Environmental Prices Handbook 2017

Methods and numbers for valuation of environmental impacts

CE Delft

Committed to the Environment

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CE Delft
Committed to the Environment

Through its independent research and consultancy work CE Delft is helping build a sustainable world. In the fields of energy, transport and resources our expertise is leading-edge. With our wealth of know-how on technologies, policies and economic issues we support government agencies, NGOs and industries in pursuit of structural change. For 40 years now, the skills and enthusiasm of CE Delft's staff have been devoted to achieving this mission.

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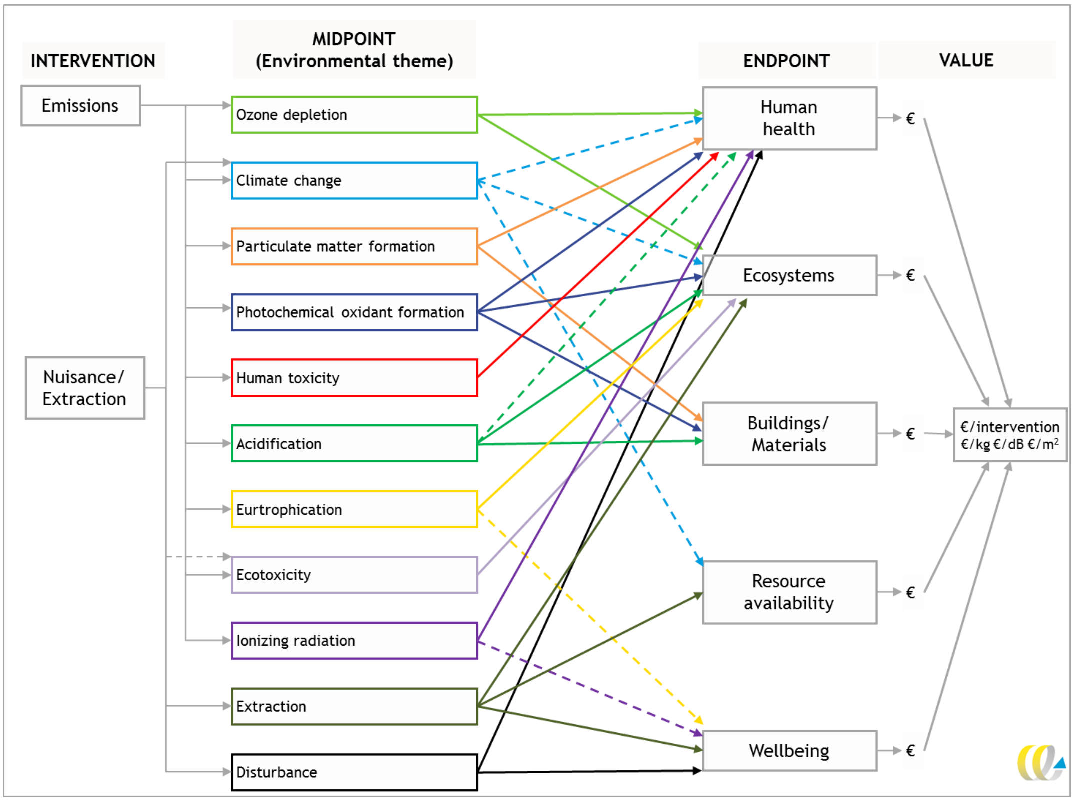
Summary

Environmental prices are constructed prices for the social cost or pollution, expressed in Euros per kilogram pollutant. Environmental prices thus indicate the loss of economic welfare that occurs when one additional kilogram of the pollutant finds its way into the environment. These prices can also be calculated for immaterial forms of pollution such as noise nuisance and ionizing radiation. In such cases the environmental price is expressed in Euros per unit of nuisance or exposure (in decibels, for example).

Environmental prices provide average values for the Netherlands, for emissions from an average emission source at an average emission site in the year 2015. In this Handbook these prices are presented at three levels:
1. At pollutant level: a value for emissions of environmentally damaging substances.
2. At midpoint level: a value for environmental themes such as climate chage or acidification.
3. At endpoint level: a value for the impacts of environmental pollution, such as damage to human health or ecosystem services.

The methodology used in this Environmental Prices Handbook is designed to harmonize the values at pollutant, midpoint and endpoint level, to achieve consistent valuation of the impacts or pollution in the Netherlands.
Figure 1 provides an overview or the relationships covered in this Handbook, with each arrow representing a relationship that has been mapped.

Figure 1 The relationships mapped in this Environmental Prices Handbook



Note: Dashed lines represent relationships examined and (partly) quantified in the context or this Handbook, dotted lines relationships that were not directly quantified because a different approach was adopted for quantifiying impacts.

Results: pollutant level environmental prices

Prices at pollutant level, giving information on the cost of environmental pollution, are the ones most frequently used in analyses. This Handbook and the associated webtool provide such environmental prices for over 2,500 pollutants. Table 1 lists the values or the substances most commonly encountered in the context of air pollution and climate change.

Table 1 **Environmental prices for average atmospheric emissions in the Netherlands (€$_{2015}$/kg emission)**

Stof		Lower	Central	Upper
Carbon dioxide*	CO_2	€ 0.014	€ 0.057	€ 0.057
Chlorofluorocarbons*	CFC_{11}	€ 99.6	€ 313	€ 336
Ultra-fine particulate matter	$PM_{2,5}$	€ 56.8	€ 79.5	€ 122
Particulate matter	PM_{10}	€ 31.8	€ 44.6	€ 69.1
Nitrogen oxides	NO_x	€ 24.1	€ 34.7	€ 53.7
Sulphur dioxide	SO_2	€ 17.7	€ 24.9	€ 38.7
Ammonia	NH_3	€ 19.7	€ 30.5	€ 48.8
Volatile organic compounds	NMVOC	€ 1.61	€ 2.1	€ 3.15
Carbon monoxide	CO	€ 0.0736	€ 0.0958	€ 0.152
Methane*	CH_4	€ 0.448	€ 1.75	€ 1.77

* The value for greenhouse gas emissions includes VAT and increases by 3.5% per annum relative to the 2015 values, as detailed in Section 6.3.

The upper and lower pollutant level values are recommended for use in social cost-benefit analyses, the central values in other applications.

Results: Environmental prices at the midpoint level

The midpoint level environmental prices relate to the familiar set of environmental themes like climate change and eutrophication. They can be used as a weighting factor in life cycle assessment (LCA) or for calculating the external cost of particular materials or products. Table 2 lists the values to be used as external costs or weighting factors.

Table 2 **Midpoint level environmental prices (€$_{2015}$/unit)**

Theme	Unit	External cost	Weighting factor
Climate change	€/kg CO_2-eq.	€ 0.057	€ 0.057
Ozone depletion	€/kg CFC-eq.	€ 30.4	€ 123
Human toxicity	€/kg 1,4 DB-eq.	€ 0.158	€ 0.158
Photochemical oxidant formation	€/kg NMVOC-eq.	€ 2.1	€ 2.1
Particulate matter formation	€/kg PM_{10}-eq.	€ 69	€ 69
Ionizing radiation	€/kg kBq U235-eq.	€ 0.0473	€ 0.0473
Acidification	€/kg SO_2-eq.	€ 5.4	€ 8.12
Freshwater eutrophication	€/kg P-eq.	€ 1.9	€ 1.9
Marine eutrophication	€/kg N	€ 3.11	€ 3.11
Terrestrial ecotoxicity	€/kg 1,4 DB-eq.	€ 8.89	€ 8.89
Freshwater ecotoxicity	€/kg 1,4 DB-eq.	€ 0.0369	€ 0.0369
Marine ecotoxicity	€/kg 1,4 DB-eq.	€ 0.00756	€ 0.00756
Land use	€/m^2*year	€ 0.0261	€ 0.037
Noise >60dB*	€/dB/person	€52-€228	-

Note: * Valuation of noise varies with noise levels and source of noise, see Chapter 6.
External costs are characterized based on an individualist perspective, weighting factors based on a hierarchist perspective. For explanation, see Chapter 3 and Annex A.

Endpoint level

This Handbook reports monetary values for the endpoint impacts human health (mortality and morbidity), ecosystem services, damage to buildings and materials, resource availability and (noise and visual) nuisance. These values form a pivotal element of this Handbook, as they are used to derive the values assigned to midpoint impacts. Table 3 provides an overview or the values adopted.

Table 3 Endpoint level environmental prices

Impact	Indicator/method	Value (lower-upper)
Human health		
Acute mortality	VOLY	€ 50,000-110,000
Chronic mortality	VOLY	€ 50,000-110,000
Morbidity	QALY*	€ 50,000-100,000
Ecosystem services		
Productive ecosystem services**	Crop productivity losses (as a proxy)	
Biodiversity loss	PDF	€ 0.16-1.23/PDF/m^2
Buildings and materials		
Buildings and materials	Restoration costs**	
Resource availability		
Environmental benefits	Environmental prices	
Scarcity and security or supply	Further study**	
Nuisance		
Noise nuisance	Source- and level-specific	
Visual nuisance	Location-specific	

* Besides QALYs other quantifications were also used, such as IQ loss (€ 17,500/lost IQ-point).

** Not fully quantified in this Handbook.

Abbreviations: VOLY: Value or Life Years; QALY: Quality Adjusted Life Years, PDF: Potentially Disappeared Fraction.

Using environmental prices

Environmental prices can be used as a calculation tool in studies and practical applications by government and industry. There are three basic uses:

1. In social cost-benefit analysis (SCBA). Environmental prices are used to assign a value to the environmental impacts of a particular measure or action. For use in this application, the upper and lower values of the pollutant level price are recommended.
2. In the context of corporate social responsibility (CSV) and benchmarking. Companies can use environmental prices to quantify their environmental footprint as well as for preparing environmental annual reports, social business cases and ecological profit-and-loss accounts. In these applications the central pollutant level value is recommended.
3. In life cycle assessment (LCA). LCA practitioners can use environmental prices to weight the calculated environmental impacts to produce a ‘single score’. Companies can determine which materials have the least average environmental impact, for example, key information for optimizing the environmental footprint or their operations.

Environmental prices are average prices for average emissions in the Netherlands and are consequently less suitable for site-specific studies and applications. When considering particular situations involving toxic substances, as with lead soil pollution or hazards relating to plastic coatings on packaging cans, for example, it is not therefore recommended to use environmental prices. In such cases it is better to perform a dedicated study to determine the environment dispersal of the pollutant, its uptake in humans, animals and/or plants, and the effects of uptake on human health and/or ecosystem services. Working with environmental prices in these kinds or situation is too coarse a methodology, given the uncertainties involved.

Reading guide

This Handbook has a three-part structure. Part 1, Chapters 1 to 3, is a User Guidel. After a general introduction, the procedures adopted in the underlying study are justified and the principal assumptions discussed. The environmental prices for the main pollutants are then presented and their use in different contexts explained. Part 2, Chapters 4 to 6 is a detailed elaboration of how the environmental prices were calculated for each environmental theme and endpoint. Here we provide accountability for the choices made in this Handbook and discuss the relevant literature. The third part comprises two Appendices. The first provides some theoretical background on valuation procedures, the second the environmental prices of emissions or over 250 air, soil and water pollutants. All in all, environmental prices for over 2,500 pollutants were calculated in the study underlying this Handbook.
These can be looked up alphabetically as well as under the relevant pollutant code (CAS code) at the Environmental prices Handbook website, www.cedelft.eu/en/environmental-prices which is online in since September 2017.

PART 1: USER GUIDE

1 Introduction

1.1 Background

In modern societies, ever more goods and services are traded in the marketplace. Whenever we go into a shop we see countless articles with a price tag. Based on these prices we decide whether to buy Product A or B, or both, or leave the shop with no purchase at all. A shop can be viewed as a market. It is not only shops where prices play a key role, though. On stock markets, too, prices are what enable trading in companies, goods, physical products and financial products like derivatives. Online, billions of prices are available at any given moment and are used by traders, investors, corporations, consumers and producers to decide on whether to buy or sell.

Market prices are thus a key variable steering the economic process, reflecting what consumers are prepared to pay for a given product or service. If the price goes up, fewer consumers will generally want to buy the product. For the marginal consumer, the price reflects precisely the amount of income he or she is willing to spend on the product or service. In principle, then, prices indicate the value that society, at the margin, thinks the product or service is worth.

Not all goods or services are traded in the marketplace. Many things, such as safety, decency, dykes, leisure time, natural beauty and a clean environment, are not traded directly in markets. But although these things do not have a direct ‘price’, everyone will agree they are important for the wellbeing of a country’s citizens. An unsafe country, with no standards of decency, where nobody has any leisure time, where floods occur in heavily polluted areas and where there is no nature left begins to approximate Dante’s inferno.

While environmental quality is to the good of human wellbeing and prosperity, then, it is unpriced. Since every society makes daily use of economic tools for analysing investments and efficiency, for weighing up costs and benefits and for a host of other purposes, a need arises to express the benefits to human welfare of a clean environment in a price, too, so these can be duly accounted for in economic decisions. This is what environmental prices do: they put a monetary value on environmental quality, by looking at what people would be willing to pay for that quality *as if* there were a market for it.

Environmental prices are implicit prices: the price of environmental quality cannot be determined directly in the marketplace and must therefore be calculated. From the late 1960s onwards, numerous studies have sought to put a price on air pollution and noise nuisance (for a review of Dutch studies see (Hoevenagel & De Bruyn, 2008)). In doing so, most such studies take as their point of departure the *damage* caused by pollution and other forms of environmental intervention. Environmental quality is then valued on the basis of the estimated damage arising as a result of emissions and other changes in the Earth’s natural capital.

Since 1997 CE Delft has been publishing ‘shadow prices’ expressing the value of the environment, calculating it in terms of the marginal costs of securing standing environmental policy targets (CE Delft, 1997; CE Delft, 1999; CE Delft, 2002; CE Delft, 2010). In the latest edition of the Shadow Prices

Handbook, dating from 2010, this set of prices was extended to include an estimate of the *damage* caused by pollution and other interventions, with shadow prices being provided for air, soil and water pollution by over 400 environmentally hazardous substances as well as for noise and land use. This 2010 Handbook has been widely used for preparing environmental annual reports (e.g. (NS, 2014)), quantifying environmental impacts in cost-benefit analyses (see e.g. (Buck Consultants, 2012), (ECN; SEO, 2013)), estimating external costs (see e.g. (Allacker & Nocker, 2012)), quantifying environmental issues in, for example, corporate mergers (Kloosterhuis & Mulder, 2013) and creating tools to increase environmental awareness in the SME sector, as with the Environmental Barometer (Stimular, 2016).

Now in 2017, however, the prices reported in the 2010 Shadow Prices Handbook (CE Delft, 2010) are no longer up-to-date, for three main reasons. The first is that revised General Guidelines for Social Cost-Benefit Analysis have been published (CPB; PBL, 2013), providing a new Dutch framework for SCBA and for assigning values to external impacts. The old framework adopted in the 2010 Handbook was consequently no longer valid. Second, the Discount Rate Working Group (Ministerie van Financiën, 2015) has issued new guidelines for the discount rate to be used in SCBA and other contexts and for valuing impacts on health care and nature, with knock-on effects on environmental prices. Third, new research has been published on the impacts of pollution and other environmental interventions on public health and other issues of relevance for social welfare.

For these reasons the Dutch Ministry of Infrastructure and Environment commissioned CE Delft to prepare an update of the 2010 Shadow Prices Handbook, setting out the subject matter in a manner accessible to a wide range of readers. In this new Environmental Prices Handbook we present a comprehensive set of environmental prices for use in the Netherlands and the methodological framework employed to develop them.

1.2 What are environmental prices?

Environmental prices are indices that calculate the social marginal value of preventing emissions, or interventions like noise and land-use changes, expressing it in Euros per kilogram pollutant or per decibel, for example. Environmental prices thus indicate the loss of welfare due to one additional kilogram of pollutant or decibel of noise being emitted to the environment. In this sense, environmental prices are often the same as external costs

Because a market for environmental quality is lacking, environmental prices cannot be observed directly, i.e. empirically, but must be calculated using the results of studies on human preferences for avoiding the impacts of pollution. This new Environmental Prices Handbook provides a research framework and methodology for putting a numerical price on the value that society attaches to environmental quality.

1.3 Using environmental prices

Environmental prices are used in a wide variety of studies and practical applications by, or commissioned by, government, industry and NGOs for many purposes. Three main areas of application can be distinguished:

1. **Social cost-benefit analysis (SCBA).** Environmental impacts play a key role in economic decision-making in countless areas. A typical example is road construction, where it is not only the cost-effectiveness of the transport link that needs considering, but also pollution impacts and land-use changes. By assigning a value to these impacts using environmental prices, these impacts can be numerically compared with financial-economic data, to establish whether or not the overall impacts of road construction lead to net gains in economic welfare.
2. **Corporate Social Responsibility (CSR) and benchmarking.** Companies and other organizations do not operate as islands, but are embedded in society as a whole. In recent years companies have come under growing pressure to put a numerical value on their impact on the wider environment and for this purpose, too, environmental prices are a useful tool. In environmental annual reports they can be used for social or ecological profit and loss accounts. Environmental prices can also be used to benchmark the environmental performance of a company or organization against that of competitors or other organizations, as with the Environmental Barometer referenced above.
3. **Weighting in Life Cycle Assessment (LCA)**. In LCAs and other kinds of environmental analysis such as Environmental Impact Assessments (EIAs) the impacts of a product are expressed at 'midpoint' or 'endpoint' level, the former referring to environmental themes like climate change or ecotoxicity, the latter to the issues affected, like human health or ecosystems. Environmental prices allow midpoint impacts to be summed to a single figure. This involves an implicit 'weighting' of midpoint and/or endpoint impacts[1]. This provides companies with a quantitative handle for improving the lifecycle environmental impact of their products and factoring in the environment in procurement and production strategies.

1.4 Aim and scope

1.4.1 Aim

The study underpinning this Handbook had a fourfold objective:

1. To develop a set of scientifically robust and consistent environmental prices for the Netherlands for pollutant emissions and environmental impacts at midpoint and endpoint level.
2. To make this set of values as comprehensive as possible in terms of types of impact and number of pollutants included.
3. To make this set of values applicable for use in SCBA, CSR and LCA and, where necessary, adjust them specifically for use in these domains.
4. To make this set of values widely available by means of an interactive online interface guaranteeing consistent use of the Environmental Prices Handbook across all types of user.

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[1] The ISO standard 14040-44 for LCA does not support weighting for comparative LCAs. The recommendation is to compare at midpoint level.

1.4.2 Scope

The environmental prices reported here are based on damage costs. By calculating and valuing the damage caused by environmental pollution (or other such interventions) with respect to a range of endpoints, a value can be assigned to the *additional* overall damage caused by a additional kilogram of a given emission (or equivalent).

The environmental prices reported here are *average* prices for the year 2015 per kilogram emission (and other units for land use and noise) from an *average* source at an *average* location (with average population density and average income, for example). Environmental prices are thus rough-and-ready estimates that are not necessarily valid in specific situations. For particulate matter and noise, specific values are also reported for traffic. In principle, these prices represent the social value of environmental pollution for 2015 emissions. For use in future years, specific guidelines are provided (see Chapters 3 and 5).

1.4.3 Application

This Handbook reports four sets of environmental prices:

A+B): An upper and lower value of estimates derived according to the economic principles employed in SCBA and elsewhere. The ranges in these estimates reflect the uncertainties in people's valuation of environmental quality, and should be explicitly included in SCBAs, as laid down in the new Dutch General SCBA Guidelines (CPB; PBL, 2013).

C): A central value calculated according to standard economic principles, suitable for use by companies in CSR settings.

D): A central value that can be used as a weighting factor in LCA. This value is very similar to C, but impacts for future generations are discounted at a lower rate and thus count for more.[2]

1.5 Limitations

This Handbook presents sets of environmental prices and weighting factors for use as indices in economic and environmental analysis. These prices are average values for emissions from an average source in the Netherlands in 2015. The Handbook provides guidelines on *which* set of environmental prices or weighting factors are to be used in a given context, distinguishing three analytical settings: external cost estimates and Social Cost–Benefit Analysis; Life Cycle Assessment; and tools like benchmarking used in the context of Corporate Social Responsibility. This Handbook is not concerned with the design of such analyses, though. These is thus no discussion of characteristic issues like system boundaries, sensitivity analyses, distribution effects, allocation and so on. For cost-benefit analyses, readers are referred to the General SCBA Guidelines (CPB; PBL, 2013) and the specific SCBA Guidelines for Environment (CE Delft, 2017) and Nature (Arcadis & CE Delft, forthcoming).

Nor is this document to be regarded as an all-inclusive manual for valuing environmental goods or as a textbook for weighting environmental impacts. The aim of this project was to create concrete and consistent sets of environmental prices and weighting factors that can be used in day-to-day practice. The price estimates have been drawn up by CE Delft based on the

[2] This value is included because in LCA characterization is usually from a 'hierarchist' perspective, with most impacts included over an undiscounted time frame of 100 years, while economic valuation corresponds more with an 'individualist' perspective. See also Annex A.

best available scientific understanding. They have been put to and discussed with an Advisory Committee comprising representatives of the Netherlands Bureau for Economic Analysis (CPB) and the Netherlands Environmental Protection Agency (PBL) and other scientific experts, and adjusted as necessary in reponse to their remarks (cf. Section 1.8). In choosing our methods we based ourselves on what is currently held to be *mainstream opinion* in the sciences of environmental valuation, characterization and weighting – with some preference for the most *recent* findings. This means there are *alternative* valuation and weighting methods which, while mentioned here (along with references), are discussed only briefly in terms of how they compare with the methodology adopted here. Given the very extensive literature on valuation and weighting, it would indeed be unfeasible to summarize all the methods in current use. Those using the environmental prices or weighting factors developed in this Handbook must therefore themselves judge whether the figures presented here are preferable to those cited in other publications.

Unless otherwise stated, the environmental prices presented here are expressed in €/kg emission.[3] These prices have been calculated as average values for the Netherlands. Users should make their own judgment as to whether these averages can be used in a particular application like SCBA or LCA. As justification for such choices will always depend on the specific issue for which the environmental prices are being used, the question of whether use of national averages is justified cannot be answered by us here. Local circumstances like population density, existing pollution levels and local pollution limits may mean the data presented here cannot always be applied at the local level (e.g. municipal or provincial). Nor can additional impacts in other countries, including developing nations, be determined using these environmental prices.[4] Finally, use of these environmental prices is also highly contingent on the pollution source or sources involved: transport emissions are far more damaging to human health than average emissions, for example, because they occur closer to the ground. Using these average values for determining the damage due to transport emissions will consequently always lead to an underestimate. We consider these important issues when using environmental prices and in this Handbook we therefore assess the implications of using the figures in transport contexts.

All the environmental prices and weighting factors presented here are (ultimately) expressed as upper, lower and central values. We are all too aware that this implies a degree of quasi-certainty. The environmental prices themselves have been calculated on the basis of multitude of uncertain factors. The formal treatment of uncertainty (detailed in an annex to the original Dutch language version) shows variations to be very substantial – so substantial that use of environmental prices should in fact be discouraged in the first instance. This holds not only for the prices developed here but also for other methods for valuing and weighting environmental goods (few of which include any formal treatment of uncertainty, it may be added). It is a question of choosing the lesser evil, though: either one refrains from using environmental prices, which means financial data cannot be compared with environmental impacts and those impacts cannot be mutually compared, or one does use them, but recognizes that the results have a degree of uncertainty. This choice will depend in part on the issue for which the environmental prices are being used and how rock-solid one wants the final

[3] Voor noise nuisance, ionizing radiation and resource depletion other units are used.

[4] For this purpose CE Delft has developed the Benefito model (CE Delft, 2011).

results to be. In some cases sensitivity analyses can help make the uncertainties more transparent.

1.6 Relation to other environmental valuation methods

1.6.1 The 2010 Shadow Prices Handbook

The values in the present Environmental Prices Handbook 2017 replace those in the old Shadow Prices Handbook from 2010. The principal changes compared with the previous Handbook are as follows:

- Just one environmental pricing method is now used, based mainly on damage costs. The use of abatement costs for standing policy targets has been abandoned, except for climate change.[5]
- This method is designed to cater for the three perspectives of SCBA, CSR and LCA.
- Because uncertainties must now be explicitly included in SCBAs, alongside a central value, upper and lower values are also provided, in accordance with the recommendation of the General SCBA Guidelines in the Netherlands. For corporate CSR calculations and LCA weighting the central value will suffice.
- For health impacts the prices have been assumed to remain constant over time in real terms. In other words, positive income elasticity is no longer deemed relevant for environmental quality. This is in line with the recommendations of the Netherlands' Discount Rate Working Group, which have been adopted by the Dutch Cabinet and which we endorse. The possibly higher value assigned to health in light of income is thus cancelled out by the increased 'supply' of health owing to technological advance[6].
- Damage to agricultural crops has been added to valuation of nature rather than valuation of damage to buildings, as was previously the case. Irreversible impacts on nature have been assumed to have a relative price rise of 1% per annum, in line with the recommendations of the Discount Rate Working Group.
- Climate change has been valued using the least-cost prices (or so called "efficient prices") and have been based on the WLO scenarios published by CPB and PBL at the end of 2015 (cf. Section 3.4.4).This is in line with the recommendations of the Discount Rate Working Group.
- Two additional endpoints have been included: mineral resource availability and nuisance. While these endpoints are described and the valuation methods explained, no characterization factors for these endpoints are provided here establishing a relationship between production processes, emissions and endpoint impacts. In SCBAs or valuations of resource savings by industry these should therefore be independently quantified.
- In this Handbook the valuation method used for biodiversity has been adapted, with specific values for the Netherlands being elaborated.
- Values for ecotoxicity have been calculated at midpoint level, while the characterization factor for land use has been recalculated and adjusted to the midpoint characterization factor.

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[5] This is in line with the General SCBA Guidelines (CPB; PBL, 2013) and the recommendations of the Discount Rate Working Group (Ministerie van Financiën, 2015).

[6] According to the Discount Rate Working Group it is unkown which impact is greater. Alternatively, one can also say that demand for health has decreasing marginal utility: the more health there is available, the lower the marginal utility of an additional unit. This reasoning could give another justification for not factoring in a positive income elasticity of demand.

In addition, all the environmental prices have been thoroughly revised and adjusted to incorporate the latest findings on environmental damage reported in the literature and adjusted for inflation to give 2015 price levels.

1.6.2 Handbook on External Costs of Transport

Under the umbrella of the IMPACT project, in 2008 CE Delft and partners were commissioned by the European Commission to produce the Handbook on External Costs of Transport (CE Delft; INFRAS; Fraunhofer-ISI; University of Gdansk, 2008), which was updated in 2014 by Ricardo-AEA and partners (Ricardo-AEA; DIW econ; CAU, 2014). The aim of this Handbook was to review the methods recommended for valuing the external costs associated with transportation and provide a list of environmental prices to be used for the impacts concerned. Included in this publication are recommended prices for air-pollutant emissions (particulate matter, NO_x, SO_2, NMVOC), greenhouse gas emissions (CO_2) and noise.

For air-pollutant emissions, this European Handbook recommends using the prices developed in the NEEDS project. For CO_2 emissions the values published by (Kuik, et al., 2009) are recommended; these are the abatement costs for securing the 2°C target for 2050 agreed to in the 2015 Paris Agreement. Finally, the environmental prices for noise presented in the European Handbook are based on HEATCO values (as are the values in the 2010 Shadow Prices Handbook).

The environmental prices elaborated in the present Handbook can be used for traffic-related issues. As these have a major local component, though, with emission sources closer to the ground than the overall Dutch average, it may be necessary to adjust the prices developed here at a later date, elaborating them to obtain better estimates directly valid for traffic.

1.6.3 The concepts of true cost, true price and true value

Over the last few years, industries have begun to show major interest in use of environmental prices. One area of application is assessing the environmental consequences of alternative investments using financial parameters. This is in line with increasingly common application of Corporate Social Responsibility, with companies quantifying their environment impact and taking this on board in decision-making. Environmental prices are also used to obtain numerical data for use in social and environmental annual reports. There are currently numerous agencies advising companies on their social impact and putting a value on the environmental damage they cause. The results of such analysis are published in reports for third parties (KPMG, 2015); (True Price, 2017).

The methods adopted for quantifying environmental impacts are by no means always transparent and in many cases no tangible link with specific environmental prices can be established. It is therefore impossible to compare the results obtained using our methods and the methods used in these other reports.

1.7 Reading guide

1.7.1 Environmental price units

All the environmental prices presented in this Handbook relate to pollutant emissions (or other environmental interventions) in 2015 from the Netherlands' mainland territory. All prices are expressed in €/kg emission (etc.), in 2015 prices (abbreviated to $€_{2015}$) and, unless otherwise specified, can be considered to include (average) VAT.[7]

Some fraction of the emissions occurring on Dutch territory will cross the border and impact other countries. Impacts on populations there have been valued the same as for the Dutch population. Certain impacts will take time to manifest themselves. The health impacts of today's air-pollutant emissions will only emerge after several years or decades, for example, while for carbon emissions the impacts will extend over many generations. All future impacts of today's emissions have been implicitly and explicitly discounted in our calculations, with a 3% p.a. discount rate being employed in explicit discounting, in line with the recommendations of the Discount Rate Working Group (Ministerie van Financiën, 2015).

1.7.2 Rounding of values

The environmental prices reported in this Handbook have been rounded to three decimal places when expressed in floating-point notation.[8]
The suggested degree of precision is obviously illusory. However, as these prices will be used in settings like cost-benefit analysis, where they will often need to be multiplied by a million or more, we leave it to users to decide how the results obtained using these prices should be rounded, depending on the application concerned. We feel this is more appropriate than our recommending a preferred degree of rounding.

1.7.3 Structure of this Handbook

This report consists of two parts. Part 1, comprising Chapters 1 to 3, is the User Manual. It explains and justifies the methodology adopted, discusses the main premises and presents the environmental prices for key pollutants. Chapter 2 discusses the general methodological background. Chapter 3 provides concrete advice on when and how the reported prices can be used by specific user groups, distinguishing between use in SCBAs, use as weighting factors in LCAs and use by companies in a CSR context.

Part 2 of this study, Chapters 4 to 6, looks in more detail at the methods employed to calculate these environmental prices. Chapter 4 is an overall review of the changes relative to the 2010 Shadow Prices Handbook, in both general terms and for specific calculations. The value estimates for endpoint and midpoint impacts are then elaborated in more detail in Chapters 5 and 6, respectively. Chapter 5 considers valuation of impacts on the endpoints human health, ecosystem services, buildings and materials, resource availability and nuisance, discussing the premises underlying valuation and how these sometimes differ from those adopted in 2010. Chapter 6 then indicates, for each environmental theme like acidification, eutrophication and noise, how our environmental prices have been constructed.

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7 This is because the prices are based on consumers' willingness-to-pay, which they express in prices inclusive of VAT. For further discussion, and explanation of the exception, see Section 3.4.3. This does not mean, however, that 'net' environmental prices may be estimated by deducting a VAT percentage.

8 Rounded to three decimals 145; 14.5; 1.45; 0.145 then all have the same degree of precision when written in floating-point notation.

The original Dutch language report had nine appendices on a wide range of issues. This English translation has just three. Annex A is a brief introduction to the perspectives adopted from cultural theory in modelling environmental impacts and is a shortened version of the Dutch Annex A. Annex B contains some in-depth information on estimation of the various impacts and is an abridged version of Annex C in the Dutch handbook. Annex C, finally, lists the environmental prices for emissions of over
250 pollutants to air, water and soil.

1.7.4 Web-based tool

Parallel with publication of this Environmental Prices Handbook, an online tool has been launched where users can look up and work with the prices calculated and reported here.
It is available at www.cedelft.eu/en/environmental-prices and can be used free of charge. It contains environmental prices for over 2,000 environmental pollutants. The present Handbook can be regarded as a technical background document for the online tool.

1.8 Accountability

1.8.1 Supervision and support

The research and writing for this Handbook were carried out between January 2016 and April 2017. The project was under the overall supervision of the project principal, the Dutch Ministry of Infrastructure and Environment (I&E), represented by Karel Zeldenrust and Robin Hamerlinck.

The study was regularly discussed and commented on by an Advisory Group, who provided oral and written comments on draft versions of all texts.
This group comprised Karel Zeldenrust, Mark Overman and Frans Duinhouwer (I&E), Joop van Bodegraven and Marcel Klok (Ministry of Economic Affairs), Eric Drissen and Gusta Renes (Netherlands Environmental Assessment Agency, PBL), Gerbert Romijn (Netherlands Bureau for Economic Policy Analysis, CPB), Marian Bertrums, Rob van de Veeren and Anna Krabbe Lugner (Directorate-General for Public Works and Water Management, *Rijkswaterstaat*),
Rob Maas (National Institute for Public Health and the Environment, RIVM) and Martin Linssen (Ministry of Finance).

Besides the Advisory Group, a formal expertise group was also appointed, consisting of Mark Goedkoop (PRé Consultants) and Bert van Wee (Delft Technological University), who contributed by providing helpful comments and suggestions.

We are very grateful to the supervisors, the members of the Advisory Group and the experts for all their work and input. It goes without saying, though, that we alone bear ultimate responsibility for the ideas and results presented here.

1.8.2 Expertise

While this Handbook derives most of its underlying information from literature study, we were unable to find all relevant data in this way. In elaborating the numerous issues involved we therefore also made grateful use of information provided by (international) experts in this field, often via email. In the framework of this study the following people furnished us with important data:

- Prof. dr. Ari Rabl, ARMINES/*Ecole de Mines;*
- Prof. Ståle Navrud, Agricultural University or Norway;
- Prof. dr. Christopher Murray;
- Daniel Sutter, INFRAS, Zurich;
- Till Bachman, Jonathan van der Kamp, EIFER, Karlsruhe, Germany;
- Kees Peek, RIVM;
- Rob Aalbers, CPB;
- Hans Nijland, Hans Hilbers, Gerben Geilenkirchen and Arjan Ruijs, PBL;
- Milan Scasny, Charles University, Prague.

We thank them all for their willingness to answer our questions and discuss our premises. Again, though, they bear no responsibility for the results presented here.

1.8.3 English language version

This English-language version, translated by Nigel Harle, contains environmental prices for the Netherlands and will be used as input for a new version of the Environmental Prices Handbook reporting estimated EU28-average environmental prices, to be published in the summer of 2018.

Most of the text in the original Dutch handbook has been translated. However, some of the Appendices have not been translated, or have been abridged in this English version. Table 4 below summarizes which Appendices have been translated and which have been omitted. In general, we have omitted those appendices with a literature review concerning valuation of specific endpoints, since the issues concerned are already elaborated in Chapters 5 and 6 – though far more briefly. Some of the appendices included in the English version have been elaborated in slightly more detail, as with the treatment of uncertainty and the impact-pathway modelling used in this handbook.

Table 4 Indication of annexes that have been translated and which once have been omitted from the English language version

Annex in Dutch version	Translated into English?
Annex A Characterization	Largely included in Annex A
Annex B Valuation of human health	Not translated
Annex C Impact pathway modelling	Largely included in Annex B
Annex D Valuation of raw material scarcity	Not translated
Annex E Valuation of biodiversity	Not translated
Annex F Valuation of noise	Not translated
Annex G Assignment of damage estimates to midpoints	Some parts included in Annex C
Annex H Treatment of uncertainty	Some parts included in Annex C
Annex I Overview of environmental prices	Included in Annex D

2 Methodological framework

2.1 Introduction

Environmental prices are indices expressing the willingness-to-pay for less environmental pollution in Euros per kilo pollutant. Environmental prices thus indicate the loss of economic welfare that occurs when one additional kilo of the pollutant enters the environment. In many cases they equal external costs. These prices can also be calculated for immaterial forms of pollution like noise nuisance and ionizing radiation, then being expressed in Euros per unit nuisance or exposure (in decibels and kBecquerel, respectively, for example).

In this chapter we discuss the main aspects of the methodological framework of calculation and use of environmental prices. As an introduction, in Section 2.2 we first consider the economic and environmental significance of these prices. Then, in Section 2.3, we set out the basic framework employed here for valuation. In Section 2.4 we consider the use of environmental prices in more detail.

2.2 Introduction to environmental prices

2.2.1 Significance in welfare economics

Valuation of environmental quality means expressing the value society assigns to that quality in monetary terms. Since in many cases that value cannot be directly established – via market prices, for example – it must be calculated.

Research on financial valuation of environmental impacts goes back to the 1930s, when US citizens sought compensation in the courts for the sulphur dioxide emissions of a Canadian mining company (Read, 1963). In the Netherlands, valuation of environmental impacts was first carried out by academics In the 1970s in the context of noise nuisance (see Opschoor, 1974). Since then, valuation has become an integral part of environmental economic research, with a great deal of work being undertaken on both methodological development and numerical valuation (Hoevenagel & De Bruyn, 2008).

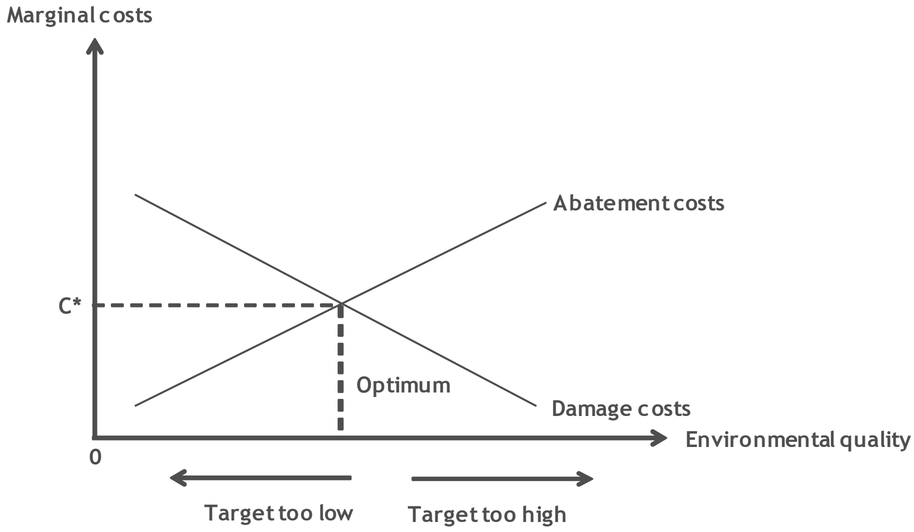
In economic terms, most environmental services cannot be provided through market mechanisms. Clean air, biodiversity and avoidance of environmental risks are not things that can be bought in the supermarket. Such services are nonetheless scarce, given their limited availability and the numerous impacts of our consumption and production patterns on that availability (Hueting, 1980). In economic terminology, we are faced with negative external impacts: side-effects of production and consumption that affect the welfare of others without them receiving financial compensation for their loss of welfare.

2.2.2 Environmental prices as equilibrium prices

It is instructive to imagine that a market for environmental services did exist. How much clean air would we then buy? According to standard economic theory, society would arrive at a point where the benefits of one additional unit of clean air equal the cost of an additional unit of pollution reduction. In other words, the moment pollution abatement becomes more expensive than the value assigned to clean air, we have reached the ‘optimum’ pollution level. In economic terms, this level of pollution is referred to as Pareto-

optimal, or Pareto-efficient, because there is no pollution level with a higher level of welfare, defined as the sum of producer and consumer surplus.
The associated marginal costs are known as the equilibrium price of the environmental impact category concerned. They indicate the value assigned by society to the impact in question. At this point, marked C* in Figure 2, the marginal abatement costs equal the marginal damage costs of pollution.

Figure 2 Optimum pollution level and associated equilibrium environmental price according to standard economic theory



Note that as environmental quality improves, marginal abatement costs rise; this reflects the general tendency for pollution control to become increasingly costly the further it goes. At the same time, damage costs decline as more environmental quality becomes available. This downward trend illustrates the declining marginal utility deriving from improvements to environmental quality.

This optimum pollution level and the associated equilibrium price obviously differ depending on the pollutant involved. This is due in the first place to abatement costs differing for the various categories of environmental impact. A 50% reduction in SO_2 emissions, causing acid rain, for example, is cheaper to achieve than the same reduction in CO_2 emissions, causing climate change. This is due to the different costs of the abatement technologies required. Secondly, society values different environmental impacts differently, perhaps viewing climate disruption as more important than acid rain, implying that the marginal damage of climate change is greater than that of acidification. The consequence of this (hypothetical) reasoning would be that society attaches greater value to reducing CO_2 emissions than to reducing SO_2 emissions.[9]

2.2.3 Environmental prices as (external) damage costs

Equilibrium prices express the true economic value of pollution *if all external impacts were internalized*. Although these prices could in principle be calculated and used to assign a value to emissions, this is not generally done. The main reason is that such prices indicate the external cost to society of a particular project *only if* the pollution level at the time is 'optimum'.

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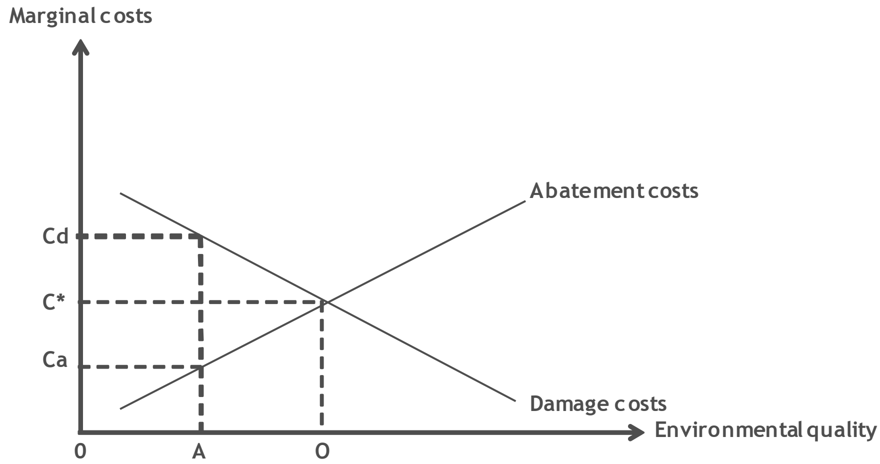
[9] Under the assumption of linear marginal damage cost functions.

The more the actual situation deviates from the optimum, however, the less correct the estimates of the extra cost will be. In most cases, actual environmental quality wil be 'suboptimal' as a result of insufficiently effective environmental policy. The damage costs will therefore generally be higher than the abatement costs (for a comparison of damage and abatement costs see (CE Delft, 2010)).

This is illustrated in Figure 3. Imagine that environmental quality is currently at level A as a result of environmental policy with a marginal abatement cost Ca. The current level of environmental quality (interpreted here as the inverse of pollution) is below the optimum, marked O. The marginal damage cost associated with the current situation is therefore Cd. In this case the damage cost Cd indicates the value to be assigned to a small change in environmental quality. It represents the marginal cost of the infinitely small increase (decline) in damage resulting from an infinitely small decline (increase) in environmental quality.[10]

Figure 3 **Price of environmental quality at a suboptimal pollution level**



Environmental prices thus indicate the value of emissions relative to one another and relative to other goods in society. In addition, environmental prices are in most cases also equal to the value to be given to the external costs of pollution and other environmental interventions. This value is equal to the 'Pigovian tax' required to internalize external impacts (Pigou, 1952). Other things being equal, internalization of external costs, so they can be included in policy deliberations, leads to greater economic welfare.

[10] In the 2010 Shadow Prices Handbook this was referred to as the shadow price. Formally speaking, the shadow price is the value of a controlling factor (the 'Lagrange factor') at the optimum, which means it is the infinitely small change in the objective function due to an infinitely small change in the controlling factor. 'Shadow price' is thus the proper name for 'abatement cost'. For the damage cost function, though, this is a derived shadow price for the limited availability of environmental quality due to policy. To avoid getting embroiled in a semantic debate, in this study we use the more neutral term 'environmental price'.

2.2.4 Environmental prices as weighting step in characterization

Today over 10,000 pollutants are known that can cause environmental damage and for a long time environmental scientists have been looking for a way to condense the vast amount of data often yielded in environmental analyses into a single indicator. This compression of data can be achieved in two ways: via characterization and weighting.

Characterization is a process in which an index, known as a characterization factor, is used to express how much a standard amount of a given substance contributes to a particular environmental impact. The higher the characterization factor, the greater the contribution. The gas methane has a higher characterization factor for the environmental impact 'climate change' than carbon dioxide, for example. This means a kilo of methane causes more global warming than a kilo of carbon dioxide.

Using characterization factors, emissions can be grouped into a series of aggregated environmental themes like climate change, acidification and human toxicity, referred to as 'midpoints' (cf. Section 2.3.3). These impacts on the various environmental themes cannot then be mutually weighted, however. All a researcher can conclude is that a given recycling policy will impact positively on climate, say, but negatively on eutrophication.
The question is then: Is the policy good or bad for the environment? In other words: Which environmental theme is more important? To answer this question the various environmental impacts can be individually weighted, allowing a 'single score' to be calculated as a final result. This score indicates whether the net result of the LCA signifies environmental gains or losses.

Weighting is thus a process in which midpoint scores are combined to yield a single, uniform indicator. For weighting environmental impacts at midpoint level, various methods have been proposed in the literature, including methods based on 'distance to target' (VROM, 1993), expert panels (Huppes, et al., 2007) and impacts on endpoints (Goedkoop, et al., 2013). In this context, environmental prices can be seen as a further method for mutually weighting environmental themes and combining environmental impacts into a single, uniform indicator. This indicator then provides information on whether a particular measure, purely from an environmental perspective, is to be recommended because it leads to greater 'welfare'.

This means environmental prices can also be used to weight environmental impacts. They express the relative value of emissions (etc.) relative to one another and relative to other goods circulating in society. When valuing emissions, in the context of SCBA for instance, one is generally looking at the value of emissions compared with other financial figures. When it comes to the weighting of emissions, though, we are concerned primarily with how emissions compare to one another. These weighting factors can then be regarded as the socio-economic weight to be attributed to the various environmental impacts.

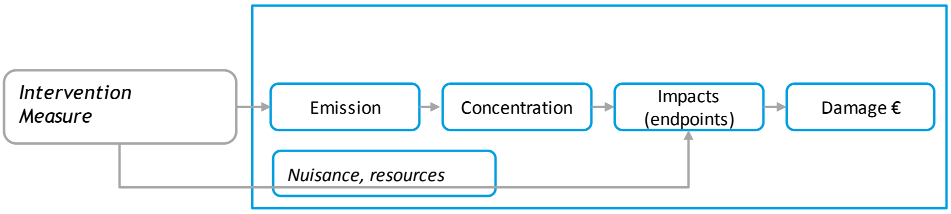
The environmental prices developed in this Handbook are derived from damage costs and are consistent with standard practice in welfare economics. However, there are also other approaches using monetization, such as the Environmental Priority Strategies (EPS) method developed in Sweden (Steen, 1999).[11]

2.3 Valuation framework in this Handbook

2.3.1 Overall framework

The overall framework adopted in this Environmental Prices Handbook is schematically summarized in Figure 4.

Figure 4 Relationships relevant in this Handbook



A given activity leads to a certain intervention in the environment. That intervention, or policy measure, results in a change in emissions, nuisance or resource extraction. In the case of emissions, these are transported via air, soil or water to other areas, where they are added to existing emission concentrations. This concentration then leads to changes in 'endpoints' relevant to human welfare. These changes can be monetarily valued by quantifying the amount of damage caused at the endpoints. The entire chain from emissions, nuisance and resources through to damage in monetary terms is the subject of this Environmental Prices Handbook. The effectiveness of interventions or policy measures is beyond the scope of the Handbook.

2.3.2 Relevant endpoints

In this Environmental Prices Handbook we distinguish five endpoints:
1. Human health (morbidity, i.e. sickness and disease, and premature mortality).
2. Ecosystem services (including agriculture).
3. Buildings and materials (man-made capital).
4. Resource availability
5. Wellbeing (aesthetic and ethical values).

The issues captured in this fivefold categorization are broader than found in the literature. ReCiPe, for example, distinguishes three endpoints: human health, ecosystem services and resource scarcity (Goedkoop, et al., 2013). The chosen endpoints are described in detail in Chapter 5.

[11] The systematics of the EPS method come close to the concept of 'unpaid cost', with the researcher deriving values for willingness-to-pay via a hierarchy of 'principles'. There is no discounting of future impacts. Particularly for resource depletion, the method establishes a relatively high value, based on 'restoration costs'.

2.3.3 Relevant midpoints

Midpoint categories capture the impact of emissions on aggregated environmental themes. In the various handbooks used by environmental scientists and practitioners there is variation in the number and scope of the midpoints employed. In this Handbook we closely follow the categories used in ReCiPe (Goedkoop, et al., 2013), distinguishing the following eleven midpoints:

1. Ozone depletion.
2. Climate change.
3. Particulate matter formation.
4. Photochemical oxidant formation.
5. Acidification.
6. Eutrophication.
7. Human toxicity.
8. Ecotoxicity.
9. Ionizing radiation.
10. Nuisance (noise and visual nuisance).
11. Extraction (land use).

These midpoints are described in detail in Chapter 5 and are largely in line with what is cited in the literature for midpoint characterization (see (Guinée, et al., 2002); (Goedkoop, et al., 2013); (JRC, 2012). Compared with ReCiPe (Goedkoop, et al., 2013) this means we have added one midpoint: nuisance (in particular, noise nuisance), and combined several ReCiPe midpoints, as with our treatment of ecotoxicity, eutrophication and land use. In contrast to ReCiPe, impacts on the availability of mineral resources, water and fossil fuels are not included as separate midpoints in this Handbook, but valued only at endpoint level (see Chapter 5).

In the systematics adopted in this Handbook, a number of midpoints also cited in the literature (Guinée, et al., 2002) have not been included. These relate primarily to interventions on the interface between nature and the environment:

- erosion of farmland soils;
- salinization of farmland soils;
- light pollution;
- stench;
- visual impact (‘horizon pollution’);
- spread of invasieve species.

These all impact primarily on the endpoints ‘ecosystems’ and ‘wellbeing’. In many cases there is no directly observable relationship between emissions and these midpoints. In addition, no ‘Dutch average’ can generally be calculated for these kinds of environmental impact, which are often project-specific. Nor are they usually included in LCA calculations. For these reasons they have not been taken as midpoints in this Handbook. Methods and studies concerning valuation of these impacts are described in relation to the endpoint ‘nuisance’ (see Section 5.7).

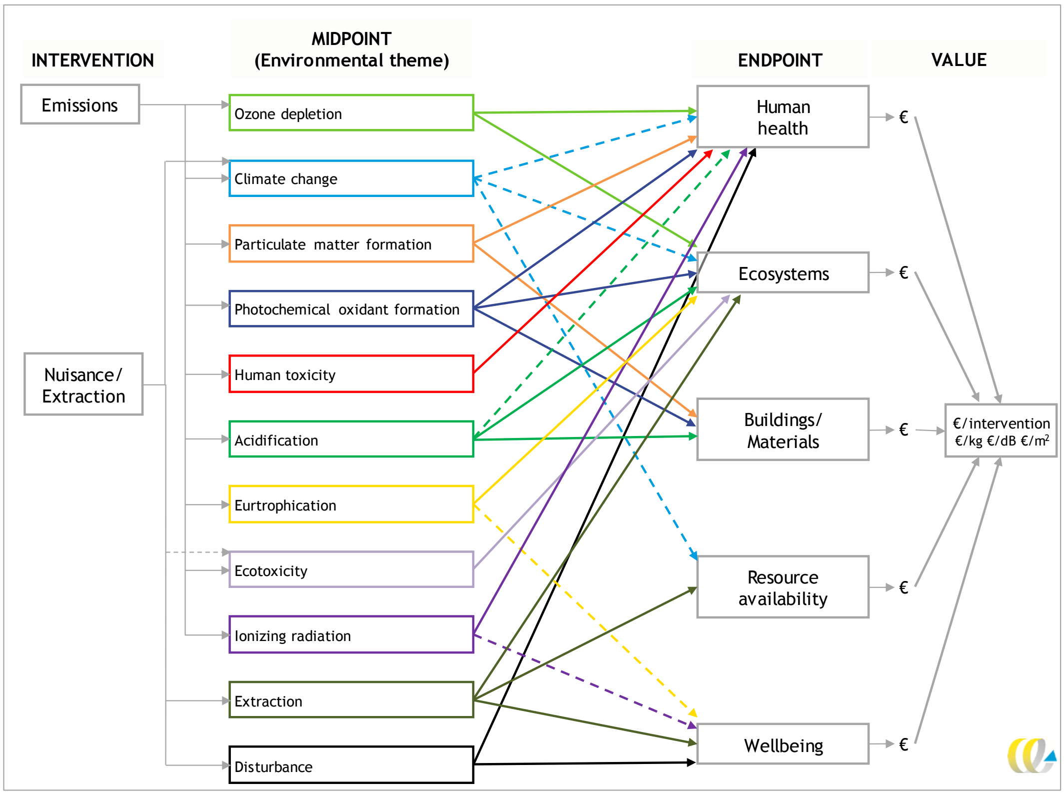
2.3.4 Relations between pollutant, midpoint and endpoint level

At the core of this Environmental Prices Handbook are two steps:

1. Establishing the relationships between environmentally hazardous substances (emissions) or causes of ‘disturbance’ (noise, land-use change) and their impacts on midpoints and endpoints.
2. Valuing these endpoints and translation back to damage per intervention.

