGEPA, 1996**
Sas, H., G. Huppes, E. de Haan. and J. Kuyper, J.
Weegfactoren voor bedrijfsmilieuplannen NOGEPA (Weighting factors for firms' environmental policy plans in NOGEPA)
Den Haag : NOGEPA, 1996

**Nijland and van Wee, 2008**
H. Nijland and B. van Wee
Noise valuation in ex-ante evaluations of major road and railroad projects
In: European Journal of Transport and Infrastructure Research, vol. 8, (2008); p. 216-226

**Ott et al., 2006**
W. Ott, M. Baur, R. Iten and A. Vettori
Konsequente Umsetzung des Verurserprinzips : Umwelt-Materialien Nr. 201
Bern : BUWAL, 2005

**PBL, 2009**
G.J.M. Velders, J.J.M. Aben, W.F. Blom, J.D. Dam, H.E. Elzenga, G.P. Geilenkirchen, P. Hammingh, A. Hoen, B.A.Jimmink, R.B.A. Koelemeijer, J. Matthijsen, C.J. Peek, C.B.W. Schilderman, O.C. van der Sluis, W.J. de Vries
Concentratie kaarten voor grootschalige luchtverontreiniging in Nederland Rapportage 2009
Bilthoven/Den Haag : Planbureau voor de Leefomgeving (PBL), 2009

**Power, 1996**
T.M. Power
Environmental protection and economic well-being : The economic persuit of quality
New York : M.E. Sharpe, Publishers, 1996

**PRé, 2000**
M. Goedkoop and R. Spriensma
The Eco-indicator 99 : A damage-oriented method for Life Cycle Impact Assessment, Methodology Report, second edition
Amersfoort : PRé Consultants B.V., 2000

**Rabl, 1999**
A. Rabl
Air Pollution and Buildings: an Estimation of Damage Costs in France
In : Environmental Impact Assessment Review, vol. 19, no. 4 (1999); p. 361–385

**Rabl and Spadaro, 1999**
A. Rabl, J.V. Spadaro,
Environmental Damages and Costs: an Analysis of Uncertainties
In : Environment International, vol.25, no. 1 (1999); p. 29-46

**Ready et al., 2002**
R.C. Ready, J. Malzubris, S. Senkane
The relationship between environmental values and income in a transition economy surface water quality in Latvia
In : Environment and Development Economics, vol. 7, no. 1 (2002); p. 147-156

**RECORDIT, 2000**
S. Weinreich (ZEW) et al.
Real Cost Reduction of Door-to-Door Intermodal Transport (RECORDIT
Deliverable 1: Accounting Framework
Mannheim : ZEW Center for European Economic Research, 2000

**Read, 1963**
J.E. Read
The Trail Smelter Dispute
In : The Canadian Yearbook of International Law, 1 (1963); p. 213-229

**RIVM, 2000**
B.J.E. Ten Brink, A.J.H. van Vliet,C. Heunks, D.W. Pearce, A. Howarth
Technical Report on Biodiversity in Europe : An integrated economic and environmental assessment
Bilthoven : RIVM, 2000

**RIVM , 2008**
A. van Dijk, P.N. de Outer, H. Slaper
Climate and ozone change effects on ultraviolet radiation and risks (COEUR)
Using and validating earth observation
Bilthoven : RIVM, 2008

**RIVM, 2009**
Risico's van stoffen : Overzicht Nederlandse prioritaire stoffenlijst
http://www.rivm.nl/rvs/stoffen/prio/totale_prior_stoffenlijst.jsp
accessed August 2009

**RMNO, 2008**
Aart de Zeeuw, Roel in 't Veld, Daan van Soest, Louis Meuleman and
Paul Hoogewoning
Social Cost Benefit Analysis for Environmental Policy Making
Den Haag : RMNO, 2008

**Robbins, 1935**
L. Robbins
An Essay on the Nature and Significance of Economic Science, 2nd edition
London : Macmillan, 1935

**Ruijgrok et al., 2004**
E.C.M. Ruijgrok, R. Brouwer, H. Verbruggen, E.J. Bos
Waardering van natuur, water en bodem in maatschappelijke kosten baten analyses: een handreiking ter aanvulling op de OEI-Leidraad
Den Haag : Ministerie van Verkeer en Waterstaat ; Ministerie van Economische Zaken, 2004

**RWS, 2009**
Discontovoet
Den Haag : Ministerie van Verkeer en Waterstaat, Rijkswaterstaat, 2009
http://www.rws.nl/dvs/themas/leefbaarheid/economie/see/15e_Nieuwsbrief/Artikel_1_Discontovoet.jsp