The overall framework is shown in Figure 5, setting out all the relationships between emissions, midpoints and endpoints and their valuation that are of relevance for this Handbook.[12]

Figure 5 Relationships between intervention, midpoints, endpoints and valuation in this Handbook



Note: Dashed lines represent relationships examined and (partly) quantified for this Handbook, dotted lines those that were not directly quantified because a different approach was used for impact quantification. ‘Depletion’ includes land use, among other things, and ‘nuisance’ noise nuisance. For further details see Chapter 6.

The endpoint level is that at which there are no longer any ‘feedback’ effects. This level thus forms the basis for valuation. For the five endpoints, in Chapter 5 we examine people’s willingness-to-pay for improvement in the form of pollution abatement. Via the midpoints these values can then be calculated back to a value for reducing the emissions themselves, or avoiding an environmental intervention (as with noise or land-use changes).

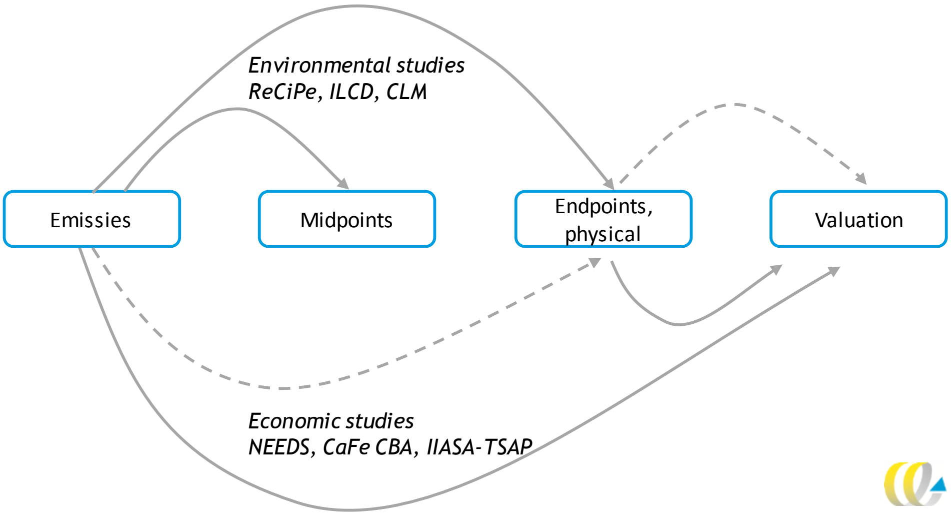
12 This is not to say that all these relationships have indeed been quantitatively determined.

There is currently no single methodology or study that brings all these elements together in a consistent environmental and welfare-economic framework. The 2010 Shadow Prices Handbook endeavoured to do so by combining two kinds of studies:

1. Environmental studies like ReCiPe (Goedkoop, et al., 2013) in which physico-chemical models supplemented by impact studies are used to establish relationships between pollutants and midpoints, on the one hand, and midpoints and endpoints, on the other. Environmental studies typically use both characterization models which define impacts of pollutants relative to each others with impact pathway models that describe the relationship between emissions, via dispersion and concentration, towards impacts.
2. Economic valuation studies like NEEDS (2008a) and Holland (2014), in which dispersion models, concentration-response functions and valuation tools are used to establish a relationship between a pollutant level (emissions) and a monetary value per endpoint level. Economic modelling thus typically combines impact pathway models with economic valuation techniques.

Each type of study models part of this chain of relationships, as shown schematically in Figure 6.

Figure 6 Relationships between emissions, midpoints, endpoints, valuation and relevant fields of study



Note: Dashed lines indicate that these steps are used in the studies, but only infrequently.

What one thus sees is that, on the one hand, there are environmental studies like ReCiPe (Goedkoop, et al., 2013) that establish the primary relationships between emissions, midpoints and endpoints. This information is used in LCA software packages like SimaPro. Environmental studies focus very strongly on describing as precisely as possible the physico-chemical impacts of emissions and how they relate to endpoints. Many economic studies, on the other hand, are concerned with putting a price on pollution, as holds for the major European research programmes NEEDS (NEEDS, 2008a), CAFE-CBA (AEA, 2005) and IIASA-TSAP (IIASA, 2014);(Holland, 2014), the results of which are used in European cost-benefit analyses. These kinds of studies are concerned above all with establishing as accurately as possible a relationship between emissions and valuation according to the premises of neoclassical welfare economics.

Both kinds of studies thus chart a relationship between emissions and endpoints, but with different accents in terms of premises and details. The great advantage of ReCiPe, for example, is its attempt at consistency between midpoint and endpoint impacts (Goedkoop, et al., 2009); (Goedkoop, et al., 2013). ReCiPe, funded partly by the Dutch government, is indeed the first major project guaranteeing, to a certain extent at least, such consistency.

The drawback of ReCiPe for monetary valuation, however, is that the relationship between emissions and endpoints is reported solely as an average global value (or, if global data are lacking, a European average). In addition, impacts are not time-discounted, which means ReCiPe cannot be used to derive values consistent with the premises of welfare theory.

Economic studies, on the other hand, also establish a relationship between emissions and their impacts, but do so in a way that paints with a broad brush in environmental terms. In addition, economic studies have as their main weakness that the relationship between emissions and endpoint damage is established for a mere 20 or so environmental pollutants. For the thousands of other environmentally hazardous substances these studies provide no useful information at all.

2.3.5 Combining modelling approaches

The Environmental Prices methodology employed in this Handbook combines characterization models, impact pathway analyses and valuation methods to arrive at a consistent estimate of the welfare costs associated with emissions at the pollutant, midpoint and endpoint levels. *The key feature of the Environmental Prices methodology thus lies in its harmonization of the premises of the three research methods.*

The manner in which this has been achieved can best be explained with reference to the scheme shown in Figure 7. As can be seen, characterization models as well as the impact pathway approach both establish a relationship between emissions and endpoint impacts. Characterization models like ReCiPe distinguish the endpoints resources, ecosystems and health.[13] Impact pathway approaches like NEEDS distinguish the endpoints ecosystems, health, buildings and nuisance.[14] The impact pathway approach establishes no explicit relationship for resources, while characterization models fail to do so for buildings or nuisance. For two endpoints, there is overlap between the two approaches: ecosystems and human health. For this Handbook it was therefore necessary to balance the two approaches or decide which was preferable.

By combining and harmonizing the environmental and economic models, we obtain a uniform framework in which emissions (etc.) can be valued, with the following advantages:
- A final step is added to environmental characterization: monetization. By clearly defining this monetization as impacts on welfare, a uniform framework is created in which all environmental impacts can be systematically weighed up against one another.

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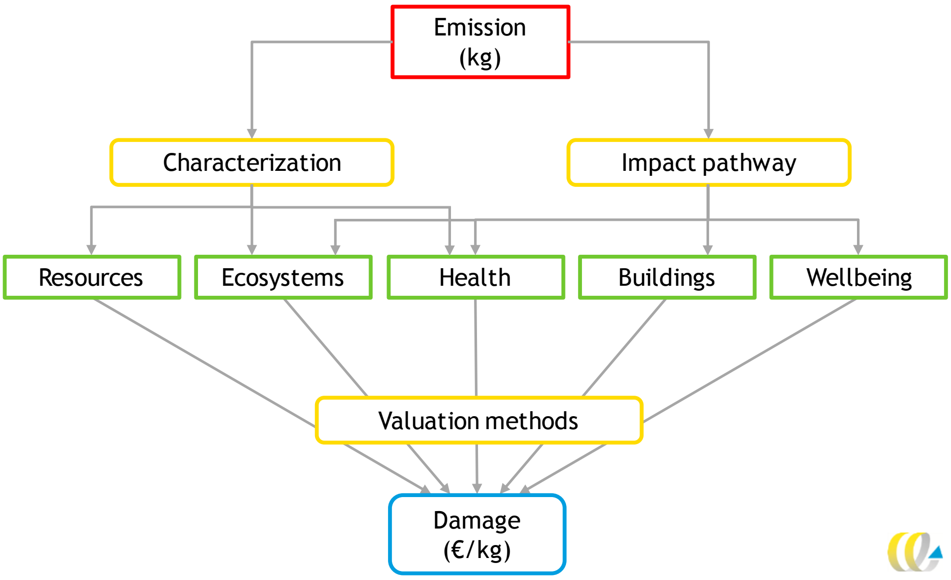
13 Throughout this discussion we base ourselves on ReCiPe (Goedkoop, et al., 2013). While drawing up the present Handbook a new ReCiPe version was published, in January 2017. This was not taken on board in determining our environmental prices, because this was too late into our study.

14 Nuisance has generally received relatively litte attention, however. The impact pathway approach also treats climate problems as midpoints that generate impacts on the endpoints health and ecosystems.

- Through due use of characterization factors, economic valuation of an individual pollutant can be extended to valuation for all pollutants with a similar environmental endpoint impact. This means the number of pollutants to which an economic value can be assigned is expanded enormously.
- By working with the economic valuation studies that contain data on individual countries, environmental characterization can be land-specific.

Figure 7 Characterization models, impact pathway analyses and valuation methods as a basis for the Environmental Prices Handbook



It is sometimes queried whether it would not be better to proceed directly to valuation at endpoint level, rather than expressing damage costs in kilogram emissions. However, pollution impacts need to be considered from a multi-emission/multi-impact perspective, since the various pollutants contribute to different environmental problems (the ‘environmental themes’), which in turn have distinct impacts on the various receptor groups (humans, ecosystems, etc.). Thus, SO_2 contributes both to acidification, impacting ecosystems, and to formation of secondary aerosols, impacting human health. This means there can be no simple one-to-one translation from emissions to impacts but that models must be used.

In Section 4.2 we look in more detail at how the various modelling approaches have been harmonized in this Environmental Prices Handbook. In Chapter 5 an extensive review is provided of the valuation methods adopted.

2.4 Perspectives and use

2.4.1 Lower, upper and central values

This Handbook presents environmental prices at pollutant level (SO_2 and NO_x emissions, for example), midpoint level (environmental themes like climate change and acidification) and endpoint level (indicators for human health and ecosystems).

At pollutant level the prices are expressed as a lower, central and upper value. This approach has been adopted so the reported prices reflect the uncertainties inherent in assigning a value to pollution. The upper and lower values are mainly for use in social cost-benefit analyses (SCBA), to calculate the impacts of (government) policy, for example. Since publication of the General SCBA Guidelines (CPB; PBL, 2013), uncertainties may no longer be 'concealed' in discount rates or sensitivity analyses, but must be explicitly treated as a core element of the SCBA. To duly account for the uncertainties and gaps in our knowledge when valuing the welfare impacts of emissions (etc.) the Guidelines therefore recommend working with ranges. This means that in this Handbook we have developed an **upper** and **lower value** for use in SCBAs. These upper and lower bounds have been set at the level of endpoint valuation and work back, via the web of relationships between pollutants and endpoints, to upper and lower values at the pollutant level.

For other users, **central values** are reported and recommended. The central value has been elaborated differently for each endpoint and represents the best possible estimate given the uncertainties in endpoint value (cf. Chapter 5). For companies using our environmental prices for detailing business cases or environmental annual reports, it is recommended to use these central values. Corporate financial annual reports do not generally give ranges or upper or lower bounds, and use of our central values is thus in line with standard practice.

For use as weighting factors in LCAs, two values have been developed: a value based on external costs that is a good match for the individualist pespective in ReCiPe and a value for use as a weighting factor that is entirely in line with the hierarchist perspective in ReCiPe. For an explanation of these perspectives, see Annex A.

2.4.2 Objections to the use of environmental prices

The environmental prices presented here are average values for pollution from an average emission source at an average location in the Netherlands. *These prices can consequently only be taken as approximate averages and should not be used in concrete situations*. In such cases it is recommended to perform a dedicated study to establish potential pollutant and other impacts.

Economic valuation of impacts on nature, the environment and human health may also elicit moral objections, with some holding that it is undesirable, inappropriate or morally reprehensible to put a price tag on health or nature. Economic valuation does not justice, they say, to 'intrinsic values' like the mere existence of plant and animal species, or moral values like caring about one's neighbours. This is is no way the case, though. Economic valuation merely facilitates and rationalizes choices between alternative ways of allocating scarce resources (time, money). Money spent on Alternative A cannot be spent on Alternative B. When weighing up these choices it is perfectly feasible to recognise and duly allow for intrinsic or moral values. Even the most dedicated environmental warrior must ultimately decide how much they are willing to spend on environmental aims and how much on their lunch. When deciding what fraction of our money to spend on development cooperation, we do not deny the intrinsic value of those living in the developing world. Economists look at how much people are willing to pay for various goods and objectives and use this information to deduce the economic value of those goods. People may obviously disagree with other people's preferences and (moral) values and thus with their willingness-to-pay, but all economists are doing is observing and noting what is ocurring in society at large.

Environmental prices can only be used to derive marginal values. They cannot be used to calculate the total value of the Earth's biodiversity, say (cf. (Constanza, et al., 1997). At the margin, decisions are taken that affect nature, the environment and human health. Environmental prices can be used to include these impacts in decision-making, but not to justify or legitimize pollution. Illegal pollution must always be tackled according to the law of the land.

When it comes to valung human health, there are sometimes misconceptions. In putting a value on health it may seem as if judgment is being made on the value to be assigned to a human life, which some people deem immoral. From an ethical perspective, however, there is no moral obligation to save a life at any cost (at the expense of one's own life, for example). More importantly, though, economic valuation makes no pronouncement on an individual human life, but on so-called *statistical* lives. In the policy decisions in which economic valuation is employed, we are concerned with marginal changes in the risks to which people are exposed. If a certain risk is reduced from fifteen in a million to fourteen in a million for a population of one million, for example, one statistical life is saved. Economists simply note that such comparative assessments of risks and potential gains are made all the time in everyday life, such as when deciding whether or not to get into a car or plane, or pursue a certain lifestyle with its associated risks of premature death. So although no price tag can be put on life itself, when it comes to safety in the sense of statistical risk reduction, it can be. For this reason, in economic terms a problem arises in deciding which risks are aceptable and which are not. With environmental prices, this weighing up of choices is rendered explicit, for use in tandem with other decision-making procedures.

Some critics object to economic valuation on the grounds that by putting the emphasis on the goods owned by individuals, it is only self-interest that is factored in. They hold that issues like environmental protection should be evaluated based on the 'public interest', i.e. with reference to what is best for society as a whole (cf Mouter & Chorus, 2016). Whether this public interest is the same as the sum of all individual self-interests is still an unanswered, controversial question in political philosophy. We can only stress that environmental prices based on willingness-to-pay that can be used for cost-benefit analyses are no substitute for the political process; all they do is provide information on people's preferences, i.e. how much people are willing to pay for a given change in environmental quality. It is then up to politicians whether and to what extent they opt to deviate.

3 Use of environmental prices

3.1 Introduction

In practice, environmental prices can support decision-making in two ways:
- When analysing the social impacts of investment decisions, environmental impacts can be included along with financial data because they can be assigned a monetary value using environmental prices. A case in point is Social Cost-Benefit Analysis (SCBA), where environmental prices are used primarily for **valuation**, providing a means of comparing environmental impacts with financial items to arrive at integral consideration of all the impacts associated with an (investment) decision. In principle, **valuation** of environmental impacts using environmental prices occurs in every SCBA in which external impacts are also monetized (see e.g. (ECN; SEO, 2013)) and by companies in calculating social business cases.
- In environmental analyses like Life Cycle Assessment (LCA), Environmental Impact Assessment (EIA) and benchmarking, environmental prices can be used to weight the various environmental impacts identified. The main aim here is environmental **weighting**, as a means of comparing the contribution of different environmental themes. **Weighting** of environmental impacts is sometimes carried out as a final step in LCAs in order to express the results in a 'single-score indicator'. In line with the methodology employed in this Handbook, the welfare impacts of emissions are monetized within a standard welfare-economics framework. The EPS system (Environmental Priority Strategies in product design; (Steen, 1999)) also involves monetary weighting, but using premises based more on monetization of a hierarchy of principles than on welfare economics. Financial valuation is often applied as a weighting method in various LCAs and in concrete calculation tools like the Environmental Barometer (for small and medium-sized enterprises, SME), DuboCalc (used in the construction industry) and GreenCalc (for comparing the environmental profile of buildings).

In this chapter we present the environmental prices developed for this Handbook and discuss their use in more detail. First of all, in Section 3.2 we report the environmental prices for a series of common air, soil and water pollutants. We then go on to explain the use of environmental prices with reference to three groups of users:
1. Companies calculating their environmental impact (Section 3.3).
2. Practitioners carrying out a Social Cost-Benefit Analysis (Section 3.4).
3. Practitioners weighting LCA environmental impacts to arrive at a single-score indicator (Section 3.5).

For each user group this chapter provides concrete guidelines on how the environmental prices can be applied and what specific issues are likely to be encountered in that particular setting. In doing so, we list prices for a handful of pollutants only. The full list is provided in Annex C. Further information and an online tool for calculating the environmental prices of over 2,500 pollutants is available at: www.cedelft.eu/en/environmental-prices.

3.2 Environmental prices: a brief synopsis

This section reports environmental prices for several commmon pollutants. The majority are expressed in €/kg pollutant, in 2015 prices. The two exceptions are noise and ionizing radiation, expressed respectively in € per decibel and € per kiloBequerel (measuring the intensity of emitted radiation).

As stated earlier, the environmental prices reported in this chapter are average values for the Netherlands. The damage costs of environmental pollution (etc.) can vary widely according to local circumstances (particularly population density) and the nature of the emission (from industrial stacks versus vehicle tailpipes, for example). Environmental prices make no allowance for these differences.[15] For this reason, these environmental prices cannot simply be applied to specific cases of local pollution, for pollution in other countries or for pollution by non-average emission sources. In Chapter 6 these issues are considered in more detail, as well as the background to the calculations (neither of which issues are discussed in the present chapter).

3.2.1 Environmental prices for emissions to the atmosphere

Table 5 reports the values for the most frequently encountered atmospheric emissions in €/kg emission.

Table 5 Environmental prices for key atmospheric emissions (€$_{2015}$ per kg emission)

Pollutant		Environmental price (€/kg emission)			Relevant midpoints[1]							Endpoints[1]		
		Lower	Central	Upper	PM formation	Smog formation	Acidification	Climate change	Ozone layer	Human toxicity	Ecotox./ Eutrophication	Human health	Ecosystem services	Materials/Buildngs
Carbon dioxide[2]	CO_2	€ 0.014	€ 0.057	€ 0.057				x				nc	nc	nc
Chlorofluorocarbons[2]	CFC11	€ 99.6	€ 313	€ 336				x	x	x	x	x	x	
Fine particulates, 2.5 μ or less	$PM_{2.5}$	€ 56.8	€ 79.5	€ 122	x			nc				x		x
Coarse particulates, 10 μ or less	PM_{10}	€ 31.8	€ 44.6	€ 69.1	x			nc				x		x
Nitrogen oxides	NO_x	€ 24.1	€ 34.7	€ 53.7	x	x	x	nc			x	x	x	x
Sulphur dioxide	SO_2	€ 17.7	€ 24.9	€ 38.7	x	x	x	nc				x	x	x
Ammonia	NH_3	€ 19.7	€ 30.5	€ 48.8	x		x				x	x	x	
Volatile organic compounds	NMVOC	€ 1.61	€ 2.1	€ 3.15		x						x	x	X
Carbon monoxide	CO	€ 0.0736	€ 0.0958	€ 0.152		x						x		
Methane[2]	CH_4	€ 0.448	€ 1.75	€ 1.77		x		x				nc	nc	nc
Cadmium	Cd	€ 798	€ 1159	€ 1,831						x	x	x	x	
Arsenic	As	€ 703	€ 1033	€ 1,228						x	x	x	x	
Lead	Pb	€ 3,967	€ 5,908	€ 6,596						x	x	x	x	
Mercury	Hg	€ 24,770	€ 34,480	€ 53,630						x	x	x	x	
Formaldehyde	CH_20	€ 19.3	€ 26	€ 40		x				x	x	x	x	

1 An x indicates the pollutant has been characterized on the midpoint or endpoint; nc = not calculated, climate emissions being priced using abatement costs rather than damage costs.

15 With the exception of $PM_{2.5}$; see Section 6.4.

[2] The value reported for greenhouse gases includes VAT and increases at 3.5% per annum from the 2015 baseline. These values can therefore only be be used for 2015 emissions. For valuation in later years, see Section 6.3.

These environmental prices are average prices for the Netherlands. For particulate matter the specific emission site is crucially important. In Sections 6.4.9 and 6.4.10, PM damage costs in specific industry and traffic settings are considered in more detail.

3.2.2 Environmental prices for emissions to water

For emissions to water, prices were calculated for the 'prioritary pollutants' for which targets are laid down in the European Water Framework Directive, supplemented by total nitrogen,total phosphorus and phosphate, key factors in eutrophication. Table 6 reports the lower, central and upper values.[16]

Table 6 Environmental prices for emissions to water of prioritary and eutrophying pollutants (€$_{2015}$ per kg 2015 emission)

Pollutant	Environmental price (€/kg emission)			Relevant midpoints		
	Lower	Central	Upper	Eutrophic.	Human tox.	Ecotox.
1,2-Dichloorpropane	€ 18.3	€ 25.1	€ 38.7		x	x
Atrazine	€ 3.3	€ 11	€ 20.9		x	x
Aldrin	€ 1645	€ 2256	€ 3487		x	x
Benzene	€ 0.0568	€ 0.0794	€ 0.124		x	x
Beryllium	€ 7.44	€ 26.9	€ 52.3		x	x
Captan	€ 0.0208	€ 0.0805	€ 0.156		x	x
DDT	€ 47.3	€ 67.4	€ 106		x	x
Dichloromethane	€ 1.78	€ 2.44	€ 3.77		x	x
Dichlorvos	€ 0.19	€ 0.344	€ 0.583		x	x
Dicofol	€ 249	€ 342	€ 529		x	x
Ethylbenzene	€ 0.00546	€ 0.0129	€ 0.0232		x	x
Hexachlorobenzene	€ 408	€ 559	€ 865		x	x
Naphthtalene	€ 0.188	€ 0.289	€ 0.466		x	x
Pentachlorophenol	€ 2.15	€ 8.66	€ 16.9		x	x
Phosphate (PO_4)	€ 0.156	€ 0.629	€ 1.22	x		
Tetrachloroethylene	€ 7.45	€ 10.2	€ 15.8		x	x
Total nitrogen (N)	€ 3.11	€ 3.11	€ 3.11	x		
Total phosphorus (P)	€ 0.473	€ 1.9	€ 3.71	x		
Trichloromethane (chloroform)	€ 3.12	€ 4.27	€ 6.6		x	x
Trifluralin	€ 13	€ 18.4	€ 28.8		x	x
Zinc	€ 0.168	€ 1.14	€ 2.96		x	x

3.2.3 Environmental prices for emissions to the soil

Emissions to the soil can occur via waste dumping or leakage or eutrophication, potentially impacting ecosystems and/or human health. Table 7 reports the environmental prices of several key pollutants with respect to soil pollution. The impacts they may have on IQ have not be quantified. For heavy metals there is a substantial difference between the upper and lower value. This is explained further in Section 6.8 and is, amongst others, due to scientific uncertainty about dispersion of these pollutants in the food

[16] No environmental price could be established for di(2-ethylhexyl)phthalate (DEHP), as ReCiPe provides no characterization factor.

chain (via uptake by crops and animals) and the resultant impacts on human health. The lower value is based on far more conservative assumptions than the upper value.

Table 7 **Environmental prices for key emissions to the soil (€$_{2015}$ per kg emission)**

Pollutant	Lower	Central	Upper
Cadmium	€ 24.3	€ 2,039	€ 6,248
Arsenic*	€ 21.6	€ 69.3	€ 168
Lead*	€ 0.107	€ 14.2	€ 43.6
Mercury*	€ 864	€ 1,549	€ 2,959
Nickel	€ 0.0326	€ 0.342	€ 0.965
Formaldehyde	€ 1.51	€ 2.06	€ 3.19
P-fertilizer**	€ 0.0251	€ 0.101	€ 0.196
N-fertilizer	€ 0.227	€ 0.227	€ 0.227

* These values do not include loss of IQ associated with soil pollution.

** These are European values that are not necessarily representative for the Netherlands.

3.2.4 Environmental prices for other impacts

Environmental prices have also been derived for noise nuisance and land use. Those for noise nuisance indicate the external costs of both health damage and noise-related nuisance. Those for land use are the external costs of the biodiversity loss associated with the land use.

For road-traffic noise the environmental prices reported in Table 8 can be used; these increase with rising noise levels. The decibel units are explained in Section 6.11.3.

Table 8 **Environmental prices for road-traffic noise nuisance (€$_{2015}$ per dB (Lden) per person per year)**

Noise nuisance	Lower	Central	Upoer
50-54 dB(A)	21	26	31
55-59 dB(A)	40	48	58
60-64 dB(A)	43	52	64
65-69 dB(A)	80	97	117
70-74 dB(A)	84	103	125
75-79 dB(A)	89	108	134
>= 80 dB(A)	91	111	138

Rail-traffic noise nuisance is generally valued lower, air-traffic noise higher. Precise values for these two variants, including a breakdown into damage costs for nuisance and health, are reported in Section 6.11.

Environmental prices for land use are shown in Table 9. These are the annually recurring costs to be attributed to use of an average m^2 of land in the Netherlands.

Table 9 **External costs of land use (€$_{2015}$/m^2 per year)**

	Lower	Central	Upper
Netherlands	€ 0.007	€ 0.026	€ 0.050

External cost estimates for several specific types of land are reported in Section 6.12.

3.3 Use of environmental prices by companies

3.3.1 Why environmental prices?

For companies, sustainability is a key constraint on production. Developments and challenges in the realm of sustainability are coming thicker and faster all the time. One of the challenges is suitable integration of the whirlwind of developments in research, technology and policy into everyday business operations (Figure 8).

A growing number of companies now view sustainability not merely as a constraint but as an opportunity to be grasped. By saving on energy and raw materials and by recycling they can add economic value to their production processes while at the same time contributing to a sustainable world. They are appreciating that innovation is not only possible in the production phase but up and down the entire supply chain. In all of this, financial value (price) plays a key role. Prices provide information on the value society assigns to products, but also on the costs of getting the product onto the market.

Some products are unpriced, but still of value to society. The environment is a case in point. There is a risk of gains and losses to society being inadequately reflected in product prices, while at the same time it is hard for sustainability issues to be weighed against financial. This is where environmental prices offer a solution, for they reflect the price society is prepared to pay to avoid pollution or to produce more sustainably. Environmental prices mean a price is put on the impacts of pollution on human beings, plants and animals: the financial value people would assign to a clean environment if it were on sale in a shop like other goods.

Figure 8 Challenges and potential solutions for companies in relation to sustainability

3.3.2 User applications of environmental prices

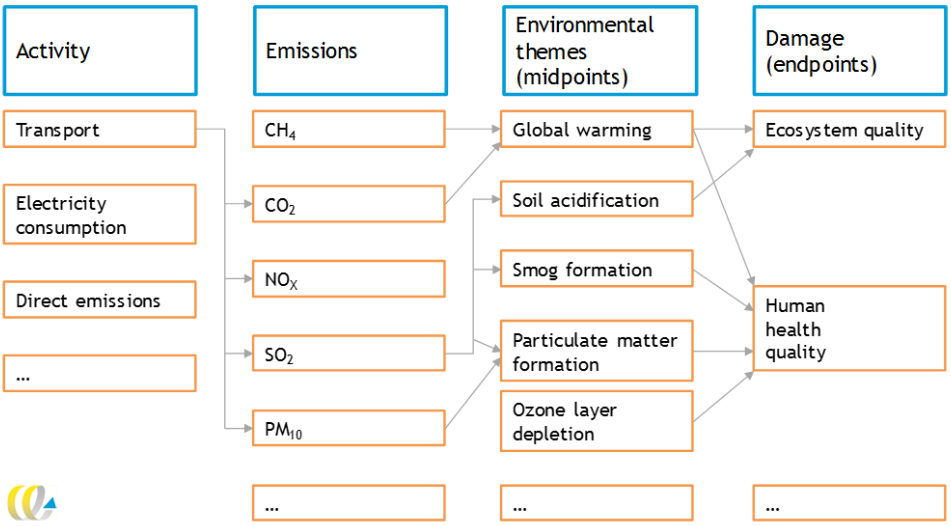
Environmental prices can be used by companies in the context of Corporate Social Responsibility (CSR) to quantify progress on certain sustainability issues, viz. those associated with the environment and the health and wellbeing of human beings and living nature.

Environmental prices can be used in the following situations, for example:
– to improve insight into where in a company's value chain the greatest environmental gains can be made;
– to calculate the sustainability gains achievable via improved procurement policies;
– to assess how additional energy use for recycling compares with reduced primary resource consumption;
– to crunch the numbers on the environmental impacts of alternative investments, equivalent to doing calculations on a social business case;
– to calculate a uniform environmental score for use in an environmental annual report.

3.3.3 How do environmental prices work?

Business activities like transportation, gas and power consumption and material and feedstock production lead to emissions of toxic and otherwise damaging substances. These pollutants have different environmental impacts. Some contribute to global warming, others to soil eutrophication, yet others to ozone layer depletion, while some are toxic to humans or animals. Sometimes an emission of a particular pollutant has several different environmental impacts. Sulphur dioxide, for instance, causes particulate matter formation, photochemical smog and soil acidification. Figure 9 shows a typical example of an industrial activity with a range of environmental impacts causing ultimate damage to human health and ecosystems.

Figure 9 Relationship between industrial activities, emissions and the environment



Note: This figure is merely an illustrative example and is not intended to provide a full picture of environmental cause-effect relationships.

Environmental prices put a price on the chain from emissions to ultimate damage. They are of no use for translating a particular industrial activity into emissions, however. For this purpose there are dedicated tools available, such as the SME Environmental Barometer developed by the Stimular Foundation.

Companies can of course also carry out their own analysis. Frameworks available for this purpose include existing reporting obligations to local or national authorities, emissions registration under the EU Emissions Trading Scheme (EU ETS) and reporting to the European Pollutant Release and Transfer Register (E-PRTR).

3.3.4 Concrete use of environmental prices

If quantitative emissions are known, environmental prices can be used to calculate the environmental damage of the company activity concerned or the environmental benefits of an envisaged investment. To do so, the physical emissions (in kg pollutant) are multiplied by the relevant environmental prices (in €/kg pollutant) to express the aggregate resultant impacts in Euros. This price figure stands for the sum total of the environmental impacts of the pollutant concerned. In the case of SO_2 emissions, for example, it accounts for soil acidification, smog formation and particulate matter formation. In this way all the environmental impacts resulting from the company's various -activities can be expressed in monetary terms.

For use by companies, we recommend taking the central values of the environmental prices reported in this Handbook. Table 10 lists these for a (small) selection of emissions covered by the E-PRTR.

Table 10 Central values for emissions to air and water of common E-PRTR pollutants (€/kg emission)

Air compartment	€/kg	Water compartment	
Coarse particulates (PM_{10})	€ 44.6	Aldrin	€ 2,256
Nitrogen oxides	€ 34.7	Arsenic	€ 433
Sulphur dioxide	€ 24.9	Atrazine	€ 11
Ammonia	€ 30.5	DDT	€ 67.4
Volatile organic compounds (NMVOC)	€ 2.1	Dichloromethane	€ 2.44
Methane	€ 1.75	Hexachlorobenzene	€ 559
Cadmium	€ 1,159	Mercury	€ 1,980
Arsenic	€ 1,033	Naphthalene	€ 0.289
Lead	€ 5,908	Pentachlorophenol	€ 8.66
Mercury	€ 34,480	Trichloromethane	€ 4.27
Toluene	€ 3.66	Zinc	€ 1.14

Environmental prices for other emissions can be found in the tables in Section 3.2 and Annex D or retrieved from the online tool available at www.cedelft.eu/en/environmental-prices. We reiterate that companies should always take the central value.

In the case of greenhouse gas emissions, companies must make their own choices on what perspective to adopt. There are two possibilities:

- proceed from current policy;
- proceed from policy required to achieve the 2°C Paris target.

If the company opts to calculate the costs of greenhouse gas emissions based on current policy targets, the associated prices must be taken. Continuing with current policy, i.e. linearly rising emissions cuts under the EU ETS and a continuation of policies favouring renewables, will in all likelihood lead to the planet warming by 2.5-3.5°C by the end of the present century (the High Scenario in the WLO calculations: (Aalbers, et al., 2016).

If, alternatively, the company is keen to participate in efforts towards achieving the 2°C target, the CO_2 prices for that scenario must be taken. In both scenarios the CO_2 prices increase linearly over time by 3.5% per annum (excluding inflation) relative to 2015 values, as shown in Table 11.

Table 11 **CO_2 prices (excl. VAT) associated with two policy targets for various years (€/tCO_2, 2015 constant prices)**

	2015	2020	2030	2040	2050
Current policy	48	57	80	113	160
2°C policy	80	95	130	180	260

For companies undecided as to which prices to use, we would recommend taking those for the 2°C target, since these correspond best with the premises of Corporate Social Responsibility. If the CO_2 prices are used together with other environmental prices, 18% VAT must be added (the average rate for consumer expenditures in the Netherlands as has been calculated in SEO, (2016b).[17]

3.4 Use of environmental prices in SCBA

3.4.1 General framework

Social Cost-Benefit Analysis (SCBA) is a decision-support tool that can be used to clarify the considerations at work in government policy elaboration. Most policy alternatives have a range of impacts, and by expressing as many of these as possible in monetary terms, they can be compared, providing valuable information on the pros and cons of each alternative (CPB; PBL, 2013).

In 2013 General Guidelines for SCBA were published in the Netherlands (CPB; PBL, 2013), prescribing how such analyses are to be carried out. These Guidelines were subsequently elaborated in more detail for individual policy domains (in so-called *Werkwijzers*). In 2017 CE Delft drew up SCBA Guidelines for the Environment (CE Delft, 2017). The guidelines, indices and recommendations in this document can be used in environmental policy-making as well as in other policy areas with major environmental implicatiions or impacts.

3.4.2 User applications in SCBA

SCBA can be performed for a wide variety of purposes, including the following:
- Concrete government investments, such as motorway construction or introduction of separated household waste collection. In this case there are (government) investment costs and social benefits in the form of reduced pollution, which SCBA allows to be compared.
- Environmental policy instruments, such as a waste charge or renewable energy subsidy. In this case the government is setting a framework for compelling or 'nudging' industries and consumers to invest or change their behaviour. In such cases, besides policy costs there are above all private costs to industries and/or consumers and social benefits through reduced pollution.

17 In all the tables in which CO_2 prices are reported along with other prices, the CO_2 prices have already been increased by 18% VAT to make them comparable with those of other pollutants. See also the discussion in Section 3.4.3.

– Exploration of policy options, such as whether air-quality standards need to be tightened or recycling targets increased from the perspective of social welfare. In this case SCBA supports the problem analysis and explores whether additional environmental policy is desirable in welfare terms.

3.4.3 How are environmental prices used in SCBA?

In SCBA environmental impacts are quantified whenever possible as volume changes in pollutant emissions to soil, air and water.[18] Emissions are dispersed through the environment, leading ultimately to impacts on endpoints: human health (morbidity and mortality), ecosystem services, buildings and materials, resource availability and nuisance. Environmental prices establish a link between emissions and endpoint impacts and assign a value to those impacts.

Environmental prices can thus be used in a SCBA and are recommended in situations in which it is unknown where in the Netherlands the environmental impacts occur, or if such impacts are only a minor, secondary issue in the SCBA. If the SCBA is concerned with a measure or policy with markedlly regional or local impacts, use of environmental prices is not to be recommended. Environmental impact assessment using a method like the impact pathway approach is then recommended. Also, if substantial funds are available for the SCBA and major environmental impacts are anticipated it is recommended to carry out a dedicated environmental impact assessment to explicitly model and estimate the relationships between emissions and impacts on all endpoints. In doing so, the endpoint impact values reported in Chapter 5 of this Handbook can be used.

The recomendation in the SCBA Guidelines for the Environment referenced above (CE Delft, 2017) is to value environmental impacts as far as possible, but to explicitly identify the uncertainties surrounding the monetary values by working with upper and lower values. These upper and lower values must then also be applied in both the High and Low WLO scenarios (CPB; PBL, 2015a).

The environmental prices presented here have been constructed to implicitly include VAT. This is because most of these prices are based on willingness-to-pay studies, where consumers base their preferences on prices that include VAT.

When it comes to climate change this is different: here prices are based on the costs of policy measures calculated by CPB and PBL exclusive of VAT. However, it is not entirely clear what VAT rates should be added to the CO_2 prices to make them comparable with the other prices.[19] We therefore propose that 'climate prices' be used in SCBAs with 18% VAT added (the average VAT rate; see SEO, 2016b) until further research has shown what net impact VAT has on these prices.

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18 SCBA also includes nuisance impacts (noise nuisance, visual nuisance), which while not quantified as emissions can still be valued using environmental prices.

19 No pronouncement can be made, however, as to whether these 'climate prices' including VAT would be higher or lower than those reported in Section 3.3. On the one hand they may be higher, because the technology costs of the measures do include VAT, but on the other they may be lower, if savings on revenues are raised by VAT (including the VAT on the energy tax). In the framework of this Handbook it cannot therefore be stated a priori which effect will dominate and to what extent.

3.4.4 Overview of prices

The SCBA Guidelines for the Environment (CE Delft, 2017) review the prices to be used in SCBAs. They are also reported as the upper and lower values in the tables in Section 3.2.
In addition, there are adjusted 'climate prices' for use in SCBA. These are adjusted for use in the High and Low WLO scenarios (see Text Box 1), which use specific prices derived in (Aalbers, et al., 2016), which can be described as least-cost prices to attain climate policy targets, or in the wording of the authors "efficient CO_2 prices". Table 12 summarizes the prices to be used in any SCBA with major climate impacts.

Table 12 Efficient CO_2 prices (excl. VAT) according to WLO (€/tCO_2, constant prices)

	2015	2030	2050
Low scenario	12	20	40
High scenario	48	80	160
2°C	60-300	100-500	200-1,000

Box 1 WLO scenarios and climate

The WLO ('Prosperity and Living Environment') scenarios were published at the end of 2015 by the Netherlands Bureau for Economic Analysis (CPB) and the Netherlands Environmental Protection Agency (PBL) for use as standard reference scenarios. The scenarios reflect future trends with their associated uncertanties. While few assumptions are made with regard to specific policies, this does not hold for the climate and energy field, where it is assumed there will be major international progress on climate policy, with inescapable future consequences for the Netherlands. The High and Low scenarios reflect differences as to which of the pledges under the Paris Agreement are honoured. In the Low scenario it is assumed that only the unconditional pledges are fulfilled, resulting in 45% lower emissions in 2050 for the EU (and thus the Netherlands) relative to 1990 and a temperature rise of 3.5-4°C by the end of this century. In the High scenario the conditional and unconditional pledges are all honoured, leading to 65% lower emissions in 2050 relative to 1990 and a temperature rise of 2.5-3°C post-2100. It should be noted, though, that the stated goal of 'Paris' was to restrict this rise to 2°C at most, and preferably to less than 1.5°C.

3.5 Use of environmental prices as midpoint weighting factors in LCA

3.5.1 General framework

Environmental prices can also be used for weighting environmental impacts in Life Cycle Assessment and allied applications. These prices signify the relative value of emissions compared with one another and with other goods circulating in the marketplace. When emissions are valued in exercises like SCBA, their value is usually considered relative to other financial parameters. When weighting emissions in LCA, though, the primary interest is mutual comparison among emissions. These weighting factors can then be regarded as the socio-economic weight attributed to the various environmental impacts.

Weighting factors depend on the characterization method adopted, and the factors developed in this Handbook were developed on the basis of the characterization adopted in ReCiPe.[20] To a certain extent these factors can

[20] The environmental prices for use by companies and in SCBAs are based on 3% annual discounting of future developments and the individualist perspective for environmental characterization. While this is in line with standard economic discounting practice, in LCAs

also be adopted in other characterization methods, such as CML2 (Guinée, et al., 2002) or the PEF methodology ILCD (JRC, 2012).One problem, though, is that on a number of environmental themes these methods base their weighting on different pollutants than the ones considered here. Simple conversion is often unfeasible, because the midpoint environmental prices developed here were calculated in conjunction with characterization. The main elements of our method for calculating weighting factors are described in Section 4.2 (and in more detail in Annex G of the original Dutch language report).
A key issue to appreciate is that the weighting factors in this Handbook are a weighted average of the relative damage caused in the Netherlands by the various pollutants with respect to the midpoint concerned. For other countries and other characterization methods, different weighting factors thus hold.[21]

3.5.2 Environmental prices as weighting factors

The environmental prices that can be used as weighting factors in Life Cycle Assessment are reported in the third column of Table 13. These factors are based on the environmental prices calculated in this project by CE Delft and are specifically suited for use in LCAs according to the ReCiPe methodology under the hierarchist perspective, the one most commonly adopted in LCAs (cf. Annex A).

If the purpose of the LCA is to obtain estimates of external costs, however, the weighting factors must be taken from the external cost set in the last column. Apart from ozone layer depletion, acidification and land use, these are identical to the values used as weighting factors. As explained in Annex A of the Dutch language Handbook, in calculating external costs we opted for a mix between the individualist and hierarchist perspective, as this is is most in line with the premises of economic damage estimation.

Table 13 Environmental prices per impact category, for use in LCA

Impact category	Unit	Environmental price as weighting factor	Environmental price as external cost
Climate change	€/kg CO_2-eq.	€ 0.057	€ 0.057
Ozone layer depletion	€/kg CFC-eq.	€ 123	€ 30.4
Human toxicity	€/kg 1,4 DB-eq.	€ 0.158	€ 0.214
Photochemical oxidant formation	€/kg NMVOC-eq.	€ 2.1	€ 2.1
Particulate matter formation	€/kg PM_{10}-eq.	€ 69	€ 69
Ionizing radiation	€/kg kBq U_{235}-eq.	€ 0.0473	€ 0.0473
Acidification	€/kg SO_2-eq.	€ 8.12	€ 5.4
Freshwater eutrophication	€/kg P-eq.	€ 1.9	€ 1.9
Marine eutrophication	€/kg N	€ 3.11	€ 3.11
Terrestrial ecotoxicity	€/kg 1,4 DB-eq.	€ 8.89	€ 8.89
Freshwater ecotoxicity	€/kg 1,4 DB-eq.	€ 0.0369	€ 0.0369
Marine ecotoxicity	€/kg 1,4 DB-eq.	€ 0.00756	€ 0.00756
Land use	€/m^2a	€ 0.037	€ 0.0261

the hierarchist perspective is generally used. To guarantee consistent use of environmental prices in LCAs, we have also calculated a central value according to hierarchist principles, which we propose using for LCA weighting. This value is reported solely as an environmental price at midpoint level and can be used with the weighting factors.

21 In principle it should also be possible to develop weighting factors for impacts at endpoint level using the monetary values presented in Chapter 5, by elaborating values for the ReCiPe endpoints DALY and PDFs. To do so would in all likelihood involve specific conversion steps that are beyond the scope of this Handbook. One aspect requiring special consideration would be the different discounting procedures adopted in ReCiPe and here.

3.5.3 How are environmental prices used in a LCA?

These weighting factors can be used in Life Cycle Asssessment of products or raw-material supply chains. For this purpose, the environmental prices must be multiplied by the outcomes of the LCA at midpoint level. As stated above: if the purpose of using environmental prices is weighting, we recommend the "weighting factor" set. If the purpose is the calculation of external costs, we recommend the "external cost" set. The "weighting factor" set is entirely based on the Hierarchistic Perspective from ReCiPe whereas the "external cost" is based on a combination of hierarchist and individualistic perspectives as stated in Annex A.

In Table 14, we provide an example of a (fictional) woven and dyed textile product weighing 200 grams and made of 60% cotton/40% polyester. As can be seen, the environmental price of this product sums to a total of € 0.51, with the environmental impacts PM formation and climate change contributing most.

Table 14 Example: individual impact scores in LCA and weighted environmental score using environmental prices

Impactcategorie	LCA score for 200-g garment, woven and dyed, 60% cottton/40% polyester A	Unit x	Environmental price per environmental impact/ indicator B	Unit =	Result
Climate change	2.44	kg CO_2-eq.	€ 0.0566	€/kg CO_2-eq.	€ 0.14
Ozone depletion	1.78E-07	kg CFC-11-eq.	€ 30.4	€/kg CFC-eq.	€ 0.00
Acidification	0.010	kg SO_2-eq.	€ 5.4	€/kg SO_2-eq.	€ 0.05
Freshwater eutrophication	2.99E-04	kg P-eq.	€ 1.9	€/kg P-eq.	€ 0.00
Marine eutrophication	3.37E-04	kg N-eq.	€ 3.11	€/kg N	€ 0.00
Human toxicity	0.13	kg 1,4 DB-eq.	€ 0.214	€/kg 1,4 DB-eq.	€ 0.03
Photochemical oxidant formation	0.0047	kg NMVOC	€ 2.1	€/kg NMVOC-eq.	€ 0.01
Particulate matter formation	0.0032	kg PM_{10}-eq.	€ 69	€/kg PM_{10}-eq.	€ 0.22
Terrestrial ecotoxicity	4.27E-04	kg 1,4 DB-eq.	€ 8.89	€/kg 1,4 DB-eq.	€ 0.00
Freshwater ecotoxicity	8.97E-04	kg 1,4 DB-eq.	€ 0.0369	€/kg 1,4 DB-eq.	€ 0.00
Marine ecotoxicity	1.96E-03	kg 1,4 DB-eq.	€ 0.00756	€/kg 1,4 DB-eq.	€ 0.00
Ionizing radiation	0.22	kBq U_{235}-eq.	€ 0.0473	€/kg kBq U_{235}-eq.	€ 0.01
Land use	1.8	m^2a	€ 0.0261	€/m^2	€ 0.05
Total weighted LCA score using environmental pricing:					**€ 0.51**

PART 2: METHODOLOGICAL PART

4 Calculating environmental prices

4.1 Introduction

This chapter discusses how the environmental prices in this Handbook were calculated. First, in Section 4.2 we set out the general methodology, which is based on harmonizing the premises of existing valuation methods, impact pathway analyses and characterization models. We then explain the changes made to the methodology followed in the 2010 Shadow Prices Handbook. This is done in Section 4.3 for the main elements of valuation, in Section 4.4 for the characterization models and in Section 4.5 for the impact pathway approach. Section 4.6, finally, discusses the the use of the environmental prices in the present Handbook now and in the future.

This chapter focuses on the main methodological changes. The precise considerations and literature underpinning these changes are discussed in further detail in Chapters 5 and 6. (In the appendices of the original Dutch language edition there is more comprehensive treatment)

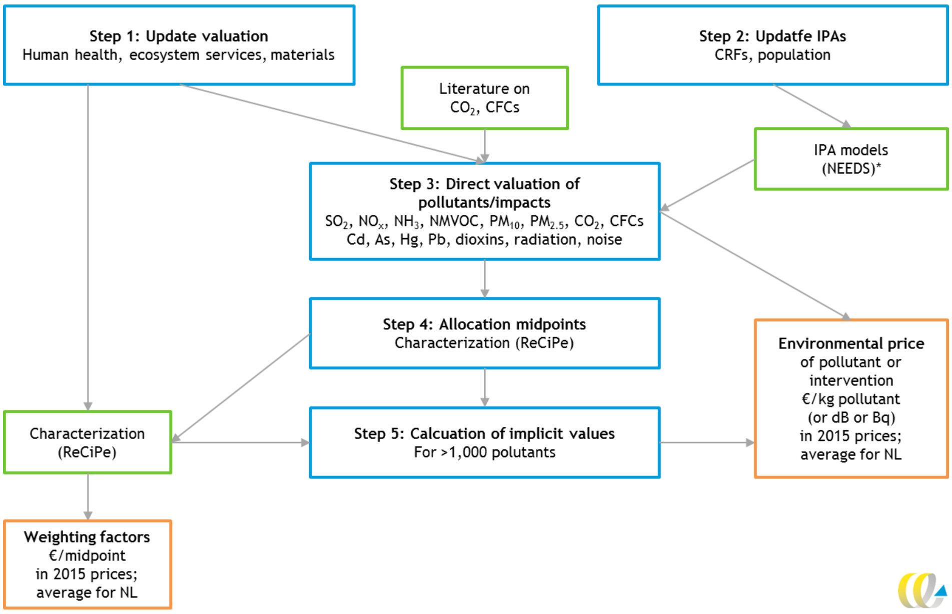
4.2 General methodology

As explained In Section 2.3, the environmental prices presented in this Handbook have been derived by combining three kinds of models/methods:

1. Characterization models defining physico-chemical relationships between interventions like emissions and midpoint impacts (midpoint characterization) and between midpoint impacts and endpoints (endpoint characterization).
2. Impact pathway models describing the relationships between emissions and endpoint impacts, mapping environmental dispersal of emissions and the impacts of the resultant concentrations on humans, animals, plants and buildings/materials.
3. Valuation techniques establishing a financial relationship between endpoint impacts and the changes in economic welfarde resulting from altered availability of the endpoint.

Just as in the Shadow Prices Handbook (CE Delft, 2010) the methodology employed in the present Environmental Prices Handbook combines work in all three fields of research. The process adopted to update the former prices is shown schematically in Figure 10.

Figure 10 The Environmental Prices Handbook methodology



Five steps can be distinguished.

In the first step, for each of the five endpoints adopted here monetary values were established that are in accordance with recent international literature and the premises laid down in the General SCBA Guidelines (CPB; PBL, 2013)) and the recommendations of the Discount Rate Working Group. This yielded values for human health, biodiversity, agricultural crops and material restoration costs, all in 2015 prices. These values are broadly discussed in Section 4.3, with a more detailed examination in Chapter 5.

Next, in Step 2, the impact pathway analyses (IPAs) were updated. These specify the relationship between emissions in the Netherlands and impacts on endpoints and are built around concentration-response functions (CRFs). These adjustments are broadly discussed in Section 4.5 and in more detail in Chapter 5.

In Step 3 the updated values and IPAs, combined with the results of literature analyses on CO_2 and CFCs, were used for direct valuation of fifteen pollutants or pollutant groups. These values constitute the environmental prices presented for these substances in this Handbook.

Step 4 then consists of allocating these fifteen pollutants or pollutant groups across the various midpoints. Most of these pollutants (the exception being PM) have impacts on multiple environmental themes. The manner of allocation is the same as used in the 2010 Shadow Prices Handbook and is briefly described in the present Annex C.

Next, in Step 5, the damage cost of the various pollutants on each environmental theme was weighted using 2015 Dutch emissions (converted to ReCiPe equivalency factors) to arrive at a weighted average value for damage at the midpoint level. This allows the damage due to all the pollutants characterized in ReCiPe to be calculated and a weighted average midpoint damage factor to be derived. An implicit environmental price is thus calculated for all the pollutants characterized in ReCiPe with respect to the endpoints adopted here.

The main elements of the methodology are shown Figure 10. In Chapters 5 and 6 more detailed descriptions are provided on how the various choices involved in the respective steps were made. Below, we look in more detail at how the valuation methods, characterization and IPAs were adapted compared with the 2010 Shadow Prices Handbook.

4.3 Methodology update: valuation

First of all, the extent to which the valuation principles adopted in the Shadow Prices Handbook needed adjusting was assessed in general terms. As this is reported at length in Chapter 5, here this issue is considered only briefly.

In general, three changes have been introduced:
1. All prices have been brought in line with the recommendations of the Discount Rate Working Group, viz. to no longer apply a positive income elasticity for valuation of health (cf. Section 5.3). For ecosystems a relative price rise of 1% per annum was adopted.
2. All prices have been adjusted to 2015 prices.
3. All prices are presented as an upper and lower value and have been adjusted to incorporate the latest findings reported in the literature.

The main changes in the valuation system are summarized in Table 15.
In Chapter 5 the choices made are explained in more detail and the values for pollutant impacts on materials and buildings, wellbeing and resource availability are also discussed.

Table 15 Changes to the valuation system

	2010 Shadow Prices Handbook	This Handbook
Price level	2008 prices	2015 prices
Income elasticity	0.85%	0%. Prices also not adjusted to income elasticity between 2005 and 2015.
Value for human health	VOL = € 40,000 chronic in 2000 prices, € 55,0000 in 2008 prices	For mortality a range from € 50,000 to € 110,000 (2015 prices), for morbidity from € 50,000 to € 100,000.
Value for ecosystems	€ 0.56/PDF/m^2 based on average European values from (Kuik, et al., 2008)	A range from € 0.16/PDF/m^2 to € 1.23/PDF/m^2 (2015 prices) based on own calculations for the Netherlands proceeding from (Kuik, et al., 2008)

4.4 Methodology update: characterization

4.4.1 2010 Shadow Prices Handbook

In the previous Handbook, characterization was based on ReCiPe (November 2009 version), adopting the hierarchist perspective. In this respect no changes have been made here, except for the midpoint 'particulate matter formation', where ReCiPe was not used to derive the contributions of the various pollutants, with these being estimated directly using the NEEDS impact pathway approach. The relative contributions of $PM_{2.5}$ and PM_{10} was estimated using our own calculations.

4.4.2 New developments

Since the previous Handbook there have been developments in how emissions are characterized with respect to impacts and the associated indicators. On the one hand, ReCiPe was updated in 2012 and 2013. On the other, the ILCD (International Life Cycle Data) method has been developed by the European Commission's Joint Research Centre (JRC, 2012).This method is now widely applied in Western Europe and has been used to develop the Product Environmental Footprint and Organization Environment Footprint (PEF/OEF) frameworks, among other things.

In calculating environmental impacts and environmental damage these 'umbrella' analysis methods all make frequent use of the same underlying methods. Climate impacts, for example, are calculated using the method developed by the Intergovernmental Panel on Climate Change (IPCC). Nevertheless, the ReCiPe and ILCD methods each adopt a slghtly different approach for human toxicity and land use, for example. In ReCiPe, environmental impacts can be characterized with respect to endpoint damage to human and ecosystem health within a consistent framework. This is not currently feasible in the ILCD approach, though there are plans for further characterization to endpoint damage in the future. In Annex A more information is provided on the differences between ReCiPe and ILCD.

4.4.3 Choices made in this Handbook

In this new Handbook it was opted to again perform characterization on the basis of ReCiPe, for the following three reasons:

1. ReCiPe has multiple characterization methods, depending in part on the perspective adopted (individualist, hierarchist, etc.), but none of these methods is entirely compatible with the impact pathway approach. The individualist perspective in ReCiPe does, however, show some similarity with the discounting adopted in NEEDS.[22]
2. ReCiPe works with harmonized characterization from midpoint to endpoint. This is a major advantange compared with ILCD, where endpoint characterization in particular is still in an early stage of development.
3. ReCiPe is regularly maintained and updated.

In this Handbook, characterization is based on the ReCiPe characterization factors (Version 1.12, April 2016). In contrast to the 2010 Shadow Prices Handbook, the individualist perspective has now been adopted throughout, apart from a few environmental themes discussed in more detail in Chapter 6.

[22] NEEDS works with a 50-year horizon and a 3% discount rate. In the ReCiPe individualist perspective, on certain themes impacts only count for the first 20 years post-emission. If a given value is discounted at 3% p.a. over a 30-year period, this gives approximately the same result as a 20-year horizon with no discounting. A cut-off point therefore works similarly to using a discount rate. Because of the net price increase of 1% p.a. (see Section 5.3.6), for land-use changes it was opted to adopt the ReCiPe hierarchist perspective for characterization.

4.5 Methodology update: impact pathway approach

4.5.1 2010 Shadow Prices Handbook

Traditionally, two approaches have been adopted for expressing emissions in monetary terms: NEEDS and CAFE-CBA. Both proceeded from the impact pathway approach (IPA), in which emissions are related via atmospheric transport and dose-effect relationships to endpoint impacts, which are then assigned a monetary value. NEEDS and CAFE-CBA essentially consist of four interlinked models/databases:

1. Emission databases (and/or projections).
2. Dispersion models converting emissions to concentrations, using a combination of meteorological and atmospheric-chemistry models.
3. Concentration Response Functies (CRFs) converting concentrations to physical endpoint impacts on health, ecosystem services, buildings, etc.
4. Monetary valuation of these physical impacts.

In the 2010 Shadow Prices Handbook the two methods are described in more detail. The handbook Shadow Prices uses then subsequently the NEEDS impact pathway approach.

4.5.2 New developments in the literature

Since 2009 there has been no further development of NEEDS. CAFE-CBA, however, has been further elaborated by, amongst others, Holland (2014); IIASA (2014) and used in policy estimation for European agreements on transboundary air pollution.

In the framework of the present project it was examined whether it would be possible to link up with CAFE-CBA models to value the damage costs of air-pollutant emissions. This proved difficult, as the reporting method is not very transparent and the authors have not made the underlying models and assumptions publically available. For air pollution It only proved feasible to calculate an EU-average value (see the Dutch Annex C).

It is also striking is that recent shadow price manuals for Ireland, Belgium and Germany (under development) are still based on the NEEDS methodology owing to its far greater transparency. This methodology has the added advantage of allowing a certain amount of adjustment, since the underlying spreadsheets and modelling runs were made available to us for the purposes of this project.

In this Handbook it was therefore opted to base environmental prices on NEEDS, adjusting the estimates wherever possible to the 2015 context.

4.5.3 Choices made in this Handbook

It was thus opted to use the NEEDS model, with the following three adjustments being made:

1. Dutch pollutant emissions in 2015 were far lower than in 2005, leading to changes in atmospheric chemistry. The NEEDS results therefore had to be adjusted to the lower background concentrations.
2. In 2015 the Dutch population had grown and was, on average, older compared with 2005. The NEEDS results therefore had to be adjusted to current population size and composition.
3. In 2015 more research results had become available on air-pollution impacts with, in particular, (WHO, 2013); 2014) publishing new recommendations on how these impacts should be included in calculations.

Below we explain how exactly these adjustments were made.

Lower background concentrations
Parts of the NEEDS model, such as the dispersion and atmospheric-chemistry models, could not be explicitly unpacked by us. However, because there are numerous NEEDS modelling runs available for estimating emission reduction scenarios, the underlying model structure can to a certain extent be derived. It was opted to proceed from the 2010 and 2020 emission scenarios in the NEEDS Excel tool (as used in the Ecosense dispersion model). Actual 2015 Dutch emissions were then scaled to the difference between the 2010 and 2020 values. These results were put to and discussed with atmospheric-chemistry experts and explanations for a rise or fall in damage costs per kg pollutant elaborated. In this way an adjustment was made for the lower background pollutant levels in 2015 and their influence on damage estimates.

Diferent population composition
The NEEDS modelling results are rooted in a grid-based distribution of the population and an assumed population of 16.4 million in 2015. In addition, they are based on a European average for the population composition in 2004. The Dutch population currently stands at 17 million and is also meanwhile on average older than assumed in NEEDS. Using CBS population data, a further estimate was made of the impacts of this on the CRF functions used in NEEDS (see Annex B).

New findings on air-pollution health impacts
Understanding of air-pollution health impacts has also improved in recent years (WHO, 2013), 2014), which means not all the CRFs adopted in NEEDS (2008a) are still valid. In the present study all these CRFs were individually checked and discussions held on whether they still reflect the latest scientific understanding. On this basis the CRFs for NMVOC and NO_x were adjusted upwards. In Section 6.5 (and in Annex B) this issue is discussed in more detail.

To assess whether the results obtained after these adjustments approximated recent IPA modelling results, as reported in IIASA (2014), in Annex C of the Dutch edition we carried out a rough conversion of our premises for the EU27 and compared these with the recent results in Holland (2014). The calculation in the annex shows that our method yields values lying within the uncertainty margins of the recent studies. It can therefore be concluded that our proposed adjustments probably give a realistic picture of air-pollution impacts in the Netherlands in 2015.

4.6 Use of environmental prices

4.6.1 Use of midpoint environmental prices and extension to over 2,500 pollutants

Following the procedure described in Section 4.2, and the adjustments made according to the Sections 4.3 to 4.5, the environmental price per midpoint was determined. Table 16 provides a synopsis.

Table 16 **Environmental prices for midpoints, based on individualist characterization perspective (€$_{2015}$ per kg, unless otherwise specified)**

Midpoint	Unit	Lower	Central	Upper
Climate change	€/kg CO_2-eq.	€ 0.014	€ 0.057	€ 0.057
Ozone depletion	€/kg CFC-eq.	€ 22.1	€ 30.4	€ 45.7
Human toxicity	€/kg 1,4 DB-eq.	€ 0.157	€ 0.214	€ 0.331
Smog formation	€/kg NMVOC-eq.	€ 1.61	€ 2.1	€ 3.14
Particulate matter formation	€/kg PM_{10}-eq.	€ 49.3	€ 69	€ 106
Ionizing radiation	€/kg kBq U235-eq.	€ 0.0305	€ 0.0473	€ 0.0614
Acidification	€/kg SO_2-eq.	€ 1.19	€ 5.4	€ 10.7
Freshwater eutrophication	€/kg P-eq.	€ 0.473	€ 1.9	€ 3.71
Marine eutrophication	€/kg N	€ 3.11	€ 3.11	€ 3.11
Land use	€/m^2 a	€ 0.00647	€ 0.0261	€ 0.0507
Terrrestrial ecotoxicity	€/kg 1,4 DB-eq.	€ 2.21	€ 8.89	€ 17.3
Freshwater ecotoxicity	€/kg 1,4 DB-eq.	€ 0.00917	€ 0.0369	€ 0.0719
Marine ecotoxicity	€/kg 1,4 DB-eq.	€ 0.00188	€ 0.00756	€ 0.0147

Using these environmental prices for each midpoint, as a final step an extensive list of implicit environmental prices can be drawn up. This is done by using the environmental ratio between pollutants contributing to the same environmental theme as determined in ReCiPe. Annex D lists the main values for air, water and soil pollution with over 250 substances.
The values for over 2,500 pollutants are provided online at:
www.cedelft.eu/en/environmental-prices

These values can be used under the following assumptions:
- there is a linear relationship between the pollutant's contribution to the midpoint and the associated damage;
- as ReCiPe characterization is based on European averages, use in the Netherlands assumes the impact of the pollutant in relation to its midpoint environmental price is the same in the Netherlands as in Europe.

The correctness of these assumptions has not been further examined, as this was beyond the scope of the present study.

It may be noted that Table 16 can also be used for weighting environmental impacts in LCA, but as the hierarchist perspective (see Annex A) is generally adopted in LCA, different weighting factors are proposed for this purpose (see Section 3.5).

4.6.2 Use of these prices for valuing future emissions

The environmental prices reported here are valid for emissions in 2015 and it may be queried whether they change over time. After all, such prices may be used for valuing future emissions, particularly in SCBAs. So can environmental prices calculated as average prices for average Dutch emissions in 2015 also be used for valuing emissions in 2030, say?

For environmental prices relating to climate change Aalbers et al. (2016) suggests how these can be convered to annual figures. This boils down to a 3.5% price increase per annum, starting from the 2015 values (which are calculated back from the 2050 values). In this way the value of greenhouse gas emissions can be calculated for each year in the future (see also Section 6.3).

For the prices of other emissions there is no similar rule of thumb. In general, though, we advise considering the 2015 environmental prices as remaining the same for future emissions. This is based on the following considerations:

- For human health, valuation has been assumed constant in time, in line with the recommendations of the Discount Rate Working Group, which have been adopted as standard Dutch government policy.
- For the impacts of emissions on ecosystems, a 1% annual price increase can be assumed.This would lead to higher environmental prices, particularly for pollutants with severe impacts on ecosystem services. It should be noted, though, that for pollutants with combined impacts on human health and ecosystem services (e.g. SO_2, NO_x, NH_3) impacts on the former are valued far higher than on the latter.
- Paradoxically, in the Netherlands air-pollution impacts generally worsen with declining emissions. This is due to the formation of secondary aerosols and the excessively high levels of nitrogen compounds in the Dutch atmosphere (cf. Section 6.5).
- The impact of certain pollutants may decline if emissions fall below a certain threshold. It should be noted, though, that the existence of such thresholds is by no means always assumed in the toxicological and epidemiological literature and has in most cases not been assumed in calculating the impacts in Chapter 5.

In summary, there is some indication that environmental prices may increase over time, but not to any substantial degree. As a conservative approach, it has therefore been assumed that the prices remain constant.

4.6.3 Use in the future and 'expiry date'

The environmental prices calculated here can be used some way into the future. If an SCBA is carried out in 2020, for example, our environmental prices can be adjusted to 2020 price levels by correcting for inflation between 2015 and 2020, preferably using the consumer price index for this purpose. Following the recommendations of the Discount Rate Working Group, no adjustment need be made for income.

Adjusting for inflation is a non-fundamental adjustment because it involves no changes to the basic system used for calculating the environmental prices.

Fundamental adjustments are, in contrast, necessary if changes are made to the systematic variables underlying the calculations of damage costs. This may be the case if a new method is used for valuing a human life or ecosystem sevices, for example. Adjustments may also be required if the WHO publishes new findings on the potential damage of certain pollutants, say. In this area, particularly, new research is being published all the time. Rejecting a threshold for the chronic impacts of NO_2 pollution may lead to NO_x damage costs rising by around 30-50%, depending on the WHO's overall health impact assessment. New insights into environmental dispersion may also mean the environmental prices need to be updated, and the same holds if characterization factors are adjusted.

At a later date it will therefore need to be reviewed whether the environmental prices reported here still reflect the latest scientific understanding.

5 Valuation of endpoint impacts

5.1 Introduction

This chapter discusses the values of endpoint impacts used for constructing the environmental prices reported here. These values are based on a literature study. First, in Section 5.2, we provide a general review of valuation methods. We then go into more detail on valuation of the various specific endpoints:
- human health (Section 5.3);
- ecosystem services (Section 5.4);
- buildings and materials (Section 5.5);
- resource availability (Section 5.6);
- wellbeing (Section 5.7).

In each section the choices made in the earlier Shadow Prices Handbook are justified and an explanation given of the changes deemed necessary in the present Environmental Prices Handbook. For each endpoint, values are then calculated for use in valuing emissions and midpoints.

5.2 General methodology

5.2.1 General

In the damage-cost approach an attempt is made to estimate the 'demand function' for environmental quality. This function hinges on how much people are prepared to pay for environmental quality: how much of their income they are willing to sacrifice for an additional unit of environmental quality.
This is referred to as the willingness-to-pay (WTP). An alternative option is to consider how much people are prepared to pay to accept environmental damage: their willingness-to-accept (WTA). The concepts of WTP and WTA are thus both defined in terms of individual preference.

Estimation of WTP can be approached in various ways, falling into two basic categories:
- revealed preferences, emerging from the choices people actually make;
- stated preferences, derived from questionnaires that measure people's WTP for maintaining or improving environmental quality.

For many environmental issues it is hard to establish WTP via questionnaires because most people have no real understanding of what environmental quality means for their lives. Questionnaires with questions like 'How much would you be willing to pay for a 1 kt reduction in SO_2 emissions?' will not yield meaningful results, "1 kt SO_2 emissions" being too abstract a notion. Questions therefore need to be carefully construed so respondents can pronounce on concrete issues they can personally relate to. This means WTP is estimated mainly at the endpoint level, in terms of concrete environmental impacts on human health, ecosystem damage, damage to crops, fisheries and biodiversity and so on.

In this Environmental Prices Handbook four methods have been used to estimate the willingess-to-pay for damage avoidance (on the five endpoints):

a. Damage valuation via revealed preferences.
b. Damage valuation via stated preferences.

c. Damage valuation based on restoration costs.
d. Damage valuation based on abatement costs.

In economic valuation studies there is generally held to be a 'ladder' among these methods, with direct damage valuation the most preferred method and valuation based on abatement costs the least preferred. There may be exceptions to this general rule, though. Thus, in the case of climate change the damage - referred to as the 'social cost of carbon' - is so uncertain that the abatement-cost method may sometimes provide a better price indication.

In some cases none of the above valuation methods are truly satisfactory. A different method may then be explored: damage valuation based on modelling loss of income (i.e. Gross Domestic Product). In this Handbook this approach to valuation is explored for resource depletion, among other endpoints (see Section 5.6). Below, the four main methods are discussed and it is explained which method has been adopted for which environmental theme.

5.2.2 Valuation based on revealed preferences

With methods based on revealed preferences, observed market behaviour in an existing, complementary market is used to indirectly derive the willingness-to-pay in a non-existent market. In the Netherlands this is usually done by analysing house prices (hedonic pricing).[23] By comparing house prices at locations exposed to noise nuisance, say, with prices in quieter locations an implicit value for the damage due to noise nuisance can be derived, provided due correction is made for other impacts.

Revealed-preference studies generally use econometric methods, as in the valuation of noise nuisance in the Netherlands, for example (see e.g. (Theebe, 2004)).[24] The great advantage of this method is that it proceeds from people's actual choices (in complementary markets) in light of their budgetary constraints. A drawback, though, is that it can be hard, in econometric terms, to sufficiently isolate the influence of one explanatory variable. Particularly if this variable correlates with missing variables, the method can lead to over- or underestimates.[25] In addition, the method is sensitive to missing-variable bias. If a spoiled view and noise nuisance go hand in hand, for example, the valuation of noise nuisance may be an overestimate if the welfare loss due to the spoiled view is not properly corrected for. The results also need to be duly validated.

Another, more fundamental problem is that revealed-preference methods can lead to erroneous damage estimates if people are inadequately informed about the damage resulting from environmental pollution and other interventions. Experience shows that people are indeed insufficiently aware of certain kinds of health impacts, as in the case of noise, for which there is now growing evidence that it causes not only nuisance but also health damage. This kind of damage is not always fully included when people put a value on nuisance.[26]

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[23] An alternative is valuation based on travel times, whereby it is assessed how far people are prepared to travel to spend leisure time in scenic countryide, for example.

[24] See also Section 6.11.

[25] A negative correlation leads to underestimation, a positive correlation to overestimation.

[26] This is also due partly to the fact that the costs of health damage are not borne entirely by the home-owner.

In this Environmental Prices Handbook, noise nuisance is valued partly on the basis of stated preferences. In addition, the value assigned to air-pollution damage to buildings has also been partly determined using revealed preferences for building clean-up.

5.2.3 Valuation based on stated preferences

Willingness-to-pay can also be derived on the basis of stated preferences obtained via questionnaires, interviews or other methods. The most popular method is the Contingent Valuation Method (CVM), in which respondents are asked directly in a questionnaire what they are willing to pay for a given good, described precisely in the research scenario. Based on consumers' response to how they would react in a hypothetical situation in which supply of the good in question varies, an implicit value for that good is derived. If respondents are honest, well-informed and rational, stated-preference research is in principle the most reliable source of information on people's preferences for environmental quality (Arrow, 1993); (Hoevenagel, 1994). However, this theoretical, ideal situation does not usually hold in practice (as discussed below). Well-known problems include an absence of budgetary constraints, leading to people reporting a higher value than they would in reality be prepared to pay. In addition, the results are very sensitive to how the study is precisely designed and participants' perceptions of how the results will be used (cf. Section 5.2.7). People may also give answers felt to be socially desirable or strategically beneficial.

In the Contingent Valuation Method (CVM) respondents are, for example, asked to report their WTP for health or conservation of certain ecosystems threatened by development. Another option is to ask for respondents' willingness-to-accept (WTA) the loss of that ecosystem, although the WTA approach is considered to yield less credible results (as discussed in the following text box). One variant of the CVM method is the Discrete Choice Experiment (DCE) method, in which respondents are given a number of alternatives and asked to choose the most attractive. The WTP for certain attributes (mortality risk, for example) is then revelaed by econometric analysis.

Box 2 Difference between WTP and WTA in the CVM method

One criticism of the Contingent Valuation Method is that the value obtained depends very much on whether the WTP or WTA is asked for. According to standard economic theory the WTP and WTA should be equal, but empirical and experimental studies have shown that people on average put a more than seven times higher value on a sum to be paid than on a sum to be received (Horowitz & McConell, 2002). At the same time, this need not necessarily be a drawback of the stated-preferences method and a difference between WTP and WTA may indeed emerge from people's preferences, as postulated in Kahneman's Prospect Theory (Kahneman & Tversky, 1979). This is due in part to people attaching more value to material assets and being risk-averse. Research by Kahneman et al. (1990), for example, has shown that the price people ask (WTA) for an article they have just received is higher than the price they would be willing to pay for it (WTP). One reason for this is the 'endowment impact', as described by (Thaler, 1980), which states that people attach more value to a good they already possess than to one they might possibly acquire in the future. In SCBA this would mean there is an implicit preference for the 'status quo'.

In this Handbook health impacts are based mainly on studies using stated preferences. In Section 5.3 this is discussed in more detail.

5.2.4 Valuation based on (potential) restoration costs

A third method for valuing the impacts of environmental pollution is by estimating the (potential) restoration costs, i.e. what it would cost to undo the pollution damage. In the literature (NEEDS, 2008c) it is generally recognized that this is a less accurate measure of damage, for two reasons:

1. Valuation using restoration costs may potentially be based on overestimation, because it is not always economically optimal to restore all damage. In Chapter 2 we saw that the 'optimum' pollution level always exceeds zero. A certain amount of environmental damage is therefore socially optimal. In adopting the restoration-cost approach it is assumed the optimum pollution level is zero.
2. Valuation using restoration costs may lead to underestimation, because not all damage is amenable to 'restoration'.

The objection of overestimation can be parried by not taking the hypothetical restoration costs as the point of departure, but actual monetary outlay by, for example, home-owners. In that case, the restoration costs are used to derive a revealed-preference value. In all probability this will then lead to an underestimate, because not all home-owners will opt to repair the damage. For these reasons the restoration-cost method is less accurate than the revealed-preference and stated-preference methods.

In this Handbook valuation using restoration costs has been used for the impacts of air pollution on buildings and materials and to a certain degree also for assigning a value to ecosystem services. This is not to say that we hold the repair-cost approach to be superior to the revealed-preference and stated-preference methods, merely that there is currently too little research available on these issues for valuation using the latter two methods.

5.2.5 Valuation based on abatement costs

The final valuation method is based on abatement costs, also known as prevention costs. Much environmental policy is associated with quantitative targets (20% emissions reduction relative to 2010, say) and this method proceeds from the marginal cost of securing such targets. The abatement-cost method is based, more specifically, on the costliest abatement measure.

In the 2010 Handbook, abatement costs were recommended for environmental policy for which targets have already been set; this was in line with the former 'OEI Guidelines' used for valuing the impacts of infrastructure projects. These guidelines were superseded in 2013 by the General SCBA Guidelines, which means all midpoint environmental themes have now been valued using damage costs. The only exception is climate change, for which the Discount Rate Working Group has recommended using the abatement-cost method, based on the elaboration of climate policy in the WLO scenarios (see Text Box 2).
In addition, the General SCBA Guidelines also leave open the option of using the abatement-cost methodology if there is no other way to value damage. In this Handbook this proves to be the case for the impacts of nitrogen on marine ecotoxicity. So this too can be valued, we have here used the abatement-cost method, using the existing Dutch water-pollution charge as a proxy for the willingness-to-pay for damage avoidance (above all, excessive algal growth) resulting from discharge of nitrogen compounds. Here the charge reflects the marginal costs of achieving the policy target (reduced ecotoxicity).

If the abatement-cost method is used, it is important to take ‘efficient’ or ‘least-cost’ prices: the minimum price of securing a given policy target. If we assume a fully-informed and economically-rational acting government, policy targets will be designed such that an ‘optimum’ pollution level is attained. To achieve this pollution level, in welfare economics a ‘Pigouvian charge’ is introduced on the polluting activity that internalises the external impacts at least cost. What a Pigouvian charge embodies, in other words, is efficient application of policy to optimise economic welfare.

5.2.6 Synopsis of methods used

To summarize, in this Environmental Prices Handbook endpoints have been valued using the methods shown in Table 17.

Table 17 Methods used in this handbook to value endpoints and climate change (through literature)

Endpoint	Methods
Human health, mortality	Stated preferences, range also checked via revealed preferences
Human health, morbidity	Stated preferences, revealed preferences
Ecosystem services	Stated preferences, restoration costs
Buildings and materials	Restoration costs
Resources	Damage costs, abatement costs, modelling
Climate change	Abatement costs
Wellbeing (Nuisance)	Revealed preferences, CRF modelling

5.2.7 Limitations to valuation of environmental quality

Ascribing a value to environmental quality has several serious limitations. Although this issue has spawned thousands of publications over the last two decades, there are still major uncertainties about the reliability of the valuation methods employed. This is due primarily to the fact that values for environmental quality derived in a research setting are hard to verify against people’s actual preferences (cf. (Carson, 2000); (Bateman, et al., 2002)). A key factor here is the pronounced in-built bias of each research method. The principal limitations are as follows:

– Completeness: There appear to be no methods that can represent the full spectrum of human appreciation of environmental quality. In particular, optional and intrinsic values are poorly covered in valuation studies.
– Knowledge and information bias: Most people are poorly informed about how environmental pollution relates to human health, to name one example. In revealed-preference methods this results in pollution impacts being undervalued. In CVM studies it is well known that if people are given prior information on air-pollution impacts, they value these far higher.
– Study bias: CVM methods, in particular, yield widely ranging results, depending on how the study is designed. Carson et al. (1997) have shown that the sequence in which questions are asked has a key influence on valuation, a fact that has also been empirically proven (Payne, et al., 2000).While this is well understood by economists, it is often ignored when values are assigned in SCBA (cf. the discussion in Chapter 6). It may be added that this criticism is now generally recognized by researchers and in recent years more and more valuation studies are being designed as Discrete Choice Experiments, with the sequence of questions also being varied so due corrections can be made (cf. the discussion above).

In this Environmental Prices Handbook we make no pretence of our monetary values being either complete or infallible. We stress, rather, the major uncertainties that are inevitably attached to human valuation of

environmental goods. One way we do so is by citing all values as numerical ranges. In Annex C of this handbook we analyse the uncertainties associated with the various methods used. There is no denying, though, that the values presented here are not the outcome of an exact science.
The only way to avoid the scientific uncertainties surrounding environmental prices is to not value environmental goods at all. Although such a course may at first seem to solve the problem of scientific uncertainty, it stands in stark contradiction to the fact that each and every day consumers, industries and governments make decisions involving *implicit* weighting of financial data and impacts that cannot be expressed in financial terms. While numerical environmental prices may not really change this state of affairs, at least they mean these decisions can now be made more explicit. To our mind, this seems to be the main benefit of using environmental prices.

5.3 Valuation of human health

Human health impacts are broken down into morbidity, i.e. illness, and mortality, i.e. premature death, with a distinction made between acute and chronic mortality. Three kinds of pollution-related health impacts can consequently be distinguished:

1. Chronic mortality, expressed as a reduction in life expectancy. Epidemiological studies have shown that people in polluted areas have shorter lives than those in cleaner areas, a relationship that also holds at lower air-pollutant concentrations (OECD, 2012). The main causes of death are cardiovascular and pulmonary disease.
2. Acute mortality, expressed as an increased risk of death. Certain kinds of pollution, including smog, have also been correlated with acute heart failure. This means an increase in the risk of premature death.
3. Morbidity, expressed as an increased incidence of illness at the population level, or 'disease burden'. Environmental pollution leads to an increased incidence of asthma and pulmonary disorders. In addition, there are numerous other health problems associated with pollution, including allergies, eczema and so on. Reduced IQ development due to lead pollution, among other causes, is another element of the morbidity impact.

Following earlier attempts in transport and health care, in the 1970s the health impacts of environmental pollution were also monetarily valued.
In most of the studies published to date, health damage emerges as the single largest cost item in the overall costs of environmental pollution.

5.3.1 Midpoint-to-endpoint relationships

The following midpoints have an impact on the endpoint 'human health':

- particulate matter formation;
- photochemical oxidant formation;
- ionizing radiation;
- human toxicity;
- nuisance (noise nuisance);
- ozone depletion;
- acidification*;
- climate change*.

With the exception of acidification and climate change, all these impacts have been included in the present study. In the case of acidification, the only direct health impacts are probably very minor.[27] The indirect health impacts of acidifying emissions associated with formation of secondary aerosols and ozone have been included under particulate matter formation and photochemical oxidant formation, respectively. In this Handbook the impacts of climate change have been determined on the basis of abatement costs. This means the health impacts of climate change are not treated separately, but integrally included (as a proxy) in the valuation of climate change policy.

5.3.2 Measuring health impacts

Health impacts are usually expressed using a physical indicator expressing the number of life years (mortality) or certain quality of life (morbidity) 'lost'. The most commonly indicators used are: YOLL, DALY and QALY.[28] Table 18 provides a brief explanation of each indicator.

Table 18 Indicators for human health impacts

Indicator	Meaning	Explanation	Used for environmental impacts in:
YOLL	Years of Lost Life	Number of years of life lost due to premature mortality	NEEDS, IIASA-TSAP, CAFE-CBA
DALY	Disability-Adjusted Life Years	Number of years of life lost due to impaired health	ReCiPe
QALY	Quality-Adjusted Life Years	Number of years of perfect health	Certain individual studies (e.g. Hubbell, 2006)

With these indicators, mortality is expressed in 'number of life years lost'. Morbidity (illness) is normally also expressed in these indicators using a conversion table in which illness and disability are expressed as partial mortality, as in Hubbell (2006) for the QALY framework, for example. Generally speaking, morbidity is more usually expressed in QALYs rather than DALYs or YOLL. Studies employing YOLL, such as NEEDS (2008a), often use the QALY framework for valuing the relative disease burden.[29]

YOLL, DALY and QALY essentially each measure a different aspect of health impacts. All the main European studies on the social costs of air pollution have adopted YOLL for premature mortality, with morbidity valued separately using the QALY framework. The reasoning is that the YOLL framework is more congruent with the actual action of environmental pollution, which tends to shorten life span, particularly through respiratory and cardiovascular disease towards the end of a person's life. YOLL then most accurately reflects mortality impacts. DALY and particularly QALY are used more in the realm of health care. Annex B in the Dutch language version provides detailed information on each of these indicators and how they relate to each other.

[27] Apart from NO_2, but the impact of this pollutant has been added to the chronic impacts of photochemical smog formation; see Section 6.5.

[28] YOLL is sometimes also expressed in LYL (Life Years Lost).

[29] Here the assumption is made that 1 additional YOLL equals the loss of 1 QALY. For more information see Annex B in the Dutch language version.

5.3.3 Valuation of health impacts

All three indicators in Table 18 are quantified in ‘years’. For use in SCBA, in the CSR context or for final weighting in LCAs they therefore need to be assigned a monetary value. The valuation methods most often used for this purpose are the VSL (Value of a Statistical Life) and VOLY (Value Of a Life Year) frameworks. The former is often used in the context of transport policy, but also in health-care and environmental settings. OECD (2012) has carried out a meta-analysis of valuation using VSL. The results show that the median value of VSL for valuing the health impacts of pollution is around € 2.5 million. In NEEDS (2008c) it is rightly stated that, in the air-pollution context at any rate, mortality valuation via VOLY is better than via VSL, for the following reasons:

1. Air pollution can rarely be identified as the primary cause of an individual death, only as a contributing factor.
2. VSL makes no allowance for the fact that the loss of life expectancy through death is far less for mortality associated with air pollution (around six months) than for typical accidents (30-40 years), the figure on which the VSL calculations are based. In other words, the main mortality impact of air pollution occurs later in life, while accidents are more likely to occur at an earlier stage.

For this reason, in the NEEDS project VOLY is used for valuing the mortality impacts of air pollution. This Value Of a Life Year is the value assigned to a life year on the basis of estimated life expectancy. It can be calculated using stated or revealed preferences.

In the NEEDS project, VOLY was valued using the Contingent Valuation Method by asking people for their willingness-to-pay for a three or six month longer life span as a result of improved air quality. One innovative feature of NEEDS was that people were asked explicitly how they value small changes in life expectancy. As a result, a lower value for VOLY was found than in other projects in which people were asked (in Discrete Choice Experiments) about their risk of dying prematurely. As argued in NEEDS (2007a), an analysis based on changes in life expectancy yields a better estimate than one based on changes in mortality risk, because in epidemiological studies the impacts of air pollution manifest themselves as reduced life expectancy.[30]

In NEEDS, VOLY was based more specifically on a 2006 WTP research study in which people were asked, in face-to-face interviews and payment-card experiments, how they valued a few extra months at the end of their life.[31] Based on the empirical results, augmented by literature reviews, the NEEDS team arrived at an average VOLY for the EU25 (plus Switzerland) of € 40,000. This figure is for chronic mortality, i.e. shortening of life expectancy. For the risk of acute mortality the team deemed the results of earlier studies on mortality risk valid, and for acute mortality a VOLY of € 60,000 was thus adopted. In addition, on the basis of an earlier WTP study a QALY-based valuation was used for various kinds of morbidity such as respiratory problems, cancer, lost working hours due to illness and costs of hospital visits.

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30 This is in line with the evidence of health risks due to PM_{10}, which shorten life span and thus also life expectancy. These impacts have been primarily proven in epidemiological rather than toxicological studies.

31 This method thus combines asking people for their valuation with a simple experiment. By using payment cards, this method is considered more reliable than simply requesting willingness-to-pay, because the physical action of payment makes people more aware of the fact they must pay.

There have been numerous studies showing a strong correlation between the value people give to mortality risk or reduced life expectancy and their financial income. In NEEDS (2006) it was opted to primarily adopt a European perspective and calculate pan-European averages, but differentiating between new member states in Central and Eastern Europe and 'old' states. The central value for the new member states was calculated as € 33,000, with the value for the EU15 plus Switzerland slightly higher: € 41,000 (NEEDS, 2008c).

5.3.4 Methodology in the 2010 Handbook

In the 2010 Shadow Prices Handbook damage to human health was valued using the values reported in NEEDS. In line with the above, an EU25-average VOLY of € 40,000 per annum was therefore taken for chronic mortality. For acute mortality due to smog-related heart failure a higher VOLY was adopted in NEEDS, viz. € 60,000 per annum.[32] The value for morbidity (e.g. hospital costs) was also taken from NEEDS.

In the 2010 Manual these values were thus adopted, but adjusted to 2008 prices and real income, and assuming, in accordance with NEEDS (2008c), that the NEEDS valuation was based on income (and price) levels for the year 2000 (for further discussion, see below). In line with NEEDS, we there applied a positive income elasticity of 0.85, yielding a VOLY of € 55,021.[33] This value was used for all the environmental themes, with the exception of ozone layer depletion, where a direct valuation was based on the ReCiPe characterization, which reports human-health impacts for this theme in DALYs. Based on a VOLY of € 40,000 we calculated a DALY in which, after extensive deliberations, we opted conservatively for taking 1 VOLY as equal to 1 DALY (though there are indications (as discussed in Annex B of the original Dutch version of this Handbook and in the Shadow Prices Handbook (CE Delft, 2010) that a DALY should be assigned a higher value than a VOLY.

5.3.5 New research for this Handbook

For the present Handbook, additional research on valuation of human health was carried out with respect to four issues:
- the implications for VOLY valuation of the recommendations of the Discount Rate Working Group;
- the implications for VOLY valuation of the QALY-value adopted in the new SCBA Guidelines for the Social Domain (SEO, 2016a);
- new insights and interpretations based on NEEDS (2008c);
- new literature and studies on VOLY valuation and deliberations on whether in the light of this literature and more recent insights the VOLY used in the 2010 Handbook is to be deemed too low or too high.

Below, we look more closely at these four aspects.

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32 The higher value for acute mortality can be justified because in NEEDS people were asked for the influence on their life expectancy, which generally yielded lower values. People are prepared to pay more when they are asked about the risk of mortality. Acute heart failure can obviously be seen as a mortality risk.

33 Furthermore, in NEEDS another assumption was made: that premature mortality of babies implies a two times higher VSL than for adults, based on several studies in the literature. In NEEDS adults have an implicit VSL of € 1.5 million, children € 3 million. According to Rabl et al. (2014) this is also justifiable, parents being acutely sensitive to their childen's health. It should be added, though, that in the totality of impacts this effect is very small, with Rabl et al. (2014, p. 502) stating that for PM_{10} child mortality accounts for no more than 3.4% of total damage, while for $PM_{2.5}$ the damage is negigible. A different assumption on this point would therefore not make any great difference in environmental prices.

5.3.6 Implications of the Discount Rate Working Group

In line with the recommendations of the Discount Rate Working Group, health benefits may no longer be boosted using a positive income elasticity, because the higher willingness-to-pay for health is cancelled out (over time) by increased 'supply' of health.

We interpret this to mean that technological advances make it ever less expensive to stay healthy, with the overall health of the population improving as a result. According to the standard economic theory of declining marginal utility value, it can then be stated that the value of an additional unit of health is continually falling as a result of declining environmental pollution.[34] In economic terms, this statement is accurate. The relationship between baseline health level and the value to be assigned to health is also evidenced in empirical studies. Istamto et al. (2014), for example, found that the WTP for a reduction in air pollution is negatively correlated with the baseline health level measured by the RAND-36 questionnaire: the healthier one is, the lower the value accorded to measures to curb air pollution.[35]

In this Handbook we have adopted the recommendations of the Discount Rate Working Group, applying no positive income elasticity to VOLY valuation. This means the NEEDS valuation from 2005 has only been increased to correct for inflation and no longer by a factor for income growth. In doing so we thus assume that people in 2015 are prepared to spend a lower fraction of their income on pollution prevention than in 2005 because their health situation has improved.

5.3.7 New interpretation of NEEDS

In the 2010 Shadow Prices Handbook it was stated, in line with NEEDS (2008a), that the prices and incomes adopted in NEEDS were based on the situation in the year 2000. In that Manual the correctness of this assumption was not further examined. In work on the present study, however, it was concluded that this is perhaps erroneous. The VOLY valuation study was based on questionnaires conducted in 2005 and 2006, and respondents therefore answered their questions using 2005 prices and incomes, implying that mortality impacts were likewise expressed in 2005 prices. In the final NEEDS report (NEEDS, 2008a) all impacts are cited in 2000 prices, though. The reason for this course of action is not clearly explained.

The year adopted for expressing prices is important, because precisely between the year 2000 and 2005 there was rapid monetary deflation, i.e. inflation, due to introduction of the Euro. This makes it plausible that respondents in the WTP study conducted after introduction of the Euro worked mentally with different prices than those in WTP studies prior to the Euro.

The value assigned to lost work-days due to sick leave can also be queried. The figure of € 295/day (in 2000 prices) adopted in NEEDS (2008a) is far higher than had previously been used. If this effect is combined with the total number of hours worked (and the result inflation-adjusted), one arrives at a figure in excess of Dutch GDP. Why this higher value was adopted is again not entirely clear. In our opinion it is correcter (and simpler) to base valuation on

[34] Here we note that the Working Group does not adopt the argument of declining marginal utility, merely that the costs of health maintenance become cheaper over time.

[35] Other things remaining equal, one would therefore also expect a lower VOLY as the air becomes cleaner. In NEEDS (2008c) this was not taken into account, as a single, constant VOLY-value was used in all the various scenarios, regardless of the baseline pollution level.

the reward for labour as a production factor (salaries and social security payments) as cited in the National Accounts. This yields a value of € 175/day in 2015 prices.

5.3.8 Implications of QALY-value in SCBA Guidelines

The SCBA Guidelines for the Social Domain (SEO, 2016a) adopt a value of € 50,000-100,000 for a QALY. This QALY-value, from the realm of curative health care, cannot simply be transferred to VOLY.[36] In the first place, QALY takes health benefits as its metric, while VOLY considers impacts on life expectancy. VOLY is thus more in line with *preventive* than *curative* health care. In the appendices of SEO (2016a) it is discussed whether a QALY for preventive health care should be lower than for curative health care. Although there are studies positing that this is indeed the case, SEO (2016b) argued that there are no theoretical grounds for such a move. They therefore recommend that this not be valued separately. The implication is then also that there is no reason to value environmental pollution differently from health-care interventions.

Secondly, valuation using VOLYs is concerned primarily with involuntary risks, valuation using QALYs with voluntary interventions. Willingess-to-pay for avoiding involuntary risks is generally higher, implying higher valuation with VOLYs than with QALYs.

In Annex B of the Dutch language version, we look at a discussion that goes into whether a VOLY-to-QALY conversion factor might be found. Our best guess for such a factor is based on a ratio of 1.087 between DALY and QALY (so that 1 DALY = 1,087 QALY) and a ratio of 1 between DALY and VOLY. The higher value of a VOLY compared with a QALY is due primarily to the value assigned to avoiding premature death being higher than that for avoiding sickness, so that a QALY of zero (no utility value for state of health) per annum does not equal 1 YOLL (no longer alive due to death). Based on age weights and relative disease burden, conversion is then feasible. In Annex B of the Dutch version it is argued that a conversion factor of 1.087 is the best possible estimate at present.

This means the QALY values prescribed in the SCBA Guidelines for the Social Domain result in a value of € 54,350-108,700 per annum for a VOLY. This range is precisely in the middle of the range reported by Desaigues et al. (2011) for the stated preference method used in NEEDS, where, converted to 2015 prices, lower and upper values of € 33,500 and 134,000 are adopted.[37]

5.3.9 New literature and debate on the VOLY-value

Finally, we investigated whether new literature provided any grounds for adopting a higher or lower VOLY than in the 2010 Shadow Prices Handbook.

Since 2011 there have been several studies on the costs and benefits of clean-air policy In the EU (see for example Holland, 2014; IIASA, 2014) and these studies have worked with a far higher VOLY: € 58,000 median and € 135,000 average.
It should be noted, though, that these values are based on the inflation-corected values calculated in the NewExt (2004) study, which are in turn based

[36] The QALY is obviously inverse to the VOLY; see later in this chapter.

[37] In this calculation, income elasticity has been taken as zero. In 2000 prices the lower and upper values are € 25,000 and 100,000.

on mortality risk. As argued above, the value to be assigned to air pollution is better represented by quantifying the impact on life expectancy. For this reason, we do not necessaily see this as a better valuation in scientific terms. What we do observe is that the central value of NEEDS (2007a) is no longer used in a number of important European cost-benefit analyses and has been rejected in the European Commission's 2009 Impact Assessment Guidelines (EC, 2009a); (EC, 2009b), which recommend standard usage of a range of € 50,000-100,000 for a VOLY if pollution-related health damage is being quantified in such assessments.

Chanel & Luchini (2014) posit that the VOLY values adopted in NEEDS lead to underestimation of the true value of prolonged life expectancy. In their WTP study on the benefits of emissions reduction in France they write that the WTP for air-pollution prevention leads to an underestimate if only impacts on one's own life are taken on board. Many people want cleaner air not only for themselves but above all also for the ones they love. When they include this fact, they arrive at a far higher VOLY of € 140,000, in France, for avoiding premature death due to the impacts of air pollution. This is similar to the criticism of Mouters & Chorus (2016) that stated-preference studies yield an underestimate if only the impacts on one's own life expectancy are included.

In addition, Bijlenga et al. (2011) bring forward that WTP studies using questionnaires, as with NEEDS, generally arrive at a lower value for a VOLY than Discrete Choice Experiments in which stated preferences are established for multiple aspects at the same time. However, they also state that there are no theoretical grounds for arguing which of the methods is better. (Istamto et al., 2014), on the other hand, arrive at a 3-5 times lower value for air-pollution health impacts than NEEDS (Desaigues, et al., 2011), reporting that this is due to their using a web-based survey compared with the face-to-face interviews plus payment-card experiments used in NEEDS, which they state are known to yield higher values. In our opinion the survey method used by Istamto et al. (2014) is indeed less comprehensive than the NEEDS study and cannot therefore simply be adopted without further ado as a basis for revising VOLY values.

Besides new empirical studies, other research has also been published, in particular several meta-analyses and comparisons of results from the environmental and other domains (such as transport) associated with health impacts. OECD (2012) is a meta-analysis of the values assigned to human health based on the VSL (Value of a Statisical Life) metric. This study concludes that the median VSL used in the environmental domain is approximately € 2.4 million.[38] Based on an average VSL-to-VOLY ratio of between 20 and 40 for pollution (as argued in Annex B of the Dutch language version), this means a VOLY should be valued at between € 60,000 and 120,000. OECD (2012) analyses the differences in values obtained using the VOLY and VSL metrics and reports that use of VSL in combination with QALY generally leads to pollution being valued higher than when VOLY is used, with this due to the fact that VOLY underestimates the price of morbidity (illness). Only if a high value of € 130,000 is adopted for a VOLY is the value assigned to morbidity in line with studies using VSL, according to OECD (2012). Based on the OECD study, the French government has recommended adopting a figure of € 115,000 (in 2010 prices) for a VOLY in cost-benefit analyses (Quinet, 2013).

[38] According to OECD (2012), the VSL used in the environmental context is half that used for victims of traffic accidents. One reason for this, they state, is the fact that in WTP studies people assign a lower value to 'public good'-type issues such as the environment.

Finally, research also shows that, apart from income, education level also influences people's valuation of health. The more educated they are, the higher people value a VOLY (see e.g. OECD (2012) for a general discussion and Istamto et al. (2014) for the Netherlands). Because the average education level in the Netherlands was higher in 2016 than in 2005, one would anticipate a higher value being assigned to a VOLY.

5.3.10 Choices in this Handbook

As the basic point of departure in this Handbook we have here taken the VOLY-value given in the NEEDS project for the EU15: € 41,000, in 2005 prices. Converting this to 2015 prices gives a figure of € 48,000, slightly less than the lower value of the QALY, expressed in VOLY (see above), from the SCBA Guidelines for the Social Domain (SEO, 2016a), which would give a VOLY of € 55,000 (argumentation for which is provided in Annex B of the Dutch language version). The EU study on the costs and benefits of clean-air policy works with a VOLY of € 58,000 (Holland, 2014) ; (IIASA, 2014). It would seem plausible, therefore, that the true lower value of a VOLY is somewhere around € 50,000 (in 2015 prices). There is indeed evidence that this is only a lower bound.
The values reported in the latest studies are generally higher. Assuming the same range adopted in SEO (2016b), we arrive at an upper VOLY-value of € 110,000, which is similar to the value of € 115,000 recommended by the French government. This value is slightly below the upper bound adopted in the EU studies (Holland, 2014); (IIASA, 2014). It is therefore well feasible that there is a ceiling for the value of a VOLY that lies somewhere between € 110,000 and 120,000.

Based on these consideraions, we have opted to take € 50,000 as the lower bound of a VOLY and € 110,000 as the upper bound.

The VOLY is the most important metric for valuing the health impacts of environmental pollution because pollution has a greater impact on mortality than on morbidity. For morbidity calculations we proceeded from a QALY as formulated in the social domain with a lower value of € 50,000 and an upper value of € 100,000. At the lower bound a VOLY thus equals a QALY. For the upper bound, though, we distinguish between € 100,000 for a QALY and € 110,000 for a VOLY.

These upper and lower values for human health are recommended for use in SCBAs. Industries and environmental scientists generally make less use of ranges, preferring a central value instead. Because the VOLY in all probability does not have a normal distribution, we have opted to take a central value of € 70,000 for both a VOLY and a QALY.

In this Handbook we have also chosen to no longer adopt a separate value for acute mortality. This is because acute mortality due to pollution generally affects senior citizens. There are indications that an extra life-year at the end of one's life is valued less than an extra year of life expectancy earlier on. This is why people approaching the end of their life indeed put a lower value on an additional life-year than the average population. It is not unusual to take the step of valuing acute mortality due to elevated ground-level ozone no differently from chronic mortality; in the Ecosense-model, too, there is assumed to be no difference between chronic and acute mortality (NEEDS, 2008b). In this Handbook, this is the approach that has been adopted.

For infant mortality we followed the approach of NEEDS (2008a), using a VSL twice as high as that for adults. This results in a VSL of € 3 million (in 2005 prices) for the lower bound of health impacts. For loss of working hours we based ourselves on the National Accounts, dividing the sum total of rewards to labour as a production factor (salaries and social security payments) by the aggregate number of hours worked. In 2015 prices the reward for the production factor labour works out at € 175 per day (incl. VAT).

5.4 Valuation of ecosystem services and biodiversity

Ecosystems, i.e. assemblages of organisms in a particular environment, contribute in a multitude of ways to human prosperity. Known as 'ecosystem services', this contribution consists of all the various products and services supplied by the natural world and benefiting our lives. Emissions and land use (changes) can affect the functioning of ecosystems and thus the availability of the services they provide.

Besides ecosystem services, biodiversity, i.e. the diversity of plant and animal species, is also important in its own right. On the one hand, human society considers it of value to pass on this 'rich tapestry' to future generations. On the other, biodiversity is of critical importance for the quality and very survival of nature, because it supports fundamental processes like soil formation and the hydrological cycle, which in turn supply humans with all manner of (ecosystem) services.

This section explains how damage costs due to environmental pollution have been valued for the theme of ecosystem services.

5.4.1 Categorization of ecosystem services

Ecosystem services are defined and categorized acording to the various services and benefit they provide to humanity. CICES (EEA, 2011)[39] distinguishes three classes of service:
- provisioning services (e.g. food from agricultural crops, biomass as fuel, fisheries, forestries, freshwater);
- cultural services (e.g. recreation, aesthetic value of the environment, spiritual values);
- regulation and maintenance services (e.g. climate regulation, soil formation, biological pest control, water purification).

In the Netherlands and the EU there has been copious research on categorizing and quantifying ecosystem services, but relatively little on how these services are affected by emissions, with the exception of carbon emissions, which in this Handbook are treated via abatement costs, however (cf. Section 6.3).[40] For cultural and regulation services there are virtually no useful studies on how these are impacted.

For the provisioning services of ecosystems, in contrast, there is a certain amount of research available, particularly for agricultural crops. NEEDS (2007a), for example, quantifies the impacts of sulphur dioxide and ozone levels on crop yields. The relationships between calcium and acidification and between nitrogen emissions and nutrient requirements have

[39] The Common International Classification of Ecosystem Services (CICES) system, used by the Netherlands Environmental Assessment Agency (PBL) as well as the EU.

[40] See for example (Wheeler & Braun, 2013). In this Handbook, climate impacts are treated using abatement costs, with valuation based on the marginal costs of achieving policy targets. This means it is no longer necessary to determine and value the impacts of carbon emissions at endpoint level.

also been studied. By multiplying changes in crop yields by market prices for the crop in question, damage costs can be quantified (see also Annex B). In addition, for certain environmental themes like ozone depletion a relationship has also been established between the emissions causing the environmental problem and the ensuing damage to agricultural crops and forestry (see e.g. (Hayashi, et al., 2006)). However, we know of no research that has systematically quantified the impact of emissions on *all* provisioning services. In addition, impacts on fisheries, for example, are often quantified via the concept of biodiversity (see below).

5.4.2 Biodiversity and its relationship with ecosystem services

Biodiversity can be defiried as the variety, number and quality of species, populations and ecosystems, which, apart from their functional significance, also engender ethical and moral considerations. Biodiversity loss leads to loss of ecosystem functions (intrinsic and extrinsic) and loss of ecosystem resilence. People attach value, furthermore, to maintaining the world's rich diversity of natural species and conserving them for future generations.

There is therefore debate as to whether biodiversity should be considered an independent ecosystem service or an indirect contributor to the creation of other ecosystem services. The latter stance appears to be gaining ground (Kuik, et al., 2007). Science for Environment Policy (2015) concludes on the basis of the available literature that, even after 20 years of research, the exact relationship between biodiversity and ecosystem services is still not entirely clear.

Nonetheless, several broad conclusions can be drawn:
– Although biodiversity clearly plays a fundamental role in ecosystem functioning, its exact relationship to ecosystem services cannot be adequately quantified.
– The relationship between biodiversity and the various ecosystem functions is non-linear. Generally speaking, regulation functions benefit from greater biodiversity. Provisioning functions like agriculture and forestry have, on average, the highest yields at relatively low biodiversity, though. In the case of cultural functions, the relationship differs according to the function. In general terms, cultural functions benefit from greater biodiversity, though this does not hold for recreational functions at very high biodiversity levels.
– Regulation and maintenance functions are important in the sense that biodiversity is a precondition for maintaining ecosystem servives. In the longer term, high biodiversity is a precondition for maintaining provisioning functions, for example.
– In all of this there is synergy as well as trade-off among ecosystem services, particularly between provisioning services like crop production and regulation and maintenance services.