**Sassi, 2006**
F. Sassi
'How to do (or not to do)… : Calculating QALYs, comparing QALY and DALY calculations'
In : Health Policy and Planning, vol. 21, no. 5 (2006); p. 402-409

http://heapol.oxfordjournals.org/cgi/reprint/21/5/402

**SEDAC, 2006**
Center for International Earth Science Information Network (CIESIN), Columbia University; and Centro Internacional de Agricultura Tropical (CIAT)
Gridded Population of the World, version 3 (GPWv3)
Columbia University Palisades (NY) : Socioeconomic Data and Applications Center (SEDAC), 2006
http://sedac.ciesin.columbia.edu/gpw
accessed February 2009

**SEI, 1999**
Costs and strategies presented by industry during the negotiation of environmental regulations
Stockholm : Stockholm Environmental Institute (SEI), 1999

**SenterNovem, 2009**
Nederlandse Emissierichtlijn Lucht
Den Haag : SenterNovem (InfoMil), 2009

**Simon, 1981**
J.L. Simon
The Ultimate Resource
Princeton (NJ) : Princeton University Press, Princeton,1981

**Spadaro and Rabl, 2008**
J.V. Spadaro and A.Rabl
Global Health Impacts and Costs Due to Mercury Emissions
In : Risk Analysis, Vol. 28, No. 3 (2008); p. 603-613

**Staatsblad van het Koninkrijk der Nederlanden, 2003**
230 Besluit van 25 augustus 2003, houdende nieuwe regels betreffende stoffen die de ozonlaag afbreken (Besluit ozonlaagafbrekende stoffen Wms 2003)\
Den Haag : SDU, 2003

**Steen, 1999**
B. Steen
A systematic approach to environmental priority strategies in product development (EPS) version 2000 – models and data of the default method
Göteborg : Chalmers university of technology, 1999

**Stern, 2006**
N. Stern
The Economics of Climate Change : The Stern Review
London : Cabinet Office - HM Treasury, 2006

**Thompson et al., 1990**
M.Thompson, R., J. Ellis and A. Wildavsky
Cultural Theory
Boulder : Westview, 1990

**TNO, 2002**
A.K. van Harmelen, H.J.G. Kok, A.J.H. Visschedijk
Potentials and costs to reduce $PM_{10}$ and $PM_{2.5}$ emissions from industrial sources in the Netherlands
Apeldoorn : TNO-MEP, 2002

**Tol, 2005**
R.S.J. Rol
The marginal damage costs of carbon dioxide emissions : an assessment of the uncertainties
In : Energy Policy, vol. 33, no. 16 (2005); p. 2064-2074

**Tol, 2008**
R.S.J. Tol
The Social Cost of Carbon : Trends, Outliers and Catastrophes
Kiel : Kiel Institute for the World Economy, 2008

**UNITE, 2003b**
P. Bickel, S. Schmid, J. Tervonen, K. Hämekoski, T. Otterström, P. Anton, R. Enei, G. Leone, P. van Donselaar, H. Carmigchelt
Unification of accounts and marginal costs for transport efficiency (UNITE)
Deliverable 11 : Environmental marginal cost case studies
Stuttgart : IER, University of Stuttgart, 2003

**UNSCEAR, 1993**
United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR)
Sources and effects of ionizing radiation : Report to the General Assembly, with Scientific Annexes.
http://www.unscear.org/unscear/en/publications/1993.html
accessed June 2009

**UNSCEAR, 2000**
United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR)
UNSCEAR 2000, Report Vol. I, UNSCEAR 2000 Report to the General Assembly, with scientific annexes, Volume I: Sources
http://www.unscear.org/unscear/en/publications/2000_1.html

**Vogtlander and Bijma, 2000**
J. Vogtlander, A. Bijma
The Virtual Pollution Prevention Costs '99 : a single LCA-based indicator for emissions
In : The International Journal of Life Cycle Assessment 5(2): 113–124

**VROM, 1993**
Albert Adriaanse
Environmental policy performnace indicators
Den Haag : Ministerie van VROM, 1993

**VROM, 1994**
Emissies in Nederland, trends, thema's en doelgroepen 1993 en ramingen 1994
Den Haag : Ministerie van VROM, 1994

**VROM, 2000**
Eco-indicator 99. Manual for designers. A damage oriented method for Life Cycle Impact Assessment
Den Haag : Ministerie van VROM, 2000
http://www.pre.nl/download/EI99_Manual.pdf
accessed on 27 October 2008