Despite the relative paucity of studies on the impacts of emissions on biodiversity, NEEDS (NEEDS, 2008c) and ReCiPe (Goedkoop, et al., 2013) made an attempt at quantification. In ReCiPe it was assumed that species diversity is an adequate proxy for ecosystem functioning and the relationship between emissions and species extinction was quantified. NEEDS, for its part, stated that biodiversity loss leads to loss of ecosystem functions and a deterioration of ecosystem resilience. This is in line with Science for Environment Policy (2015).

There is a certain justification in taking biodiversity as a proxy for the intrinsic and extrinsic value of ecosystems (i.e. nature), given the pivotal role of biodiversity in the quality of ecosystem services. At the same time, though, there may be a negative correlation between biodiversity and agricultural yields, with this arguing for subtracting a figure for crop losses from the value adopted for biodiversity loss. This is the solution adopted in this Handbook, with the welfare impacts of damage to ecosystem services being quantified as biodiversity losses minus crop losses (including forestry and livestock fodder crops, but excluding livestock farming itself and fisheries).

5.4.3 Midpoint-to-endpoint relationships

The following midpoints have an impact on the endpoint 'ecosystems':
- eutrophication;
- acidification;
- smog formation;
- ecotoxicity;
- ozone depletion;
- land use;
- Ionizing radiation*;
- climate change*.

In this Handbook all these impacts have been monetized except for ionizing radiation and climate change. For ionizing radiation no good method could be found for quantifying the impacts of radionuclides on species diversity. As mentioned above, in this Handbook climate change has been approached via the abatement-cost method, so that no additional distinction can be made between health and ecosystem impacts (cf. Section 6.3. In the case of acidification, ozone depletion and smog formation, impacts on both crop yields and biodiversity have been taken on board in calculating environmental prices. For the other themes, only the impacts on biodiversity have been included here, under the implicit assumption that impacts on crop yields cannot be considered external impacts.[41]

5.4.4 Methodology in the 2010 Handbook

In the 2010 Shadow Prices Handbook the endpoint *damage to ecosystems* was valued only in terms of impacts on biodiversity, with impacts on crop yields quantified on the endpoint 'damage to buildings and materials' and agricultural crops thus considered as 'materials'. There, ecotoxicity impacts were not monetized at all.

In the 2010 Handbook the impacts of emissions on biodiversity were based on NEEDS (2007) for the themes acidification and smog formation, on Hayashi (2006) for ozone depletion and on ReCiPe (Goedkoop, et al., 2009) for the other environmental themes. In line with ReCiPe, biodiversity loss was expressed using a specific indicator: PDF/m^2/y, where PDF stands for Potentially Disappeared Fraction (of species). This indicator expresses annual species loss in a given area and was used by Goedkoop and Spriensma (PRé, 2000) as one of the first as a metric for biodiversity loss. In ReCiPe (Goedkoop, et al., 2013) a certain reference number of species was established for the various types of land use. If there is land-use change from a type with lower species diversity, biodiversity declines, allowing a 'delta-PDF' to be calculated.
This delta-PDF approach was also applied in NEEDS (Ott, et al., 2005) for determining the ecosystem impacts of acidification and eutrophication.

[41] Land use changes may affect crop market prices, for example. We here assume, however, that this is a induced economic effect that is not incorporated in land prices. Land use changes consequently have no external effect.

Complementing this approach, the 2010 Handbook followed (Hayashi, et al., 2006) for direct valuation of crop damage due to ozone-layer depletion.

Box 3 PDF as a measure of biodiversity

PDF is an indicator of ecosystem damage that expresses the risk of species extinction as a result of emissions, land-use changes and other deleterious factors. The current assemblage of plant and animal species under a certain land-use regime (Si) is compared with a reference regime (Sref) to give the relative species richness, the inverse of which is PDF:
PDF = 1-Si/Sref
For emissions, PDF : 1 – POO (Probability of Occurance). PDF, the Potentially Damaged Fraction, is the fraction of species that is most probably absent owing to unfavouraable environmental conditions due to acidification, eutrophication and other such factors. A PDF of 0.2 PDF.m^2.jaar, for example, means a loss of 20% of the species on 1 m^2 of land for 1 year.

In NEEDS (2007a) and the Shadow Prices Handbook, *valuation* of biodiversity impacts was based on Kuik et al. (2008), who carried out a meta-study on the willingness-to-pay for biodiversity found in WTP studies. The meta-study took in international studies that valued various aspects of biodiversity (forest conservation, preservation of ecosystem values, tourism). The value is a proxy for welfare. Kuik et al. (2008) arrive at a value of €$_{2004}$ 0.47 per PDF/m^2/jaar. This is an average value for average damage in Europe. In the 2010 Handbook this average was adopted, with no attempt to adjust it specifically to the Netherlands.

In the 2010 Handbook, damage to agricultural crops was valued in combination with the endpoint ‘damage to buildings and materials’. The valuation of crop damage was based on NEEDS. The impacts of SO_2 and ozone were modelled using concentration-response functions. Changes in crop yields due to elevated SO_2 concentrations were calculated for wheat, barley, potatoes, sugarbeet and oats. For ground-level ozone the relative change in yields of rice, tobacco, sugarbeet, potatoes, sunflowers and wheat was calculated. Monetary valuation of crops was based on price per tonne, quantified as an unweighted average of the prices of the above crops.

5.4.5 New developments: valuation and impact quantification

A number of initiatives are underway to value both biodiversity and ecosystem services, such as TEEB and the ‘Natural Capital’ programme set up by the Dutch Ministry of Economic Affairs. These initiatives aim to quantify the value of biodiversity and ecosystem services to society, so they can be properly accounted for in policy decisions and projects. For the forthcoming SCBA Guidelines for Nature (cf. Section 5.4.8) it is being examined to what extent these initiatives are succeeding in providing a workable handle for quantifying the welfare losses resulting from interventions impacting biodiversity.

TEEB

TEEB (The Economics of Ecosystems and Biodiversity) is a global initiative to put a robust figure on the value of nature. Under this umbrella a variety of studies have been published in recent years that value ecosystem services like timber harvesting, fisheries, recreation and so on. Additional research has also been carried out on valuation of nature as ‘natural capital’ in the Dutch government programme ‘Natural Capital Netherlands’ (PBL, 2015). Under this programme the Ministry of Economic Affairs has commissioned research on the economic and social value of nature in the Netherlands.[42] These studies have, to our knowledge, yielded no basis for establishing a relationship between emissions and a physical indicator of ecosystems.

European initiatives

In the framework of the European Biodiversity Strategy a considerable amount of work has been done on developing biodiversity indicators and inventorying and categorizing ecosystem services. Studies include MAES (Mapping and Assessment or Ecosystem Services) and SEBI (Streamlining European Biodiversity Indicators). These types of initiatives are yielding a huge amount of data that may be relevant in the future for updating current indicators (species numbers). These projects are concerned more with assessing the current status of biodiversity in the EU, however, and once again provide no basis for linking emissions to biodiversity and ecosystem services. The results are therefore of no direct use for developing environmental prices.

5.4.6 New insights: characterization

Pollutant characterizion reflects the relationship of one pollutant relative to another in terms of ecosystem impacts. Besides ReCiPe, the International Reference Life Cycle Data System (ILCD) Handbook has been developed by the Institute for Environment and Sustainability at the EU Joint Research Centre (JRC). ILCD is an analysis of best practices dating from 2009 and has been used to elaborate the Product Environmental Footprint (PEF) and Organization Environmental Footprint (OEF). The ILCD method measures changes in land use in terms of kg C-deficit, the degree to which the soil contains and retains carbon. The method makes no allowance for species diversity, nor does it link impacts to biodiversity, so does not enable midpoint-to-endpoint translation. ILCD is therefore less suitable for assigning a value to land use. In other repects the method is in line with ReCiPe.

For ozone layer depletion and freshwater and marine eutrophication, the PROSUITE project[43] recommends the ReCiPe approach (Ecofys, 2014).
This project itself uses PDFs for valuing the endpoint ‘natural environment’ using the ReCiPe characterization factors.

One limitation of using characterization factors as a basis for valuation, as in the PROSUITE project, is that these factors represent typical, European-average relationships for the relative damage of pollutants. For the Dutch situation, with relatively serious problems in the realm of eutrophication, these data may not always be representative.

5.4.7 New insights: PDF valuation

In the 2010 Shadow Prices Handbook the value adopted for biodiversity was the average value of an EDP[44] per m^2 per annum of $€_{2004}$ 0.4706, based on Kuik et al. (2008). This value is the average value from a meta-analysis encompassing a number of European countries. The median value in this study

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42 Of the seven studies four have now been completed. One example is a report on the Dutch overseas territory Bonaire in which all ecosystem functions have been valued to yield a ‘Total Economic Value’ using various methods, including surveys, WTP and avoided damage costs (IVM, 2013).

43 PROspective SUstainability assessment of TEchnologies, a large-scale EU FP 7 project (2009-2013) aimed at developing methods to determine the lifecycle social, economic and environmental impacts of technologies.

44 Ecosystem Damage Potential, which is a slightly different measure, but (Kuik, et al., 2008) state that for all practical applications EDP and PDF can be considered identical.

is $€_{2004}$ 0.0604, a factor 8 lower. This implies that the overall distribution of values comprises relatively many high values (Kuik, et al., 2008). In a study on the external costs of energy production, Ecofys (2014) takes Kuik's median value rather than the average. Generally speaking, in meta-analyses more value is attached to the median than to the average. On the other hand, an earlier study by NEEDS (2006) arrived at a PDF-value of € 0.45/€ 0.49 per PDF/m^2, the same as the average value in Kuik et al. (2008).[45] NEEDS (2006) use the restoration-cost approach.The Éclaire project (Holland, 2014); (IIASA, 2014) investigated the economic value of air pollution impacts on ecosystem services, with biodiversity valued using WTP (as with Kuik), restoration costs (as with Ott) and revealed preferences (costs of legislation). This project indicates that WTP-based values are conceptually the most robust, but that data availability may be a problem. In that case, use can be made of restoration costs. Restoration costs can also be used to validate WTP-values. Holland (2014); IIASA (2014) report that restoration costs represent a minimum value for biodiversity, because even after recovery genetic information may still be lost, for example. Rabl (1999) raises the value of NEEDS (2006) by a factor 2 to capture the true damage. Brink and Grinsven (2011) work with a range, multiplying the value of NEEDS (2006) by an (arbitrary) factor 5 to obtain an upper bound and taking the value of NEEDS (2006) as a lower bound. This approach was also adopted by Grinsven et al. (2013). Holland (2014); IIASA (2014), too, state that restoration costs represent the best possible estimate.

One route that has not yet been explored is to adapt the meta-analysis of Kuik et al. (2008) to the specific Dutch situation. In all likelihood, Dutch nature areas have a higher value because nature is relatively scarcer here.[46] One thing to emerge from the meta-analysis of Kuik et al. (2008), for example, is that the value assigned to nature increases with population density and declines as the surface taken up by nature areas grows. Numerous smaller nature areas in a densely populated country therefore yield the highest value for nature. Using the regression analysis in Kuik et al. (2008) it would be possible to derive values specifically for the Netherlands.

5.4.8 Choices in this Handbook

Dose-effect relationships

In this Handbook, relationships between emissions and impacts on PDF have been calculated in the same way as in the 2010 Shadow Prices Handbook. For NO_x, SO_2 and NH_3 these were determined on the basis of NEEDS (2008a). For ozone depletion we based ourselves on Hayashi et al. (2006) and for other midpoints (eutrophication, ecotoxicity) on ReCiPe. More detail on midpoint valuation is provided in Chapter 6.

Valuation

We have now opted to adjust the methodology from the 2010 Shadow Prices Handbook to bring it in line with the results of Kuik et al. (2008), using the regression analysis reported in that study to obtain specific values for the Netherlands. This yields an estimated average value of € 0.93/PDF/m^2 for

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45 In the Shadow Prices Handbook it was reported that the average value of NEEDS (2006) was € 0.45, which is an EU-average. In NEEDS (2006) it is stated that minimum restoration costs in Germany are € 0.49/PDF/m^2. The figure of € 0.45 is a conversion from the German price level (using purchasing power parities) to an average European price level.

46 On the other hand, emissions have relatively less impact on nature in the Netherlands because there is relatively less of it, and emissions consequently more often impinge on urban areas, where there is no nature to damage.

Dutch nature in 2004 prices (the full calculations are provided in an annex in the original Dutch version). This is a factor 2 higher than the European-average estimate reported in Kuik et al. (2008).

To err on the safe side, we nonetheless propose that this be used as an upper value. The central value can then be based on restoration costs, which, following Ott et al. (2008), we take to be € 0.48/PDF/m^2 for the Netherlands (2004 prices).

To calculate a lower value we make use of the fact that the median value is a factor 8 less than the average value obtained in the random sampling of Kuik et al. (2008). By decreasing the value for the Netherlands by a factor 8 we obtain a lower (rounded) value for our country of € 0.12 in 2004 prices.

In addition, we have adjusted our values as follows:
– Translation to 2015 prices.
– Annual inflation has been taken as 1%, in line with the recommendations of the Discount Rate Working Group for 'irreplaceable' nature. PBL is currently investigating which nature counts as such and to what extent this should then be incorporated in SCBAs using a lower inflation figure. Because the results of this study are not yet available, it was decided to apply the 1%-per-annum figure to all Dutch nature, given that prices are based on valuation in 2004.[47]

As in the 2010 Shadow Prices Handbook, no positive income elasticity has been assumed for biodiversity. If deemed necessary, this assumption can be discussed under the umbrella of the SCBA Guidelines for Nature that are currently being drawn up.

From the above, the values reported in Table 19 emerge.

Table 19 Valuation of PDF.m^2.yr (€)

	$€_{2004}$	$€_{2015*}$
Upper value	€ 0.934	€ 1.240
Central value	€ 0.480	€ 0.635
Lower value	€ 0.119	€ 0.158

* 2015 prices based on 1% annual inflation in real terms.

The upper and lower values provide upper and lower bounds for valuing the impacts of emissions on biodiversity and can be used in SCBAs. The central value is the recommended value for use by industry and has also been used for arriving at a characterization factor.

Addition of crop damage

Damage to agricultural crops has been added to the valuation of ecosystems. For the valuation itself the same method was employed as in the 2010 Handbook, adjusting prices to present-day levels in the markets concerned.

New prices have been based on average European producer prices for the EU28 as reported by FAO (see Table 20). Prices in USD_{2014} were converted to EUR_{2014} using the average 2014 exchange rate and then converted to EUR_{2015} using the

[47] As acidifying and eutrophying emissions often have a long-term impact on soil quality, we assume that emissions in principle have an impact on irreversible nature (functions).

general Harmonized Index of Consumer Prices (HICP). These prices were then weighted by consumption of the crop concerned to determine the average price rise between 2000 and 2015. Finally, 18% VAT was added.

Table 20 **Average EU producer crop prices (€/t crop yield, excl. VAT)**

	2000 Prices	Source	2015 Prices	Source
Sunflower	273	FAOSTAT € (2001)	335	FAOSTAT ($€_{2015}$)
Wheat	137	IFS € (2003)	179	FAOSTAT ($€_{2015}$)
Potato	113	FAOSTAT € (2001)	214	FAOSTAT ($€_{2015}$)
Rice	200	IFS € (2003)	305	FAOSTAT ($€_{2015}$)
Rye	99	FAOSTAT € (2001)	142	FAOSTAT ($€_{2015}$)
Oats	132	FAOSTAT € (2001)	145	FAOSTAT ($€_{2015}$)
Tobacco	2,895	IFS € (2003)	3,508	FAOSTAT ($€_{2015}$)
Barley	93	IFS € (2003)	153	FAOSTAT ($€_{2015}$)
Sugarbeet	64	FAO € (2002)	34	FAOSTAT ($€_{2015}$)

5.5 Valuation of buildings and materials

5.5.1 Description of endpoint

Pollution can affect the quality of man-made capital goods, leading to higher maintenance costs. Acidification, for example, leads to accelerated erosion of calcareous building materials (gypsum, cement and concrete)[48], iron and steel (reinforced concrete) and zinc gutters (VMM, 2013a). This shortens the useful life of these materials and means additional maintenance costs, as well as potentially causing permanent damage to historic buildings, monuments and suchlike (Watt et al., 2009). Another example is particulate matter soiling windows and causing visual damage to buildings, and thus giving rise to welfare losses. Because of the catalytic action of the soot particles, this pollution also accelerates the erosion of building surfaces.

Aacidification and ozone pollution (photochemical oxidant formation) also corrode rubber and paint, again pushing up maintenance costs. Discharges of toxic and corrosive materials also impact surface waters and sewers, burdening operators of water-treatment and sewage plants with extra costs.

Damage to buildings, materials and machinery is usually modest compared with impacts on other endpoints and has been given relatively little attention by researchers. Although these costs are cited in several comparative valuation studies, it is as a ‘memorandum item’ (see for example (AEA, 2005)). In the 2010 Shadow Prices Handbook the damage costs of these pollutants were partially monetized, but combined with damage to agricultural crops. Because crop damage is now included with damage to ecosystem services, in this Handbook we sought to make a dedicated estimate of air-pollution damage to buildings and materials, which indeed proved feasible.
For emissions to water this was not the case, though.

48 Cement and concrete react with atmospheric carbon dioxide to form calcium carbonate, which is then washed out by acidifying emissions. This calcium carbonate and atmospheric NO_x also react with cement to form calcium nitrate, which is rapidly flushed out.

5.5.2 Midpoint-to-endpoint relationships

Damage to buildings and materials is caused primarily by air and water pollution on the following midpoints:
- acidification;
- particulate matter formation;
- photochemical oxidant formation.

The other midpoints have no direct impacts on this endpoint.

5.5.3 Methodology in the 2010 Handbook

In the Shadow Prices Handbook these impacts were taken together with impacts on agricultural crops, in line with NEEDS. Only the impacts of acidifying emissions were quantified, with no damage costs included for photochemical oxidant or PM formation. For SO_2, which impacts mainly on buildings (and scarcely crops), a damage figure of € 0.43 per kilogram was taken, for example, based on NEEDS. This is approximately 3% of the total damage of SO_2 on all endpoints (including health and biodiversity). The values in the 2010 Handbook were discounted at 3% p.a. and expressed in 2008 prices. In line with treatment in Watkiss, et al. (2006), damage to buildings was not scaled up with a positive income elasticity, there being no empirical basis for such a step. This issue was examined by Rabl (1999), who in France found no correlation between damage costs and income.

5.5.4 New findings

A number of case studies have been published in which air-pollution impacts on a specific object or region have been calculated and monetized (see for example (Watt, et al., 2009). Since completion of the NEEDS project, however, no new estimate has been published of damages per kg emission. In an estimate of external costs in Switzerland (Ecoplan and INFRAS, 2014) air-pollution damage to buildings due to traffic was estimated to be about 20% of damage to human health. This is far more than the contribution estimated in NEEDS, which came to a maximum of 2% relative to health damage for the EU27. This can be explained partly by the fact that traffic emissions occur at a lower level, making them more damaging to buildings than emissions at *average* height. Another reason is that in NEEDS only one kind of damage was monetized: acidification impacts on normal, 'utilitarian' buildings.

For this Handbook we have therefore sought to calculate a more comprehensive estimate of external costs, particularly for the upper value. Watkiss, et al. (2006) distinguishes four cost categories associated with this form of damage:
1. Damage due to acid corrosion of metals, paint and stone in utilitarian buildings.
2. Damage due to acid corrosion of calcareous building stone in historic buildings.
3. Damage to paint and rubber due to ground-level ozone.
4. Damage to buildings due to particulate pollution.

In addition, damage due to reduced visibility is sometimes also distinguished (Watkiss, et al., 2001). Although there are a number of American studies allocating such costs to PM_{10} (notably in cities located in valleys), this is not frequently encountered in Europe. In the (largely flat) Netherlands this problem is virtually absent and so has not been quantified here. Cost estimates in the literature (Rabl., 1999); (Holland, et al., 1998); (Bal, et al., 2002); (Watkiss, et al., 2006); (VMM, 2013b) for damage per unit emission are generally based on additional expenditure on building maintenance. While PM pollution is eminently suitable as an issue for CVM studies on the visual 'nuisance' of soot-soiled buildings, in practice such studies are few and far between (cf. (Rabl., 1999)). Using restoration costs is a less accurate measure, because, as also argued in Chapter 2:
1. For impacts on buildings, valuation on the basis of restoration costs may potentially lead to overestimation, as it is not always economically optimal to repair all damage (see also Chapter 2).
2. If valuation based on restoration costs proceeds from real-world expenditure on building repair by property-owners, this objection is removed, as we then have a 'revealed preference'. This is the route adopted by Rabl (1999), among others. For rented buidings this leads to an underestimate, though, as scarcity and regulations mean this market segment is not entirely efficient. Here, the party renting a soot-soiled building may suffer a loss of welfare but see the landlord unwilling to clean it as he can still rent it for the set price. Without providing any supporting evidence, Rabl (1999) states that expenditure on restoration costs amounts to approximately half the total loss of welfare.
3. Finally, not all damage can be restored: besides the damage there is thus also potentially loss of value in monuments and other objects of cultural heritage. According to VMM (2013b), case studies show that aesthetic impacts on such objects are of the same order of magnitude as restoration costs.[49]

An extra complication is mentioned in Watkiss, et al. (2006) and VMM (2013b), where the point is raised that in determining damage to specific cultural heritage national averages may not simply be taken. This is because the various types of traditional materials used in such objects vary very widely when it comes to air-pollution impacts. Limestone is far more sensitive to damage by acid deposition than brick, for example. This means a study on one particular region or country cannot just be applied to another. For this reason, Watkiss, et al. (2006) proposes not quantifying this impact. At the same time, though, the impacts of acid emissions on concrete, brick and cement are far more uniform, making rough estimates of damage to these materials feasible.

5.5.5 Choices in this Handbook

For this Handbook we have worked with a range: the low/central estimate includes the damages that are certain, the high estimate those that are uncertain, too. Because for the impacts on buildings and materials we found more evidence for the low estimate being correct, in this Handbook we have also taken this as a central value.

49 Since no references are given for the case studies, this statement is hard to verify.

The following four cost categories have been adopted here:

1. **Corrosion due to acidification.** As in the 2010 Shadow Prices Handbook, the corrosive impacts of acidifying emissions on metals, building stone and paint are based on NEEDS (2008a). NEEDS itself derives its prices from maintenance costs per square metre for a number of different materials. These prices have not been adjusted to the slightly higher density of buildings in 2015 compared with 2000, because we assume this has been offset by use of less corrosion-sensitive materials in buildings (including renovations).
2. **Particulate pollution.** The impacts of particulate pollution are based on Defra (2016), who in turn derive their calculations from Rabl (1999), who analysed expenditure on restoration of pollution-soiled buildings in fifteen French cities. Applying a regression analysis, Rabl estimated damage costs, defined a CRF-function and calculated damage costs as € 0.21/kg PM_{13} in 1998 prices. This value has been taken as the basis for Dutch restoration costs, correcting for population density and inflation and the fact that Rabl took PM_{13} rather than PM_{10}. This results in an estimate of € 0.8 for 1 kg PM_{10} in the Netherlands.[50] It should be noted, though, that this value holds only for primary particles, because this is the fraction containing soot.
For secondary particulates, eventual damage has been set at zero.[51] Given a ratio of 1/2 for $PM_{2.5}/PM_{10}$, this means the value for $PM_{2.5}$ is € 0.4/kg $PM_{2.5}$.
3. **Corrosion impacts on cultural heritage.** In line with the British and Belgian handbooks, impacts on cultural heritage have not been valued using a central value, as the uncertainties are too great. VMM (2013b) states that these are about the same as the restoration costs under category (1). For Paris, Rabl (1999) calculates these to be 62% of the combined restoration costs under (2) and (3). This is in line with the approach adopted in VMM (2013b). We have therefore taken this as the upper damage value.
4. **Impacts on paint and plastics.** For the costs of damage to paint and plastics due to ozone, we adopted the values reported in Watkiss, et al. (2006), who state that paint damage is unlikely have any major impact as average ozone concentrations are generally too low. According to Watkiss, et al., evidence of such impacts dervies mainly from US studies carried out in the late '60s. For damage to rubber materials empirical evidence does exist, though.
For the UK a central value of £ 85 million/yr has been estimated, with a range from £ 35 million to 189 million (1997 data). If this is compared with total 1997 UK emissions – 2,032 kt – this is a modest sum.[52] Since then there has been a further decline in the use of natural rubber, moreover, which has been largely superseded by synthetic materials. Given these facts, we opted for a central value of zero on this impact. For the upper value we took the CRF-function from the literature underpinning Watkiss, et al. (2006), giving a damage figure of € 0.1/kg NMVOC.

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50 The damage function in Rabl (1999) is: {E*4.14 FF/(person·year·mg/m^3) x 1.05 x D} /K, where E = emission in kg/jaar, FF = French Franc, D = population density in 10^{-4} capita/m^2 and K = deposition velocity, set at 0.01 m/s. Assuming a linear CRF-function, this yields a damage estimate of 31.7 mg/s for France.

51 As acidifying pollutants like SO_2, NO_x and NH_3 also have an impact on PM_{10}, the impact of these emissions has been deducted from the total damage costs of PM_{10}.

52 Under the two simplest assumptions of a linear relationship between emission, concentration and damage and no international transport of ozone-forming pollutants, the damage would thus be about 5 €ct per kg NMVOC.

It can still be queried to what extent (2) overlaps with (1) and (4), since the study used for (2) is based on outlay on restoration costs and as the other emissions also generate restoration costs there may potentially be double-counting. Rabl (1999) carried out a regression analyses to assess whether expenditure on restoration costs correlates with atmospheric SO_2 levels, too, and found this variable to be insignificant. His conclusion was therefore that restoration costs in France were due primarily to particulate pollution rather than acidification. He nonetheless hesitates to make an unequivocal call on the issue. VMM (2013b), too, states that including both categories may potentially lead to double-counting, given one-off decisions on repair and the fact that the associated costs cannot be allocated linearly to acidification or PM formation. In light of these uncertainties, we have opted to exclude Rabi's data from the central, low estimate, on the assumption that these costs do not come on top of costs in the other categories. In the high estimate, though, these costs have been included.

5.5.6 Values adopted in this Handbook

Table 21 reports the values for emissions in Euro per kg in 2015 prices for emissions in 2016.

Table 21 Values of emissions with impacts on buildings and materials (€$_{2015}$ per kg emission)

Midpoint	Indicator used	Lower value (= central value)	Upper value
Particulate matter formation	kg PM_{10}-eq.	0	€ 0.8
Acidification	kg SO_2-eq.	€ 0.6	€ 1.2
Photochemical oxidant formation	Kg NMVOC-eq.	€ 0	€ 0.1

Based on the literature used, we recommend taking these values as constant, even if emissions decrease in the future, because most background studies assumed a linear relationship between emissions and damage with respect to this endpoint and, given the empirical evidence, this also seems most plausible.

5.6 Valuation of resource availability

Security-of-supply of mineral resources is generally seen as being of major value to society. Over 50 years ago Barnett and Morse (1963) already reported that this issue had been garnering the interest of US politicians and researchers since the late 19th century. Since then that interest has certainly not declined, as evidenced by innumerable reports, from the Club of Rome's 'Limits to Growth' (Meadows, et al., 1972), through to contemporary EC policy documents on 'sustainable use of natural resources' (EC, 2005); (EEA, 2005), 'critical materials' (EC, 2011) and the 'circular economy' (EC, 2014a).
In these and similar policy publications the importance of mineral resources - particularly resources dubbed 'crucial', 'critical' or 'priority' – is generally introduced by noting their pivotal importance for our prosperity, followed by a statement that most of our resources are currently imported from abroad. In recent years reference is then geneally made to China, which today is pursuing an expansive investment policy, mainly in poor African countries, with a view to securing resource stocks. In EU member states, policies of this kind are largely lacking (see e.g. HCSS et al., 2011).

However this may be, the question of relevance here is whether, besides resource extraction, resource *consumption* also has an external impact which might be taken on board in a SCBA or which, for an industry, might be included in calculating its own social value. The idea is then that by reducing resource consumption (including water and energy) aggregate savings accrue to society that exceed the price of the unconsumed resources. But is this the case? Can an economic perspective be developed in which resource consumption induces external costs?

It should be noted that in LCA studies this issue is not deemed relevant. Depletion of abiotic resources has long been included in LCAs as a relevant endpoint of environment interventions (PRé, 2000). What we are concerned with here, though, is the risk of leaving future generations without resources. Given the importance of the 'precautionary principe' and 'stewardship' in the LCA perspective, there is logic in putting a value on this forgotten item. In ReCiPe (Goedkoop, et al., 2013) impacts on this endpoint are quantified under the assumption that current consumption will eventually lead to higher extraction costs. For a tonne of iron ore, to take an example, this leads to additional costs that are roughly equal to the price of the ore itself. From an economic perspective these extra costs can be regarded as pecuniary externalities.[53]

5.6.1 Methodology in the 2010 Handbook

In the 2010 Shadow Prices Handbook the position was adopted that resource scarcity need not, *in itself*, induce external costs. From a traditional economic perspective resource depletion is not deemed a real or technical externality, merely a financial one.[54] If resource extraction and resource price are in accordance with Hotelling's rule, then the social value of avoiding depletion of non-renewable resources is, by definition, included in the resource price. Only if it can be convincingly argued that markets are not operating efficiently can an external cost be assigned to resource consumption – if parties are operating with erroneous information, say, or if heavily polluting extraction generates external costs that are not included in prices (CE Delft, 2010).

In the 2010 Handbook the issue was also raised that in the literature there appears to be excessive focus on the importance of resources for human wellbeing. If revenues from resource extraction are invested in activities that generate more welfare than the resources themselves, even suboptimal extraction boosts welfare. In addition, besides a pronounced cyclical component, long-term price trends of most resources tend to fall, in real terms (Simon, 1981). Innovations with respect to extraction, use and/or recycling reduce demand and increase supply, which means cyclical price rises virtually always causes prices to fall in the longer term (Bruyn, 2000).
A decrease in price is a sign of declining, not growing scarcity.

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53 The reason for including abiotic resources here is the importance of this theme for recycling issues, which are under the gambit of environment ministeries.

54 Financial externalities are determined by prices and in the context of the General SCBA Guidelines (CPB; PBL, 2013) are definied as indirect impacts with no effect on welfare. If person A buys a lot of cheese, for example, the cheese price rises, which is bad for person B who also wants cheese. This is part and parcel of an efficient market, though, and is not therefore seen as an externality affecting welfare. Financial externalities do not affect market efficiency, but do influence welfare distribution.

5.6.2 New research for this Handbook

As part of the research for this Handbook, it was reappraised whether or not resource scarcity gives rise to external impacts with implications for welfare. To that end, the following four issues were examined more closely:

1. Is the assumption valid that resource markets operate efficiently, in an intertemporal sense? This was elaborated by quantifying the social cost of rent-seeking behaviour using Hotelling's rule.
2. Does security of supply come with external costs? This was examined by considering actual expenditure on maintaining strategic oil reserves.
3. Do environmental impacts in the extraction phase have external costs not passed on in resource prices? These costs can be quantified using LCA methods.
4. Can a WTP-value be derived from the 'precautionary principle' or the notion of 'stewardship'? This leads to a recommendation for further study, as consumers cannot simply be assumed to translate moral values into a 'willingness-to-pay'.

Our research on these issues is described at length in an annex of the Dutch version of this Handbook. It emerges from the discussion there that, while it is certainly possible to estimate external costs for these impacts, the resultant figures are very uncertain.

5.6.3 Choices in this Handbook

As our research in the Annex G of the Dutch version of this handbook shows, it proves difficult to put a robust value on resource scarcity. In this Handbook we therefore recommend that further research be conducted on this important issue. Hotelling's rule does not provide a solid enough basis for calculating an interim value, as unambiguous empirical data on which to base such calculations is lacking. In addition, Hotelling's extraction model provides a very simplified picture of reality.

A lower bound would appear to be given by the economic damage associated with resource price volatility. The abatement-cost and damage-cost approaches both yield very low values, with additional costs amounting to less than 1% of the resource's market value.

The upper bound is more uncertain. For setting an absolute upper value, consideration might be given to adopting the method used in ReCiPe. It seems probable, though, that the upper bound is very much lower than the value reported there. Without additonal study, no precise conclusions can be drawn.

5.7 Valuation of wellbeing

Environmental interventions can also cause nuisance by affecting people's general wellbeing, by disturbing their peace and quiet, spoiling valued views, affecting the smell of the countryside, or degrading other aesthetic or spiritual values. In many cases there is no directly observable relationship between emissions and this endpoint. Nor, indeed, is this type of nuisance often included in LCA calculations. For these reasons these issues cannot be quantified as a unique endpoint in this Handbook, but are instead grouped together under the theme 'wellbeing'.

Here, two categories of nuisance are valued:
- noise nuisance;
- visual nuisance.

5.7.1 Noise nuisance
Ambient noise is a major environmental problem that has a variety of deleterious impacts on human wellbeing and health, as well as on nature. As traffic noise is far and away the main source, most studies concerned with valuing noise nuisance have focused on the transport sector (Navrud, 2002). Studies on the valuation of noise from other sources like industry and neighbours are scarce, though several studies have investigated the noise nuisance of wind turbines (see below).

There is growing evidence that noise can have a range of adverse effects on human health, with WHO (2011) distinguishing the following: cardiovascular disease, disturbed sleep patterns, reduced cognitive performance and various hearing problems. In addition, noise can therefore also lead to productivity losses. All these impacts have already been included under the theme ‘human health’, however.

Even if noise does not cause health impacts or productivity losses, though, it can still be experienced as irritating or annoying, when one is enjoying a summer’s day in the garden, for instance. This is the kind of nuisance that is captured in the endpoint ‘wellbeing’.

In addition, noise also has impacts on ecosystem services, by disturbing quiet areas, for instance, thus reducing the recreational value of parks and nature and possibly even impacting the ecosystems themselves. There has been very little research on these last two impacts, however, and they are not generally included in analyses.

In this section we consider how nuisance is to be valued. First, we briefly discuss the three methods generally used for this purpose, going on to examine the environmental prices they yield. Finally, we present our own conclusions on valuing noise nuisance with respect to wellbeing.

Valuation methods
Three basic methodologies can be distinguished for valuing nuisance due to ambient noise:
- **Stated preference (SP) methods**, in which people are asked, via surveys or experiments, to state their WTP for noise reduction. This method leads directly to a WTP per dB per person (or household). SP methods have the advantage of allowing the researcher to control for all external factors and thus isolate the value of noise nuisance. One challenge, though, is to define ‘nuisance’ in such a way that the respondent understands it in the same way as the researcher. In addition, respondents may answer questions strategically.
- **Revealed preference (RP) methods**, in which the value assigned to noise nuisance is derived from actually observed market impacts. By far the most frequently used RP method for valuing the impacts of noise is hedonic pricing, deriving the WTP for noise reduction from variation in house prices. The great advantage of RP methods is that valuation is based on people’s actual behaviour (Andersson, et al., 2013). On the other hand, though, it is difficult to isolate the impact of noise on house prices (methodologically, confounding variables, etc.).

– **Environmental Burden of Disease (EBD)**: in recent years there have been several studies valuing noise nuisance using DALYs (Bruitparif; ORS Ile-de-France; WHO, 2011); (Defra, 2014); (WHO, 2011). In the broad definition of health adopted by the WHO (*"a state of complete physical, mental and social well-being and not merely the abscence of disease or infirmity"*, WHO, 2011) nuisance can be deemed a health impact and can therefore be expressed in DALYs. The advantage of this method is that the risk of double-counting with certain other health impacts (e.g. disturbed sleep) can be avoided, since the number of DALYs can be determined for each 'health endpoint' individually. The greatest drawback of this method is the major uncertainty surrounding the 'disability weight factor' to be adopted. Because nuisance is a less clear-cut health effect, it is hard for experts to assign an appropritate factor. In addition, there is as yet little literature on this issue. The range of values proposed by WHO (WHO, 2011) for the disability weight factor is consequently fairly broad: 0.01 to 0.12, with 0.02 as a central value.

In the literature there is no clear agreement as to which of the three methods is preferable (Andersson, et al., 2013). Here, we therefore take a closer look at the literature on all three methods.[55]

Results of SP studies
In the 2010 Shadow Prices Handbook the damage costs from HEATCO (HEATCO, 2006) were recommended for valuing noise. In that study a review of (six) SP studies published by Navrud (Navrud, 2002) was adopted as the basis for valuation of noise nuisance. The latter study arrives at a range from € 2 to 32 per dB per household per annum (in 2001 prices). Based on this result, the EU Working Group on Health and Socio-Economic Aspects (2003) recommended using a shadow price for noise nuisance of € 25 per dB per household. This value was adopted by HEATCO and converted to national values. Corrected for inflation and average household size, for the Netherlands this gives a (constant) value of € 16 per dB per person per annum for road and rail noise and € 25 for aviation noise (in 2015 prices). The higher price for aviation noise reflects the fact that people experience aircraft noise as 'worse' than road-traffic noise (see e.g. (Miedema & Oudshoorn, 2001)). For rail traffic, HEATCO (2006) applied a 'rail bonus' of 5 dB (a threshold of 55 dB rather than 50 dB), because rail noise is experienced as less of a nuisance than road-traffic noise.

Since HEATCO, 2006/Navrud, 2002, one extenstive meta-analysis of SP studies in this field has been published, by Bristow et al. (2015).
For higher noise levels, in particular, this new study reports higher values than HEATCO (2006). In contrast to HEATCO (2006), which uses a constant value per dB, Bristow et al. (2015) work with a value for noise nuisance that rises with noise levels. This rising value is in line with the valuation applied in other European countries (see Table 22; the decibel units are explained in Section 6.11.3).

55 A more elaborated treatment of this can be found in Annex F of the Dutch version of this handbook.

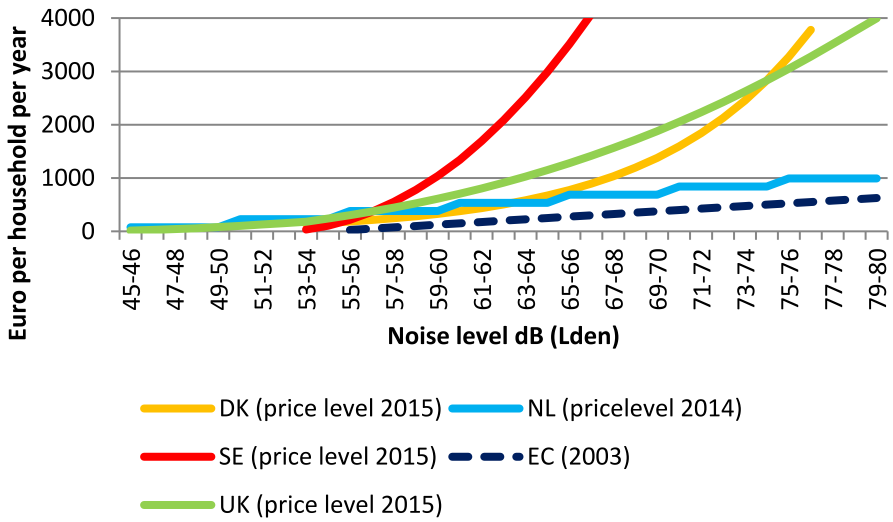
Table 22 **Results of SP studies (€2015 per person per dB(L_{den}))[a]**

	< 55 dB	55 - 64 dB	> 65 dB
Road traffic			
HEATCO (2006)/Navrud (2002)	16	16	16
Bristow et al. (2015)[b]	22 (18-25)	43 (36-50)	82 (69-95)
Rail traffic			
HEATCO (2006)/Navrud (2002)	0	16	16
Aviation			
HEATCO (2006)/Navrud (2002)	25	25	25
Bristow et al. (2015)	52 (43-60)	103 (86-119)	196 (164-227)

a In converting household values to values per person an average household size of 2.2 persons was assumed for 2015 (CBS).

b The range in environmental prices presented by Bristow et al. (2015) depends on how consumer surplus is measured: the lower bound is based on WTP-values for a loss (higher noise level), the upper bound on WTA-values for a gain (lower noise level). The central value is the average of the two.

Figure 11 **Valuation of nuisance due to road-traffic noise as a function of noise levels in various EU countries (€ per household per annum)**



Source: CEDR, Technical report 2017-03. State of the art in managing road traffic noise: cost-benefit analysis and cost-effectiveness analysis, January 2017.

As in HEATCO (2006); Bristow et al. (2015) assign a significantly higher value to aircraft noise than to road-traffic noise. In contrast to the acoustic literature, however, no evidence is found for a lower value for rail noise relative to road-traffic noise.

Results of RP studies

Table 23 provides a synopsis of the values assigned to noise nuisance in various studies based on hedonic pricing. The results are reported here in terms of a Noise Sensitivity Depreciation Index (NSDI), which gives the average percentage decline in house prices for a 1 dB increase in noise. An NSDI of 0.55 therefore means that house prices fall on average by 0.55% for every decibel increase in noise.

Table 23 Results of hedonic pricing studies

Study	Noise Sensitivity Depreciation Index (NSDI)
Road traffic	
Dekkers & Van der Straaten (2008)	0.16
Theebe (2004)	0.0-0.5
Udo et al. (2006)	1.7 (1.1-1.9)
Anderson et al. (2010; 2013)	1.15-2.19
Bateman et al. (2001)	0.55 (0.08-2.22)
Day et al. (2007)	0.18-0.55
Navrud (2002)	0.08-2.22
Nelson (2008)	0,4-0,6
SAEFL (2003)	0.6-1.2
Nellthrop et al. (2007)	0.20-1.07
Rail traffic	
Dekkers & Van der Straaten (2008)	0.67
Theebe (2004)	0.0-0.5
Udo et al. (2006)	1.7 (1.1-1.9)
Anderson et al. (2010; 2013)	0.08-1.03
Day et al. (2007)	0.67
Aviation	
Dekkers & Van der Straaten (2008)	0.77
Lijesen et al. (2010)	0.8
Theebe (2004)	0.0-0.5
Getzner & Zak (2012)	0.85 (0.5-1.3)
Nelson (2008)	0.7-0.9

The NSDI varies from 0.08 to 2.22[56], with both Bateman (Bateman, et al., 2002) and Navrud (2002) reporting that the average NSDI is probably towards the lower end of this range (0.55). This is in line with the latest studies. There is also no evidence that the studies on the Netherlands (Dekkers & Van der Straaten, 2008); (Lijesen, et al., 2010); (Theebe, 2004); (Udo, et al., 2006) yield significantly higher or lower values than the international studies.

For a comparison with the values derived from the SP studies, we took an illustrative test case to determine the value (per person). This yielded an NSDI of 0.55. Based on an average house price of € 230,000, an average household of 2.2 persons, a 5% p.a. discount rate and a 10-year discount period, this NSDI corresponds with a WTP of approximately € 75 per person per dB per annum. This value is in relatively close agreement with the values reported by Bristow et al. for higher noise levels.

Many of the RP studies cited in Table 23 assume a linear relationship between noise level and NSDI. There is a paucity of literature on the possibility of this being non-linear (Blanco & Flindell, 2011). Two Dutch studies (Udo, et al., 2006); (Theebe, 2004) have done so explicitly, however, and both conclude that the value increases with rising noise levels. Theebe (2004) only finds this effect at noise levels over 65 dB(A), while Udo et al. (2006) observe it over the entire range.

56 The relatively large differences in estimated NSDI can be (partly) explained by methodological differences among the studies (e.g. the functional form employed), the various methods used for controlling for confounding variables (e.g. air quality), or differences in preferences among those in the cohorts investigated (Blanco & Flindell, 2011).

Finally, the results shown in Table 23 also support the acoustic literature (e.g. (Miedema & Oudshoorn, 2001)) in which people experience aircraft noise as 'worse' than road-traffic noise.

On the comparison between road-traffic and railway noise there is less agreement. The results of Andersson et al. (2010; 2013) indicate people put a higher price on the former, which is in line with the acoustic literature. Day et al. (2007) and Dekkers and Van der Straaten (2008), in contrast, report higher NSDI-values for rail than road, with Day et al. positing that this might be explained by the small number of observations for rail, rendering the results for this category less reliable.

Results of EBD studies
In recent years a number of studies have been carried out that seek to put a price on noise nuisance by estimating how many DALYs correspond with the noise nuisance experienced (e.g. (Bruitparif; ORS Ile-de-France; WHO, 2011); (Defra, 2014)). In doing so, these studies base themselves on the WHO recommendations (WHO, 2011). We converted the results of Defra (2014) to Dutch values; see Table 24.

Table 24 **Results of EBD studies (€$_{2015}$ per person per dB (L_{den}))**

	< 55 dB	55-64 dB	> 65 dB
Road traffic	11	20	40
Rail traffic	6	14	37
Aviation	21	38	55

Comparison of these results shows that the Defra (2014) values are considerably lower than those of Bristow et al. (2015). This is due (partly) to the EBD method adopting a conservative approach in which only the most serious kinds of nuisance ("highly annoyed") are included. WHO (2011) provides no disability weight factor that can be applied to cases with less nuisance, moreover, making it impossible to correct the estimates on this point.

Like Bristow et al. (2015) and some of the RP studies, Defra (2014) states that the value to be assigned to noise nuisance increases with noise levels. Also, the differences found in the value of noise from the various types of transport are in line with the acoustic literature.

Conclusion
Based on the above analysis, for the Netherlands we recommend using the environmental prices found by Bristow et al. (2015). Compared with the prices recommended in the 2010 Shadow Prices Handbook (which were based on HEATCO, 2006), these values have the great advantage of increasing with rising noise levels. This means these values are more in line with both the latest literature and the valuation indices used in other European countries (Denmark, UK, Sweden). Moreover, using SP rather than RP results has the benefit of these being easy to use in a wide range of research and policy settings, as they are already expressed in € per dB per person. Finally, compared with the EBD results, the SP results of Bristow et al. (2015) have the advantage of including a greater fraction of the nuisance and are also based on more reliable methods.

No values for rail-traffic noise are reported by Bristow et al. (2015). However, in line with the acoustic literature (and some of the valuation literature), we recommend basing these values on those for road-traffic noise, but applying a 5 dB ‘rail bonus’.

As a threshold we propose taking 50 dB(A), in line with the recommendations in the 2010 Handbook. Although it is known that lower noise levels also cause nuisance (WHO, 2011); (EEA, 2010) it is insufficiently clear to what extent the valuation studies yield reliable indices for lower noise levels, too.

Table 25 provides a synopsis of the recommended values for noise nuisance.

Table 25 Recommended values for noise nuisance (€$_{2015}$ per dB (L_{den}) per person per annum)

	< 55 dB	55 - 59	60- 64 dB	65-69 dB	≥ 70
Road traffic	22	43	43	83	83
Rail traffic	0	22	43	43	83
Aviation	52	103	103	196	196

These values are added to those for health impacts (see above) to arrive at an integral value for noise nuisance. In Section 6.11 more information on valuation of noise is presented.

5.7.2 Visual nuisance

Visual nuisance, too, can impact welfare. This may the case when a new development reduces local environment quality, by directly blocking a view, for example, or by changing the nature of the landscape and making the view less attractive. Factors affecting the degree to which visual nuisance is experienced are the height, shape and size of the object deemed a nuisance, its proximity to homes and its disharmony with landscape morphology.
In addition, the amount of visual nuisance depends on how well the new development is consciously blended into its surroundings.

Visual nuisance may lead to a decline in the value of the area concerned, making it less attractive to live or be there. As visual nuisance is always highly context-specific, it is impossible to draw up generally valid valuation guidelines. In the Dutch edition of this Handbook a specific study is cited (VU, 2014) that uses revealed preferences to establish the drop in house prices near wind farms. As wind farms also cause noise nuisance, though, a universally valid indicator for visual nuisance still remains unfeasible.

This category of nuisance is consequently not included in the environmental prices in this Handbook.

6 Valuation of midpoint impacts

6.1 Introduction and general methodology

This chapter discusses how environmental prices have been set at midpoint level, i.e. for each of the individual environmental themes. In this Handbook eleven midpoints are distinguished:
1. Ozone depletion.
2. Climate change.
3. Particulate matter formation.
4. Photochemical oxidant formation.
5. Ionizing radiation.
6. Human toxicity.
7. Ecotoxicity.
8. Acidification.
9. Eutrophication (freshwater and marine).
10. Nuisance (noise).
11. Extraction (land use).

These midpoints are described in Sections 6.2 to 6.12, along with the methods used to arrive at the estimated impacts and the values assigned to them. First, though, in Section 6.1.1 we briefly review which midpoints have been taken to relate to which endpoints.

6.1.1 Midpoint-to-endpoint relationships

There is a vast web of potential relationships between the eleven midpoints and five endpoints distinguished in this Handbook. Table 26 summarizes which of them are covered here. For a schematic picture, see Figure 5 in Section 2.3.4.

Table 26 **Relationships between midpoints and endpoints covered in this Handbook**

Endpoint Midpoint	Human health	Ecosystems	Buidings & materials	Resource availability	Wellbeing
Ozone depletion	YES	partly			x
Climate change	diff	diff	diff	diff	diff
Particulate matter formation	YES		YES		
Photochemical oxidant formation	YES	partly	partly		
Ionizing radiation	YES	x			x
Acidification	diff	YES	YES		
Human toxicity	YES				
Ecotoxicity		YES			
Eutrophication		partly			x
Nuisance (noise)	YES				partly
Extraction (land use)		partly		diff	partly

Explanation: YES (green): impact included virtually entirely and monetized accordingly.
partly (orange): impact partly monetized.
x (red): characterization from midpoint to endpoint, but result not incorporated here.
diff: impact determined differently. For climate change (blue) abatement costs were used, while for acidification (yellow) impacts were allocated under the headings of particulate matter formation and smog formation. For extraction (violet), no definitive method was found for the impact on resource availability.
An empty cell means the theme was not characterized with respect to the endpoint or that impacts are negiligble.

An empty cell means there is no midpoint-to-endpoint characterization in our methodology. An 'x' in a cell means that while such characterization is in principle feasible, no estimate is provided in this Handbook. Many of these impacts are still being explored by researchers, while others are so location-specific that no universally valid averages can be given for the Netherlands. 'YES' means midpoint-to-endpoint characterization is relevant and that a quantitative average for the Netherlands is included in this Handbook, while 'partly' indicates only some of the impacts have been quantified.

6.2 Ozone depletion

6.2.1 Description of midpoint

The ozone layer is a layer of the atmosphere about 15 to 30 km up in the stratosphere that is relatively rich in ozone (O_3). It filters out much of the incoming solar ultraviolet radiation (UV), which is hazardous to life on Earth. In the 1980s the thickness of the ozone layer was found to be declining, reducing the effectiveness of this shield. Variations in the thickness of the ozone layer are in part a natural phenomenon, caused among other things by volcano eruptions, but are also due to human activity, most specifically emissions of chlorine- and bromine-containing chemicals. These compounds react with stratospheric ozone, reducing its effectiveness as a UV-filter.

While ozone-layer depletion is a global environmental problem, the impacts are not the same everywhere, as the layer's thickness depends very much on latitude. At the equator it is thinner and less subject to variation. This is the source region for production of stratospheric ozone and here emissions have the least impact on ozone levels. In polar regions, in contrast, the layer is thickest but also most subject to fluctuation and depletion through the action of chemicals. This is because the ozone is not produced here, but accumulates after transport from the equator. If transport remains constant while depletion intensifies, a deficit arises, observed as a 'hole' in the ozone layer.

Global emissions of ozone-depleting substances (ODS) peaked in the mid-90s and have been slowly declining since (Fraser, et al., 2015). Despite successful international agreements, ODS are still used in a range of applications and are released as emissions (e.g. through leakage). Because of the time lag between emissions and resultant ozone levels, on average 15 years (VMM, 2013e), it is only recently that the thickness of the ozone layer has begin to recover. With continued decline in ODS emissions, recovery should eventually proceed more effectively than at present.

6.2.2 Sources

Stratospheric ozone is broken down by chlorine, bromine and nitrogen compounds, with CFCs, halons, HCFCs and methyl bromide constituting the main human sources. These chemicals, which have been in production since the early 20th century, are used principally as coolants in refrigerators and air-conditioning systems, as chemical 'dry cleaning' agents, in aerosol cans, as fire retardants, in foam manufacture and for soil defumigation (methyl bromide). Global production of ODS has declined substantially since the mid-'90s thanks to measures implemented under the Montreal Protocol.