**VROM, 2002**
Ontwerp Landelijk Afvalbeheerplan 2002-2012, inspraakversie 11 januari 2002
Den Haag : Ministerie van VROM, 2002

**VROM, 2007a**
Schoon en Zuinig = Clean and Efficient' programme
Den Haag : Ministerie van VROM, 2007

**VROM, 2007b**
Memorie van Toelichting bij de Vaststelling van de begrotingsstaten van het Ministerie van VROM en van de begrotingsstaat van het Waddenfonds voor het jaar 2008, kamerstuknr. 31 200 XI
Den Haag : Ministerie van VROM, 2007

**VROM, 2008a**
Ontwerp Landelijk afvalbeheerplan 2009 – 2021 : Naar een materiaalketenbeleid (inspraakversie 8 december 2008)
Den Haag : Ministerie van VROM, 2008

**VROM, 2008b**
Fijn stof en BBT : Achtergrondrapportage Actieplan fijn stof en industrie
Den Haag : Ministerie van VROM, 2008

**VROM, 2008c**
Arthur D. Little
Indicatieve kosteneffectiviteit en nationale impact van relevante bestaande oplossingen voor $PM_{2,5}$ reductie in 2030
In : Verkenning milieutechnologieën NMVOS, ammoniak en fijn stof P.41
Den Haag : Ministerie van VROM, 2008

**Wang and Mullahy, 2006**
H. Wang and J. Mullahy
Willingness to pay for reducing fatal risk by improving air quality :
A contingent valuation study in Chongqing, China
In : The Science of Total Environment, vol. 367, no.1 (2006); p. 50-57

**Warringa, 2003**
G. Warringa
De beoordeling die weegmethoden maken van de kosteneffectiviteit van milieumaatregelen in de olie- en gaswinningsindustrie
Utrecht : Utrecht University, Department of Science, Technology and Society, 2003

**Watkiss et al., 2005a**
Paul Watkiss, Tom Downing, Claire Handley, Ruth Butterfield
The Impacts and Costs of Climate Change, final Report
Prepared as task 1 of the project 'Modelling support for Future Actions : Benefits and Cost of Climate Change Policies and Measures'
Brussels : European Commission DG Environment, 2005
http://europa.eu.int/comm/environment/climat/pdf/final_report2.pdf
accessed on 10 August, 2008

**Watkiss et al., 2005b**
Paul Watkiss with contributions from David Anthoff, Tom Downing, Cameron Hepburn, Chris Hope, Alistair Hunt and Richard Tol
The Social Cost of Carbon (SCC) Review: Methodological Approaches for Using SCC Estimates in Policy Assessment, Final Report
London : Department for Environment, Food and Rural Affairs (Defra), 2005

**Weitzman, 2007**
M.L. Weitzman
A Review of the Stern Review on the Economics of Climate Change
In : Journal of Economic Literature, vol. 45, nr. 3 (2007); p. 703-724

**WHO, 2008**
Health statistics and health information systems
Geneva : World Health Organisation (WHO), 2008
http://www.who.int/healthinfo/statistics/en/
accessed: 28 July 2009

**Witmond et al., 2004**
B. Witmond, S. Groot, W. Groen and E. Dönszelmann
The impact of REACH : Overview of 36 studies on the imipact of the new EU chemicals policy on society and business
The Hague : Ecorys and Opdenkamp Adviesgroep for Dutch Presidency, 2004
http://www.eu2004-reach.nl/downloads/Comprehensive_Overview-v2.pdf

**WMO, 2003**
Scientific Assessment of Ozone Depletion : 2002, Global Ozone
Geneva : World Meteorological Organization, 2003

**World Bank, 1993**
Investing in Health : World Development Report 1993
New York : Oxford University Press, 1993

**World Bank, 2006a**
D.T. Jamison, J.G. Breman, A.R. Measham, G. Alleyne, M. Claeson, D.B. Evans, P. Jha, A. Mills, P. Musgrove
Disease Control Priorities in Developing Countries, second edition
New York ; Washington DC. : Oxford University Press and The World Bank, 2006

**World Bank, 2006b**
A.D. Lopez, C. D., Mathers, M. Ezzati, D.T. Jamison, C.J.L. Murray, (ed) Global Burden of Disease and Risk Factors
New York : Washington D.C. : Oxford University Press ; The World Bank, 2006
http://www.dcp2.org/pubs/GBD
accessed August 2009