Besides chlorine and bromine compounds there are also other pollutants that can impact the ozone layer, such as nitrogen compounds. The main nitrogen compound reaching the stratosphere is nitrous oxide, or laughing gas (N_2O). Although most of this comes from natural sources, there is also a sizeable anthropogenic component, particularly from agriculture.

6.2.3 Impacts

Ozone depletion impacts humans, plants and animals. UV-radiation can damage DNA and proteins in the skin and eyes, and over time cause skin cancer and cataracts. It also affects the physiological functioning of wild plants and agricultural crops and can cause radiation damage (VMM, 2013e).[57] Ozone depletion thus negatively affects both human and ecosystem health.

Most ozone-depleting substances are also greenhouse gases, thus contributing to climate change. These impacts are characterized under the endpoint 'climate change', however, and are included there in this Handbook.
In addition, there are impacts on photochemical oxidant formation, with a decrease in stratospheric ozone sometimes leading to an increase in ground-level ozone. This impact is not included in ReCiPe and has consequently not been taken on board in the present Handbook, either.

6.2.4 Midpoint indicator unit

Substances with an impact on the theme 'ozone depletion' were characterized according to ReCiPe (Goedkoop et al., 2009, 2013). In ReCiPe impacts on this midpoint are expressed in kg CFC-11-equivalents. CFC-11, a chlorinated fluorocarbon formerly used mainly as a refrigerant, has the highest ozone-depleting potential (ODP) of any compound in this family.[58] It is defined as having an ODP of 1.

6.2.5 Valuation in this Handbook

Valuing the impact of ODS was not an issue covered by the NEEDS project. Our estimates of human health impacts are therefore based on the ReCiPe methodology (Goedkoop et al., 2013). There, the impact of a change in UV-B-radiation on human health is calculated using the AMOUR model. The resultant damage factor is expressed in DALYs per unit change in the Effective Equivalent of Stratospheric Chlorine (EESC), with this figure then converted to a characterization factor in DALYs/CFC-11-eq. for each class of ODS. This is the same approach as adopted in the 2010 Shadow Prices Handbook.

For human health impacts, a monetary value was obtained using a standard value for a DALY, under the assumption that 1 DALY = 1 VOLY.

For impacts on ecosystem services, only endpoint damage to agricultural crops was included. For a selected series of crops this damage was multiplied by the estimated production cost, based on Hayashi et al. (2006). This is identical to the approach adopted in the 2010 Handbook.

Table 27 reports the average midpoint characterization factors adopted for the Netherlands on this theme. As can seen, there is a substantial difference between valuation according to the individualist perspective and the hierarchist perspective. This is primarily because there is no discounting of longer-term impacts in the latter case, while in the hierarchist perspective other health impacts besides skin cancer are also included, such as cataract. These impacts are more uncertain and are consequently ignored in the individualist perspective.

57 In the Antarctic seas, for example, excessive UV exposure is damaging phytoplankton, affecting both growth and DNA. Phytoplankton form the basis of the marine food chain.

58 CFC-11 is also an important greenhouse gas.

Table 27 **Average damage costs for ozone depletion for an average Dutch emission source in 2015 (€$_{2015}$ per kg emission)**

Pollutant	Perspective	Lower	Central	Upper
CFC-11	Individualist	22.1	30.4	45.7
CFC-11	Hierarchist	NA	**123**	NA

6.3 Climate change

6.3.1 Description of midpoint

Climate change refers to anthropogenic changes to the Earth's climate (temperature, weather). The climate is currrently changing as a result of rising atmospheric concentrations of greenhouse gases, which let through incoming solar radiation but prevent escape of the infrared radiation reflected from the Earth's surface. This phenomenon, the greenhouse effect, is causing global temperatures to rise. The principal greenhouse gases (GHG) are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), but there are many others, including ozone-depleting chemicals like HCFCs (see above).

Rising atmospheric GHG levels and the resultant rise in global temperatures are already having major effects on ecosystems and weather patterns.
If current emission levels remain unchecked, average global temperatures are set to increase to 6°C above preindustrial levels. This will have extremely grave and partly unpredictable impacts on weather systems, sea levels and the environments on which plants, animals and humans depend. Other likely consequences will include increased flooding, droughts and the spread of diseases like malaria.

6.3.2 Sources

The single largest source of GHG emissions is fossil fuel combustion. Fossil fuels are used intensively in every economic sector and over the past century their consumption has spiralled. This generates huge emissions of CO_2, as well as nitrous oxide, which has a far greater Global Warming Potential (GWP) molecule-for-molecule. GHG emissions also arise in agriculture and in landfills, where it is methane and nitrous oxide that are released. Some industries are also characterized by high GHG emissions, such as cement and aluminium production. And then there are refrigerants and aerosol gases that also end up the atmosphere, during production and usage and as waste; this holds for both traditional CFCs and their newer replacements. All these GHG emissions lead to increased atmospheric levels of the gaes in question and consequently to global temperature rise.

Besides the cited GHG emissions there are also other pollutants that play a role in global warming. Black carbon (soot) in the atmosphere, for example, affects the amount of sunlight the Earth can reflect. The dark colour of soot means it absorbs more sunlight, leading to further temperature rise. This is particularly relevant when the particles are deposited on snow-covered surfaces, as it is precisely here that so much of the sunlight reaching the Earth is reflected back into space. There are also emissions with a cooling effect, including sulphur dioxide (SO_2). This has both a direct and an indirect cooling effect, the former due to SO_2 aerosols reflecting sunlight, the latter due to atmospheric SO_2 contributing to cloud formation and thus having a cooling effect (Fuglestvedt, et al., 2010). Aircraft emissions contribute, too, in terms of both cooling and warming (CE Delft, 2014).

6.3.3 Impacts

Without effective climate policy, temperatures are projected to rise to 6°C above prehistoric levels by the end of this century (IPCC, 2007). This kind of extreme climate change will have a major and in many respects irreversible impact on ecosystems, on human health and on the very fabric of our socio-economic systems. The impacts will not be distributed equally around the world, but will be far graver in developing nations, which moreover have less capacity to adapt (GHF, 2009).

The impacts have been described frequently and at length in the various IPCC reports and elsewhere:

- Sea-level rise will lead to major losses of farmland, particularly in river deltas, where the vast majority of the world's population lives. This will in all likelihood lead to major migrations and disrupt societies around the world. It may also lead to additional loss of farmland and wetlands.
- Direct effects on human health stem from reduced cold stress in the winter months and higher temperatures in the summer. Up to a point these impacts will cancel one another out. Additional impacts include an increased risk of exposure to certain parasitic diseases like malaria that are currently restricted mainly to the tropics.
- There will be considerable shifts in global food production, with a loss of agricultural potential in warmer countries being only partly compensated by increased potential in colder regions. These changes are expected to be rapid and may therefore lead to major socio-economic problems in terms of adaptation, with famines and mass migration increasingly common.
- There will be impacts on water supplies, with shortages aggravated in certain regions, not only through drought but also through further salinization of ecosystems. In other regions, in contrast, there will be more water available than has historically been the case.
- Impacts on ecosystems and biodiversity are the most complex and difficult to assess. Potential impacts include an increased risk of extinction of vulnerable species, altered distribution patterns and catastrophic damage to isolated ecosystems like coral reefs.
- Extreme weather events like heat waves, droughts, storms and tornadoes do not depend linearly on temperature rise and the damage they can potentially cause is very hard to estimate. There may also be catastrophic impacts like the loss of the West Antarctic or Greenland ice sheets, methane escape from melting tundra and the sea-bed, instability or collapse of the Amazon rainforest, tipping-over of ocean currents and disruption of the Indian monsoon. All these impacts are very hard to estimate, but their consequences would be enormous.

6.3.4 Mispoint indicator unit

ReCiPe characterizes the various greenhouse gases using their Global Warming Potential (GWP), based on IPCC (2007), with the GWP of CO_2 set at 1.

6.3.5 Environmental prices using the damage-cost method

In welfare economics the preferred method for valuing external costs is to base them on damage estimates (cf. Chapter 2). Ever since the emergence of anthropogenic climate change, economists have been working hard to estimate the damage it is likely to cause. By summing the various forms of damage, discounting them over time and relating them to CO_2 and other emissions, an attempt has been made to determine the Social Cost of Carbon (SCC). This SCC can be interpreted as the net present value of the future costs and benefits associated with emission of one additional so-called CO_2-equivalent.

The SCC is usually calculated using climate economic models in which assumptions about impacts are combined with assumptions on global income trends and distribution. A meta-analysis of 211 studies on the SCC has been carried out by Tol (2008), who showed that the spread in results is enormous: from less than € 1/tCO_2 to over € 500/tCO_2.[59] Taking a 3% annual discount rate, he arrives at an average figure of around € 5/tCO_2 and argues on this basis that the damage estimates reported in the influential Stern report (Stern, 2006) are outliers. He also states there is a less than 1% chance of the average damage estimate exceeding € 20/tCO_2.

Van den Bergh and Botzen (2015) consider such pronouncements premature, however, as the SCC literature is characterized by a very high degree of uncertainty. They cite four main sources of uncertainty in estimating the damage costs of climate change:
– A number of key cost categories are either ignored or only partially included: these include biodiversity losses, potential impacts on economic growth trends, political instability, violent conflicts and migration. The main reason for these omissions is a lack of reliable methodologies to estimate their magnitude.
– Uncertainties regarding impacts: there are major uncertainties about the full extent of the problematique and its impacts on the Earth's climate, sea-level rise and extreme weather events. In particular, potentially extreme events like a weakening or collapse of the Gulf Stream, complete melting of the Greenland and West Antarctic ice sheets, or changes in climate subsystems like the El Niño Southern Oscillation are frequently omitted from the analyses, or insufficiently accounted for.
– Uncertainties as well as widespread debate on the social discount rate to be adopted when calculating the damage costs of climate change.
– Insufficient allowance for people's aversion to losses and risk: people are generally risk-averse and prefer not to suffer loss. In most studies, however, this is scarcely allowed for, if at all.

Given these uncertanties, Van den Bergh and Botzen (2015) conclude that using a damage-cost method to calculate a shadow price for CO_2 will by definition yield highly uncertain results. A beter alternative in their view would be to decide on a safe atmospheric concentration of CO_2 and then perform a cost-effectiveness analysis of policies to achieve it. This boils down to the abatement cost approach.

Taking on board the criticisms of Van den Bergh and Botzen (2015), Gerlach et al. (2014) consider whether it might be feasible to simplify damage estimates, by examining the most crucial parameters. In their study they present a formula said to combine core insights from economic and climate models.

As its input, this formula takes the estimated damage resulting from a given temperature rise, the climate sensitivity (the temperature rise due to a doubling of atmospheric CO_2), gross global income, discount rate, the decay of atmospheric CO_2 and the rate at which the Earth's surface temperature adjusts. Using this simplified formula they carried out Monte Carlo calculations and then determined the extent to which uncertainties in parameters translate into uncertainties with respect to the SCC.[60] On this basis Gerlach et al. (2014) conclude that the average damage costs of CO_2 are € 37/tCO_2, with a median

59 For current CO_2 emissions, with impacts discounted at 3% per annum.

60 Based on the available literature, an estimate was made of the probability distribution of climate sensitivity, projected damage factor and discount rate.

value of € 17/tCO_2. New in their analysis is that they show the distribution of estimates is not only skewed, but also has a very long tail. They estimate there is an 8% probability of the SCC exceeding € 100/tCO_2, a considerably higher figure than concluded in the earlier work of Tol (2008). What Gerlach et al. (2014) in fact do is underscore the conclusion of Van den Bergh and Botzen (2015) that there is a major spread in results, because the underlying calculations require decisions to be made about parameters for which scientists are hard-pressed to provide precise values.

6.3.6 Environmental prices using the abatement-cost method

The 2010 Shadow Prices Handbook took as a rule that the abatement-cost method was to be given preference for pollutants on which international agreements had been reached. In this method (cf. Chapter 2) valuation is based on the marginal costs of securing the relevant policy target. To this end a so-called Pigouvian charge is taken that is precisely enough to achieve the target and is given by the cost of the most expensive measure that needs to be taken as part of the most cost-effective policy package for securing the target.

In the 2010 Handbook, valuation was based on the then-valid target of a 20% reduction in carbon emissions in 2020 compared with 1990. For CO_2 emissions a price of € 25/tCO_2 was taken for impacts up to the year 2020, followed by an incremental annual price rise to € 85/tCO_2 in 2050. This increase was based on a meta-analysis that also considered shadow costs.

Today, the EU has agreed to a far more ambitious target of 40% emissions reduction in 2030, a target that has also been adopted by the Dutch government.[61] Although there are not yet any binding targets for post-2030, European leaders have voiced an ambition to reduce the EU's carbon emissions by 80-95% relative to 1990 as part of efforts by the group of developed nations to reduce their aggregate emissions by a similar amount.[62] In addition, on 5 October, 2016 the European Parliament ratified the Paris climate agreement, under which countries are obliged to do all they can to reduce greenhouse gas emissions to such an extent that average planetary temperature rise remains far below 2°C, with 1.5°C the current objective. At the moment, global distribution of the now vastly shrunken 'emissions space' is not yet entirely clear, but for the EU the total reduction by the year 2050 is anticipated to be closer to 95% than to 80% (PBL, 2016).

The abatement-cost method calculates the marginal cost of achieving a policy target. The problem with carbon emisisons, though, is that it is not entirely clear at the moment what target holds for the Netherlands:
– Is it the 20% reduction in 2020 relative to 1990 under standing EU policy for the years up to 2020, to be achieved through concrete policies like the EU Emissions Trading Scheme?
– Is it the 25% reduction in 2020 laid down by a court in The Hague in its landmark 'Urgenda' ruling of 24 June, 2015?
– Is it the 40% reduction policy already agreed to at the EU level for 2030 and cited by the Dutch Cabinet as being the *minimum* target for that year?
– Is it the implicit target under the Paris Agreement ratified by the European Parliament, based on a maximum global temperature rise of 2°C or even 1.5°C, which (though it is not yet precisely clear how the global 'emissions

[61] See for example (COM 2014/15 final) and the ministerial letter to the Dutch parliament of 26 October, 2014 (IENM/BSK-2014/213064).

[62] See for example (COM/2011/0112).

space' will be divided) will probably lead to a policy challenge of 80 to 95% reduction in 2050 relative to 1990.

In principle one could say that, because policy ambitions extend over a range of timeframes, an overall policy 'roadmap' can be adopted running through the individual targets for 2020, 2030 and 2050. One problem with this, though, is that, in order to achieve a 2°C target at least cost, one would have to achieve more than a 20% reduction in 2020, and more than 40% in 2030. In other words, the targets agreed for 2020 and 2030 are not 'time-efficient' in light of the targets likely to stem from the Paris Agreement.

Partly for this reason, in this Handbook we propose adopting two alternative abatement-cost methods for carbon emissions, based on:

1. The current policy path, using the existing targets for 2020 and 2030 and extrapolating these to 2050.
2. The 2°C policy path, using the targets for 2050 to interpolate targets for earlier years such that an efficient price trajectory is achieved.

This is essentially the same approach as adopted in the Dutch WLO scenarios (see Text Box 4), where for climate policy, besides the Low and High scenarios, a 2°C scenario was explored based on trends in carbon prices if the global community seriously pursues drastic emissions cuts. Below we explain how environmental prices have been calculated in the two approaches. Since efforts to control climate change will intensify over time, with least-cost measures soon exhausted, the price tag on greenhouse gas reduction will rise year on year.

Environmental prices for the current policy path

The Netherlands Bureau for Economic Policy Analysis (CPB) and the Netherlands Environmental Assessment Agency (PBL) have also calculated the costs of securing various climate targets within the framework of the WLO scenarios published at the end of 2015 (CPB; PBL, 2015a). Around the same time the Dutch Cabinet adopted the recommendations of the Discount Rate Working Group, which specify that CO_2 emissions are to be valued using the price trends calculated by CPB and PBL in the two WLO scenarios: Low and High (see Text Box 4). In the High scenario the policy challenge is in line with the policy for 2030 adopted by the EU in 2014 (and currently elaborated in a range of concrete policy measures like the EU ETS. In the Low scenario the target is weaker than current policy ambitions, under the assumption that around 2025 it is realised that international climate policy is not working, leading to a further weakening of such policy (CPB ; PBL, 2015c).

Box 4 **The WLO scenarios**

At the end of 2015 PBL and CPB published their 'WLO scenarios' setting out future trends, with their associated uncertainties, for both the Dutch and the global economy (CPB; PBL, 2015a). Two scenarios were elaborated: Low and High.

In the Low scenario there is limited further globalization, resulting in lower economic growth of around 1% per annum and slower population growth. In this scenario, climate and energy policy is based as far as possible on standing arrangements, with policy targets that have already been elaborated in concrete measures and policies.

In the High scenario globalization continues apace. There is greater (international) confidence in the future than in the Low scenario, creating greater willingness to cooperate and conclude agreements. Through trade agreements there is further market integration and continued growth of migration. There is also a greater willingness to conclude international agreements on issues like climate. The High scenario combines relatively high population growth with high economic growth of around 2% per annum. In policy terms, the High scenario is based not only on standing climate and energy policy through to 2030, but also on proposed policy (such as the EU's intention to reduce CO_2 emissions by 40% in 2030 relative to 1990).

In addition to these two scenarios, a specific sensitivity scenario on carbon prices has been developed in case the world decides to pursue the efforts related to the 2-degrees target.

The WLO scenarios are based on CO_2 price trends derived from the EU ETS. These cannot simply be adopted in an SCBA, because the EU ETS is not an economy-wide instrument. In a background document CPB and PBL therefore explain how the WLO scenarios can be employed to calculate a price path that can be used in SCBAs and does offer an economy-wide perspective (Aalbers, et al., 2016). To do so they proceed from the High scenario for the year 2050 and extend the ETS to *all* sectors of the economy.[63] The High scenario thus essentially has an economy-wide CO_2 price, with marginal costs amounting to € 160/tCO_2 in 2050.

In the Low scenario, too, the 2050 CO_2 price can be considered economy-wide, because in rounded terms the prices for securing the reduction targets in Low are virtually the same as the ETS prices.[64] In both the High and the Low scenarios, then, the 2050 prices are the marginal costs of achieving the set targets for the economy as a whole. On this basis an *efficient* price path can be calculated for the intervening years with the aid of Hotelling's rule, with the CO_2 emissions space being understood as a kind of 'stock' and the prices in 2050 being discounted using the relevant discount rate.[65] For the two scenarios this yields the price paths for efficient CO_2 pricing shown in Table 28.

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63 In their exploration of the 2°C target this is already the case in 2030.

64 According to Aalbers et al. (2016), in the Low scenario the economy-wide efficient CO_2 price in 2050 equals the EU ETS price (after rounding), because the ETS abatement-cost curve is virtually flat over a large range, which means the overall reduction target in Low can be achieved through additional measures at more or less the same marginal cost.

65 Because the ETS prices are European prices, Aalbers et al. (2016) adopt a slightly higher discount rate in Hotelling's rule, resulting in 3.5% annual increase in the carbon price. In justifying this higher discount rate, the authors point to the slightly higher growth rates in Eastern and Southern Europe compared with North-West Europe and the Netherlands, implying a higher discount rate for the EU as a whole than for the Netherlands. The discount rate can also be considered as a price rise.

Table 28 Efficient CO_2 prices in the WLO secnarios (€/t_{2015} CO_2, constant prices, excl. VAT)

WLO scenario	2015	2030	2050	GHG emissions reduction in 2030/2050 rel. to 1990
Low	12	20	40	-25%/-45%
High	48	80	160	-40%/-65%

These prices can be interpreted as the marginal social costs of securing the reduction percentages in the WLO scenarios in 2050.

Up to a point, similar prices have also been calculated in other studies (though by different methods). The European Commission's Impact Assessment of the 2030 targets (EC, 2014b), for example, states that in a 40% reduction scenario EU ETS prices may rise to € 40-53 t/CO_2, under an assumption of minimum additional policy on energy efficiency and renewable energy in 2030. In the period up to 2050 prices will range from € 85 to € 264, depending on the emissions cuts that need to be achieved under the EU ETS.

In the PRIMES Reference scenario it is calculated that a standstill of the 2020 targets (as in WLO Low), with only the ETS sectors securing the agreed 1.74% annual reduction, will lead to an EU ETS carbon price of € 35 in 2030. The PRIMES Reference scenario also calculated that if annual ETS reductions up to 2020 are extrapolated to 2050, this will give an EU ETS price of around € 90/tCO_2 and economy-wide carbon emissions cuts of about 48%.

Environmental prices for the two-degree path

If politicians decide to put their shoulders behind the policies required to secure the 2°C target, carbon prices will rise substantially. In that case WLO projects prices rising to € 200 or even € 1,000/tCO_2 in 2050. Using these figures, efficient prices can be calculated for the intervening years; in 2030 they will already have to be between € 100 and € 500.

These figures may seem high, but they are backed up by other studies. In a meta-analysis of the costs of the greenhouse gas abatement required for long-term stabilization of atmospheric levels, Kuik et al. (2009) show that these costs may in fact rise far more sharply yet. From their meta-analysis of 62 studies they estimated abatement costs as a function of targets (ranging from 450 to 650 ppm CO_2-eq.). For a long-term target of 450 ppm CO_2-eq. (giving a temperature rise of approx. 2°C) they report abatement costs of € 129/tCO_2, with a range of € 69-241. For 2050 their central estimate is € 225/tCO_2, with a range of € 128-396. These values are in constant 2005 Euros.

To correct for inflation, these figures must be increased by 17% to obtain 2015 prices. This gives a central value of € 263/tCO_2, with a range of € 150-463. The central (median) value is thus closer to the lower bound for the 2°C target in the WLO scenarios than to the upper bound.

6.3.7 Valuation in this Handbook

In this Handbook valuation of the impacts of climate change is based on abatement costs. There were two reasons for opting for this approach:
1. Since publication of the 2010 Shadow Prices Handbook, to our mind the literature has only underlined the uncertainties in damage estimates. There is at any rate no trend towards uncertainty margins decreasing.
2. The Dutch Cabinet has adopted the recommendations of the Discount Rate Working Group vis-à-vis mandatory use of the values from the WLO

scenarios in SCBAs. This means these values must at any rate be adopted as our upper and lower estimates, for use in SCBAs. It means it is only logical to set the central values, recommended for use by industry and as weighting in LCA, in the same way.

In this Handbook the values from the WLO High and Low scenarios (Aalbers, et al., 2016) thus provide the upper and lower price estimates. For industry and for use in LCA we have adopted a central value. It seems likely that industries using carbon prices to calculate the impact of their activities will want to factor in the tighter policy regime post-2020. This is closer to the High scenario. We therefore propose adopting the value from the High scenario for this central value, too. This value is the same for use by companies and for LCA weighting.

For the two-degree path, in this Handbook we propose setting the lower and upper values on the basis of the WLO 'two-degree scenario'. These values should be used in SCBA sensitivity analyses in the Netherlands. As the point of departure for the central values we propose taking Kuik et al. (2009). In line with Aalbers *et* al., 2016, the 2030 value from Kuik et al. (2009) has been discounted by 3.5% p.a. to arrive at efficient prices for 2015.

Table 29 shows the CO_2 prices for 2015, 2030 and 2050 proposed in this Handbook for two different policy contexts and exclusive of VAT.

Table 29 **Environmental prices for the theme climate change (€/t CO_2 emission, excl. VAT)**

	2015	2030	2050
Current policy			
Lower	12	20	40
Central	48	80	160
Upper	48	80	160
Two-degree path			
Lower	60	100	200
Central	80	130	260
Upper	300	500	1,000

Because the environmental prices for other pollutants are (usually) based on willingness-to-pay, which is measured inclusive of VAT, these prices should be raised by the average VAT rate if they are being used together with other environmental prices in SCBAs, for example. Following SEO (2016b), a figure of 18% can be taken for this purpose (VAT and other indirect charges that increase cost-price). The impacts of other greenhouse gases can be calculated by means of characterization factors. The IPCC publishes such factors for the various gases, expressed in CO_2-equivalents, updating them at regular intervals. The most recent update is the Fifth Assessment Report from 2014. In determining the environmental prices we have based ourselves on the latest IPCC data. As an illustration: for fossil methane IPCC has a characterization factor of 30.5 kg CO_2-eq. for a 100-year time horizon: a basic figure of 28 kg CO_2-eq. plus 2.5 kg CO_2-eq. as a correction factor because methane degrades partly to CO_2.[66] We have here worked with a 100-year horizon because the

66 ReCiPe (v.1.12) currently still uses a somewhat older characterization factor of 25 kg CO_2-eq.; nor are so-called feedback impacts included. There is still debate, though, on what values can be derived for other GHG from an efficient CO_2 reduction path (cf. (PBL, 2016)). In the

policy targets agreed to under the auspices of the IPCC are also based on this perspective.

As the basis for calculating the midpoint price we propose taking the central value associated with current policy for the year 2015, viz. € 48/tCO_2-eq.

6.4 Particulate matter formation

6.4.1 Description of midpoint

Airborne particulate matter (PM) is a mixture of particles (liquid or solid) of varying size and composition. A gas containing suspended PM is known as an aerosol. PM can be categorized in various ways, the most important being:

– By origin (anthropogenic or natural). Anthropogenic emissions are caused by human activity and include soot and smoke formed in combustion, while natural (biogenic) emissions arise through natural processes like sea salt being blown onto coasts.
– By source (primary or secondary). Primary particles are emitted directly into the atmosphere by a wide range of sources. Secondary particles are formed in the atmosphere in chemical reactions involving gaseous compounds like ammonia (NH_3), sulphur dioxide (SO_2), nitrogen oxides (NO_x) and organic chemicals.[67]
– By size/diameter, usually with a breakdown into PM_{10}, $PM_{2.5}$ and PM_1, standing for particles with a diameter less 10, 2.5 and 1 µm, respectively. The smaller particles are more damaging.
– By chemical composition (PM comes in hundreds of forms). Although there are indications that PM toxicity depends not only on diameter but also on composition, there is as yet insufficient solid evidence except in the case of black carbon, which appears more hazardous than other forms (cf. Chapter 5).

6.4.2 Sources

Anthropogenic particulates are emitted from many different sources, the main being combustion processes, which give rise to fine soot particles as well as gases. The PM from combustion reactions generally belongs to the finest categories. PM also arises in certain mechanical processes, such as the milling of grain. The material blown up in these proceses usually belongs to the coarser fractions. Particles deposited on the ground are transferred back to the atmosphere by the wind or by human activity. Examples of such sources include open-air storage of sand or other bulk goods, and dirt and tyre particles blown up from roads and verges. There are also natural sources of wind-blown coarse PM, such as wind erosion of soils and atmospheric dispersion of sea-salt.

6.4.3 Impacts

Airborne particulates impact on human health and damage buildings and monuments.They also cause visual nuisance in the form of haze.

Health impacts

Of all the environmental pollutants to which humans are exposed, it is primary and secondary particulates that cause the greatest health damage, because

future these prices may therefore be adjusted to ensure the overall GHG emission cuts required can be achieved at least cost.

67 These gases are less volatile and are consequently blown downwind, where they generate aerosols by forming new particles (nucleation) or by coalescing with already existing particles (coagulation).

they transport a wide range of toxic substances directly into the air passages and lungs. Depending on the particle size, they lodge in the nose, throat and mouth cavity or in the lungs and alveoli. The smaller particles penetrate deepest into the lungs, where they can cause both immediate and later damage.

According to the World Health Organization (WHO, 2005), the $PM_{2.5}$ fraction of airborne particulates poses a greater health risk than the PM_{10} fraction. The $PM_{2.5}$ fraction is also more directly related to anthopogenic particulate emissions than PM_{10} and thus more amenable to policy action (RIVM, 2015).

Pope et al. (2004) have found evidence for three possible pathophysiological mechanisms for explaining the impacts van $PM_{2.5}$ on mortality and morbidity:
1. $PM_{2.5}$ aggravatess the severity of COPD (chronic obstructive pulmonary disease) and asthma.[68]
2. $PM_{2.5}$ causes inflammatory reactions and intensifies arteriosclerosis, which can lead to coronary heart disease.
3. $PM_{2.5}$ leads to reduced heart rhythm variability and an elevated risk of heart arrhythmia and mortality (via cardiac arrest).

In addition, toxicological studies show that particulates can also cause genetic damage as well as allergic and inflammatory reactions (VMM, 2013b).

It has been shown in a number of studies (see (VMM, 2013b)) that reduced PM levels lead to a decline in premature mortality. There are several indications that PM toxicity is influenced by both the size and chemical composition of the particles. The ultrafine size of some PM increases its toxicity and explains (in part) the health impacts (see (VMM, 2013b)). There is also evidence that certain heavy metals and black carbon have additional toxic impacts.

Although it is also often held that primary particulates are more damaging than secondary particulates, the WHO (2013) holds there are no scientific grounds for such a distinction. They therefore recommend that the two categories should be considerd equally harmful: in calculating the damage cost, we have followed this recommendation.

Impacts on buildings
Airborne particulates cause visually observable damage to buildings and monuments. Soot soils both streets and buildings, which means they have to be cleaned more often.

6.4.4 Midpoint indicator unit
ReCiPe expresses impacts on this theme in kg PM_{10}-equivalents. ReCiPe (Goedkoop, et al., 2013) has no separate characterization for the relationship between PM_{10} and $PM_{2.5}$.

6.4.5 Treatment in the 2010 Handbook
In the 2010 Handbook the theme of PM formation was modelled entirely using the NEEDS Exceltool. To obtain the weighting factor, the relative damage of each component of PM_{10} was weighted using the 2006 Dutch emission. On this basis a weighted weighting factor for use in LCAs was developed. The ReCiPe characterization was not employed in the 2010 Handbook. Based on the relative emissions of $PM_{2.5}$ and PM_{10} in the Netherlands a

[68] Although this was not strictly proven in the study, this is probably due to COPD patients usually being diagnosed with pneumonia or flu at the time of death.

characterization factor was developed expressing the relative damage of PM_{10} compared with $PM_{2.5}$. For the theme of PM formation the only impacts covered were those on human health.

6.4.6 Update: characterization factors

The ReCiPe midpoint indicator is expressed in kg PM_{10}-equivalent. The list of pollutants contributing to the PM_{10} formation is relatively short: besides PM_{10}, the variants of NO_x and SO_x as well as NH_3, a precursor for secondary aerosol formation. To a very limited extent, NMVOC also feeds in to PM_{10} formation; in ReCiPe this was valued at zero, though. More recent epidemiological studies (cf. WHO, 2013, 2014) have identified chronic health impacts for NMVOC too, however, besides the familiar acute impacts (see Section 6.5). The IIASA-TSAP project (IIASA, 2014) (WHO, 2013)[69] consequently adopted a positive value for NMVOC, equivalent to a characterization factor of 0.09 relative to PM_{10}. For particulates we have opted to follow ReCiPe, however, and include the impact of NMVOC under photochemical oxidant formation (see Section 6.5).

It should be noted that ReCiPe works with PM_{10} and not $PM_{2.5}$. One problem when using this indicator is therefore that it is not so much PM_{10} as $PM_{2.5}$ that poses human health risks. Health damage from particles larger than $PM_{2.5}$ is minor and in all likelihood negligible (Mcdonnell, et al., 2000). It is therefore important to know how much $PM_{2.5}$ there is in 1 kg of PM_{10}. In the 2010 Shadow Prices Handbook (CE Delft, 2010) this was determined on the basis of the share of $PM_{2.5}$ emissions in PM_{10} emissions. Analysis of Dutch emission data for 2008 showed this to be 61%. This figure is in agreement with the monitoring-station analysis of RIVM (2001, p.32), who concluded that the $PM_{2.5}$ to PM_{10} ratio was reasonably consistent for the six monitoring stations included in their study, ranging between 0.6 and 0.7. At the same time there are indications that $PM_{2.5}$ emissions have declined more than than PM_{10} emissions: in 1995 70% of Dutch PM_{10} emissions were $PM_{2.5}$, while in 2015 that figure had fallen to 53% (data: Dutch Emissions Register). Based on these recent emission data a characterization factor of 1.88 seems appropriate.[70]

Because the coarse fraction of PM (the share of PM_{10} with a diameter greater than $PM_{2.5}$) in NEEDS also has a modest impact on human health, we have opted to take the *health damage* of $PM_{2.5}$ relative to PM_{10} as our basis for characterization. This leads to a characterization factor of 1.79 for $PM_{2.5}$ for 1 kg PM_{10}-equivalent.

6.4.7 Discussion: adjustment of valuation for black carbon?

Several recent WHO studies report new scientific findings on black carbon, also known as black smoke. A WHO literature study shows that it has health impacts that are in many cases greater than those due to PM_{10} (by a factor 6-14), but the same within the interquartile range[71].

As a rough approximation, airborne particulates can be taken to comprise three chemical categories:
- carbon: mainly primary anthropogenic PM, which can have health impacts;
- organic: mainly secondary PM deriving from hydrocarbons;
- inorganic: mainly primary natural and secondary sources, making up a large fraction of total PM emissions.

69 The reference is to the IIASA-TSAP Clean Air Europe project.

70 1.88 = 1/0.53.

71 Interquartile range is used as a measure of data variability, If the data is divided into four quartiles, the interquartile range is the central 50% of the data.

Particle size and chemical composition are related. Smaller particles like $PM_{0.1}$ consists mainly of black carbon. In principle, in this Handbook we have valued health impacts in relation to $PM_{2.5}$ levels rather than PM_{10} levels. On average, using the characterization factors described above, $PM_{2.5}$ causes twice as much damage as PM_{10}. According to WHO (2014) there is no unambiguous evidence that black carbon has a greater impact than $PM_{2.5}$, but at the same time they state that in some cases it may serve as a useful additional indicator.
There must then be information available on the fraction of black carbon in PM and on emissions of black carbon, however. Because such data are not systematically collected as Dutch averages, in this Handbook we propose making no separate adjustment for this issue.

6.4.8 Valuation in this Handbook

For the impacts of particulate matter formation on endpoints we adjusted the impacts of $PM_{2.5}$, NO_X, NH_3 and SO_2 from the NEEDS project (2008a) using the updates cited in Section 6.4.6. In addition, the following corrections were implemented:
- lower emissions and consequently lower background concentrations;
- higher population growth and change in age cohorts (see Annex B);
- inflation;
- addition of restoration costs for soiled buildings in the upper price, as per Section 5.5.

For the endpoint 'health', NO_x and to a lesser extent also SO_2 contribute not only to PM formation but also to photochemical smog formation. Here we have allocated acute impacts entirely to the latter theme and chronic impacts besides those of NO_2 (for which see Section 6.5.7) entirely to the former.
Table 30 reports the average values for the Netherlands for the pollutants of relevance for this midpoint.

Table 30 **Average damage costs for PM formation for an average Dutch emission source in 2015 (€$_{2015}$ per kg emission)**

Pollutant	Lower	Central	Upper
PM_{10}	€ 31.8	€ 44.6	€ 69.1
$PM_{2.5}$	€ 56.8	€ 79.5	€ 122
SO_2	€ 17	€ 23.8	€ 36.5
NO_x	€ 10.4	€ 14.5	€ 22.2
NMVOC*	€ 0.287	€ 0.401	€ 0.614
NH_3	€ 18.5	€ 25.9	€ 39.7
MPF** ($kgPM_{10}$-eq)	€ nb	€ 69	€ n.c.

* Values for the chronic impacts of photochemical smog formation; cf. Section 6.5.

** MPF = midpoint characterization factor; the prices cover not only damage to human health but also damage to buildings.

Apart from the upper values for PM_{10} and $PM_{2.5}$, all these environmental prices are determined entirely by human-health impacts. Besides these pollutants, there are no others that have impacts on this midpoint. For this reason the ReCiPe characterization factors are of no further relevance here.

The damage costs per unit emission are higher in the Netherlands than the European average. This is due, on the one hand, to the high population density in this country, which means emissions cause more damage here than elsewhere in Europe. This relationship is not linear, though, as Dutch emissions are transported partly to other countries with a lower population density.

The damage caused by a unit emission of $PM_{2.5}$ in the Netherlands is generally twice the European average (EU27).

On the other hand, these higher costs are also due to the specific atmospheric reactions involved. The Netherlands has relatively high atmospheric NH_3 levels. NH_3, NO_x and SO_2 all react to form particulates, but in the case of NO_x the relationship is linear, while for NH_3 it is quadratic. In this country the emissions of all three pollutants are set to decline between 2010 and 2020, but the reduction will be more pronounced for NO_x and SO_2 than for NH_3. This means there will be relatively more atmospheric NH_3 for the NO_x and SO_2 to react with. This is the main reason that lower emissions of NO_x and SO_2, if unaccompanied by an equal decline in NH_3 emissions, lead to higher damage costs per kg emission for these pollutants. Until such time as NH_3 emissions are tackled more effectively, this situation will persist in the Netherlands.

Based on the same systematics and characterization according to the hierarchist perspective (see Annex A), a midpoint environmental price of € 69/kg PM_{10} has been taken as the characterization factor for PM_{10}-equivalent.

6.4.9 Specific values for power stations and industry

The values cited above are averages for an average Dutch emission. As a substantial share of PM emissions are traffic-related and emissions height is a particularly important factor in PM distribution and impacts, these average values are not always applicable when the specific emission source is known. Especially for power stations and industry, the height of stacks is a major factor determining further emissions dispersion and dilution. In the densely populated Netherlands this is particularly important: a higher stack makes it more likely that a fraction of the emissions will end up in less populated areas.

Based on the NEEDS modelling runs we can now make a conversion for emissions from stacks over 100 metres high.[72] This is typically the case for coal-fired power stations and refineries. Table 31 summarizes the damage costs to be used in such cases. As can be seen, damage costs are almost 50% lower if emissions are from a stack over 100 metres high.

Table 31 Average damage costs for PM formation for an average Dutch emission from a >100 m stack in 2015 (€$_{2015}$ per kg emission)

Polutant	Lower	Central	Upper
PM_{10}	n.c.	€ 22.7	n.c.
$PM_{2.5}$	€ 26.2	€ 36.6	€ 56.2
SO_2	€ 7.84	€ 11	€ 16.8
NO_x	€ 4.77	€ 6.67	€ 10.2
NMVOC*	€ 0.132	€ 0.185	€ 0.283
NH_3	€ 8.52	€ 11.9	€ 18.3

* Values for the chronic impacts of photochemical smog formation; cf. Section 6.5.

72 In the 2010 Handbook this conversion could not be properly performed, as too little information was available on the NEEDS modelling runs.

6.4.10 Specific values for traffic

With traffic, too, a specific emission source is involved with a different damage factor than the average for the Netherlands as a whole.
There are two reasons that traffic emissions are more harmful:
1. They occur close to the ground, so the PM is more readily inhaled.
2. They occur mainly in densely populated areas. The damage per unit emission will be greater in built-up areas, as more people are exposed there.

In the 2010 Shadow Prices Handbook we were unable to properly differentiate according to emission height and population density. In the appendices of that Handbook we referred to the HEATCO study (HEATCO, 2006), where such values were calculated. That study worked with modelling runs using the EcoSense dispersion model to calculate the health damage due to traffic and power-station emissions. One of the findings was that 1 kg $PM_{2.5}$ emitted by traffic in an urban area in the Netherlands is over 25 times more damaging than 1 kg emitted from a power station. A study by CE Delft and Vrije Universiteit (2012) made calculations using the values reported in HEATCO (2006), yielding values 1.3 to 7 times higher than the central values in the table in Section 6.4.7.
The values from that study cannot simply be adopted in this Handbook, though, because the environmental prices calculated here proceed from different assumptions on income elasticities and population composition.

Precise calculation of specific prices for Dutch traffic emissions is beyond the scope of this Handbook, where the main focus is on average prices. In contrast to the case of power-station emissions, there is no straightforward way of calculating the additional damage caused by traffic emissions. We recommend researching this issue in more detail.

Based on HEATCO (2006) it is, however, possible to approximately estimate the impact of traffic emissions, distinguishing between urban and rural areas.[73] Although HEATCO gives no precise definition of these two categories, it can safely be assumed that emissions in cities with over 500,000 inhabitants count as 'urban', yielding the rough estimates reported in Table 32.

Table 32 Approximate average damage costs for $PM_{2,5}$ from Dutch traffic emissions, differentiated by emissions location (€$_{2015}$ per kg emission)

	Lower	Central	Upper
Traffic: highly urbanized areas*	€ 383	€ 536	€ 823
Traffic: rural areas	€ 92.1	€ 129	€ 198

* Cities with over 500,000 inhabitants.

As can be seen, on this basis damage due to traffic $PM_{2.5}$ emissions in urban areas is 6-7 times greater than the national averages adopted in the 2010 Handbook. For rural areas the factor is 1.6. For the central estimates, these

[73] The following procedure was adopted. The figures reported in HEATCO (2006) for the health damage due to power-station PM_{10} emissions (at factor prices) were converted to equivalent $PM_{2.5}$ emissions using the calculated characterization factors described above and this damage compared with the damage due to $PM_{2.5}$ emissions reported in HEATCO for Dutch traffic emissions. The resultant figure was then multiplied by the results from Section 6.4.8 for the damage due to power-plant $PM_{2.5}$ emissions to yield a rough estimate of the damage due to traffic emissions. Additional research is needed to obtain more accurate information on the health impacts of traffic emissions.

values are in the ranges reported in CE Delft and Vrije Universiteit (2012). For further information and possible values for semi-urbanised areas, readers are referred to these publications.

6.5 Photochemical oxidant formation (smog)

6.5.1 Description of midpoint

Photochemical oxidant formation, otherwise known as photochemical smog or 'summer smog' formation, refers to pollution of the lower atmosphere (troposphere) with compounds like ozone (O_3), peroxyacetylnitrate (PAN), nitrogen dioxide (NO_2) and hydrogen peroxide (H_2O_2) that act as oxidizing agents (VMM, 2013d).

Ozone is the most representative as well as most important component of photochemical smog. It is a strong oxidizing agent and is hazardous to humans, plants and materials. It has an adverse impact on respiratory and cardiac functions, reduces crop yields and erodes certain materials and monuments.

Ozone is not emitted direcly, but is created in the presence of nitrogen oxides (NO_x) and non-methane volatile organic compounds (NMVOC) under the influence of sunlight. Carbon monoxide and methane also play a part in ozone formation.[74] Ozone is itself fairly unstable and reacts constantly with NO to form NO_2 and oxygen. At the same time, NO_2 and oxygen also react to form O_3 and NO. The presence of NMVOC means this equilibrium is being continually upset, however: on balance, more NO is converted to NO_2, leading to rising ozone concentrations.

The relationship between the amount of ozone formed and initial NO_x and NMVOC concentrations is by no means linear (VMM, 2013d). There is a 'worst-case' NO_x-to-NMVOC ratio at which ozone formation is highest (VMM, 2013d). In densely populated areas like Belgium and the Netherlands, where NO_x levels are relatively high, this means the most effective way to lower ozone levels is to reduce NMVOC. In the more thinly populated south and east of Europe it is the other way round. This means that in the Netherlands a reduction in NO_x does not always necessarily mean that ozone levels fall. Particularly if NO_x emissions are relatively low, a rise in NO_x emissions may even induce a drop in ozone levels (VMM, 2013d).

6.5.2 Sources

The main source of NO_x emissions are high-temperature combustion processes in vehicle and other engines, heating plant and industrial processes. NMVOC comes from a variety of sources, including fuel combustion and evaporation of industrial solvents, as well as from biogenic sources, in the form of isoprene and terpenes emitted by forests and other vegetation. CH_4 emissions derive primarily from agriculture and landfills, while CO arises through incomplete combustion of fossil fuels.

6.5.3 Impacts

Elevated tropspheric ozone levels, and particularly the peak concentrations that then often arise, cause respiratory damage. These 'ozone episodes' are more likely to occur in stagnant weather, particularly on hot, sunny days. Acute health impacts include respiratory disorders and inflammatory reactions

[74] Because of the greater transport distances involved, CO and CH_4 emissions are above all important for background ozone concentrations.

in the lungs. During these episodes, anyone – including healthy people – exerting themselves outdoors will suffer from decreased lung capacity and run the risk of inflammation of the respiratory system. The risk is greatest for those already suffering from respiratory disorders. Health effects can be avoided, or at any rate reduced, by refraining from heavy physical activity or remaining indoors.

In epidemiological studies, impacts have generally been quantified above an ozone threshold of 35 ppb or 70 µg/m^3 (known as SOM035) (NEEDS, 2007c). At higher concentrations there is considered to be a risk of acute mortality during physical exercise. As argued by WHO (2013 and 2014), there also appears to be a chronic health impact above this threshold. This is discussed further in Sections 6.5.5 and 6.5.6.

Besides health impacts, elevated ground-level ozone levels also cause damage to crops, ecosystems and certain materials. Plants take up atmospheric ozone through the stomata (microscopic openings) in their leaves. Within the plant cells, ozone damages cell membranes and causes oxidative stress. The plant responds by producing anti-oxidants (vitamins C and E) and ethylene (a plant hormone). This interrupts normal cell processes, causing crops and other plants to die back or fail to ripen, or lose their foliage early (VMM, 2013d).

The effective ozone dose received by a plant depends on the species and growing conditions. For agricultural crops, Humblot et al. (2013) have demonstrated that yields can be affected very differently depending on the crop, with wheat yields suffering but barley being positively affected.

Certain materials are also sensitive to ozone pollution. Natural rubber cracks more readily in the presence of ozone, and under the influence of ultraviolet radiation and temperature plastics, textile fibres, textile dyes and paints are also degraded.

6.5.4 Midpoint indicator unit

ReCiPe expresses impacts on this theme in kg NMVOC-equivalents.
The characterization factors reported are European averages and thus too coarse for distinguishing the background concentrations important for predicting ambient ozone levels. This is explained further in Section 6.5.8.

6.5.5 Treatment in the 2010 Handbook

In the 2010 Handbook, the impacts of pollutants causing photochemical smog were calculated using the NEEDS models (2008a), with both health impacts and crop damage included. Impacts on materials were not quantified.

6.5.6 Discussion: damage due to ozone

Based on recent toxicological and epidemiological data, the WHO (2013, 2014) is now of the opinion that ozone is more damaging than previously assumed. Besides the health risks in the form of acute mortality and morbidity, the WHO also reports an elevated risk of chronic mortality for the population as a whole.

WHO (2013) recommends that this be included when assessing the health impacts of air pollution. Because in the Netherlands it is above all NMVOC that triggers ozone episodes, we have opted to allocate chronic impacts to this group of pollutants. For a population 30 years old or older, the WHO (2013) recommends adopting a relative risk factor (RR) of 1.014 per 10 µg/m^3 in the summer months (April–September) for 8-hours concentration higher than

35ppb.[75] This value is thus about 3 times higher than the value adopted in NEEDS for the (year-round, whole population) acute health impacts of ozone. It is probably in agreement with the health impacts adopted in the IIASA-TSAP project (IIASA, 2014) for NMVOC for the theme of PM formation. Following IIASA, we have here opted to add this value to the theme of PM formation. For acute mortality no adjustment is necessary, as the Concentration-Response Function (CRF) used in NEEDS (2008a) is in line with the values recommended in WHO (2013). The CRF-value for 'limited-activity' days due to photochemical smog (morbidity) is outdated, though, and has been adjusted on the basis of Rabl et al. (2014) (see also Annex B).

6.5.7 Update: CRF for damage due to NO_2

A number of recent studies have yielded new information on the health effects of NO_2. When inhaled, nitrogen oxides are converted to nitric acid in the respiratory tract, paralyzing the cilia (hair-like structures) in these passages. This reduces the body's self-cleansing capacity and resistance to bacterial infection, among other knock-on effects (VMM, 2013a). Exposure to NO_2 can have irreversible impacts on pulmonary and respiratory functions, particularly in those already suffering from COPD and similar disorders, and also contribute to cardiovascular disease, leading to premature mortality. The REVIHAAP project (WHO, 2013) reports that since 2004 a growing number of studies have been publisherd identifying short- and long-term correlations between NO_2 and mortality and morbidity that come on top of the impacts of NO_2 on PM formation and of NO_2 on acute mortality due to ozone formation. There is thus a third category that is not associated with particulate matter formation or ozone formation and that has here been added to the theme of acidification.

At the time of the NEEDS project these impacts were not included because the team was unable to identify sufficient studies that properly quantified these epidemiological impacts (NEEDS, 2007b). Today (2016) the situation has changed and the WHO (2013) recommends adopting a higher CRF for NO_2 than was previously used. The HRAPIE experts (WHO, 2014) recommend including the long-term mortality impacts (all-cause and cardiovascular) of NO_2 and advise adopting a linear CRF for NO_2 for all-cause mortality, translating to an RR of 1.055 per 10 µg/m^3 (WHO, 2014). In this context the WHO (2014) notes that when employing this RR-value in multi-emission studies due care should be taken to avoid double-counting with respect to the impact of NO_2 on PM formation, which they state can be as much as 33%.

This is particularly relevant in comparatively polluted areas with NO_2 levels over 20 µg/m^3, which in the Netherlands means the Randstad conurbation and Eindhoven (RIVM, 2015). In the country as a whole, the average concentration in 2015 was lower: 15.3 µg/m^3 (RIVM, 2015). The WHO (2013) warns that working with country-specific averages in cost-benefit analyses may lead to an underestimate and therefore recommends making these costs explicit. This seems logical: if the average concentration in the Netherlands is below the threshold, this does not necessarily mean the average exposure of inhabitants is likewise below a threshold.

To make this double-counting explicit, we examined the contribution of NO_2 to the RR-value for PM formation. For PM, NEEDS (2007b) uses an overall RR for premature mortality of 1.06 per 10 µg/m^3. The relative contribution of NO_2 to

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75 RR is the ratio between the risk of an ailment in a group exposed to a given pollutant and the risk in an unexposed group. An RR of 20 thus means that the risk of the ailment is 20x higher in the exposed group.

PM formation can be derived from the characterization factors.
For characterizing NO_2 with respect to PM formation, ReCiPe takes a value of 0.22. This means the RR-value for PM formation via NO_2 can be set equal to 1.013 per 10 μg/m^3.[76] Assuming, in line with WHO (2014), a linear CRF for NO_2-values over the 20 μg/m^3 threshold, it can be concluded that the *additional* NO_2 RR-value must be 1.042 per 10 μg/m^3. This implies that the chronic health damage attributable to NO_2 should be a factor 3 higher than assumed in NEEDS, based on its contribution to PM formation.

To this figure an additional correction needs to be applied, because part of the Dutch population does not live in a relatively polluted area with average NO_2 levels over 20 μg/m^3. Based on RIVM (2015) we only take on board the population of the Randstad conurbation and Eindhoven as being exposed to such NO_2 levels. From CBS population data we estimate that 7 million people are subject to exceedance of this threshold as an average annual concentration, which is 40% of the Dutch population. This leads to a figure of 1.017 for the net additional RR of NO_2. This means the additional damage of NO_2 via the chronic impacts of smog formation are around 130% of those of NO_2 via PM formation.

It is important to realise, though, that this value decreases as the air becomes cleaner, as there will be increasingly fewer people living in a polluted area.

The calculated additional damage due to NO_2 has been added to the theme of photochemical oxidant formation because the impacts are similar to those resulting from ground-level ozone. In doing so, the following assumptions were made:
- Given that in the Netherlands ozone pollution is triggered mainly by NMVOC levels rather than NO_x, we assume that in this country NO_x does not translate to acute mortality due to ozone formation.
- With respect to chronic mortality, NO_2 has a direct health impact, for which we take the RR-value of 1.042 for 7 million people, which coverts to an average RR-value of 1.017 per 10 μg/m^3.
- We asssume NO_2 dispersion is the same as for NO_x.

6.5.8 Update: characterization
Characterization factors have been taken from ReCiPe (2013 version), but introducing two changes:
1. ReCiPe takes the NO_x characterization factor equal to that of NMVOC. Although this may hold as a European average, this is not the case for the Netherlands. As explained above, owing to the relatively high NO_x emissions compared with NMVOC, it is the latter that drives ozone smog formation. For acute mortality this means a characterization factor of zero.
2. On the other hand, NO_2 leads to chronic mortality, as argued above. As it is only NO_2 that leads to chronic mortality, it has been opted to ignore this value in the characterization factors, so that pollutants like SO_2 and CO are included only with respect to their contribution to acute mortality and morbidity.

The characterization factor is expressed in the same terms as in ReCiPe: kg NMVOC-equivalents, which is the same as in the 2010 Shadow Prices Handbook.

[76] This estimate is feasible because in ReCiPe PM formation is considered only in terms of its impacts on the endpoint ‘human health’.

6.5.9 Valuation in this Handbook

The endpoint impacts of photochemical oxidant formation are based on adjustment of NEEDS, supplemented by a value for NO_2, as described above. Impacts on materials like rubber have been included only in the upper value, as explained in Chapter 5. For NMVOC and NO_x the environmental prices were calculated directly. All the other environmental prices were derived from ReCiPe characterization factors. The resultant prices are shown in Table 33.

Table 33 Environmental prices for photochemical smog formation for an average Dutch emission source in 2015 (€$_{2015}$ per kg emission)

Pollutant	Lower	Central	Upper
SO_2*	€ 0.112	€ 0.157	€ 0.241
NO_x	€ 13.4	€ 18.7	€ 28.7
NMVOC	€ 1.61	€ 2.1	€ 3.15
CO*	€ 0.0736	€ 0.0958	€ 0.152
CH_4*	€ 0.0163	€ 0.0212	€ 0.0399
Formaldehyde*	€ 1.42	€ 1.84	€ 2.76

* Determined via valuation of the characterization factor.

Now that the chronic health impacts of NO_x are included, emissions of this pollutant are responsible for the majority of damage on this theme, followed by NMVOC. This also means NO_x health impacts are substantially higher than assumed in earlier studies (see for example Grinsven et al., 2013). On this theme, the environmental prices are due entirely to health impacts, except for the upper value, which also includes damage to buildings and materials.

Using the same methodology, the price for the midpoint characterization factor was also calculated for the hierarchist perspective: € 2.10/kg NMVOC-equivalent.

6.6 Acidification

6.6.1 Description of midpoint

Acidification refers to the collective impacts of airborne pollutants that are converted to sulphuric and nitric acid and deposited on soils and vegetation by means of wet or dry deposition. Unpolluted, natural clouds and rainwater have a pH (acidity) of 5.65 (VMM, 2013a), which means a lower pH is a sign of acidification. Acidifying pollutants have a long atmospheric residence time and can consequently be transported over long distances. This is particularly true of SO_2 and NO_x. This makes acidification a transboundary environmental problem requiring a coordinated international abatement strategy. In the EU the National Emission Ceilings were introduced for this purpose. Although ammonia (NH_3) also contributes to acidification, it soon disappears from the atmosphere, through dry deposition near the emission source or conversion to ammonium salts (VMM, 2013a).

6.6.2 Sources

The main source of potentially acidifying emissions are anthropogenic activities like agriculture (particularly livestock farming, NH_3) and fossil fuel consumption (SO_2, NO_x). There are also natural sources. Vulcano reuptions, for example, are accompanied by major releases of sulphur dioxide.

6.6.3 Impacts

Acidification has impacts on human health, climate change, ecosystems and buildings. In addition, NH_3 can cause stench nuisance.

Damage to human health

Sulphur dioxide acts on the mucus membranes in the mouth, nose and lungs. Its main impact is on respiratory functions (VROM, 2001). This is because when the gas comes into contact with water in the respiratory tract it is converted to sulphuric acid, which causes the air passageways to contract, leading to bronchitis and, if exposure is chronic, even to elevated mortality. Given today's low concentrations compared with the past, it is unlikely that sulphur dioxide still has any significant health effects (VMM, 2013a).

Although ammonia can in itself affect the respiratory system, this will only be the case at relatively high levels that are only likely to occur in certain working situations, most specifically on intensive livestock holdings (VROM, 2001). As the prices given in this Handbook are for an average Dutch concentration, the figures reported here cannot be applied in such situations.

Damage to ecosystems

Soils start to acidify when their acid-buffering capacity is exceeded. Soil acidification results from both anthropogenic and natural processes. Natural soil acidification can occur when an area receives more rain than it loses. In the Netherlands there is more rain than can be absorbed by natural vegetation and crops. This surplus drains away in the soil, carrying dissolved acid-buffering elements like potassium, calcium and magnesium down into deeper layers. Deposition of anthropogenic SO_2, NO_x and NH_3 can accelerate this process. Soil acidification leads to reduced plant growth and a greater incidence of crop diseases. Earthworms, moulds and other soil organisms can also be negatively impacted, with a variety of knock-on effects. When deep-burrowing earthworms disappear, for example, there is less intermixing of humus and mineral soil and reduced soil aeration (VMM, 2013a). As calcium is leached out from the soil through acidification, reduced availability of this vital element may also impact the health and survival of snails and birds.

Acidification with NH_3 and NO_x also increases soil nutrient levels, which may sometimes have a positive impact on biodiversity. In certain vulnerable ecosystems like heaths, bogs and chalk grasslands, though, such emissions lead to eutrophication and consequently damage (see the following section).

Damage to buildings

Acidifying emissions can lead to accelerated erosion of buildings and monuments, particularly those made of limestone and other calcium-rich stone or concrete.

6.6.4 Midpoint indicator unit

The three pollutants considered to have a capacity to cause acidification are SO_2, NO_2 and NH_3, each with its own 'potential acid equivalent': one mole H+ ions equals one acid-equivalent. In ReCiPe SO_2-equivalents are used as a unit, with acid-equivalents being converted to the amount of acid that can be formed from the SO_2.

6.6.5 Valuation in this Handbook

In this Handbook the environmental prices for acidification are based on the sum of impacts on agricultural crops and biodiversity reported in NEEDS (2008a), adjusted as described in Section 5.4. To this figure was added a price for damage to buildings, as described in Section 5.5. For the reasons set out in 6.6.3, no health impacts have been allocated to acidification.

Table 34 reports the environmental prices for the three main pollutants on this theme.

Table 34 **Environmental prices for atmospheric emissions contributing to acidification for an average Dutch emission source in 2015 €$_{2015}$/kg emission)**

Pollutant	Lower	Central	Upper
SO_2	€ 0.6	€ 0.933	€ 1.99
NO_x	€ 0.324	€ 1.44	€ 2.83
NH_3	€ 1.2	€ 4.63	€ 9.16

6.7 Eutrophication

6.7.1 Description of midpoint

Eutrophication refers to excessive nutrient enrichment of soil, water and air with nitrogen, phosphorus (and to a lesser extent potassium), disturbing ecological processes and natural cycles. It leads to changes in the amount of biomass and in species composition in plant and animal communities at various trophic levels. This increased nutrient availability may be due to external nutrient inputs or to changes in water or mineral balances (internal eutrophication). This increase must always be considered in relation to the 'natural' nutrient situation in the ecosystems concerned.

6.7.2 Sources

In the Netherlands It is agriculture that is the largest source of eutrophying emissions, due to fertilizer application and livestock manure. Other sources include wastewater discharge, NO_x emissions from combustion processes and dumping of effluent sludge. Eutrophying emissions can thus have an impact on air, water and soil quality.

6.7.3 Impacts

On land, eutrophication is a major threat to natural ecosystems, where interspecies competition is generally governed by limited nitrogen availability. Heaths, unimproved grassland and certain types of woodland are particularly sensitive to nitrogen eutrophication via deposition or water infiltration (VMM, 2013c). Eutrophication of surface waters can lead to algal bloom, which can in turn cause deoxygenation of the water and ultimately fish death.

6.7.4 Midpoint indicator unit

ReCiPe distinguishes eutrophication of freshwaters and marine waters. For the former kg P (phosphorus) is taken as the midpoint indicator unit, for the latter kg N (nitrogen). According to ReCiPe, in regions with a temperate climate like the Netherlands, P and N are the critical nutrients in freshwaters and marine waters, respectively. This means N-emissions to freshwaters imply no increased eutrophication burden as long as P-emissions are not substantially reduced. Conversely, P-emissions to marine waters will not lead to eutrophication unless N-emissions are substantially reduced.

6.7.5 Treatment in the 2010 Handbook

In the 2010 Handbook only eutrophying emissions to freshwater were quantified, with those to marine waters not valued. For the former category the value reported in Kuik et al. (2008) for PDF/m^2 on land was converted to m^3 water, making due allowance for the difference in species density between land and water. This type of approach can only be adopted under the assumption that a species on land represents as much 'welfare value' as an aquatic species. Although this assumption was queried at the time, there were then no better methods available.

6.7.6 Update: reappraisal of phosphates

In the 2010 Handbook the price of phosphorus was calculated from the monetary value of the ReCiPe characterization factors for the endpoint 'ecosystems'. For this Handbook this issue was re-examined. Now, the price of P has been derived directly from the ReCiPe characterization factors and the value reported in species.yr, with a conversion being made from the number of species to PDF/m^2/yr (see Section 5.4 and the annex on biodiversity valuation in the Dutch Handbook). This is identical to the treatment of ecotoxicity. Use of this method leads to an environmental price for phosphate from animal manure of € 0.16 as lower value, € 0.62 as central value and € 1.22 per kilogram phosphate as upper value.

Because ReCiPe characterization factors are based on *average European values*, the environmental price derived from them possibly leads to an underestimate of the specific, problematical Dutch situation. The abatement costs were therefore also examined. In the Netherlands there is a system of allowances in force for poultry farms designed to keep phosphate emissions within European limits, while a similar system for dairy farms is soon to be introduced (probably on 1 January, 2018). On online trading platforms, poultry allowances are currently leased for about € 2.50/year. Assuming a manure load of 0.5 kg phosphate per poultry unit, this translates to a price of € 5 per kg phosphate. The price of a dairy phosphate allowance, in which there is already some trading, is currently even higher. Here, phosphate allowances were being bought for about € 120/kg phosphate in early 2017. Assuming allowances are bought for 8-10 times the price of leasing them, this would translate to about € 12-15/kg phosphate/year. At the same time, market analysts (Jacobsen, 2016) anticipate that the actual market price may drop by a factor 3-4 once the market has settled down. We therefore judge a price of € 3-5/kg phosphate/year to be probably in line with long-term costs in the livestock sector to meet current standards.

The question is whether these costs can be used if pricing is based on an abatement-cost approach. As an alternative, one can consider the charge levied for effluent emissions to surface waters. In the Netherlands this charge stands at € 37.28 per 'pollution unit', representing annual consumption of 54.8 kg oxygen in the water. For phosphorus, discharge of 20 kg phosphorus amounts to 1 pollution unit. The shadow price of the charge is thus € 1.86 per kg phosphorus for emissions to water. This translates to € 0.61 per kg *phosphate*: precisely the central value calculated for phosphate above.

For these reasons we consider the values found to be in line with what we would expect on the basis of abatement costs. When dealing with measures in agriculture in SCBAs, for example, we nonetheless recommend that impacts on phosphate allowances be quantified, too.

6.7.7 Update: reappraisal of nitrates

To determine the environmental price for nitrogen the first method above could not be used, because ReCiPe provides no endpoint characterization for nitrogenous eutrophication of freshwater. We therefore adopted the abatement-cost method, using the charge paid for discharges to Dutch surface waters: € 37.28 per pollution unit. Adopting the same procedure as for phosphorus (see previous section), the environmental price of 1 kg N can be calculated as € 3.11.[77] This figure, taken here as the estimated environmental price of nitrate emissions to surface waters, is in line with the ReCiPe midpoint characterization factor for 1 kg N-total discharged at a non-specific location. If the nitrogen is discharged directly to sea, the environmental price is 43% higher.

6.7.8 Valuation in this Handbook

Table 35 shows the environmental prices for N and P on the theme of eutrophication.

Table 35 Environmental prices of emissions of eutrophying pollutants to air, water and soil from an average Dutch emission source(€2015/kg pollutant), with ReCiPe characterization factors in bold type

Pollutant	Theme	Compartment	Low	Central	High
NO_x	Eutrophication	Air	€ 0.121	€ 0.121	€ 0.121
N-artif. fertizer	Eutrophication	Soil	€ 0.227	€ 0.227	€ 0.227
N-manure	Eutrophication	Soil	€ 0.246	€ 0.246	€ 0.246
P-artif. fertizer	Eutrophication	Soil	€ 0.0251	€ 0.101	€ 0.196
P-manure	Eutrophication	Soil	€ 0.0237	€ 0.0952	€ 0.185
N-total*	**Eutrophication**	Water, general*	€ 3.11	€ 3.11	€ 3.11
N-total	Eutrophication	Marine waters	€ 4.45	€ 4.45	€ 4.45
P-total*	**Eutrophication**	Water, general*	€ 0.473	€ 1.9	€ 3.71
PO_4	Eutrophication	Water, general*	€ 0.156	€ 0.629	€ 1.22

* This characterization factor is based on 'water, unspecified' in ReCiPe and can be used if it is not precisely known where the pollution occurs. .

6.8 Human toxicity

6.8.1 Description of midpoint

Human toxicity covers all other pollutants that are potentially hazardous to human health, characterized primarily by their toxicity. The most important of these are heavy metals and chemical products used, among many other applications, as agricultural pesticides and flame retardants in consumer products, for example.

[77] The Dutch 'pollution unit' (veO) is defined as Q / 1000 ∗ (COD + 4.57 ∗ KjN) / 54.8, where Q = stream flow in m^3/a, COD = chemical oxygen demand in mg/l, and KjN = amount of Kjeldahl-nitrogen bound in ammonia or organic matter. The formula thus converts effluent concentrations to kg COD and N-Kjeldahl. The factor 1,000 converts grams to kilos, as COD and N-Kjeldahl concentrations are expressed in mg/l, or gram/m^3. From this formula it follows that 1 kg N = (4.57/54.8) VeO. Multiplying this ratio by the wastewater levy yields a charge of € 3.11 per kg N.

Their toxic impacts fall into five categories:
- acutely poisonous substances;
- substances that can cause cancer (carcionogenicity);
- substances that can cause genetic mutations (mutagenicity);
- substances that can impact reproduction (teratogenicity);
- substances that can irritate and damage skin, eyes or the respiratory tract.

6.8.2 Substances and sources

The main substances with impacts on the theme 'human toxicity' are heavy metals, chlorinated hydrocarbons, pesticides and other biocides and a wide range of specific chemicals used primarily in consumer and other products.

The most important sources of heavy metals are emissions from industrial production plants, from mining and oil refining. These pollutants are discharged in low concentrations in effluents or released as trace elements during combustion, roasting and incineration of fossil fuels, ores and wastes and subsequently dispersed via the atmosphere. In addition, heavy metals are contained in numerous products, including paints, phones, building materials and fertilizers. In the waste phase or via leaching they can then end up in the environment.

In the case of chlorinated hydrocarbons the main pollution source is waste incineration. These compounds are not only inhaled, but can also be ingested in food. Pesticides and other crop protection agents escape to air, soils and water during and after farm application and may remain on edible crops as residue.

6.8.3 Impacts

The toxic impacts of heavy metals have been researched in greatest detail. The most toxic of these are arsenic, cadmium, chromium, copper, mercury, lead, nickel, platinum and zinc. Besides being carcinogens, they can also have specific physiological impacts, including damage to the liver (copper), brain and cognitive learning abilities (lead) and nervous system (mercury). Heavy metals can impact human health through direct inhalation or ingestion via the food chain following uptake by plants and animals. Heavy metals in the soil can also infiltrate groundwater.

A growing body of data is also available on the toxicity of countless chemicals used in a wide range of consumer products, packaging materials and countless other materials. With many of these chemicals the damage they cause only manifests itself with the passage of time, particularly when it comes to non-acute health impacts like damage to organs, metabolism and reproduction. It was only in the 1970s, for example, that the toxic impacts of dioxins, a particularly hazardous class of chlorinated hydrocarbons, became apparent, following a series of incidents in chemical plants in Seveso and Amsterdam, among other places, where workers came to suffer acute and chronic health problems after exposure to high dioxin concentrations. Later that decade it was realised that dioxins are also toxic in lower concentrations and slowly accumulate in the bodies of both humans and animals, being soluble in fatty tissue. Later still it became clear that the class of chlorinated hydrocarbons to which dioxins belong contains many other compounds that are also toxic, including such widely used chemicals as polychlorinated biphenyls (PCBs).

The use of pesticides and other biocides also has human health impacts, which have been unravelled by reseachers in growing detail over the past few decades. They are used to protect farm crops against pests, diseases and weeds, as well as elsewhere.

Numerous consumer products also contain chemicals with potential health impacts, such as bromine-containing flame retardants, softening agents in plastics and additives in products like printing inks. Many of these products at first appeared to pose no health threat to humans, but as more data became available on leaching, intake via food or skin contact and potential for long-term damage, their toxic properties came to the fore.

6.8.4 Midpoint indicator unit

ReCiPe (Goedkoop, et al., 2013) uses kg 1,4-dichlorobenzene as the midpoint indicator unit for human toxicity, the same as for ecotoxicity. 1,4-dichlorobenzene is a chlorinated hydrocarbon that is poorly degradable and therefore accumulates in the environment, posing a hazard above all to aquatic organisms. The chemical is used in such products as mothballs and (formerly) toilet fresheners. Its inhalation can lead to dizziness, fatigue and anemia and, over time, to liver and kidney complaints and it may also be slightly carcinogenic.

In ReCiPe the characterization factor is used to express the relative toxicity of other pollutants. Its value differs substantially, depending on whether the individualist or hierarchist perspective is adopted. In the former case a conservative position is adopted with respect to the burden of proof as to suspected toxicological impacts. Impacts recorded solely in animals are not included, for example, nor heavy-metal dispersal via the soil or uptake in cereals and other food crops. In the hierarchist perspective these impacts are included (see also Annex A).

6.8.5 Treatment in the 2010 Handbook

In the 2010 Handbook toxicity was valued based on the NEEDS damage costs for atmospheric emissions of six metals, formaldehyde and dioxin. Using the ReCiPe characterization factors (hierarchist perspective) these damage costs were converted to a weighted average for 1,4-dichlorobenzene, with weighting according to the relative impact of the metals, formaldehyde and dioxin in the Netherlands based on 2006 emissions.

6.8.6 Update: CRF-values

The damage estimates in NEEDS (2008a) and the 2010 Shadow Prices Handbook (CE Delft, 2010) have been reappraised to assess their current validity, principally because the NEEDS values for these toxic chemicals were very low compared with the results of later studies. The damage costs of toxic metals have recently been researched by Rabl, Spadaro and Holland (2014) and by Nedellec and Rabl (2016) as part of the AMESTIS project. The latter study assessed the damage costs of atmospheric emissions of toxic metals by European coal-fired power plants by reviewing the epidemiological literature and concludes that the estimated damage is far higher than the values used in the 2010 Handbook. Comparison with a direct valuation based on DALYs using characterization models like ReCiPe and ILCD also indicates that the values in NEEDS (2008a) are probably too low. Finally, the doctoral study by Frantke (2012) provides evidence that the damage-cost estimates in the 2010 Handbook for the toxic impacts of pesticides are probably underestimates (see also Annex B).

For the present Handbook we therefore examined several toxicity routes, ultimately opting to disaggregate impacts into two factors:
- impacts on human health (morbidity and premature mortality);
- impacts on IQ.

For the first effect we continued to base the toxicity of dioxins on NEEDS (2008a), but that of atmospheric emissions of the heavy metals arsenic, cadmium, lead and mercury on a combination of four studies (detailed in Annex B), including the model employed in Rabl and Nedellec (2016).[78] Dividing this total damage by the emissions expressed in terms of kg 1,4-dichlorobenzene (converted using the ReCiPe data, Individualist perspective) an estimate was obtained for the emission of 1 kg 1,4-dichlorobenzene. This value was then compared with other estimates, including the estimates for the health damage due to pesticide use reported in Fantke (2012). This showed that our method yields results broadly similar to Fantke's values for the impacts of the herbicide amitrol.

For emissions of arsenic, lead and mercury we furthermore quantified impacts on IQ based on the model of Rabl and Nedellec (2016). In doing so we assigned a value € 17,500 per IQ-point (2015 prices), based on valuation of associated income loss.

6.8.7 Update: characterization factors

In contrast to the 2010 Shadow Prices Handbook, in this Handbook we have in principle consistently used characterization factors based on the Individualist perspective. In the case of human toxicity, however, many of these factors are over 100 times higher in the hierarchist perspective, for two main reasons:

– in the individualist perspective there is a greater burden of proof when it comes to (suspected) human toxic impacts (see Annex A);
– in the individualist perspective environmental dispersal of toxic substances is modelled less comprehensively, with uptake of toxic heavy metals in food crops not included, for example.

For this Handbook we sought to adopt a perspective in line with that adopted for the other themes. To this end we took citations in WHO studies as evidence of toxicological impacts, but, following the Individualist characterization perspective, taking only IARC Categories 1 and 2 as toxicological proof and not Categories 3 and 4 (see Annex A). For uptake via food crops the individualist perspective is incomplete, however, as NEEDS (NEEDS, 2008) and more recently Nedellec and Rabl (2016) show that this is an important route for the health impacts of heavy-metal emissions. For heavy metals we therefore opted to base ourselves on the characterization factors for the hierarchist perspective, corrected for the difference in burden of proof for toxicity.[79]

It was decided to use these higher characterization factors for heavy metals solely in the upper-value estimates. The lower value is thus still based on the individualist perspective. For the central value we took the average of these two characterization factors. For heavy-metal emissions to soil, in particular, this leads to substantial differences between the upper and lower values.

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78 Their model is based on more extensive dispersion routes of toxic substances in food chains than previously quantified. Using a 3% discount rate and the VOLY and QALY values adopted here (see Section 5.3) we calculated the total damage due to emissions of these four metals in the Netherlands in 2015.

79 In doing so, we used substance-specific correction factors based on the model of Nedellec and Rabl (2016).

6.8.8 Valuation in this Handbook

Table 36 reports the environmental prices for atmospheric emissions of various toxic substances.

Table 36 **Environmental prices for atmospheric emissions of toxic substances on the midpoint human toxicity for an average Dutch emission source in 2015 (€$_{2015}$/kg pollutant), with ReCiPe characterization factor in bold type**

Pollutant	Lower	Central	Upper
Cadmium	€ 798	€ 1,159	€ 1,831
Arsenic	€ 703	€ 1,033	€ 1,228
Lead	€ 3,967	€ 5,908	€ 6,596
Mercury	€ 24,770	€ 34,480	€ 53,630
CFC-11*	€ 10.2	€ 13.9	€ 21.5
Nickel*	€ 75	€ 133	€ 225
Chromium*	€ 0.152	€ 0.531	€ 1.02
Formaldehyde*	€ 19.3	€ 26.6	€ 41.2
Dioxin	€ 49,020,000	€ 67,060,000	€ 103,600,000
Midpoint: 1,4 DB-equivalent	€ 0.157	€ 0.214	€ 0.331

Note: * Environmental price for this emission has been calculated from the applied characterisation factor (see also Section 6.8.7).

It should be emphasized that the environmental prices for human toxicity reported here are more uncertain than for other themes. In studies with a specific focus on toxicity we do not therefore recommend using these prices, but rather a decicated toxicity analysis. In a future edition of this Handbook a more extensive analysis can hopefully be carried out encompassing the latest research on the dispersion, accumulation and health impacts of toxic substances.

The midpoint price, which can be used as a weighting factor, is € 0.158 for 1 kg 1,4-dichlorobenzene. This price is based on the hierarchist characterization perspective and is lower than the central value in Table 36. This is because the characterization factors from the hierarchist perspective are many times higher than those from the individualist perspective. As the environmental price reported here is the weighted average of results from several studies (see Annex B), its value becomes lower as the characterization factor becomes higher.

6.9 Ecotoxicity

6.9.1 Description of midpoint

Ecotoxicity is the impact of toxic substances not considered elsewhere on non-human organisms in ecosystems, to the extent that non-target organisms are exposed. The main agents involved are agricultural pesticides, which are designed specifically to exterminate organisms deemed to pose a threat to crops and livestock. In addition, though, pesticides are also widely used by households as well as government agencies. Almost 80% of herbicides do not reach their intended target (VMM, 2013g).

A major difference from human toxicity is that in LCA and other such analyses individual organisms are generally ignored entirely when it comes to ecotoxicity (with the exception of certain large mammals like wolves), with consideration given only to the species and population levels (National Research Council, 2014).

6.9.2 Sources and substances

VMM (2013g) distinguishes two kinds of pesticides: crop protection agents and biocides. The first category can be subdivided into insecticides, herbicides, fungicides, bactericides, molluscicides, rodenticides, nematicides (to combat nematode worms) and acariciden (for ticks and mites). These compounds are used mainly by farmers, in allotments and in public spaces.

Biocides are pesticides used in non-agricultural settings, except in applications similar to farm use. On land, examples include hospital disinfectants, wood preservatives and agents used for household pest control. At sea, shipping vessels use anti-fouling agents to avoid hulls becoming overgrown with marine organisms like algae and polyps. These agents can impact shellfish and other non-target organisms. Tributyltin (TBT), the compound that was most frequently used for this purpose, was banned worldwide in 2008, although it is still causing damage to certain European ecosystems (Tornero & Hanke, 2016). Since the TBT ban, copper salts have become the most common alternative. While these are less toxic than TBT, the resultant elevated copper levels in seawater may still pose a risk to marine life (Tornero & Hanke, 2016). These copper-based anti-fouling agents are also often supplemented with biocide 'boosters' like Irgarol (Cybutryne), which is toxic to micro-organisms.

Heavy metals are dispersed through the natural environment as a result of effluent discharges from foundries, fossil-fuel emissions, mining activities and waste incineration (VMM, 2013h). The following metals can have a toxic impact on ecosystems: arsenic (aquatic organisms), cadmium (food chains), chromium (fish), copper (plants), mercury (fish) and lead (aquatic organisms) (VMM, 2013h).

6.9.3 Impacts

Crop protection agents impact on ecosystems through their toxicity to non-target organisms, pollution of surface water, grondwater, aquatic sediments and soils, and bio-accumulation (accumulation in food chains). As pesticide residues often become dispersed throughout the environment, these side-effects occur not only close to the original source but also over far greater distances. The persistence of impacts varies from a few days to several years. The longer a toxic substance remains active, the greater the risk of bio-accumulation. In such cases a low concentration in the aquatic environment may ultimately lead to far higher concentrations in animals further up the food chain. As a result, there may also be knock-on effects on public health (VMM, 2013g) which are treated further under the theme 'human toxicity'.

For non-target invertebrates, exposure to crop protection agents can lead to mortality, a reduced lifespan, changes in growth and fertility rates, changes in sex ratios and a wide range of behavioural changes. The recent decline in populations of honeybees and other pollinating insects may be due in part to pesticides. In vertebrates, certain crop protection agents can lead to hormonal disbalance, as has been observed with reptiles, birds and mammals exposed to organochlorine and organophosphorus pesticides. Pest control may cause mammal mortality, particularly when organochlorine pesticides are involved. These pesticides are also associated with increased mortality and morbidity among marine mammals. Perinatal (just before or after birth) or neonatal (after birth) exposure to pesticides like aldrin, atrazine, chlordane and dieldrin can cause anomalous sexual development in mammals. Bird exposure to pesticides has been extensively studied. In the past, seeds treated with DDT (an organochlorine pesticide) led to the poisoning of millions of birds, with populations of prey animals also being decimated by these kinds of pesticides (VMM, 2013g).

The main impact of the biocide TBT was its effect on the endocrine system of shellfish (Tornero & Hanke, 2016). Copper is an essential trace element for many organisms, but is toxic at high concentrations. It damages the immune system of molluscs and interferes with coral reproduction. The booster biocide Irgarol disturbs photosynthesis and is highly toxic to autotrophic organisms like cyanobacteria and dinoflagelatte symbionts in coral reefs. Heavy metals burden food chains (arsenic, cadmium, chromium, mercury, lead), limit plant growth (copper) and poison aquatic biota (lead) and certain land animals like sheep (copper) (VMM, 2013g).

In our treatment of ecotoxicity all these pollutants have been included. Using ReCiPe data (Goedkoop, et al., 2013) the impacts of over 1,000 chemicals discharged to water or dispersed in soils via waste streams and their ecotoxicity impacts have been included.

6.9.4 Midpoint indicator unit

In ReCiPe (Goedkoop, et al., 2013) the ecotoxicity of a substance is expressed as toxicity relative to 1,4-dicholorobenzene discharged to the marine environment. This is the same indicator as used for human toxicity. 1,4-dichlorobenzene is a poorly degradable chlorinated hydrocarbon that consequently accumulates in the environment, with impacts mainly on aquatic organisms. This explains why the damage for this pollutant, in Euros, on the theme of ecotoxicity is greater than on the theme of human toxicity.

In ReCiPe the characterization factor is used to express the relative toxicity of different pollutants. For some substances this factor differs substantially according to whether the individualist or hierarchist perspective is adopted. This is because the impacts of metals occurring naturally in ocean water are not quantified in the individualist perspective, but are in the hierarchist perspective (Annex A).

No values for the ecotoxicity of pollutants (known as 'Hazard Property 14') have yet been set in the European Union. The European Commission has initiated a project on how this impact is to be quantified.

6.9.5 Update: characterization factors

For the characterization factors on this theme we have based ourselves on the individualist perspective in ReCiPe (Goedkoop, et al., 2013). Similarly to the discussion on human toxicity, in ReCiPe the differences between the individualist and hierarchist perspective derive from the choice of studies used for assessing ecotoxicity and the environmental compartments modelled.

Here, we have opted to work with the studies associated with the individualist perspective, but adopting the hierarchist perspective for a limited number of heavy metals (cobalt, copper, manganese, molybdenum and zinc) for estimating an upper value for marine ecotoxicity. For these pollutants we have taken the average of these two characterization factors as the central value.

6.9.6 Valuation in this Handbook

On this theme, monetary valuation is based on ReCiPe endpoint characterization. As explained in Section 5.3, to this end a relationship was established between the value of biodiversity from the economic literature and the unit of the ReCiPe characterization factor (Goedkoop, et al., 2013).

This led to the values reported in Table 37 for ecotoxicity in the various environmerntal compartments, expressed in terms of the compound used for characterization: 1,4-dichlorobenzene (DB).

Table 37 **Environmental prices for ecotoxicity for average Dutch emissions in 2015 (€$_{2015}$ per kg pollutant)**

Midpoint	Lower	Central	Upper	Unit
Ecotoxicity, terrrestrial	€ 2.21	€ 8.89	€ 17.3	€/kg 1,4 DB-eq.
Ecotoxicity, freshwater	€ 0.00917	€ 0.0369	€ 0.0719	€/kg 1,4 DB-eq.
Ecotoxicity, marine	€ 0.00188	€ 0.00756	€ 0.0147	€/kg 1,4 DB-eq.

It should be emphasized that the environmental prices given for ecotoxicity, like those for human toxicity, involve greater uncertainty than those for the other themes. We therefore advise against using them in studies concerned explicitly with ecotoxicity. It is then preferable to perform a dedicated assessment of the impacts of the toxic substances on the particular ecosystems involved and value these using specific values for these particular ecosystems.

6.10 Ionizing radiation

6.10.1 Description of midpoint

The subatomic particles and electromagnetic waves produced by certain materials are sufficiently energetic to eject electrons from other atoms or molecules, a process known as ionization. If living tissue is exposed to ionizing radiation this can cause damage to DNA, leading to apoptosis (cell death) or genetic mutation. Ultimately this may lead to the development of cancer or genetic defects that are passed on to subsequent generations. The ionizing radiation emitted by radionuclides is measured in Becquerels (Bq), expressing the number of radioactive decays per second.

6.10.2 Sources

We are all exposed to natural ionizing radiation. The two main natural sources are cosmic radiation and radioactive minerals occurring naturally in the Earth's crust. One major source of natural exposure is radon, a gas emitted from soils that can build up in crawl spaces in homes and may be responsible for between 100 and 1,200 additional cases of lung cancer per year in the Netherlands, according to the Dutch Health Council (Gezondheidsraad, 2000).

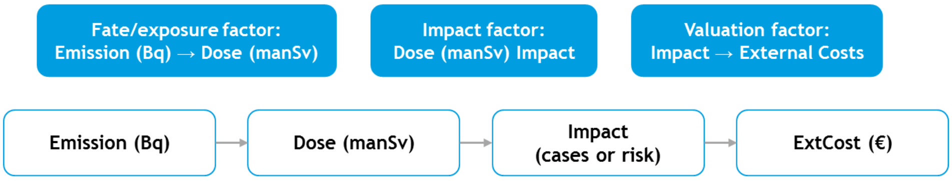
Human activities involving use of radiation (X-ray machines) and radionuclides also expose us to ionizing radiation over and above the natural background. Medical use of radiation is the largest - and growing - anthropogenic source of exposure (UNSCEAR, 2000). In addition, environment pollution with radioactive waste from nuclear power facilities and weapons testing are an important source of exposure worldwide. In some parts of the world, production of fissile material for military ends has left behind vast amounts of radioactive waste. Nuclear power stations, reprocessing plants and other nuclear facilities release radioactive substances to the environment on an everyday basis and produce large volumes of radioactive waste requiring long-term storage. In addition, radioactive materials are emitted in minor amounts from fossil-fuel combustion and the use of certain materials in industry and agriculture.

6.10.3 Impacts

The health impacts of exposure to ionizing radiation manifest themselves as fatal and non-fatal cancers and genetic damage. Human exposure as a result of anthropogenic emissions depends on the medium to which the radionuclide is emitted: surface water or the atmosphere.

6.10.4 Midpoint indicator unit

To calculate the external costs van radionuclide emissions NEEDS (2008a) uses the following simplified model:



NEEDS calculated exposure factors using the method set out by UNSCEAR (1993, 2000), in which a radionuclide emission (in Bq) is converted to a 'radiation dose equivalent' at the population level, expressed in man-Sieverts (manSv). This dose equivalent is obtained by multiplying the amount of absorbed radiation (in J/kg) by a 'quality factor' that depends on the type of radiation involved (e.g. photons vs. alpha particles) and a factor for the exposed part of the body and the duration and intensity of the radiation.

6.10.5 Treatment in the 2010 Handbook

In the 2010 Handbook, characterization and valuation were based on NEEDS (2008a), with impacts valued as the number of DALYs per cancer. This gives the number of lost life years (YOLL) due to premature mortality as a result of cancer, taken as 15.95, in line with NEEDS (2008a). A Cost of Illness (COI) was also added amounting to € 480,000. This yielded a value of € 1.12 million per fatal cancer, with non-fatal cancers only entailing the COI. The external costs per unit emission were calculated by multiplying the disease-specific values by the projected incidence of radiation-induced disease, which depends on the the radiation type.

6.10.6 Valuation in this Handbook

In this new Handbook the NEEDS value of $€_{2000}$/kBq has been recalculated using a characterization factor for Uranium-235 and a correction for inflation (using HCIP) to express prices in $€_{2015}$/kg U235-eq. In addition, a high and a low scenario were created using the high and a low value for VOLY. The upper VOLY-value adopted was € 110,000 and the lower value € 55,000. Allowance was also made for population growth. In NEEDS (2008a) the value for YOLL was reduced from 15.95 to 13, based on Humbert et al. (2012). For non-fatal cancers COI was assigned a lower value of € 420,000 (2015 prices).[80] In contrast to the 2010 Handbook, characterization is now based on the individualist perspective, to include discounting more explicitly. As there is copious evidence for the carcinogenic properties of ionizing radiation, there is also less difference between the hierarchist and individualist perspectives in ReCiPe (Goedkoop, et al., 2013).

Table 38 reports prices for radionuclides with relatively high radiological impacts.

[80] In doing so, the decrease in YOLL was translated to a proportional decrease in COI.

Table 38 **Environmental prices for ionizing radiation for an average Dutch emission source in 2015 (€$_{2015}$ per kBq U235-eq.)**

Pollutant	Onder	Centraal	Boven
Aerosols, radioactive, unspecified	€ 0.00013	€ 0.00020	€ 0.00026
Carbon-14	€ 0.00190	€ 0.00295	€ 0.00383
Cesium-137	€ 0.00189	€ 0.00293	€ 0.00381
Hydrogen-3, Tritium	€ 0.00095	€ 0.00147	€ 0.00191
Iodine-129	€ 0.00077	€ 0.00119	€ 0.00154
Iodine-133	€ 0.00107	€ 0.00166	€ 0.00216
Krypton-85	€ 0.00524	€ 0.00813	€ 0.01055
Radon-222	€ 0.00002	€ 0.00002	€ 0.00003
Thorium-230	€ 0.00228	€ 0.00353	€ 0.00459
Uranium-234	€ 0.00028	€ 0.00044	€ 0.00057
Uranium-235	€ 0.00106	€ 0.00165	€ 0.00214
Uranium-238	€ 0.00295	€ 0.00457	€ 0.00594
Lead-210	€ 0.00228	€ 0.00353	€ 0.00459
Polonium-210	€ 0.00228	€ 0.00353	€ 0.00459
Radium-226	€ 0.00227	€ 0.00353	€ 0.00458

6.11 Noise

6.11.1 Description of midpoint and sources

Ambient noise is a major environmental problem with a range of impacts on people’s well-being and health as well as on the natural world. As traffic is the main source, most valuation studies are concerned with this type of noise (EY, 2016; (Navrud, 2002)with only limited research on noise from other sources like building sites, industry, public events and neighbours. Given this lack of data, this Handbook focuses solely on valuation of traffic noise, making a distinction between road, rail and air traffic.

6.11.2 Impacts

Five deleterious impacts of ambient noise can be distinguished (Defra, 2014):

– **Nuisance:** noise can cause people nuisance in many ways, discouraging or preventing them from performing certain activities, for example, and leading to a range of negative emotions like irritation, disappointment, dissatisfaction, a feeling of helplessness or depression (WHO, 2011). It can also lead to stress-related psychological and physical complaints such as fatigue, stress and abdominal pains. In some studies all these impacts are regarded as health impacts (e.g. Defra, 2014; IGCB, 2010)[81], while in others an explicit distinction is made between nuisance and health impacts (e.g.Bristow, et al., 2015; Nelson, 2008).
– **Health impacts:** there is a growing body of evidence that noise can impact human health in a variety of ways. WHO (2011) distinguishes the following:
  - *Cardiovascular disease*: ambient noise can contribute to various forms of cardiac disease (including acute heart failure) and elevated blood pressure (hypertension). Noise-related high blood pressure can also lead to strokes and dementia (Harding, et al., 2011). These health impacts have been correlated mainly with traffic noise.
  - *Sleep nuisance*: there is copious scientific evidence for sleep (quality) being adversely affected by ambient noise. Besides the direct impacts (stress responses, time slept, number of nighttime waking episodes)

81 This is in line with the broad definition of health employed by the WHO: “a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity” (WHO, 2011).

there are also impacts the following day(s) (e.g. fatique, reduced cognitive performance) and long-term impacts (chronic sleep deprivation).

- *Reduced cognitive performance*: particularly for aircraft noise, there is good evidence that this can affect children's and adolescents' school performance and memory. Exposure to such noise at crucial educational moments may influence children's cognitieve development, with potentially life-long consequences.
- *Tinnitus*: exposure to especially high noise levels can cause tinnitus or 'phantom noise', a condition in which one hears a hissing, whistling, buzzing or 'beeping' sound in one or both ears in the absence of an external noise source. This impact is scarcely treated in the literature.
- *Damaged hearing*: there is as yet little scientific evidence for ambient noise causing chronic hearing damage.

– **Productivity loss:** noise can reduce workplace performance for a variety of reasons, including concentration problems, fatigue due to noise-related sleep problems, noise-related learning issues in children and adolescents, leading to a lower education level, and absence from work due to noise-related health complaints (TRL, 2011). These impacts are scarcely treated in the literature. There is, moreover, a risk of some of the above health impacts (like sleep nuisance) being double-counted. These impacts are therefore not included separately here.
– **Nuisance in quiet areas:** Anastasopoulos et al. (2011) have pointed out that ambient noise may reduce people's enjoyment of the benefits of quiet areas like city parks and woods, implying a loss of economic welfare. These costs of ambient noise have barely been researched, however, and have consequently been ignored here.
– **Ecosystem impacts:** there is growing evidence that ambient noise has deleterious impacts on wild animals, by disturbing breeding patterns, for example (Dutilleux, 2012). Here too this is only fledgling research, though, with no reliable monetary values available. Again, these impacts have been ignored here.

Based on the above review we conclude that it is only for the first two categories, nuisance and health impacts, that there is sufficient scientific evidence for deriving cost factors. The economic valuation of these two impacts is therefore discussed in greater detail in the next two sections.

6.11.3 Noise indicators

The unit most commonly used for measuring noise nuisance is the A-weighted decibel dB(A). The decibel is a measure of noise level, and 'A-weighting' is applied to correct for the sensitivity of the human ear to noise pitch.

Besides intensity and pitch, the time and duration of the noise are also important factors, and these are also included in the noise indicator adopted here. There are numerous such indicators, differing in how they account for the influence of the various factors. In this Handbook we use the unit Lden ('den' referring to day, evening, night), the current legal standard for measuring traffic noise in the Netherlands. Lden is calculated by establishing 'equivalent' noise levels in the day (07:00-19:00 h), evening (19:00-23:00 h) and night (23:00-07:00 h), raising evening and night levels by 5 and 10 dB(A), respectively, then calculating the 24-hour average. This indicator thus takes evening and nighttime noise to be more of a nuisance than daytime noise.

6.11.4 Monetary valuation

In this section we present our main conclusions and recommendations with respect to the valuation of noise. In the Dutch Handbook an Annex is included which explains issues in a bit more detail.

Methodology in the 2010 Handbook

In the 2010 Shadow Prices Handbook valuation of traffic noise was based on HEATCO (2006), with a distinction made between road, rail and air traffic, as people experience these three kinds of noise differently. Miedema and Oudshoorn (2001) report how people consider aviation noise to be 'worse' than road-traffic noise, and rail-traffic noise to be less of a nuisance than that of road traffic (the reason why HEATCO (2006) applied a 5 dB 'rail bonus').

In their damage costs for noise, HEATCO (2006) includes only the costs of nuisance and health impacts, in the absence of reliable cost estimates for other deleterious impacts.

New findings

Since the 2010 Handbook there have been several major new findings with respect to noise valuation:

- The marginal costs of noise nuisance (in € per dB) increase with rising noise level: if the level is already high, an extra dB leads to more additional costs than at lower levels. This effect has been demonstrated in a range of studies, including Bristow et al. (2015), Udo et al. (2006), Theebe (2004) and WHO (2011).
- There is evidence of noise nuisance occurring even at noise levels below 50 dB (WHO, 2011). It is unclear, however, whether the results of most valuation studies are also applicable to these lower levels.
- There is new epidemiological literature (WHO, 2011) on the health impacts of noise, including analyses of the risk and magnitude of various forms of cardiovascular disease (including strokes and dementia due to elevated blood pressure).
- There is evidence of health impacts occurring even at noise levels below 70 dB (WHO, 2011; Defra, 2014).
- There is evidence of health impacts increasing with rising noise levels (WHO, 2011; Defra, 2014). In other words, the marginal costs of health impacts rise with noise levels.

Valuation in this Handbook

With respect to the nuisance caused by noise, following an analysis of the available literature (cf. Section 5.6), in this Handbook we have opted to base our prices on the results of Bristow et al. (2015). These results are in turn based on a recent, extensive meta-analysis of stated-preference studies on noise-nuisance valuation. These values are also reasonably in line with the average values of noise nuisance found in revealed-preference studies[82]. In an annex to the Dutch version of this Handbook an illustrative case study is used to calculate that these studies yield an average WTP of € 75 per person per dB per annum.

Table 39 **Environmental prices for noise nuisance: central values, with lower and upper values bracketed (€2015 per dB (L_{den}) per person per annum)**

Noise level	Nuisance	Health	Total
Road traffic			
50-54 dB(A)	22 (18-25)	4 (3-6)	26 (21-31)
55-59 dB(A)	43 (36-50)	5 (4-8)	48 (40-58)
60-64 dB(A)	43 (36-50)	9 (7-14)	52 (43-64)
65-69 dB(A)	83 (69-95)	14 (11-22)	97 (80-117)
70-74 dB(A)	83 (69-95)	19 (15-30)	103 (84-125)
75-79 dB(A)	83 (69-95)	25 (20-39)	108 (89-134)
>= 80 dB(A)	83 (69-95)	27 (22-43)	111 (91-138)
Rail traffic			
50-54 dB(A)	0	4 (3-7)	4 (3-7)
55-59 dB(A)	22 (18-25)	5 (4-8)	27 (22-33)
60-64 dB(A)	43 (36-50)	9 (7-14)	52 (43-64)
65-69 dB(A)	43 (36-50)	14 (11-22)	57 (47-72)
70-74 dB(A)	83 (69-95)	20 (15-30)	103 (84-125)
75-79 dB(A)	83 (69-95)	25 (20-39)	108 (89-134)
>= 80 dB(A)	83 (69-95)	28 (22-43)	111 (91-138)
Aviation			
50-54 dB(A)	52 (43-60)	8 (6-12)	60 (49-72)
55-59 dB(A)	103 (86-119)	9 (7-14)	112 (93-133)
60-64 dB(A)	103 (86-119)	13 (10-21)	127 (96-140)
65-69 dB(A)	196 (164-227)	18 (14-28)	214 (178-255)
70-74 dB(A)	196 (164-227)	23 (18-37)	220 (182-264)
75-79 dB(A)	196 (164-227)	29 (23-46)	226 (187-273)
>= 80 dB(A)	196 (164-227)	32 (25-50)	228 (189-277)

As Table 39 makes clear, the recommended values for noise nuisance increase with rising noise levels. This is in line with current scientific understanding of this issue, as well as with the values officially prescribed in certain other EU countries (Denmark, UK, Sweden).

For the health impacts of noise we used the results of Defra (2014), translated to the Dutch situation. These results are based directly on recent epidemiological findings published by the WHO (2011). In contrast to the 2010 Shadow Prices Handbook, the new values now also factor in health impacts occurring below 70 dB. The ranges in the health-impact values given reflect the range adopted in this Handbook for valuing DALYs.

In valuing these health impacts we have ignored the costs of sleep nuisance, to avoid overlap with the costs of noise nuisance itself. Like HEATCO (2006) we assume that people are aware of the sleep-nuisance impacts of noise and that the associated costs are therefore included in the WTP-values for overall nuisance.

82 These are mainly studies using hedonic pricing, with the willingness-to-pay for noise abatement being derived from variation in house prices.

As a threshold value for both health and nuisance impacts we recommend adopting 50 dB(A), in line with the recommendations in the 2010 Handbook. Although nuisance is also known to occur at lower noise levels (WHO, 2011; EEA, 2010) it is insufficiently clear to what extent valuation studies also deliver reliable values at these levels, too.

Finally, Table 39 shows that the environmental prices for noise differ according to the type of traffic involved, with the highest prices holding for aircraft noise and the lowest for rail noise. This differentiation is in line with the acoustic literature, which provides a great deal of evidence that people deem aircraft noise 'worse' than road-traffic noise, and rail noise least 'bad'.

6.12 Land use

6.12.1 Description of midpoint and impacts

Large-scale agriculture and residential and industrial development all have impacts on the theme of land use (change). If these impacts harm nature and biodiversity, this means a loss of economic welfare. By examining ecosystem services as a function of land use, a value can be assigned to land use (change).

6.12.2 Treatment in the 2010 Handbook

In 2010 the damage costs of land use were determined using the approach adopted in NEEDS for valuing ecosystems: the Potentially Disappeared Fraction of species (see Section 5.3). Data on the relative species diversity of various kinds of land use were taken from ReCiPe (Goedkoop, et al., 2013), which distinguishes 18 types of land use. These biodiversity figures are averages for Europe. For valuing land use, the average value of the PDF reported in Kuik et al. (2008) was used: € 0.47 per PDF.m^2 (2004 prices). By multiplying the impacts of land-use change on the PDF (the characterization factor, in the hierarchist perspective) by the PDF-value, the external costs associated with each type of land use were calculated. These were weighted for the Dutch situation according to the distribution of land use in this country, using CBS statistics to yield average values for Dutch land use.

Land use also affects crop revenues, as it pushes up land prices. As this impact probably counts as a pecuniary externality (and thus only a transfer of welfare), this was not included in the 2010 Handbook.

In the 2010 Shadow Prices Handbook the value provided for PDF.m^2 was erroneously set equal to the value for PDF.m^2.year. As a result, land use featured as a very dominant factor in LCA calculations performed using the old Handbook. In practice this led to land use not being adopted as a midpoint when calculating shadow prices.

6.12.3 Valuation in this Handbook

In this Handbook we employ the same method as in the 2010 Handbook, using ReCiPe characterization factors for the hierarchist perspective to derive values for species diversity for each type of land use.[83]

[83] As the individualist perspective is based solely on temporarily reversible impacts, the hierarchist perspective was deemed to be more in line with Dutch practice.

Compared with the 2010 Handbook, valuation has here been adjusted on four points:
- The value per PDF has been adjusted (see Section 5.3.5) to provide a specifically Dutch value for biodiversity.
- The value for land use is no longer reported in m^2, but in m^2/year, in line with the units used in LCAs. In converting the costs per PDF to costs per PDF per year, we based ourselves on the restoration costs NEEDS (2006). As argued in Section 5.3, the minimum restoration costs can be taken as € 0.63/PDF/m^2. Applying a 3% p.a. discount rate over 50 years, this gives a figure of € 0.025 per PDF/m^2/year for these costs.[84] The same procedure was applied for the lower and upper values of the PDF-value from Section 5.3.5.
- Using the Eurostat LUCAS database, a conversion was carried out to arrive at a value specifically for the Netherlands. This allowed more land-use categories to be distinguished, so categorization is now more refined than in the 2010 Handbook.
- In setting the price for the midpoint weighting factor (for use in LCAs) it was decided not to base valuation on specific Dutch data, but to use global data based on the ReCiPe endpoint characterization factors for agricultural and urban land occupation. This was done because in LCAs the category 'land use' also covers land use outside the Netherlands. Biodiversity was therefore calculated in relation to land use at the global level using a simplified model (discussed in an Annex E to the Dutch version). In doing so, specifically Dutch values were used, however.[85]

The PDF-values of impacts of land-use changes remain the same; see (Goedkoop, et al., 2013). Table 40 reports the adjusted values for land use in the Netherlands.

Table 40 Average values for land use in the Netherlands for use as external costs (€$_{2015}$ per m^2 per annum)

	Dutch percentage	Central value	Upper value	Lower value
Intensive crops/weeds	28%	0.033	0.064	0.008
Monoculture broadleaf, mixed forest and woodland	12%	0.016	0.030	0.004
Coniferous forest	2%	0.022	0.044	0.006
Mixed plantations	2%	0.027	0.053	0.007
Extensively-managed grassland	6%	0.017	0.033	0.004
Intensive fertile grassland	36%	0.023	0.044	0.006
Continuous urban	13%	0.035	0.067	0.009
Dutch average	**100%**	**0.026**	**0.050**	**0.007**

Source: Eurostat, own calculations.

On this basis a figure of € 0.026/m^2/yr has been taken as the central value, € 0.007/m^2/yr as the lower value and € 0.050/m^2/yr as the upper value.

84 In Kuik et al. (2008) costs are discounted at 5% p.a. over 50 years. Here we have therefore taken the same period, but adjusted the discount rate to that used in this Handbook.

85 To our mind this choice is most in line with the hierarchist perspective adopted in LCAs.

To calculate the midpoint characterization factor we proceeded from the central value and translated this into a implicit value per species. By multiplying this value by the ReCiPe endpoint factor (for “Occupation, unknown”) in species.yr/m^2a we obtained a value in line with PDF-valuation according to the hierarchist perspective. Table 41 reports the environmental prices for the midpoint characterization factor for global land occupation, valued as if that land use were in the Netherlands. This value is the same for for Agricultural Land Occupation and Urban Land Occupation (ALO/ULO) – as the characterization factor in ReCiPe in terms of species.yr is similar for these two. The resulting environmental price is equivalent to € 0.037/m^2/yr (rounded) for the hierarchist characterization perspective.

Table 41 **Average environmental prices for land use for use as midpoint characterization factor (€$_{2015}$ per m^2 per annum)**

Midpoint unit	€/m^{2*}	€/m^2yr
Agricultural Land Occupation	0.957	**€ 0.037**
Urban Land Occupation	0.957	**€ 0.037**

Note: * €/m^2 gives the undiscounted value of land-use change over a 50-year period. This is the value presented in the 2010 Shadow Prices Handbook as land-use factor. For use in LCAs this factor must be converted to an annual figure, however. In line with NEEDS (2006) the monetary value has been discounted at 3% p.a. over 50 years.

6.12.4 Discussion: implications of SCBA Guidelines for Nature

In the Netherlands, Arcadis and CE Delft are currently working on SCBA Guidelines for Nature, scheduled for completion in late 2017. A methodology for calculating and monetizing the welfare gains ensuing from nature conservation will be treated there. In Social Cost-benefit Analyses it is therefore not recommended to work with the values for land use elaborated in the present Handbook. For the same reason, these values are not cited the SCBA Guidelines on the Environment either.

For use by industry and in LCAs this proviso does not apply, though, so in these contexts the prices reported here can (provisionally) be used in calculations.

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Annex A Characterization

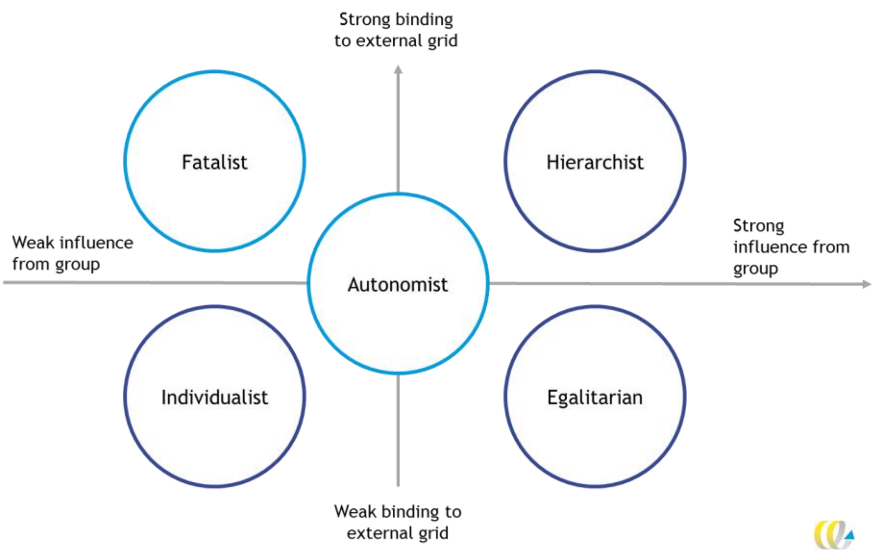
A.1 Introduction

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

A.2 Cultural theory as the basis of characterization perspectives

Thompson et al. distinguish five basic value systems by looking at the strength of the relations people have with their group and the degree to which an individual's life is circumscribed by externally imposed prescriptions (their so-called 'grid'); see Figure 12.

Figure 12 The five basic value systems according to Thompson et al.



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

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

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

A.3 The three perspectives

In ReCiPe, characterization of environmental impacts is based on scenarios representing the other three perspectives, which can be summarized as follows:
1. Individualist. In this scenario only proven cause-effect relationships are included and a short-term perspective is adopted. For human-health issues age-weighting is applied. There is technological optimism with regard to human adaptation.
2. Hierarchist. Included in this scenario are facts backed up by scientific and political bodies. The hierarchist attitude is common in the scientific community and among policy-makers.
3. Egalitarian. This scenario uses the precautionary principle and the very long-term perspective.

This Handbook generally follows the ReCiPe hierarchist perspective, though the individualist perspective has sometimes been adopted.

Table 42 indicates how these perspectives have been elaborated in ReCiPe and shows which choices we have made with respect to characterisation in this handbook. Note that for climate change we chose to base the characterisation on the IPCC 2013 values for a 100-year time-frame.

Table 42 The perspectives from ReCiPe and the chosen perspective in this Handbook

Environmental theme	Principals ReCiPe Individualistic Perspective	Principals ReCiPe Hierarchistic Perspective	Choices in this handbook for characterisation
Climate Change	N.A.	N.A.	IPCC (2013)
Ozone Depletion	Only effects of UV on skin cancer are taken into account: basal cell carcinoma (BCC), squamous cell carcinoma (SCC) and cutaneous melanoma (CM).	In addition to effects of UV on skin cancer, also additional effects such as cataract.	Individualistic perspective
Smog-formation and particulate matter	Using non-discounted impacts for a period of 20 year.	Non-discounted impacts for a period of 100 year	Individualistic perspective
Acidification	Using non-discounted impacts for a period of 20 year.	Non-discounted impacts for a period of 100 year	Individualistic perspective
Human toxicity	For metals only distribution via air and drinking water, no spreading via soil and uptake food crops. Accumulation in the environment for 100 years. Only strong scientific evidence of carcinogenic effects on humans: no evidence in animal testing. Included studies:IARC-category 1, 2A and 2B	The distribution of metals in food crops is also taken into account by emissions. Accumulation in the environment permanently. Also included evidence of tests on animals. Studies considered IARC-category 1, 2A and 2B and 3.	At the lower value the individualistic perspective and top value the hierarchical perspective. Central value is the average of the lower and upper value
Eco-toxicity	No dispersion to oceans of Cobalt, Copper, Manganese, Molybdenum and Zinc.	All substances are included	As with Human Toxicity

Environmental theme	Principals ReCiPe Individualistic Perspective	Principals ReCiPe Hierarchistic Perspective	Choices in this handbook for characterisation
Land-use	Only temporary effects on ecosystems, full recovery to natural values in 5-100 years (depending on the type of ecosystem).	More permanent damage to ecosystems, in 100 years not all damage has been restored.	Hierarchistic perspective

More information on the choices made can be found in Chapter 6.

A.4 Comparison ReCiPe and ILCD with respect to charactisation

IThe ReCiPe method was used in this Handbook on environmental prices. In addition to ReCiPe, other characterization methods exist, such as ILCD and PEF. Here we briefly indicate the differences.

Table 43 summarizes the units used in ReCiPe and ILCD characterization at midpoint level. The PEF methodology is taken directly from the ILCD characterization.

Table 43 Units in the various characterisation methods

Environmental effect	ReCiPe (2013)	ILCD/PEF
Climate change	kg CO_2-eq.	kg CO_2-eq.
Ozone depletion	kg CFC-11-eq.	kg CFC-11-eq.
Acidification	kg SO_2-eq.	mol H+-eq.
Freshwater eutrophication	kg P-eq.	kg P-eq.
Marine eutrophication	kg N-eq.	kg N-eq.
Terrestrial eutrophication		molc N-eq.
Eutrophication		
Human toxicity	kg 1,4 DB-eq.	
Non-cancer effects		CTUh
Cancer effects		CTUh
Photochemical oxidant formation	kg NMVOC	kg NMVOC-eq.
Particulate matter formation	kg PM_{10}-eq.	kg $PM_{2,5}$-eq.
Terrestrial ecotoxicity	kg 1,4 DB-eq.	
Freshwater ecotoxicity	kg 1,4 DB-eq.	CTUe
Marine ecotoxicity	kg 1,4 DB-eq.	
Ionising radiation	kBq U235-eq.	
Human health		kBq U235-eq.
Ecosystems		CTUe
Agricultural land occupation/land use	m^2a	kg-C-deficit
Urban land occupation	m^2a	
Natural land transformation	m^2	
Water depletion	m^3	m^3 water-eq.
Metal depletion	kg Fe-eq.	
Fossil depletion	kg oil-eq.	
Mineral, fossil & ren resource depletion		kg Sb-eq.
Abiotic depletion (fuel & non-fuel)		

At first glance, therefore, it appears that there may be major differences between ReCiPe on the one hand and ILCD on the other. However, a closer study of these differences showed that ReCiPe and ILCD use the same methods and literature on most themes.
For most midpoints the method of characterisation is quite similar between both methods. However, more fundamental differences exists for some midpoints, especially for human toxicity, ecotoxxicity and land use. In the Dutch version of the Handbook Environmental Prices, more information about the differences can be found.

Annex B Impact pathway modelling

B.1 Introduction

The damage calculated on the environmental themes acidification, photochemical smog formation and particulate matter formation were determined directly via an adaptation of the NEEDS modeling from 2008 (NEEDS, 2008a). In this appendix we explain which assumptions are behind the original NEEDS project and which adjustments have been made.

B.2 NEEDS project (2008)

Between 1991 and 2008, various large European research projects attempted to estimate the external costs of energy production and other activities. These research projects were called ExternE, CASES, MethodEx and NEEDS.[86] These projects, financed from European research funds, involved more than 50 research teams from more than 20 countries. The NEEDS (New Energy Externalities Developments for Sustainability) project is the most recent research carried out in this context.

To calculate environmental prices, we use an Excel application developed by the University of Stuttgart in the framework of NEEDS/CASES. This Excel application works with input of the ecological-economic model of EcoSense.This Excel tool was subsequently adjusted to more recent information concerning: concentrations response functions, affected population and background concentration of emissions.

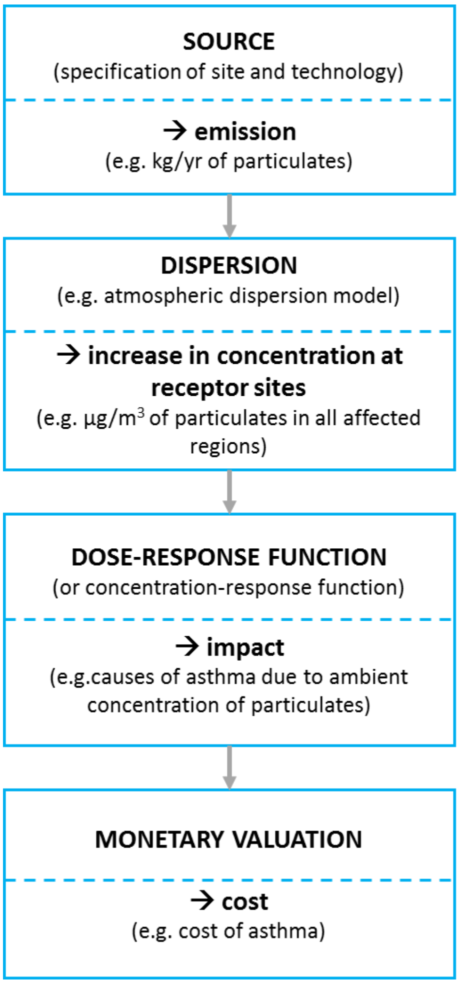
B.2.1 Impact pathway approach

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 (see Figure 13).

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

Figure 13 Impact pathway approach



The various steps will be described below.

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).

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.

B.2.2 Concentration-response functions human health from classical pollutants

This section deals with the damages due to photochemical oxidant formation and particulate matter formation, established using the Impact Pathway Approach and is largely based on estimates from the NEEDS project. Description of the methodology draws extensively on NEEDS (2008a).

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

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

It should be noted that according to the recommendations of the NEEDS project experts, human health impacts have only been defined for particulate matter (primary as well as secondary) and ozone.[87] Impacts due to emissions of SO_2, NO_x and NH_3 are factored in after chemical transformation with reactants leading to an increase of concentration of secondary particulate matter (SIA, secondary inorganic aerosols). In the scientific community there is considerable debate on whether SIA has the same toxicity as primary particles, with no consensus yet emerging. In the NEEDS project it was therefore assumed that the damage due to SIA should be the same as for primary particles.

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

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

In the Environmental Prices Handbook, this approach has been re-examined and where possible adjusted on the basis of:
– changing population composition;
– update of some CRF functions on the basis of new findings in the literature;
– new valuation of loss of working days;
– new valuation of the the VOLY.

Table 44 gives in the green shaded cells which information has been adjusted from the original NEEDS modelling.

Table 44 Overview of concentration response functions and used valuations for particulate matter formation and acute impacts from ozone. Valuations according to the LOWER estimates in this handbook

Core Endpoints	pollutant	risk group (RG)	RGF value	Age Group (AG)	AGF value	Population growth	CRF [1/ug/m3]	phys. Impact per person per ug per m3 [1/ug/m3]	unit	monet. Val. Per case or per YOLL [Euro]	External costs per person per ug per m3 [1/ug/m3]
Primary and SIA < 2.5 i.e. Particle < 2,5 um											
Life expectancy reduction - YOLLchronic	PM2.5	all	1	Total	1	1.04	6.51E-04	6.77E-04	YOLL	40000	2.71E+01
netto Restricted activity days (netRADs)	PM2.5	all	1	MIX	1	1.04	9.59E-03	9.97E-03	days	130	1.30E+00
Work loss days (WLD)	PM2.5	all	1	Beroepsb	0.423	1.04	2.07E-02	9.11E-03	days	119	1.08E+00
Minor restricted activity days (MRAD)	PM2.5	all	1	Adults_18	0.623	1.04	5.77E-02	3.74E-02	days	38	1.42E+00
Primary and SIA < 10 i.e. Particle < 10 um											
Increased mortality risk (infants)	PM10	infants	0.0019	Total	0.010	1.04	4.00E-03	8.12E-08	cases	3000000	2.44E-01
New cases of chronic bronchitis	PM10	all	1	Adults_27	0.707	1.04	2.65E-05	1.95E-05	cases	200000	3.90E+00
respiratory hospital admissions	PM10	all	1	Total	1.000	1.04	7.03E-06	7.31E-06	cases	2360	1.73E-02
cardiac hospital admissions	PM10	all	1	Total	1.000	1.04	4.34E-06	4.51E-06	cases	2360	1.07E-02
medication use/bronchodilator use	PM10	Children meeting PEACE criteria - EU average	0.2	Children_	0.105	1.04	1.80E-02	3.92E-04	cases	1.18	4.62E-04
medication use/bronchodilator use	PM10	asthmatic	0.045	Adults_20	0.791	1.04	9.12E-02	3.38E-03	cases	1.18	3.98E-03
lower respiratory symptoms (adult)	PM10	symptoma	0.3	Adults	0.812	1.04	1.30E-01	3.29E-02	days	38	1.25E+00
lower respiratory symptoms (child)	PM10	all	1	Children_	0.105	1.04	1.96E-01	2.13E-02	days	38	8.11E-01
Ozone [ug/m3] - from SOMO35											
Increased mortality risk	SOMO35	baseline_	0.0099	Total (YOL	1.000	1.04	3.00E-04	3.09E-06	YOLL	40000	1.24E-01
respiratory hospital admissions	SOMO35	all	1	Elderly_65	0.189	1.04	1.25E-05	2.45E-06	cases	2360	5.79E-03
MRAD	SOMO35	all	1	Adults_18	0.623	1.04	1.54E-02	9.98E-03	days	38	3.79E-01
medication use/bronchodilator use	SOMO35	asthmatic	0.045	Adults_20	0.791	1.04	7.30E-02	2.70E-03	cases	1.18	3.19E-03
LRS excluding cough	SOMO35	all	1	Children_	0.105	1.04	1.60E-02	1.74E-03	days	38	6.62E-02
Cough days	SOMO35	all	1	Children_	0.105	1.04	9.30E-02	1.01E-02	days	38	3.85E-01

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

Source: Ajusted from NEEDS (2008a), based on NEEDS (2007b) with own recalculations of the green cells.

B.2.3 Impacts on agricultural crops

Within the NEEDS project, the impacts of air pollution on crops have been divided into impact of SO_2, acidification of agricultural soils due to NH_3, SO_2 and NO_x, impact of ozone and effects of nitrogen deposition (NEEDS, 2008a).

Impact of SO_2
The CRF function for SO_2 assumes that yields will increase with SO_2 concentrations from 0 to 6.8 ppb (part per billion on a molecular level) and decline thereafter. The function is used to quantify changes in crop yield for wheat, barley, potato, sugar beet and oats and is defined as:

$$y = 0.74 \cdot [SO_2] - 0.055 \cdot [SO_2]^2 \qquad \text{for } 0 < [SO_2] < 13.6 \text{ ppb}$$
$$y = -0.69 \cdot [SO_2] + 9.35 \qquad \text{for } [SO_2] > 13.6 \text{ ppb}$$

with y = relative yield change; and
$[SO_2]$ = SO_2 concentration in ppb.

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

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

with dL = additional lime requirement in kg/year;
A = agricultural area in ha; and
dDA = annual acid deposition in meq/m^2/year.

Impact of ozone
For the assessment of ozone impacts, a linear relationship between yield loss and the AOT 40 value (Accumulated Ozone concentration above a Threshold of 40 ppbV) calculated for the crop growing season (May to June) has been assumed. The relative yield change is then calculated using the following equation together with the sensitivity factors given in Table 45:

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

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

Table 45 **Sensitivity factors for different crop species**

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

Fertilisation effecs from nitrogen deposition

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

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

with dF = reduction in fertiliser requirement in kg/year;
A = agricultural area in km^2; and
dDN = annual nitrogen deposition in $meq/m^2/year$.

B.2.4 Monetary valuation of crop losses

Crop losses are assessed in monetary terms using the prices of the crops damaged by air pollution. Table 46 summarises the prices per tonne used within the NEEDS project for assessing crop damage due to air pollution.

Table 46 Updated prices of major crops used in this project (€/t)

	Updated price per tonne
Sunflower	335.00
Wheat	179.00
Potato	214.00
Rice	305.00
Rye	142.00
Oats	145.00
Tobacco	3,508.00
Barley	153.00
Sugar beet	34.00

Source: FAOStat. Prices reflect price levels in the Netherlands.

It should be noted that prices have fluctuated significantly over the last 15 years.

B.3 Impacts on biodiversity

Within ExternE, the environmental impact of air pollution on biodiversity has been estimated for emissions of SO_2, NO_x and NH_3. This impact is associated with acidification and eutrophication of soils. An approach using the measure 'potentially disappeared fraction' (PDF), i.e. biodiversity losses due to acidification and eutrophication, was used (NEEDS, 2008a).

Acidification is caused mainly by emissions of sulphur oxide (SO_x), nitrogen oxides (NO_x) and ammonia (NH_3) and the attendant deposition of acidifying substances like H_2SO_4 as well as a range of sulphates. Eutrophication due to airborne pollutants is due mainly to NO_x and NH_3.

B.3.1 Concentration-response function

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

In EcoSense the following information is used to model the loss of biodiversity due to SO_2, NO_x and NH_3 emissions, using the following data:
- Values of PDF per deposition of N and S on natural soils are taken from NEEDS (2006); see Table 47.
- Depositions per 50 x 50 km^2 grid cell are available from regional dispersion modelling.
- In addition, for each grid cell the share of natural soil is available.
- Finally, a (country-dependent) 'pressure index' is used to account for differences in soil sensitivity.

Table 47 PDF per deposition of N and S on natural soil

Air pollutant	Deposition increase in kg/m^2 * year on natural soil (10 mol/ha)	Average PDF of natural land for the Netherlands	PDF * m^2 * year per kg deposition
Reference Value (Background Level)	--	0.746429	--
SO_x	6.4 * 10-5	0.74654	1.73
NO_x	4.6 * 10-5	0.746867	9.52
NH_3	1.7 * 10-5	0.74687	25.94

Source: (NEEDS, 2008a).

B.4 Impacts of human toxicity

B.4.1 Approach followed in this study

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

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

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

The concentration response functions for these substances were taken from the literature and valued with a VOLY value of € 40,000 and an IQ point value

of € 8,600. This approach was also followed in the Shadow Prices Handbook 2010. Based on a review of the more recent literature, it was decided that:
– on the one hand, the CRF functions from NEEDS (2008a) appear to be an underestimation compared to more recent toxicity information, as shown by the characterization models ReCiPe and ILCD;
– on the other hand, the appreciation for the loss of IQ points seems to be higher in more recent research.

Based on this insight, we have estimated the valuation of human toxicity as an average of four methods:
– The original NEEDS approach that has been followed in the Handbook Shadow Prices (CE Delft, 2010).
– ReCiPe (Goedkoop, et al., 2013) that is a characterization model and relates the midpoint effect to the endpoint effect by means of characterization factors and emission response functions. The indicator for human toxicity is expressed in kg 1,4-DB-eq. The effect on health of the population is shown in DALYs. This can then be valued by means of a monetary value per DALY where we assumed that 1 VOLY is 1 DALY.
We calculated here with a VOLY of € 55,000 to assure the comparison with the Handbook Shadow Prices.
– The International Reference Life Cycle Data System (ILCD) method that provides the characterization factors for metals for human toxicity. However, this method does not provide a monetary valuation at the endpoint level, but characterization factors in the unit CTUh/kg. This can then be valued using a DALY factor per CTUh and a monetary valuation per DALY. Also here we calculated the impacts with a DALY of € 55,000.
– A recent study by Nedellec and Rabl (2016) that has provided a spreadsheet model calculating the dispersion and valuation of heavy metals.

Summarizing thess approaches gave the following results:

Table 48 **Valuation of heavy metals according to four characterisation studies using a VOLY/DALY of € 55,000 in €/kg emission to air**

	ILCD-midpoint	ReCiPe-endpoint	Handbook Shadow Prices 2010	Nedellec en Rabl, 2016
Cd	€ 3.408	€ 1.384	€ 127	€ 61.376
As	€ 1.439	€ 1.972	€ 811	€ 2.229
Ni	€ 37	€ 17	€ 5,37	N.v.t.
Pb	€ 694	€ 607	€ 408	€ 8.267

This shows that the valuations in the Shadow Prices Handbook for the Netherlands are considerably lower than the average values in Europe according to ILCD and ReCiPe. Nedellec and Rabl again produce much higher values, in particular for cadmium and lead. The reason for these differences is uncertain and is probably related to the more complete uptake of the toxic substances in the food described in detail by Nedellec and Rabl. As a result, the toxic effects are also much greater than if only the inhalation of the substances is examined via the air. But the exact causes of the differences are difficult to trace.

In the Environmental Price Handbook we decided to take an average valuation for the above substances from the table. We have added the appreciation for Dioxin and mercury from NEEDS. Based on these six substances, we have explained the emissions in accordance with the methodology in Annex D, weighed on the basis of the harmfulness of the actual emissions in the Netherlands in 2015 in order to arrive at a weighted valuation for the midpoint characterization factor.

We would like to emphasize that the appreciation for human toxicity is very uncertain and should be subjected to a further investigation in future versions of this Handbook in order to arrive at a more precise calculation.

B.4.2 Toxicity and perspectives in ReCiPe

In ReCiPe (Goedkoop, et al., 2013) deals with toxicity differently in the Individualist worldview than in the Hierarchical world view.
There are two types of differences:
- difference in the distribution of impacts taken into account;
- difference in burden of proof of toxicity.

In the Hierarchic Perspective, there is a broader spread of impacts included, such as damage caused by the uptake of heavy metals in the food chain, which are not included in the Individualistic Perspective. Actually, the bulk of the damage associated with heavy metal pollution to soil seems to be related to the potential spread of emissions to groundwater and the food chain.

This fear is also expressed in the social aversion in living on former, non-cleaned garbage dumps. That is why we opt for basing the impacts of Human Toxicity of heavy metals on the hierarchist perspective.

In addition, there is an important difference in the burden of proof. Scientific studies on toxicology of materials are divided into four IARC categories according to the WHO:

Table 49 Classification of substances to toxicity according to WHO

Group	Classificatie WHO
Group 1	Carcinogenic to humans
Group 2A	Probably carcinogenic to humans
Group 2B	Possibly carcinogenic to humans
Group 3	Not classifiable as to its carcinogenicity to humans
Group 4	Probably not carcinogenic to humans

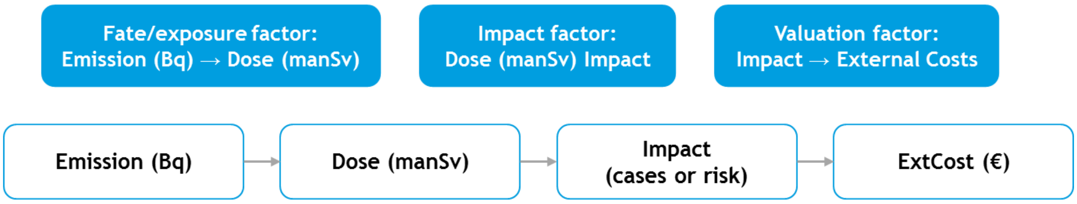
In total, the WHO has categorized nearly 1000 substances (or groups of substances) in this way. ReCiPe Individualist Perspective takes Category 1 and 2 in consideration, but in Hierarchistic Perspective, Category 3 is added plus other studies and substances that have not been approved by the WHO.

If we compare this with the other themes, where we follow the WHO strictly (both at High and Low, with acidification, smog formation, particulate matter formation and ozone depletion), the Individualistic perspective is most consistent with the general principles of our handbook and with those of the other themes. Therefore we followed here the Individualistic Perspective for all other substances that characterize on Human Toxicity.

B.5 Ionising radiation

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

Figure 14 Scheme of assessment of exposure, physical impact and external costs due to release of radionuclides (from Needs 2008)



B.5.1 Health-related effects

The fate and exposure factors used in NEEDS have been calculated using the methodology set out in UNSCEAR (1993, 2000), whereby radionuclide emissions (in Bq) are related to the 'equivalent radiation dose' at the population level. This equivalent dose is expressed in man-Sievert (manSV), which is calculated by multiplying the amount of absorbed radiation (in J/kg) by a 'quality factor' that depends on radiation type (e.g. photons vs. alpha particles) and a factor that takes into account the exposed part of the body, the duration and level of irradiation. The resulting combined fate and exposure factors in manSv/PetaBequerel (PBq; 10^{15} Becquerel) are listed Table 50. As can be seen from these data, the human radiation exposure associated with emissions depends on the medium to which the radionuclide was emitted.

Table 50 Endpoint damages to human health caused by radionuclide emissions

Radionuclide	Emitted to	Dose (manSv/PBq)	Fatal cancers (cases/PBq)	Non-fatal cancers (cases/PBq)	Hereditary defects (cases/PBq)
Aerosols, radioactive, unspecified	Air	2,000	100	240	20
Carbon-14	Air	92,270	4,614	11,072	923
Carbon-14	Water	1,000	50	120	10
Cesium-137	Air	7,400	370	888	74
Cesium-137	Water	98	4.90	11.76	0.98
Hydrogen-3, Tritium	Air	4.1	0.21	0.49	0.04
Hydrogen-3, Tritium	Water	0.85	0.04	0,10	0.01
Iodine-129	Air	64,000	3,200	7,680	640
Iodine-131	Air	20,300	1,015	2,436	203
Iodine-131	Water	63,438	3,172	7,613	634
Iodine-133	Air	0	0	0	0
Iodine-133	Water	0	0	0	0

Radionuclide	Emitted to	Dose (manSv/PBq)	Fatal cancers (cases/PBq)	Non-fatal cancers (cases/PBq)	Hereditary defects (cases/PBq)
Iodine-135	Air	0	0	0	0
Krypton-85	Air	0.214	0.01	0.03	0.00
Krypton-85	Water	0	0	0	0
Krypton-85m	Air	0	0	0	0
Noble gases, radioactive, unspecified	Air	0.43	0.02	0.05	0.00
Radon-222	Air	2,5	0.13	0.30	0.03
Thorium-230	Air	30,000	1,500	3,600	300
Thorium-230	Water	0	0	0	0
Uranium-234	Air	8,000	400	960	80
Uranium-234	Water	198	9.90	23.75	1.98
Uranium-235	Air	0	0	0	0
Uranium-235	Water	0	0	0	0
Uranium-238	Air	7,000	350	840	70
Uranium-238	Water	1,963	98	236	20
Strontium-90	Water	4.7	0.24	0.56	0.05
Rubidium-106	Water	3.3	0.17	0.40	0.03
Lead-210	Air	1,000	50	120	10
Polonium-210	Air	1,000	50	120	10
Radium-226	Air	600	30	72	6

Source: CASES, 2008.

The health impacts of radiation absorption may manifest themselves in the form of fatal and non-fatal cancers and hereditary defects. It is estimated that each manSv equivalent radiation dose leads to 0.05 cases of fatal cancers, 0.12 cases of non-fatal cancers and 0.01 cases of hereditary defects (see NEEDS, 2008a). For each of these, the expected number of cases per unit emission are shown in Table 51, columns 3-5, for each of the relevant radionuclides.

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

The external cost per unit emission was calculated by multiplying the disease-specific valuations by the expected number of diseases. As described in Annex A, in NEEDS an uplift factor is applied to account for the positive income elasticities of demand (1.7 % until 2030, 0.85% thereafter), and a discount factor of 3% until 2030 and 2% thereafter. Importantly, the radiation emitted by a certain substance changes over time, depending on its half-life. This should be corrected for in the uplift and discount factors, which in NEEDS was only done for Rn-222, H-3 and C-14 (the most prevalent emissions associated with nuclear fuel cycles). The resulting Net Present Values of emissions in the year 2008 are listed in Table 51.

Table 51 External costs of radionuclide emissions

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

Source: (NEEDS, 2008a).

B.5.2 Nature- and capital-related effects

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

Annex C Specific themes

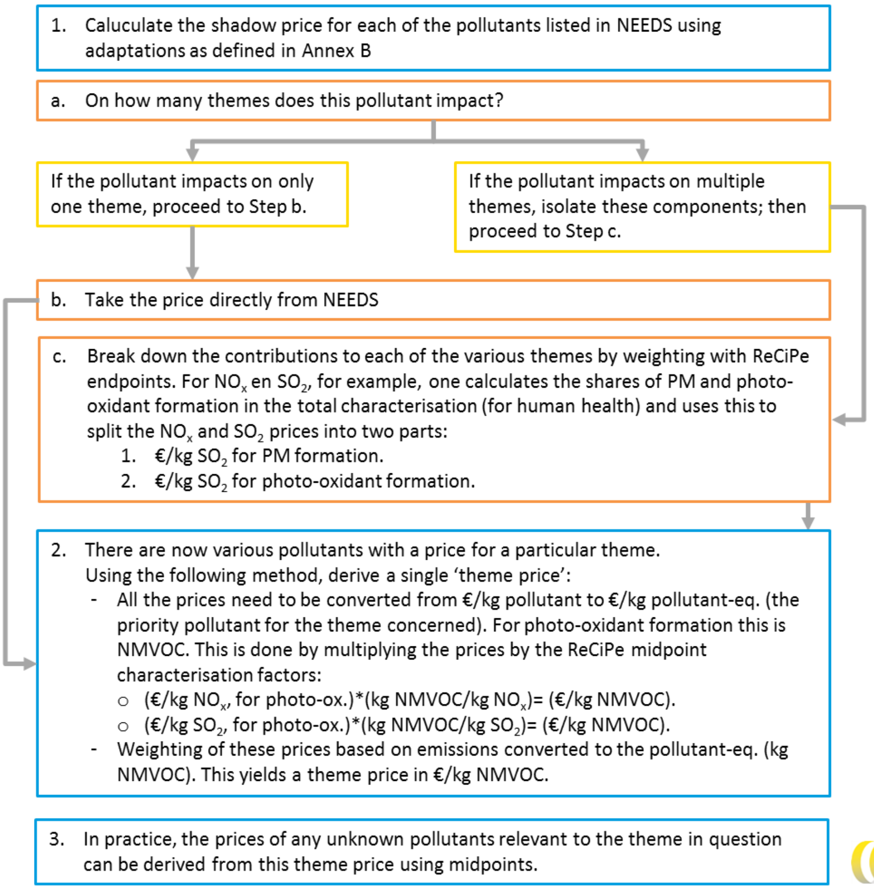
C.1 Allocation and development of weighting sets

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

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

Problem B (implicit characterisation) was resolved by taking a weighted average of the damages occurring in the Netherlands. To this end we multiplied all the damages calculated in Annex C by the respective emissions occurring in the Netherlands and then divided the figure obtained by the emissions expressed in the relevant ReCiPe midpoint characterisation factor.
In resolving Problem A (multiple environmental impacts) use was made of the ReCiPe endpoints. This is because these express, according to ReCiPe, how much each pollutant contributes to a particular endpoint. These were used to allocate the contributions of each pollutant across the midpoints.
The procedure adopted is shown in Figure 15.

Figure 15 Schematic representation of procedure adopted to calculate economic damage costs



C.2 Treatment of uncertainty[88]

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

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

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

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

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

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

Uncertainty in classical pollutants

Rabl and Spadaro (1999) have examined the uncertainties of each step of the impact pathway analysis for classical pollutants to estimate the uncertainties associated with the various components of the calculation. Table 52 reports their assumptions for the component uncertainties and the results for the damage costs for mortality. Because mortality accounts for over two-thirds of the damage costs of many pollutants, the uncertainty associated with this endpoint can be viewed as a good estimate for that associated with the sum total of impacts.

[88] This description is based on NEEDS and most of this annex has earlier been published in the Handbook Shadow Prices 2010 (CE Delft, 2010).

Table 52 Uncertainty of damage cost estimates per kg of pollutant for mortality

	Log-normal?	σ_{gi} PM	$ln(\sigma_{gi})^2$	σ_{gi} SO_2 via sulphates	$ln(\sigma_{gi})^2$	σ_{gi} NO_x via nitrates	$ln(\sigma_{gi})^2$
Exposure calculation							
Dispersion	*yes*	1.5	*0.164*	1.7	*0.282*	1.7	*0.282*
Chemical transformation	*yes*	1	*0*	1.2	*0.033*	1.4	*0.113*
Background emissions	*no*	1	*0*	1.05	*0.002*	1.15	*0.02*
Total σ_g for exposure		1.5	*0.16*	1.76	*0.32*	1.9	*0.41*
ERF							
Relative risk	*no*	1.5	*0.164*	1.5	*0.164*	1.5	*0.164*
Toxicity of PM components	*?*	1.5	*0.164*	2	*0.48*	2	*0.48*
YOLL, given relative risk	*no?*	1.3	*0.069*	1.3	*0.069*	1.3	*0.069*
Total σ_g for ERF		1.88	*0.4*	2.33	*0.71*	2.33	*0.71*
Monetary valuation							
Value of YOLL (VOLY)	*yes*	2	*0.48*	2	*0.48*	2	*0.48*
Total σ_g		2.78	*1.04*	3.42	*1.51*	3.55	*1.61*

Source: NEEDS, 2008b.

Table 52 shows sample calculations of geometric standard deviation σ_g. The relative contributions of the σ_{gi} to the total can be seen in the column $ln(\sigma_{gi})^2$.

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

Uncertainty of trace pollutants

Using the same assumption about lognormality of damage distribution, NEEDS (2008b) calculate geometric means for the trace pollutants. The results are shown in Table 53.

Table 53 Summary of geometric standard deviations for the damage costs

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

This also relates to an observation made in Annex B that the valuation of human toxicity is much more uncertain than substances in other environmental themes.

Uncertainty related to transfer to other regions

NEEDS (2008b) have also examined the uncertainties associated with the transfer of the individual components of the damage costs calculation (emissions, atmospheric modeling, dose-response functions and monetary valuation) to regions other than the EU. The results are expressed in terms of geometric standard deviations and listed in Table 54. To obtain the total uncertainty for a given region, the figures relevant to that region need to be combined with the geometric standard deviations of the damage costs for the EU15

Table 54 **Geometric standard deviations associated with the transfer of components of the damage cost calculation**

Component of calculation	σ_g
Transfer of technologies	
CO_2 emissions with CCS	1.3
Other emissions	[a]
Atmospheric modelling	
If no data for effective deposition velocity v_{dep}	1.5
If no data for stack height	2
If no data for local population or no data for wind	3
Background concentrations for sulphate and nitrate formation	1.2
Background concentrations for O_3 formation due to NO_x	2
Background concentrations for O_3 formation due to VOC	1.3
Modelling of ingestion dose	
Toxic metals	2
Exposure-Response Functions	
PM, NO_x, SO_2, toxic metals	2
Monetary values, non-market goods	
WTP for goods other than health	2
WTP for health	
$(GDP/cap)/(GDP/cap)_{ref}$ = 0.5	1.3
$(GDP/cap)/(GDP/cap)_{ref}$ = 0.2	1.7
$(GDP/cap)/(GDP/cap)_{ref}$ = 0.1	2.1

[a] Depends on site.

For example, if the transfer is to a region where no data for the effective deposition velocity v_{dep} are available, where the health system and individual sensitivities are very different from the EU15, and where the PPP-adjusted GDP/capita is 1/5 that of the EU15, the data in Table 55 indicate that the total uncertainty for the damage cost of PM_{10} can be expressed as σ_g = 4.3, which is much larger than the σ_g = 3 in the EU15. The calculations are based on Equation 1, earlier in the text.

Table 55 **Example of estimation of uncertainty with transfers**

Example for PM_{10}	σ_g	$\log(\sigma_g)$^2
In EU15	3	1.21
No v_{dep} data	1.5	0.16
CRF	2	0.48
WTP in region with $(GDP/cap)/(GDP/cap)_{ref}$ = 0.2	1.7	0.28
Total	4.3	2.13

As can be seen in the table, the total uncertainty for the damage cost of PM_{10} in the region is σ_g = 4.3, much greater than the σ_g = 3 in the EU15. If local population data are lacking, the uncertainty will increase to σ_g = 5.

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

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

Annex D List of environmental prices

D.1 Introduction

The following tables list the environmental prices calculated in this Handbook for emissions to air, water and soil. The substances listed are a selection of those considered by to be “of major concern” under the terms of the Dutch ‘Activity Decree’, which requires industries and others to avoid emissions of these pollutants to the atmosphere, soil and water, or limit them to the best of their ability. For around 20% of these substances we were able to calculate environmental prices. To this core list we have appended several other common pollutants.

D.2 Emissions to the atmosphere

Table 56 gives the environmental prices for emissions of selected pollutants to the atmosphere, listed in alphabetical order.

Table 56 Environmental prices (damage costs) for average atmospheric emissions in the Netherlands (€$_{2015}$/kg emission)

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
1,1'-Biphenyl, 3,3',4,4'-tetrachloro-, PCB-77	032598-13-3	3.71E-03	1.50E-02	2.91E-02
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	**057653-85-7**	**3.01E+00**	**1.21E+01**	**2.36E+01**
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	040321-76-4	4.61E+02	1.85E+03	3.61E+03
1,3-Dichloro-2-propanol	000096-23-1	1.32E-02	5.33E-02	1.04E-01
1,5,9-Cyclododecatriene	004904-61-4	1.64E-08	6.59E-08	1.28E-07
1-Bromopropane	000106-94-5	2.27E-06	9.13E-06	1.78E-05
2,2-Bis(4-hydroxy-3,5-dibromophenyl)propane	000079-94-7	2.37E-02	9.54E-02	1.86E-01
2,3-Dibromo-1-propanol	000096-13-9	1.40E-05	5.64E-05	1.10E-04
2,3-Dinitrotoluene	000602-01-7	1.54E-02	6.18E-02	1.20E-01
2,4,5,2',5'-Pentachlorobiphenyl	037680-73-2	1.38E-02	5.54E-02	1.08E-01
2,4,6-Tri(tert-butyl)phenol	000732-26-3	1.77E-02	7.12E-02	1.39E-01
2,4-Diaminoanisole sulfate	039156-41-7	n.c.	n.c.	n.c.
2,5-Dinitrotoluene	000619-15-8	2.06E-03	8.30E-03	1.62E-02
2-Butenal	004170-30-3	1.73E-05	6.95E-05	1.35E-04
2-Ethoxyethyl acetate	000111-15-9	1.36E-05	5.47E-05	1.06E-04
2-Methoxyethyl acetate	000110-49-6	2.06E-06	8.30E-06	1.62E-05
3,4-Dinitrotoluene	000610-39-9	2.09E-03	8.42E-03	1.64E-02
3,5-Dinitrotoluene	000618-85-9	1.07E-04	4.32E-04	8.41E-04
4,4'-Methylene di-o-toluidine	000838-88-0	7.62E-01	1.04E+00	1.61E+00
4,4'-Methylenebis-(2-chlorobenzenamine)	000101-14-4	2.33E+00	3.19E+00	4.93E+00
4,4'-Oxybisbenzenamine	000101-80-4	4.32E-01	5.91E-01	9.13E-01
4,4-Thiodianiline	000139-65-1	1.63E+00	2.23E+00	3.44E+00
4-Aminoazobenzene	000060-09-3	4.71E-04	1.90E-03	3.69E-03
Acenaphthene	000083-32-9	1.41E-01	1.93E-01	2.97E-01
Acenaphthene, 5-nitro-	000602-87-9	3.96E+00	5.42E+00	8.37E+00
Acridine	000260-94-6	4.34E-03	1.75E-02	3.40E-02
Acrylamide	000079-06-1	6.09E+01	8.33E+01	1.29E+02
Acrylonitrile	000107-13-1	1.00E+01	1.37E+01	2.11E+01

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
A-endosulfan	000959-98-8	2.23E+00	8.96E+00	1.74E+01
Aldrin	000309-00-2	5.85E+01	8.01E+01	1.24E+02
Ammonia	007664-41-7	1.97E+01	3.05E+01	4.88E+01
Aniline, p-chloro-	000106-47-8	9.41E-01	1.29E+00	2.00E+00
Anisole, pentachloro-	001825-21-4	2.84E-02	1.14E-01	2.22E-01
Anthracene	000120-12-7	5.64E-02	7.76E-02	1.20E-01
Arsenic	007440-38-2	7.03E+02	1.03E+03	1.23E+03
Azobenzene	000103-33-3	8.10E-03	3.26E-02	6.34E-02
Azocyclotin	041083-11-8	2.41E+01	3.37E+01	5.25E+01
Benomyl	017804-35-2	1.09E-01	1.55E-01	2.43E-01
Benz(a)acridine	000225-11-6	1.23E-02	4.96E-02	9.64E-02
Benz(c)acridine	000225-51-4	1.48E-01	5.95E-01	1.16E+00
Benzene	000071-43-2	8.04E-01	1.06E+00	1.60E+00
Benzene, (epoxyethyl)-	000096-09-3	3.54E-01	4.84E-01	7.48E-01
Benzene, 1-(1,1-dimethylethyl)-3,5-dimethyl-2,4,	000081-15-2	n.c.	n.c.	n.c.
Benzene, 1,2,3-trichloro-	000087-61-6	3.03E-03	1.22E-02	2.38E-02
Benzene, 1,2,4-trichloro-	000120-82-1	1.02E+00	1.40E+00	2.16E+00
Benzene, 1,3,5-trichloro-	000108-70-3	1.05E-03	4.21E-03	8.19E-03
Benzene, 1-methyl-2-nitro-	000088-72-2	8.26E-05	3.33E-04	6.47E-04
Benzene, 2,4-dichloro-1-(4-nitrophenoxy)-	001836-75-5	3.79E-01	5.35E-01	8.36E-01
Benzene, hexachloro-	000118-74-1	1.42E+02	1.95E+02	3.02E+02
Benzene, pentachloro-	000608-93-5	2.58E+01	3.55E+01	5.49E+01
Benzidine	000092-87-5	1.50E+01	2.05E+01	3.17E+01
Benzidine dihydrochloride	000531-85-1	n.c.	n.c.	n.c.
Benzidine, 3,3'-dichloro-	000091-94-1	9.47E+00	1.30E+01	2.00E+01
Benzidine, 3,3'-dimethyl-	000119-93-7	8.48E-05	3.41E-04	6.64E-04
Benzidine, 3,3'-dimethyl-, dihydrochloride	000612-82-8	n.c.	n.c.	n.c.
Benzo(a)anthracene	000056-55-3	6.64E-06	2.67E-05	5.20E-05
Benzo(a)pyrene	000050-32-8	8.36E+00	1.14E+01	1.77E+01
Benzoic acid, 4-(tert-butyl)-	000098-73-7	2.03E-04	8.18E-04	1.59E-03
Benzotrichloride	000098-07-7	9.94E+01	1.36E+02	2.10E+02
Benzyl chloride	000100-44-7	6.36E-01	8.70E-01	1.34E+00
Beryllium	007440-41-7	5.54E+04	6.59E+04	8.64E+04
beta-Naphthylamine	000091-59-8	2.91E-01	3.98E-01	6.15E-01
Binapacryl	000485-31-4	4.11E-03	1.66E-02	3.22E-02
Biphenyl, 4-amino-	000092-67-1	5.65E+00	7.73E+00	1.19E+01
Bis(chloromethyl)ether	000542-88-1	7.17E+03	9.81E+03	1.52E+04
Bisphenol A	000080-05-7	3.34E-01	4.65E-01	7.24E-01
Brodifacoum	056073-10-0	2.33E-03	9.40E-03	1.83E-02
Butadiene	000106-99-0	3.10E+00	4.09E+00	6.18E+00
Butadiene, hexachloro-	000087-68-3	1.49E-02	6.02E-02	1.17E-01
Butane	000106-97-8	9.59E-01	1.25E+00	1.87E+00
C.I. basic violet 3	000548-62-9	7.49E-02	3.01E-01	5.86E-01
C.I. disperse blue 1	002475-45-8	4.66E-02	6.38E-02	9.86E-02
C.I. solvent yellow 3	000097-56-3	5.04E+00	6.90E+00	1.07E+01
Cadmium	007440-43-9	7.98E+02	1.16E+03	1.83E+03
Carbamic acid, ethyl ester	000051-79-6	1.69E-01	2.31E-01	3.57E-01
Carbendazim	010605-21-7	4.46E-01	7.30E-01	1.20E+00
Carbon dioxide	000124-38-9	1.42E-02	5.66E-02	5.66E-02
Carbon monoxide	000630-08-0	7.36E-02	9.58E-02	1.52E-01
Chlordane , pur	000057-74-9	1.40E+03	1.91E+03	2.95E+03

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Chlorfenvinphos	000470-90-6	1.44E+02	1.98E+02	3.06E+02
Chloromethyl methyl ether	000107-30-2	3.77E+00	5.16E+00	7.97E+00
Chloroprene	000126-99-8	1.09E+00	1.49E+00	2.31E+00
Chromium	007440-47-3	1.52E-01	5.31E-01	1.02E+00
Copper	007440-50-8	1.15E+00	4.20E+00	8.25E+00
Crotonaldehyde	000123-73-9	n.c.	n.c.	n.c.
Cyclododecane	000294-62-2	5.70E-08	2.30E-07	4.47E-07
Cyclododecane, hexabromo-	025637-99-4	2.13E-02	8.60E-02	1.67E-01
Cycloheximide	000066-81-9	1.56E-03	6.30E-03	1.23E-02
Cyclopentadiene, hexachloro-	000077-47-4	2.33E+02	3.19E+02	4.93E+02
Cyhexatin	013121-70-5	1.48E+01	2.07E+01	3.23E+01
DDT	000050-29-3	6.28E+01	8.62E+01	1.33E+02
Decabromodiphenyl oxide	001163-19-5	1.28E+02	1.76E+02	2.71E+02
Delta-hexachlorocyclohexane	000319-86-8	1.67E-02	6.73E-02	1.31E-01
Dibenz(a,h)anthracene	000053-70-3	4.35E+02	5.95E+02	9.20E+02
Dibenzofuran, 2,3,7,8-tetrachloro-	051207-31-9	5.26E+02	2.12E+03	4.12E+03
Dibutyl dichloro tin	000683-18-1	3.42E-03	1.38E-02	2.68E-02
Dibutyltin oxide	000818-08-6	1.35E-08	5.43E-08	1.06E-07
Dicofol	000115-32-2	7.11E+01	9.73E+01	1.50E+02
Dieldrin	000060-57-1	7.01E+02	9.60E+02	1.49E+03
Difenacoum	056073-07-5	2.22E-04	8.95E-04	1.74E-03
Di-isobutylphthalate	000084-69-5	8.19E-05	3.30E-04	6.41E-04
Dimethyl formamide	000068-12-2	1.48E+00	2.03E+00	3.13E+00
Dimethylcarbamyl chloride	000079-44-7	3.40E+01	4.65E+01	7.18E+01
Dimethylphenol phosphate (3:1)	025155-23-1	2.59E-04	1.04E-03	2.03E-03
Dinitrogen monoxide	010024-97-2	3.75E+00	1.50E+01	1.50E+01
Dinocap	039300-45-3	5.04E+00	6.97E+00	1.08E+01
Dinoseb	000088-85-7	7.17E+01	9.93E+01	1.54E+02
Dinoterb	001420-07-1	6.97E-01	2.81E+00	5.46E+00
Dioxin, 1,2,3,7,8,9-hexachlorodibenzo-	019408-74-3	n.c.	n.c.	n.c.
Dioxin, 2,3,7,8 Tetrachlorodibenzo-p-	001746-01-6	4.90E+07	6.71E+07	1.04E+08
Diuron	000330-54-1	3.06E+00	4.55E+00	7.25E+00
Endosulfan	000115-29-7	1.03E+01	1.41E+01	2.18E+01
Endosulfan (beta)	033213-65-9	3.08E+00	1.24E+01	2.41E+01
Endrin	000072-20-8	1.28E+01	1.87E+01	2.95E+01
Endrocide (endox) (coumatetralyl)	005836-29-3	1.40E-04	5.62E-04	1.09E-03
Epichlorohydrin	000106-89-8	2.00E+01	2.74E+01	4.23E+01
Ethane, 1,2-dibromo-	000106-93-4	2.30E+01	3.15E+01	4.86E+01
Ethane, 1,2-dichloro-	000107-06-2	5.12E+00	7.00E+00	1.08E+01
Ethane, pentachloro-	000076-01-7	1.21E-05	4.89E-05	9.51E-05
Ethanol, 2-ethoxy-	000110-80-5	1.14E+00	1.49E+00	2.24E+00
Ethanol, 2-methoxy-	000109-86-4	1.20E+00	1.59E+00	2.41E+00
Ethene, bromo-	000593-60-2	2.91E+00	3.98E+00	6.15E+00
Ethene, chloro-	000075-01-4	2.41E+00	3.30E+00	5.09E+00
Ethene, trichloro-	000079-01-6	9.13E-01	1.19E+00	1.79E+00
Ethyl O-(p-nitrophenyl) phenylphosphonothionate	002104-64-5	3.93E+02	5.38E+02	8.30E+02
Ethylene oxide	000075-21-8	1.58E+00	2.16E+00	3.34E+00
Ethylene thiourea	000096-45-7	7.58E-01	1.04E+00	1.60E+00
Ethyleneimine	000151-56-4	9.94E+01	1.36E+02	2.10E+02
Fenbutatin oxide	013356-08-6	6.48E+01	8.87E+01	1.37E+02
Fenchlorazole-ethyl	103112-35-2	9.78E-02	3.94E-01	7.66E-01

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Fentin acetate	000900-95-8	3.45E+02	4.73E+02	7.31E+02
Fentin chloride	000639-58-7	3.47E+02	4.77E+02	7.38E+02
Fentin hydroxide	000076-87-9	3.29E+02	4.51E+02	6.98E+02
Fluazifop-butyl	069806-50-4	2.70E-03	1.09E-02	2.12E-02
Flucythrinate	070124-77-5	1.36E+01	2.91E+01	5.14E+01
Fluoranthene	000206-44-0	2.99E-01	4.18E-01	6.50E-01
Fluorene	000086-73-7	3.90E-01	5.38E-01	8.34E-01
Flusilazole	085509-19-9	1.26E+01	1.73E+01	2.67E+01
Formaldehyde	000050-00-0	1.93E+01	2.63E+01	4.05E+01
Furan	000110-00-9	4.77E+01	6.53E+01	1.01E+02
Glufosinate ammonium	077182-82-2	5.70E+00	7.81E+00	1.21E+01
Glycidol	000556-52-5	5.32E+00	7.28E+00	1.12E+01
Glycydyltrimethylammonium chloride	003033-77-0	2.71E-07	1.09E-06	2.12E-06
Heptachlor	000076-44-8	1.47E+01	2.02E+01	3.12E+01
Heptachlor epoxide	001024-57-3	1.66E+02	2.28E+02	3.52E+02
Heptachloronorbornene	028680-45-7	1.51E-04	6.06E-04	1.18E-03
Hexachlorocyclohexane	000608-73-1	1.44E+01	1.98E+01	3.07E+01
Hexamethylphosphoramide	000680-31-9	2.14E+02	2.93E+02	4.53E+02
Hydrazine	000302-01-2	4.02E+01	5.51E+01	8.51E+01
Hydrazine, 1,1-dimethyl-	000057-14-7	3.23E-01	4.77E-01	7.59E-01
Hydrazine, 1,2-diphenyl-	000122-66-7	3.93E-05	1.58E-04	3.08E-04
Hydrazine, phenyl-, hydrochloride	000059-88-1	n.c.	n.c.	n.c.
Isobutane	000075-28-5	8.37E-01	1.09E+00	1.63E+00
Isodrin	000465-73-6	2.24E-02	9.01E-02	1.75E-01
Isoprene	000078-79-5	3.00E+00	3.91E+00	5.87E+00
Isoquinoline	000119-65-3	4.62E-04	1.86E-03	3.62E-03
Kepone	000143-50-0	1.76E+02	2.44E+02	3.78E+02
Lead	007439-92-1	3.97E+03	5.91E+03	6.60E+03
Lindane	000058-89-9	7.19E+01	9.88E+01	1.53E+02
Lindane, alpha-	000319-84-6	1.39E+01	1.90E+01	2.95E+01
Lindane, beta-	000319-85-7	8.28E+00	1.13E+01	1.75E+01
Linuron	000330-55-2	2.37E+00	3.29E+00	5.11E+00
Mercury	007439-97-6	2.48E+04	3.45E+04	5.36E+04
Methane	000074-82-8	4.48E-01	1.75E+00	1.76E+00
Methoxychlor	000072-43-5	5.29E-01	7.25E-01	1.12E+00
Methylmercury	022967-92-6	2.58E+03	3.53E+03	5.46E+03
Mirex	002385-85-5	1.08E+04	1.47E+04	2.27E+04
Naphthalene	000091-20-3	1.13E+00	1.54E+00	2.38E+00
Naphthalene, 2-methyl-	000091-57-6	9.67E-01	1.32E+00	2.04E+00
Nickel	007440-02-0	7.50E+01	1.33E+02	2.25E+02
Nitroanisole, o-	000091-23-6	2.05E+00	2.81E+00	4.34E+00
Nitrobenzene	000098-95-3	1.96E+01	2.68E+01	4.14E+01
Nitrogen oxides	011104-93-1	2.41E+01	3.47E+01	5.37E+01
Nitrosoguanidine, N-methyl-N'-nitro-N-	000070-25-7	5.15E+01	7.04E+01	1.09E+02
Nitrous acid, 2-methylpropyl ester	000542-56-3	n.c.	n.c.	n.c.
NMVOC, non-methane volatile organic compounds, unspecified origin	999999-82-8	1.61E+00	2.10E+00	3.15E+00
N-Nitrosodiethanolamine	001116-54-7	3.98E+00	5.44E+00	8.40E+00
N-Nitrosodimethylamine	000062-75-9	2.43E+02	3.32E+02	5.13E+02
N-Nitrosodipropylamine	000621-64-7	5.29E+02	7.24E+02	1.12E+03
N-nonylphenol	084852-15-3	5.82E-03	2.34E-02	4.56E-02
Nonylphenol	025154-52-3	1.48E-07	5.95E-07	1.16E-06

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
O,p'-ddt	000789-02-6	1.39E-01	5.58E-01	1.09E+00
o-Aminoanisole	000090-04-0	1.02E-05	4.12E-05	8.02E-05
o-Toluidine	000095-53-4	1.51E-11	6.09E-11	1.18E-10
o-Toluidine, 4-chloro-, hydrochloride	003165-93-3	7.65E-01	1.05E+00	1.62E+00
Oxirane, (phenoxymethyl)-	000122-60-1	3.16E-01	4.33E-01	6.68E-01
P-(1,1,3,3-tetramethylbutyl)phenol	000140-66-9	2.66E-03	1.07E-02	2.08E-02
Particulates, < 10 um	999999-83-0	3.18E+01	4.46E+01	6.91E+01
Particulates, < 2.5 um	999999-83-3	5.68E+01	7.95E+01	1.22E+02
p-Cresidine	000120-71-8	3.60E-02	4.93E-02	7.61E-02
Pentabromodiphenyl ether	032534-81-9	1.31E+02	1.79E+02	2.76E+02
Phenanthrene	000085-01-8	2.07E-04	8.33E-04	1.62E-03
Phenanthridine	000229-87-8	2.40E-03	9.66E-03	1.88E-02
Phenol, pentachloro-	000087-86-5	6.87E+00	9.40E+00	1.45E+01
Phenolphthalein	000077-09-8	5.87E-02	8.03E-02	1.24E-01
Phenyl hydrazine	000100-63-0	3.45E-05	1.39E-04	2.70E-04
Phenylmercuric acetate	000062-38-4	1.11E+03	1.52E+03	2.35E+03
Phosphate, tris(2-chloroethyl)-	000115-96-8	2.20E-05	8.87E-05	1.73E-04
Phthalate, butyl-benzyl-	000085-68-7	1.87E-01	2.57E-01	3.98E-01
Phthalate, dibutyl-	000084-74-2	1.59E-01	2.25E-01	3.51E-01
Phthalate, dihexyl-	000084-75-3	7.02E-04	2.83E-03	5.50E-03
Phthalate, dioctyl-	000117-81-7	9.74E+00	1.33E+01	2.06E+01
P-nonylphenol	000104-40-5	3.00E-04	1.21E-03	2.35E-03
Polychlorinated biphenyls	001336-36-3	2.79E-03	1.12E-02	2.19E-02
Propane sultone	001120-71-4	8.19E+00	1.12E+01	1.73E+01
Propane, 1,2,3-trichloro-	000096-18-4	2.96E+01	4.05E+01	6.26E+01
Propane, 1,2-dibromo-3-chloro-	000096-12-8	1.69E+02	2.31E+02	3.57E+02
Propane, 1,2-dichloro-	000078-87-5	4.16E+01	5.70E+01	8.80E+01
Propane, 2-nitro-	000079-46-9	1.28E+00	1.76E+00	2.71E+00
Propiolactone	000057-57-8	1.83E+01	2.51E+01	3.87E+01
Propylene oxide	000075-56-9	6.64E+00	9.08E+00	1.40E+01
P-tert-amylphenol	000080-46-6	2.87E-04	1.15E-03	2.25E-03
Pyrene	000129-00-0	5.28E-01	7.23E-01	1.12E+00
Quinoline	000091-22-5	5.53E-04	2.23E-03	4.33E-03
Safrole	000094-59-7	1.75E-02	2.40E-02	3.70E-02
Sulfallate	000095-06-7	2.41E-01	3.30E-01	5.10E-01
Sulfur dioxide	007446-09-5	1.77E+01	2.49E+01	3.87E+01
Sulfur hexafluoride	002551-62-4	3.33E+02	1.33E+03	1.33E+03
Sulfuric acid, dimethyl ester	000077-78-1	6.17E-07	2.48E-06	4.83E-06
Tetrabutyltin	001461-25-2	2.23E-08	8.99E-08	1.75E-07
Tetraethyl lead	000078-00-2	2.46E+04	3.36E+04	5.19E+04
Tetrahydrofurfuryl alcohol	000097-99-4	7.27E-08	2.93E-07	5.69E-07
Tetramethyl lead	000075-74-1	1.16E-06	4.67E-06	9.09E-06
Tetramethyldiaminobenzophenone	000090-94-8	n.c.	n.c.	n.c.
Tetrasul	002227-13-6	1.02E-04	4.12E-04	8.02E-04
Thioacetamide	000062-55-5	1.28E+00	1.75E+00	2.70E+00
Toluene, 2,4-diamine	000095-80-7	2.47E+00	3.38E+00	5.23E+00
Toluene, 2,4-dinitro-	000121-14-2	2.30E+01	3.15E+01	4.86E+01
Toluene, 2,6-dinitro-	000606-20-2	2.36E+02	3.23E+02	5.00E+02
Toluene, dinitro-	025321-14-6	n.c.	n.c.	n.c.
Toxaphene	008001-35-2	5.09E+00	7.46E+00	1.18E+01
Tributylstannane	000688-73-3	2.01E-06	8.10E-06	1.58E-05

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Tributyltin oxide	000056-35-9	6.80E+01	1.07E+02	1.73E+02
Trichlorobenzenes	012002-48-1	2.34E-04	9.42E-04	1.83E-03
Triflumizole	068694-11-1	3.20E-02	1.29E-01	2.51E-01
Trifluralin	001582-09-8	5.54E-01	7.59E-01	1.17E+00
Trimethylaniline hydrochloride, 2,4,5-	021436-97-5	n.c.	n.c.	n.c.
Trimethylaniline, 2,4,5-	000137-17-7	n.c.	n.c.	n.c.
Vinclozolin	050471-44-8	5.26E+00	7.21E+00	1.11E+01
Warfarin	000081-81-2	5.49E+01	7.52E+01	1.16E+02
Zinc	007440-66-6	2.25E+00	1.18E+01	3.16E+01

N.c = not calculated or no impact.

D.3 Emissions to water

Table 57 gives the environmental prices for emissions of selected pollutants to water, listed in alphabetical order.

Table 57 Environmental prices (damage costs) for average emissions to water in the Netherlands (€$_{2015}$/kg emission)

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
1,1'-Biphenyl, 3,3',4,4'-tetrachloro-, PCB-77	032598-13-3	6.21E-04	2.50E-03	4.87E-03
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	057653-85-7	1.83E-01	7.36E-01	1.43E+00
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	040321-76-4	8.09E+01	3.26E+02	6.33E+02
1,3-Dichloro-2-propanol	000096-23-1	1.85E-01	7.43E-01	1.45E+00
1,5,9-Cyclododecatriene	004904-61-4	5.92E-05	2.38E-04	4.64E-04
1-Bromopropane	000106-94-5	4.23E-06	1.70E-05	3.31E-05
2,2-Bis(4-hydroxy-3,5-dibromophenyl)propane	000079-94-7	1.07E-01	4.31E-01	8.38E-01
2,3-Dibromo-1-propanol	000096-13-9	4.62E-06	1.86E-05	3.62E-05
2,3-Dinitrotoluene	000602-01-7	2.36E-01	9.51E-01	1.85E+00
2,4,5,2',5'-Pentachlorobiphenyl	037680-73-2	5.60E-03	2.25E-02	4.39E-02
2,4,6-Tri(tert-butyl)phenol	000732-26-3	1.28E-02	5.15E-02	1.00E-01
2,4-Diaminoanisole sulfate	039156-41-7	n.c.	n.c.	n.c.
2,5-Dinitrotoluene	000619-15-8	6.43E-04	2.59E-03	5.04E-03
2-Butenal	004170-30-3	2.76E-03	1.11E-02	2.16E-02
2-Ethoxyethyl acetate	000111-15-9	6.83E-04	2.75E-03	5.35E-03
2-Methoxyethyl acetate	000110-49-6	3.58E-07	1.44E-06	2.81E-06
3,4-Dinitrotoluene	000610-39-9	8.48E-04	3.41E-03	6.64E-03
3,5-Dinitrotoluene	000618-85-9	3.23E-05	1.30E-04	2.53E-04
4,4'-Methylene di-o-toluidine	000838-88-0	5.64E+00	7.71E+00	1.19E+01
4,4'-Methylenebis-(2-chlorobenzenamine)	000101-14-4	2.14E+01	2.93E+01	4.53E+01
4,4'-Oxybisbenzenamine	000101-80-4	3.63E-01	4.97E-01	7.67E-01
4,4-Thiodianiline	000139-65-1	1.53E+00	2.10E+00	3.24E+00
4-Aminoazobenzene	000060-09-3	8.74E-04	3.52E-03	6.85E-03
Acenaphthene	000083-32-9	3.56E-02	1.08E-01	2.03E-01
Acenaphthene, 5-nitro-	000602-87-9	5.98E+00	8.18E+00	1.26E+01
Acridine	000260-94-6	2.56E-01	1.03E+00	2.01E+00
Acrylamide	000079-06-1	7.61E-01	1.04E+00	1.61E+00
Acrylonitrile	000107-13-1	8.30E-01	1.14E+00	1.77E+00
A-endosulfan	000959-98-8	5.71E+01	2.30E+02	4.47E+02
Aldrin	000309-00-2	1.65E+03	2.26E+03	3.49E+03

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Aniline, p-chloro-	000106-47-8	2.14E-01	3.74E-01	6.27E-01
Anisole, pentachloro-	001825-21-4	2.87E-02	1.15E-01	2.25E-01
Anthracene	000120-12-7	3.67E-02	1.46E-01	2.84E-01
Arsenic	007440-38-2	2.02E+02	4.33E+02	9.11E+02
Azobenzene	000103-33-3	1.98E-01	7.98E-01	1.55E+00
Azocyclotin	041083-11-8	1.89E+02	2.83E+02	4.53E+02
Benomyl	017804-35-2	8.15E-02	3.00E-01	5.79E-01
Benz(a)acridine	000225-11-6	2.66E-02	1.07E-01	2.08E-01
Benz(c)acridine	000225-51-4	3.20E-01	1.29E+00	2.51E+00
Benzene	000071-43-2	5.68E-02	7.94E-02	1.24E-01
Benzene, (epoxyethyl)-	000096-09-3	8.22E-03	1.12E-02	1.74E-02
Benzene, 1-(1,1-dimethylethyl)-3,5-dimethyl-2,4,	000081-15-2	n.c.	n.c.	n.c.
Benzene, 1,2,3-trichloro-	000087-61-6	5.53E-02	2.23E-01	4.34E-01
Benzene, 1,2,4-trichloro-	000120-82-1	7.50E-01	1.12E+00	1.79E+00
Benzene, 1,3,5-trichloro-	000108-70-3	1.19E-03	4.80E-03	9.35E-03
Benzene, 1-methyl-2-nitro-	000088-72-2	1.99E-03	8.02E-03	1.56E-02
Benzene, 2,4-dichloro-1-(4-nitrophenoxy)-	001836-75-5	1.14E+00	2.06E+00	3.48E+00
Benzene, hexachloro-	000118-74-1	4.08E+02	5.59E+02	8.65E+02
Benzene, pentachloro-	000608-93-5	5.27E+01	7.26E+01	1.12E+02
Benzidine	000092-87-5	1.34E+00	1.83E+00	2.82E+00
Benzidine dihydrochloride	000531-85-1	n.c.	n.c.	n.c.
Benzidine, 3,3'-dichloro-	000091-94-1	3.60E+01	4.93E+01	7.62E+01
Benzidine, 3,3'-dimethyl-	000119-93-7	8.03E-04	3.23E-03	6.29E-03
Benzidine, 3,3'-dimethyl-, dihydrochloride	000612-82-8	0.00E+00	0.00E+00	0.00E+00
Benzo(a)anthracene	000056-55-3	3.60E-06	1.45E-05	2.82E-05
Benzo(a)pyrene	000050-32-8	2.80E-01	4.66E-01	7.70E-01
Benzoic acid, 4-(tert-butyl)-	000098-73-7	8.74E-05	3.52E-04	6.85E-04
Benzotrichloride	000098-07-7	7.20E-03	9.85E-03	1.52E-02
Benzyl chloride	000100-44-7	8.26E-02	1.36E-01	2.23E-01
Beryllium	007440-41-7	7.44E+00	2.69E+01	5.23E+01
beta-Naphthylamine	000091-59-8	5.10E-01	6.98E-01	1.08E+00
Binapacryl	000485-31-4	3.65E-01	1.47E+00	2.86E+00
Biphenyl, 4-amino-	000092-67-1	3.41E+00	4.67E+00	7.21E+00
Bis(chloromethyl)ether	000542-88-1	1.43E+03	1.96E+03	3.02E+03
Bisphenol A	000080-05-7	3.81E-01	9.04E-01	1.63E+00
Brodifacoum	056073-10-0	5.52E-03	2.22E-02	4.33E-02
Butadiene	000106-99-0	3.13E-02	4.28E-02	6.62E-02
Butadiene, hexachloro-	000087-68-3	1.21E-01	4.88E-01	9.50E-01
C.I. basic violet 3	000548-62-9	7.94E-02	3.20E-01	6.22E-01
C.I. disperse blue 1	002475-45-8	1.60E-01	2.18E-01	3.37E-01
C.I. solvent yellow 3	000097-56-3	6.15E+01	8.42E+01	1.30E+02
Cadmium	007440-43-9	5.25E+00	6.57E+00	8.91E+00
Carbamic acid, ethyl ester	000051-79-6	8.63E-03	1.18E-02	1.82E-02
Carbendazim	010605-21-7	8.52E-01	2.85E+00	5.43E+00
Carbon monoxide	000630-08-0	n.c.	n.c.	n.c.
Chlordane , pur	000057-74-9	5.25E+02	7.19E+02	1.11E+03
Chlorfenvinphos	000470-90-6	3.16E+02	4.40E+02	6.84E+02
Chloromethyl methyl ether	000107-30-2	2.05E-04	2.81E-04	4.33E-04
Chloroprene	000126-99-8	1.86E-01	2.55E-01	3.94E-01
Chromium	007440-47-3	n.c.	n.c.	n.c.
Copper	007440-50-8	1.43E+00	5.95E+00	1.20E+01
Crotonaldehyde	000123-73-9	n.c.	n.c.	n.c.

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Cyclododecane	000294-62-2	6.02E-06	2.43E-05	4.72E-05
Cyclododecane, hexabromo-	025637-99-4	2.92E-03	1.18E-02	2.29E-02
Cycloheximide	000066-81-9	6.93E-04	2.79E-03	5.43E-03
Cyclopentadiene, hexachloro-	000077-47-4	9.53E-02	1.31E-01	2.02E-01
Cyhexatin	013121-70-5	1.86E+02	2.72E+02	4.31E+02
DDT	000050-29-3	4.73E+01	6.74E+01	1.06E+02
Decabromodiphenyl oxide	001163-19-5	2.27E-04	3.10E-04	4.80E-04
Delta-hexachlorocyclohexane	000319-86-8	3.79E-01	1.52E+00	2.96E+00
Dibenz(a,h)anthracene	000053-70-3	2.00E+02	2.74E+02	4.23E+02
Dibenzofuran, 2,3,7,8-tetrachloro-	051207-31-9	3.39E+02	1.37E+03	2.66E+03
Dibutyl dichloro tin	000683-18-1	1.12E+00	4.52E+00	8.78E+00
Dibutyltin oxide	000818-08-6	4.42E-05	1.78E-04	3.46E-04
Dicofol	000115-32-2	2.49E+02	3.42E+02	5.29E+02
Dieldrin	000060-57-1	5.54E+03	7.61E+03	1.18E+04
Difenacoum	056073-07-5	1.96E-04	7.89E-04	1.53E-03
Di-isobutylphthalate	000084-69-5	6.76E-05	2.72E-04	5.29E-04
Dimethyl formamide	000068-12-2	9.27E-02	1.27E-01	1.96E-01
Dimethylcarbamyl chloride	000079-44-7	6.25E+00	8.54E+00	1.32E+01
Dimethylphenol phosphate (3:1)	025155-23-1	2.05E-05	8.24E-05	1.60E-04
Dinocap	039300-45-3	7.87E+01	1.14E+02	1.79E+02
Dinoseb	000088-85-7	1.89E+01	3.29E+01	5.50E+01
Dinoterb	001420-07-1	1.00E+01	4.03E+01	7.84E+01
Dioxin, 1,2,3,7,8,9-hexachlorodibenzo-	019408-74-3	n.c.	n.c.	n.c.
Dioxin, 2,3,7,8 Tetrachlorodibenzo-p-	001746-01-6	4.85E+06	6.64E+06	1.03E+07
Diuron	000330-54-1	3.36E+00	8.15E+00	1.47E+01
Endosulfan	000115-29-7	1.24E+01	3.13E+01	5.70E+01
Endosulfan (beta)	033213-65-9	5.27E+01	2.12E+02	4.13E+02
Endrin	000072-20-8	8.41E+02	1.33E+03	2.16E+03
Endrocide (endox) (coumatetralyl)	005836-29-3	3.72E-03	1.50E-02	2.91E-02
Epichlorohydrin	000106-89-8	1.77E+00	2.44E+00	3.77E+00
Ethane, 1,2-dibromo-	000106-93-4	1.00E+01	1.37E+01	2.12E+01
Ethane, 1,2-dichloro-	000107-06-2	2.68E+00	3.66E+00	5.66E+00
Ethane, pentachloro-	000076-01-7	2.90E-03	1.17E-02	2.27E-02
Ethanol, 2-ethoxy-	000110-80-5	1.11E-02	1.52E-02	2.35E-02
Ethanol, 2-methoxy-	000109-86-4	2.41E-02	3.30E-02	5.09E-02
Ethene, bromo-	000593-60-2	3.22E-01	4.41E-01	6.81E-01
Ethene, chloro-	000075-01-4	7.31E-01	1.00E+00	1.54E+00
Ethene, trichloro-	000079-01-6	9.96E-03	1.74E-02	2.92E-02
Ethyl O-(p-nitrophenyl) phenylphosphonothionate	002104-64-5	5.14E+03	7.04E+03	1.09E+04
Ethylene oxide	000075-21-8	2.92E-01	4.00E-01	6.19E-01
Ethylene thiourea	000096-45-7	4.81E-01	6.60E-01	1.02E+00
Ethyleneimine	000151-56-4	2.19E+01	3.00E+01	4.63E+01
Fenbutatin oxide	013356-08-6	4.18E-06	1.47E-05	2.82E-05
Fenchlorazole-ethyl	103112-35-2	2.08E+00	8.39E+00	1.63E+01
Fentin acetate	000900-95-8	2.00E+02	3.04E+02	4.88E+02
Fentin chloride	000639-58-7	1.68E+02	2.73E+02	4.47E+02
Fentin hydroxide	000076-87-9	2.01E+02	3.10E+02	5.01E+02
Fluazifop-butyl	069806-50-4	1.58E-01	6.37E-01	1.24E+00
Flucythrinate	070124-77-5	2.29E+02	5.83E+02	1.06E+03
Fluoranthene	000206-44-0	1.68E+00	5.22E+00	9.84E+00
Fluorene	000086-73-7	2.60E-01	5.32E-01	9.29E-01
Flusilazole	085509-19-9	1.60E+01	2.18E+01	3.37E+01

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Formaldehyde	000050-00-0	8.36E-01	1.15E+00	1.78E+00
Furan	000110-00-9	2.40E+00	3.28E+00	5.06E+00
Glufosinate ammonium	077182-82-2	1.91E-01	3.04E-01	4.95E-01
Glycidol	000556-52-5	2.41E-01	3.30E-01	5.09E-01
Glycydyltrimethylammonium chloride	003033-77-0	1.93E-07	7.77E-07	1.51E-06
Heptachlor	000076-44-8	3.08E+02	4.23E+02	6.55E+02
Heptachlor epoxide	001024-57-3	1.27E+03	1.77E+03	2.75E+03
Heptachloronorbornene	028680-45-7	1.13E-02	4.55E-02	8.85E-02
Hexachlorocyclohexane	000608-73-1	2.42E+01	3.69E+01	5.94E+01
Hexamethylphosphoramide	000680-31-9	8.20E+01	1.12E+02	1.73E+02
Hydrazine	000302-01-2	2.10E+00	3.31E+00	5.37E+00
Hydrazine, 1,1-dimethyl-	000057-14-7	9.24E-02	2.57E-01	4.76E-01
Hydrazine, 1,2-diphenyl-	000122-66-7	1.54E-04	6.18E-04	1.20E-03
Hydrazine, phenyl-, hydrochloride	000059-88-1	n.c.	n.c.	n.c.
Isobutane	000075-28-5	n.c.	n.c.	n.c.
Isodrin	000465-73-6	1.05E-01	4.21E-01	8.19E-01
Isoprene	000078-79-5	9.41E-03	1.43E-02	2.29E-02
Isoquinoline	000119-65-3	1.52E-02	6.14E-02	1.19E-01
Kepone	000143-50-0	1.46E+03	2.03E+03	3.16E+03
Lead	007439-92-1	9.62E-01	5.85E+00	1.60E+01
Lindane	000058-89-9	1.66E+02	2.42E+02	3.84E+02
Lindane, alpha-	000319-84-6	1.71E+01	2.47E+01	3.90E+01
Lindane, beta-	000319-85-7	1.45E+01	1.98E+01	3.06E+01
Linuron	000330-55-2	1.43E+01	2.64E+01	4.50E+01
Mercury	007439-97-6	7.88E+01	1.98E+03	5.95E+03
Methoxychlor	000072-43-5	2.46E-01	7.90E-01	1.50E+00
Methylmercury	022967-92-6	1.28E+01	1.75E+01	2.70E+01
Mirex	002385-85-5	6.20E+03	8.48E+03	1.31E+04
Naphthalene	000091-20-3	1.88E-01	2.89E-01	4.66E-01
Naphthalene, 2-methyl-	000091-57-6	3.15E-01	5.58E-01	9.38E-01
Nickel	007440-02-0	1.48E+00	5.97E+00	1.20E+01
Nitrate	014797-55-8	7.15E-01	7.15E-01	7.15E-01
Nitroanisole, o-	000091-23-6	2.88E-01	4.04E-01	6.29E-01
Nitrobenzene	000098-95-3	2.84E+00	3.91E+00	6.05E+00
Nitrosoguanidine, N-methyl-N'-nitro-N-	000070-25-7	3.94E+00	5.40E+00	8.34E+00
Nitrous acid, 2-methylpropyl ester	000542-56-3	n.c.	n.c.	n.c.
N-Nitrosodiethanolamine	001116-54-7	3.01E-01	4.11E-01	6.35E-01
N-Nitrosodimethylamine	000062-75-9	1.17E+01	1.61E+01	2.48E+01
N-Nitrosodipropylamine	000621-64-7	1.11E+02	1.52E+02	2.35E+02
N-nonylphenol	084852-15-3	1.44E+00	5.78E+00	1.13E+01
Nonylphenol	025154-52-3	2.50E-01	1.01E+00	1.96E+00
O,p'-ddt	000789-02-6	3.26E-02	1.31E-01	2.55E-01
o-Aminoanisole	000090-04-0	1.07E-04	4.30E-04	8.36E-04
o-Toluidine	000095-53-4	7.39E-04	2.98E-03	5.79E-03
o-Toluidine, 4-chloro-, hydrochloride	003165-93-3	3.30E-01	4.52E-01	6.98E-01
Oxirane, (phenoxymethyl)-	000122-60-1	8.69E-02	1.19E-01	1.84E-01
P-(1,1,3,3-tetramethylbutyl)phenol	000140-66-9	7.24E-01	2.92E+00	5.67E+00
p-Cresidine	000120-71-8	3.69E-02	5.05E-02	7.81E-02
Pentabromodiphenyl ether	032534-81-9	4.12E+01	5.63E+01	8.70E+01
Phenanthrene	000085-01-8	4.20E-02	1.69E-01	3.29E-01
Phenanthridine	000229-87-8	1.52E-03	6.12E-03	1.19E-02
Phenol, pentachloro-	000087-86-5	3.81E-01	6.07E-01	9.89E-01

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Phenolphthalein	000077-09-8	2.03E-02	2.78E-02	4.30E-02
Phenyl hydrazine	000100-63-0	5.29E-05	2.13E-04	4.15E-04
Phenylmercuric acetate	000062-38-4	4.52E+01	7.06E+01	1.14E+02
Phosphate	014265-44-2	1.56E-01	6.29E-01	1.22E+00
Phosphate, tris(2-chloroethyl)-	000115-96-8	1.53E-03	6.17E-03	1.20E-02
Phthalate, butyl-benzyl-	000085-68-7	3.59E-02	1.37E-01	2.66E-01
Phthalate, dibutyl-	000084-74-2	1.38E-01	4.79E-01	9.15E-01
Phthalate, dihexyl-	000084-75-3	8.32E-04	3.35E-03	6.52E-03
Phthalate, dioctyl-	000117-81-7	1.00E+00	1.38E+00	2.13E+00
P-nonylphenol	000104-40-5	1.56E-01	6.28E-01	1.22E+00
Polychlorinated biphenyls	001336-36-3	1.90E-03	7.65E-03	1.49E-02
Propane sultone	001120-71-4	8.91E-01	1.22E+00	1.88E+00
Propane, 1,2,3-trichloro-	000096-18-4	1.26E+01	1.73E+01	2.67E+01
Propane, 1,2-dibromo-3-chloro-	000096-12-8	5.51E+01	7.54E+01	1.16E+02
Propane, 1,2-dichloro-	000078-87-5	1.83E+01	2.51E+01	3.87E+01
Propane, 2-nitro-	000079-46-9	3.16E-01	4.33E-01	6.68E-01
Propiolactone	000057-57-8	7.34E-01	1.00E+00	1.55E+00
Propylene oxide	000075-56-9	7.17E-01	9.81E-01	1.52E+00
P-tert-amylphenol	000080-46-6	7.88E-04	3.17E-03	6.17E-03
Pyrene	000129-00-0	1.78E-01	6.46E-01	1.24E+00
Quinoline	000091-22-5	4.28E-03	1.72E-02	3.36E-02
Safrole	000094-59-7	5.28E-02	7.22E-02	1.11E-01
Sulfallate	000095-06-7	6.23E-01	8.54E-01	1.32E+00
Sulfuric acid, dimethyl ester	000077-78-1	2.38E-09	9.60E-09	1.87E-08
Tetrabutyltin	001461-25-2	1.29E-03	5.20E-03	1.01E-02
Tetraethyl lead	000078-00-2	2.36E+04	3.23E+04	5.00E+04
Tetrahydrofurfuryl alcohol	000097-99-4	2.25E-08	9.07E-08	1.76E-07
Tetramethyl lead	000075-74-1	1.11E-01	4.48E-01	8.72E-01
Tetramethyldiaminobenzophenone	000090-94-8	n.c.	n.c.	n.c.
Tetrasul	002227-13-6	3.70E-05	1.49E-04	2.90E-04
Thioacetamide	000062-55-5	8.63E-02	1.18E-01	1.82E-01
Toluene, 2,4-diamine	000095-80-7	1.26E+00	1.73E+00	2.67E+00
Toluene, 2,4-dinitro-	000121-14-2	8.96E-02	1.32E-01	2.10E-01
Toluene, 2,6-dinitro-	000606-20-2	4.21E-01	5.77E-01	8.92E-01
Toluene, dinitro-	025321-14-6	n.c.	n.c.	n.c.
Toxaphene	008001-35-2	7.44E+00	2.25E+01	4.23E+01
Tributylstannane	000688-73-3	2.82E+00	1.14E+01	2.21E+01
Tributyltin oxide	000056-35-9	1.35E+02	4.00E+02	7.49E+02
Trichlorobenzenes	012002-48-1	2.62E-04	1.05E-03	2.05E-03
Triflumizole	068694-11-1	2.42E-01	9.74E-01	1.89E+00
Trifluralin	001582-09-8	1.30E+01	1.84E+01	2.88E+01
Trimethylaniline hydrochloride, 2,4,5-	021436-97-5	n.c.	n.c.	n.c.
Trimethylaniline, 2,4,5-	000137-17-7	n.c.	n.c.	n.c.
Vinclozolin	050471-44-8	2.48E+00	3.51E+00	5.49E+00
Warfarin	000081-81-2	3.46E+01	4.74E+01	7.32E+01
Zinc	007440-66-6	1.68E-01	1.14E+00	2.96E+00

N.c = not calculated or no impact.

D.4 Emissions to the soil

Table 58 gives the environmental prices for emissions of selected pollutants to the soil, listed in alphabetical order.

Table 58 Environmental prices (damage costs) for average emissions to water in the Netherlands (€2015/kg emission)

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
1,1'-Biphenyl, 3,3',4,4'-tetrachloro-, PCB-77	032598-13-3	8.10E-05	3.26E-04	6.35E-04
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	057653-85-7	5.45E-04	2.20E-03	4.27E-03
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	040321-76-4	4.74E-01	1.91E+00	3.71E+00
1,3-Dichloro-2-propanol	000096-23-1	1.76E-02	7.10E-02	1.38E-01
1,5,9-Cyclododecatriene	004904-61-4	4.65E-08	1.87E-07	3.64E-07
1-Bromopropane	000106-94-5	1.30E-06	5.23E-06	1.02E-05
2,2-Bis(4-hydroxy-3,5-dibromophenyl)propane	000079-94-7	3.73E-05	1.50E-04	2.93E-04
2,3-Dibromo-1-propanol	000096-13-9	6.92E-06	2.78E-05	5.42E-05
2,3-Dinitrotoluene	000602-01-7	1.74E-02	7.01E-02	1.36E-01
2,4,5,2',5'-Pentachlorobiphenyl	037680-73-2	2.27E-03	9.13E-03	1.78E-02
2,4,6-Tri(tert-butyl)phenol	000732-26-3	1.00E-03	4.04E-03	7.87E-03
2,4-Diaminoanisole sulfate	039156-41-7	n.c.	n.c.	n.c.
2,5-Dinitrotoluene	000619-15-8	1.03E-03	4.13E-03	8.04E-03
2-Butenal	004170-30-3	1.26E-04	5.05E-04	9.83E-04
2-Ethoxyethyl acetate	000111-15-9	4.22E-05	1.70E-04	3.30E-04
2-Methoxyethyl acetate	000110-49-6	9.36E-07	3.77E-06	7.33E-06
3,4-Dinitrotoluene	000610-39-9	1.12E-03	4.49E-03	8.74E-03
3,5-Dinitrotoluene	000618-85-9	3.74E-05	1.51E-04	2.93E-04
4,4'-Methylene di-o-toluidine	000838-88-0	7.01E-01	9.59E-01	1.48E+00
4,4'-Methylenebis-(2-chlorobenzenamine)	000101-14-4	1.11E+00	1.52E+00	2.35E+00
4,4'-Oxybisbenzenamine	000101-80-4	3.93E-01	5.37E-01	8.30E-01
4,4-Thiodianiline	000139-65-1	6.56E-01	8.97E-01	1.39E+00
4-Aminoazobenzene	000060-09-3	3.09E-05	1.24E-04	2.42E-04
Acenaphthene	000083-32-9	4.39E-03	6.13E-03	9.53E-03
Acenaphthene, 5-nitro-	000602-87-9	1.27E+00	1.74E+00	2.68E+00
Acridine	000260-94-6	6.17E-03	2.48E-02	4.83E-02
Acrylamide	000079-06-1	1.41E-01	1.93E-01	2.98E-01
Acrylonitrile	000107-13-1	7.08E-01	9.68E-01	1.50E+00
A-endosulfan	000959-98-8	2.09E+00	8.41E+00	1.64E+01
Aldrin	000309-00-2	3.59E+01	4.91E+01	7.59E+01
Aniline, p-chloro-	000106-47-8	6.43E-02	9.51E-02	1.51E-01
Anisole, pentachloro-	001825-21-4	1.39E-02	5.59E-02	1.09E-01
Anthracene	000120-12-7	8.19E-03	1.16E-02	1.83E-02
Arsenic	007440-38-2	2.16E+01	6.93E+01	1.68E+02
Azobenzene	000103-33-3	5.05E-03	2.03E-02	3.96E-02
Azocyclotin	041083-11-8	1.32E+00	1.85E+00	2.88E+00
Benomyl	017804-35-2	7.44E-04	2.48E-03	4.73E-03
Benz(a)acridine	000225-11-6	1.23E-03	4.95E-03	9.63E-03
Benz(c)acridine	000225-51-4	1.48E-02	5.95E-02	1.16E-01
Benzene	000071-43-2	1.18E-01	1.62E-01	2.50E-01
Benzene, (epoxyethyl)-	000096-09-3	1.80E-02	2.46E-02	3.81E-02
Benzene, 1-(1,1-dimethylethyl)-3,5-dimethyl-2,4,	000081-15-2	n.c.	n.c.	n.c.
Benzene, 1,2,3-trichloro-	000087-61-6	1.63E-03	6.55E-03	1.27E-02
Benzene, 1,2,4-trichloro-	000120-82-1	2.49E-01	3.44E-01	5.33E-01
Benzene, 1,3,5-trichloro-	000108-70-3	5.34E-04	2.15E-03	4.18E-03

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Benzene, 1-methyl-2-nitro-	000088-72-2	1.16E-04	4.67E-04	9.09E-04
Benzene, 2,4-dichloro-1-(4-nitrophenoxy)-	001836-75-5	7.70E-02	1.10E-01	1.73E-01
Benzene, hexachloro-	000118-74-1	2.48E+02	3.39E+02	5.25E+02
Benzene, pentachloro-	000608-93-5	1.77E+01	2.44E+01	3.77E+01
Benzidine	000092-87-5	5.54E-01	7.58E-01	1.17E+00
Benzidine dihydrochloride	000531-85-1	n.c.	n.c.	n.c.
Benzidine, 3,3'-dichloro-	000091-94-1	3.40E+00	4.65E+00	7.18E+00
Benzidine, 3,3'-dimethyl-	000119-93-7	8.69E-05	3.50E-04	6.80E-04
Benzidine, 3,3'-dimethyl-, dihydrochloride	000612-82-8	n.c.	n.c.	n.c.
Benzo(a)anthracene	000056-55-3	1.97E-06	7.95E-06	1.55E-05
Benzo(a)pyrene	000050-32-8	6.84E+01	9.36E+01	1.45E+02
Benzoic acid, 4-(tert-butyl)-	000098-73-7	3.25E-05	1.31E-04	2.54E-04
Benzotrichloride	000098-07-7	2.32E+00	3.17E+00	4.90E+00
Benzyl chloride	000100-44-7	4.97E-02	6.81E-02	1.05E-01
Beryllium	007440-41-7	4.72E+00	9.72E+00	1.73E+01
beta-Naphthylamine	000091-59-8	3.10E-01	4.24E-01	6.55E-01
Binapacryl	000485-31-4	1.64E-03	6.62E-03	1.29E-02
Biphenyl, 4-amino-	000092-67-1	4.37E-01	5.97E-01	9.23E-01
Bis(chloromethyl)ether	000542-88-1	1.93E+03	2.63E+03	4.07E+03
Bisphenol A	000080-05-7	1.27E-02	2.34E-02	3.97E-02
Brodifacoum	056073-10-0	9.93E-07	4.00E-06	7.78E-06
Butadiene	000106-99-0	1.60E-02	2.18E-02	3.37E-02
Butadiene, hexachloro-	000087-68-3	7.50E-03	3.02E-02	5.88E-02
C.I. basic violet 3	000548-62-9	1.43E-02	5.78E-02	1.12E-01
C.I. disperse blue 1	002475-45-8	3.43E-02	4.69E-02	7.24E-02
C.I. solvent yellow 3	000097-56-3	1.72E+00	2.36E+00	3.64E+00
Cadmium	007440-43-9	2.43E+01	2.04E+03	6.25E+03
Carbamic acid, ethyl ester	000051-79-6	2.49E-02	3.40E-02	5.26E-02
Carbendazim	010605-21-7	9.52E-02	2.77E-01	5.18E-01
Carbon monoxide	000630-08-0	n.c.	n.c.	n.c.
Chlordane , pur	000057-74-9	5.10E+02	6.98E+02	1.08E+03
Chlorfenvinphos	000470-90-6	5.04E+01	6.94E+01	1.07E+02
Chloromethyl methyl ether	000107-30-2	1.19E-01	1.63E-01	2.51E-01
Chloroprene	000126-99-8	1.61E-01	2.21E-01	3.41E-01
Chromium	007440-47-3	5.39E-05	6.36E-04	1.76E-03
Copper	007440-50-8	1.18E-02	2.39E-01	6.95E-01
Crotonaldehyde	000123-73-9	n.c.	n.c.	n.c.
Cyclododecane	000294-62-2	4.01E-08	1.61E-07	3.14E-07
Cyclododecane, hexabromo-	025637-99-4	3.58E-03	1.44E-02	2.81E-02
Cycloheximide	000066-81-9	1.05E-04	4.23E-04	8.22E-04
Cyclopentadiene, hexachloro-	000077-47-4	3.58E+01	4.90E+01	7.58E+01
Cyhexatin	013121-70-5	9.75E-01	1.34E+00	2.07E+00
DDT	000050-29-3	1.53E+01	2.10E+01	3.25E+01
Decabromodiphenyl oxide	001163-19-5	7.73E+01	1.06E+02	1.63E+02
Delta-hexachlorocyclohexane	000319-86-8	5.02E-03	2.02E-02	3.93E-02
Dibenz(a,h)anthracene	000053-70-3	1.04E+01	1.42E+01	2.20E+01
Dibenzofuran, 2,3,7,8-tetrachloro-	051207-31-9	1.23E+02	4.93E+02	9.60E+02
Dibutyl dichloro tin	000683-18-1	2.69E-02	1.08E-01	2.10E-01
Dibutyltin oxide	000818-08-6	1.17E-08	4.73E-08	9.20E-08
Dicofol	000115-32-2	2.22E+00	3.04E+00	4.70E+00
Dieldrin	000060-57-1	3.00E+02	4.11E+02	6.36E+02
Difenacoum	056073-07-5	3.44E-06	1.38E-05	2.69E-05

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Di-isobutylphthalate	000084-69-5	4.74E-05	1.91E-04	3.71E-04
Dimethyl formamide	000068-12-2	2.66E-01	3.64E-01	5.62E-01
Dimethylcarbamyl chloride	000079-44-7	1.41E+01	1.93E+01	2.98E+01
Dimethylphenol phosphate (3:1)	025155-23-1	4.80E-09	1.93E-08	3.76E-08
Dinocap	039300-45-3	2.62E-01	3.61E-01	5.60E-01
Dinoseb	000088-85-7	2.31E+01	3.24E+01	5.04E+01
Dinoterb	001420-07-1	1.36E-01	5.47E-01	1.06E+00
Dioxin, 1,2,3,7,8,9-hexachlorodibenzo-	019408-74-3	n.c.	n.c.	n.c.
Dioxin, 2,3,7,8 Tetrachlorodibenzo-p-	001746-01-6	7.51E+05	1.03E+06	1.59E+06
Diuron	000330-54-1	4.02E-01	8.35E-01	1.46E+00
Endosulfan	000115-29-7	2.52E-01	3.64E-01	5.74E-01
Endosulfan (beta)	033213-65-9	1.88E+00	7.57E+00	1.47E+01
Endrin	000072-20-8	2.63E+01	3.88E+01	6.17E+01
Endrocide (endox) (coumatetralyl)	005836-29-3	2.78E-05	1.12E-04	2.18E-04
Epichlorohydrin	000106-89-8	1.63E+00	2.23E+00	3.44E+00
Ethane, 1,2-dibromo-	000106-93-4	6.75E+00	9.23E+00	1.43E+01
Ethane, 1,2-dichloro-	000107-06-2	2.50E+00	3.43E+00	5.29E+00
Ethane, pentachloro-	000076-01-7	8.33E-06	3.36E-05	6.53E-05
Ethanol, 2-ethoxy-	000110-80-5	9.38E-03	1.28E-02	1.98E-02
Ethanol, 2-methoxy-	000109-86-4	4.93E-02	6.75E-02	1.04E-01
Ethene, bromo-	000593-60-2	2.97E-01	4.07E-01	6.29E-01
Ethene, chloro-	000075-01-4	4.95E+00	6.77E+00	1.05E+01
Ethene, trichloro-	000079-01-6	1.76E-02	2.41E-02	3.73E-02
Ethyl O-(p-nitrophenyl) phenylphosphonothionate	002104-64-5	6.25E+01	8.55E+01	1.32E+02
Ethylene oxide	000075-21-8	2.10E-01	2.87E-01	4.43E-01
Ethylene thiourea	000096-45-7	1.27E-01	1.75E-01	2.70E-01
Ethyleneimine	000151-56-4	1.99E+01	2.72E+01	4.20E+01
Fenbutatin oxide	013356-08-6	7.00E+01	9.57E+01	1.48E+02
Fenchlorazole-ethyl	103112-35-2	1.89E-02	7.62E-02	1.48E-01
Fentin acetate	000900-95-8	2.72E+00	3.84E+00	6.01E+00
Fentin chloride	000639-58-7	5.28E+01	7.34E+01	1.14E+02
Fentin hydroxide	000076-87-9	1.71E+00	2.47E+00	3.89E+00
Fluazifop-butyl	069806-50-4	7.64E-04	3.08E-03	5.99E-03
Flucythrinate	070124-77-5	4.31E-01	6.83E-01	1.11E+00
Fluoranthene	000206-44-0	2.34E-02	4.12E-02	6.93E-02
Fluorene	000086-73-7	2.56E-02	3.83E-02	6.11E-02
Flusilazole	085509-19-9	9.36E-01	1.28E+00	1.98E+00
Formaldehyde	000050-00-0	1.51E+00	2.06E+00	3.19E+00
Furan	000110-00-9	7.70E+00	1.05E+01	1.63E+01
Glufosinate ammonium	077182-82-2	7.46E-01	1.03E+00	1.59E+00
Glycidol	000556-52-5	9.56E-01	1.31E+00	2.02E+00
Glycydyltrimethylammonium chloride	003033-77-0	2.97E-08	1.20E-07	2.33E-07
Heptachlor	000076-44-8	9.41E-01	1.29E+00	1.99E+00
Heptachlor epoxide	001024-57-3	1.39E+02	1.90E+02	2.94E+02
Heptachloronorbornene	028680-45-7	1.58E-04	6.36E-04	1.24E-03
Hexachlorocyclohexane	000608-73-1	1.84E+00	2.56E+00	3.99E+00
Hexamethylphosphoramide	000680-31-9	3.63E+02	4.97E+02	7.67E+02
Hydrazine	000302-01-2	5.24E+00	7.21E+00	1.12E+01
Hydrazine, 1,1-dimethyl-	000057-14-7	5.47E-02	9.26E-02	1.54E-01
Hydrazine, 1,2-diphenyl-	000122-66-7	4.55E-06	1.83E-05	3.56E-05
Hydrazine, phenyl-, hydrochloride	000059-88-1	n.c.	n.c.	n.c.
Isobutane	000075-28-5	n.c.	n.c.	n.c.

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Isodrin	000465-73-6	8.47E-03	3.41E-02	6.63E-02
Isoprene	000078-79-5	6.20E-03	8.48E-03	1.31E-02
Isoquinoline	000119-65-3	9.14E-04	3.68E-03	7.16E-03
Kepone	000143-50-0	3.40E+01	4.67E+01	7.23E+01
Lead	007439-92-1	1.07E-01	1.42E+01	4.36E+01
Lindane	000058-89-9	1.65E+01	2.31E+01	3.61E+01
Lindane, alpha-	000319-84-6	5.67E+00	7.80E+00	1.21E+01
Lindane, beta-	000319-85-7	9.33E-01	1.28E+00	1.97E+00
Linuron	000330-55-2	1.94E+00	2.92E+00	4.69E+00
Mercury	007439-97-6	8.64E+02	1.55E+03	2.96E+03
Methoxychlor	000072-43-5	4.85E-02	6.70E-02	1.04E-01
Methylmercury	022967-92-6	2.52E+02	3.45E+02	5.33E+02
Mirex	002385-85-5	4.56E+03	6.23E+03	9.63E+03
Naphthalene	000091-20-3	4.34E-02	5.95E-02	9.20E-02
Naphthalene, 2-methyl-	000091-57-6	3.16E-02	4.35E-02	6.75E-02
Nickel	007440-02-0	3.26E-02	3.42E-01	9.65E-01
Nitroanisole, o-	000091-23-6	5.25E-01	7.19E-01	1.11E+00
Nitrobenzene	000098-95-3	1.77E+00	2.42E+00	3.74E+00
Nitrosoguanidine, N-methyl-N'-nitro-N-	000070-25-7	1.64E+01	2.25E+01	3.47E+01
Nitrous acid, 2-methylpropyl ester	000542-56-3	n.c.	n.c.	n.c.
N-Nitrosodiethanolamine	001116-54-7	7.00E-01	9.57E-01	1.48E+00
N-Nitrosodimethylamine	000062-75-9	3.66E+01	5.01E+01	7.74E+01
N-Nitrosodipropylamine	000621-64-7	8.50E+01	1.16E+02	1.80E+02
N-nonylphenol	084852-15-3	9.89E-04	3.98E-03	7.75E-03
Nonylphenol	025154-52-3	1.64E-04	6.62E-04	1.29E-03
O,p'-ddt	000789-02-6	2.98E-03	1.20E-02	2.34E-02
o-Aminoanisole	000090-04-0	2.31E-05	9.30E-05	1.81E-04
o-Toluidine	000095-53-4	6.08E-08	2.45E-07	4.77E-07
o-Toluidine, 4-chloro-, hydrochloride	003165-93-3	1.16E-01	1.59E-01	2.46E-01
Oxirane, (phenoxymethyl)-	000122-60-1	7.75E-02	1.06E-01	1.64E-01
P-(1,1,3,3-tetramethylbutyl)phenol	000140-66-9	2.01E-03	8.10E-03	1.58E-02
p-Cresidine	000120-71-8	1.58E-02	2.16E-02	3.34E-02
Pentabromodiphenyl ether	032534-81-9	1.96E+01	2.68E+01	4.13E+01
Phenanthrene	000085-01-8	1.54E-04	6.19E-04	1.20E-03
Phenanthridine	000229-87-8	6.77E-05	2.73E-04	5.30E-04
Phenol, pentachloro-	000087-86-5	6.77E-03	9.42E-03	1.46E-02
Phenolphthalein	000077-09-8	3.01E-03	4.11E-03	6.35E-03
Phenyl hydrazine	000100-63-0	7.82E-06	3.15E-05	6.13E-05
Phenylmercuric acetate	000062-38-4	1.39E+02	1.92E+02	2.97E+02
Phosphate, tris(2-chloroethyl)-	000115-96-8	1.02E-04	4.12E-04	8.01E-04
Phthalate, butyl-benzyl-	000085-68-7	5.12E-03	7.59E-03	1.21E-02
Phthalate, dibutyl-	000084-74-2	1.34E-02	2.16E-02	3.54E-02
Phthalate, dihexyl-	000084-75-3	2.40E-04	9.66E-04	1.88E-03
Phthalate, dioctyl-	000117-81-7	2.61E-01	3.58E-01	5.52E-01
P-nonylphenol	000104-40-5	1.18E-04	4.76E-04	9.26E-04
Polychlorinated biphenyls	001336-36-3	1.07E-03	4.30E-03	8.36E-03
Propane sultone	001120-71-4	3.21E+00	4.39E+00	6.78E+00
Propane, 1,2,3-trichloro-	000096-18-4	1.83E+01	2.51E+01	3.87E+01
Propane, 1,2-dibromo-3-chloro-	000096-12-8	2.83E+01	3.88E+01	5.99E+01
Propane, 1,2-dichloro-	000078-87-5	2.10E+01	2.87E+01	4.43E+01
Propane, 2-nitro-	000079-46-9	2.63E-01	3.60E-01	5.56E-01
Propiolactone	000057-57-8	3.37E+00	4.60E+00	7.11E+00

Pollutant	CAS registry number	Lower value €/kg	Central value €/kg	Upper value €/kg
Propylene oxide	000075-56-9	6.37E-01	8.72E-01	1.35E+00
P-tert-amylphenol	000080-46-6	1.68E-04	6.77E-04	1.32E-03
Pyrene	000129-00-0	2.05E-01	2.82E-01	4.37E-01
Quinoline	000091-22-5	3.39E-04	1.36E-03	2.65E-03
Safrole	000094-59-7	4.62E-03	6.32E-03	9.76E-03
Sulfallate	000095-06-7	8.11E-02	1.11E-01	1.72E-01
Sulfuric acid, dimethyl ester	000077-78-1	2.11E-07	8.48E-07	1.65E-06
Tetrabutyltin	001461-25-2	3.62E-08	1.46E-07	2.83E-07
Tetraethyl lead	000078-00-2	1.16E+03	1.59E+03	2.45E+03
Tetrahydrofurfuryl alcohol	000097-99-4	2.56E-08	1.03E-07	2.00E-07
Tetramethyl lead	000075-74-1	6.96E-05	2.80E-04	5.45E-04
Tetramethyldiaminobenzophenone	000090-94-8	n.c.	n.c.	n.c.
Tetrasul	002227-13-6	7.52E-06	3.03E-05	5.89E-05
Thioacetamide	000062-55-5	1.97E-01	2.70E-01	4.17E-01
Toluene, 2,4-diamine	000095-80-7	8.86E+00	1.21E+01	1.87E+01
Toluene, 2,4-dinitro-	000121-14-2	8.08E+00	1.11E+01	1.71E+01
Toluene, 2,6-dinitro-	000606-20-2	9.16E+01	1.25E+02	1.94E+02
Toluene, dinitro-	025321-14-6	n.c.	n.c.	n.c.
Toxaphene	008001-35-2	1.25E+01	1.81E+01	2.85E+01
Tributylstannane	000688-73-3	1.60E-05	6.46E-05	1.26E-04
Tributyltin oxide	000056-35-9	1.81E+00	3.94E+00	6.98E+00
Trichlorobenzenes	012002-48-1	1.19E-04	4.77E-04	9.29E-04
Triflumizole	068694-11-1	3.61E-02	1.45E-01	2.83E-01
Trifluralin	001582-09-8	1.20E+00	1.64E+00	2.54E+00
Trimethylaniline hydrochloride, 2,4,5-	021436-97-5	n.c.	n.c.	n.c.
Trimethylaniline, 2,4,5-	000137-17-7	n.c.	n.c.	n.c.
Vinclozolin	050471-44-8	4.25E-01	5.88E-01	9.11E-01
Warfarin	000081-81-2	9.31E+00	1.27E+01	1.97E+01

N.c = not calculated or no impact.