Environmental Prices Handbook 2024: EU27 version

Methodical justification of key indicators used for the valuation of emissions and the environmental impact

CE Delft

Committed to the Environment

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Version 1.0

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Summary

What are environmental prices?

Environmental prices are constructed prices for the social cost of pollution, expressed in euros per kilogram pollutant. Environmental prices therefore reflect the loss of economic welfare that occurs when one additional kilogram of a pollutant finds its way into the environment. These prices can also be calculated for immaterial forms of pollution such as noise nuisance and ionising radiation. Environmental prices are used in analyses where the financial magnitude needs to be weighed against the environmental impact. Expressing the environmental impact in terms of damage in euros allows it to be weighted and compared to financial parameters, such as in social cost-benefit analyses, social business cases and life cycle analyses.

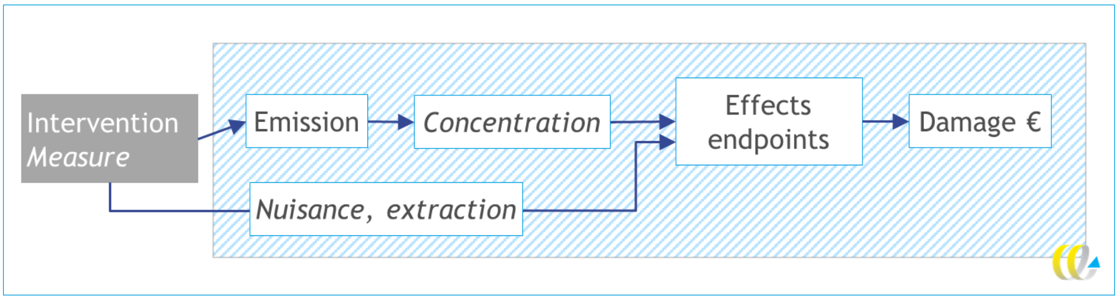
In the absence of a market for environmental quality, environmental prices cannot directly be empirically observed, but must be calculated. CE Delft has been calculating environmental prices since 1997 and presenting this in the form of handbooks since 2010. The calculations in the Environmental Prices Handbook need to be periodically updated to reflect new scientific findings on the relationship between emissions and economic welfare losses. For the Netherlands, the latest update was done in 2023. The Environmental Prices Handbook 2024 for the EU incorporates these new findings as well, and assigns a completely new valuation to all pollutants. The Environmental Prices Handbook 2024 EU version therefore replaces the earlier Environmental Prices Handbook for the EU from 2018.

The current Handbook provides key indicators for the valuation of emissions to air, water and soil for more than 3,000 environmentally damaging pollutants. In addition, this Handbook includes various prices that can be used to value the outcome of life cycle assessments and process these into a ‘single score’. The Environmental Prices Handbook is frequently used in cost-benefit analyses, in life cycle analyses and in reports by companies and other institutions on their impact on society.

Environmental prices methodology

Environmental prices are determined based on a cause-effect relationship between emissions, the environmental impact and damage. The cause-effect relationship is depicted in Figure 1.

Figure 1 - Relevant cause-effect relationships in this study

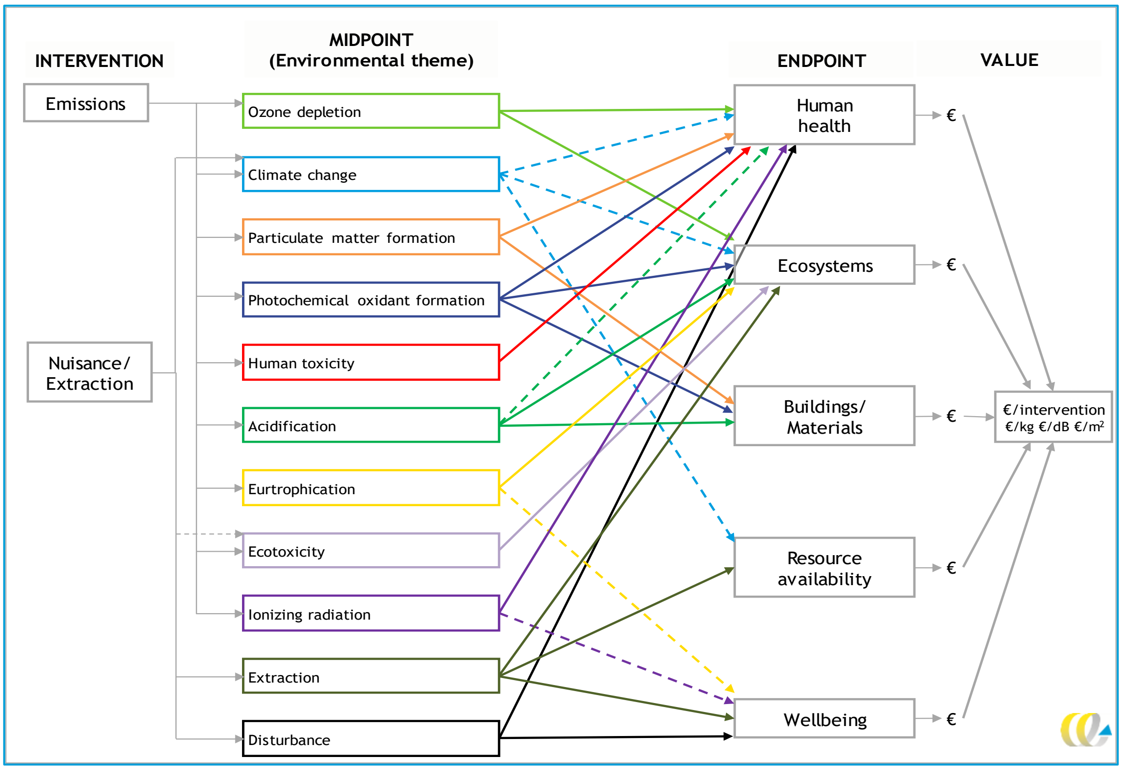


Any activity leads to a certain intervention in the environment. This could be emissions, nuisance or extraction, such as the consumption of water or the use of land or raw materials. In the case of emissions, these are transported via air, soil or water to other areas, where they contribute to a change in existing emission concentrations. This altered concentration then leads to changes in aspects relevant to human welfare, such as health or biodiversity. These 'aspects relevant to human welfare' are referred to as 'endpoints' in environmental science. All relationships located in the shaded part of Figure 1 are included within the scope of this Environmental Prices Handbook.

The environmental prices in this Handbook apply to emissions from an average emission source at an average emission location in the year 2021. The prices are presented at three levels:
1. At pollutant level, such as emissions of environmentally harmful pollutants to air, water and soil, such as CO_2, particulate matter, phosphate or cadmium.
2. At the level of environmental themes, known as midpoints, such as climate change, acidification or ecotoxicity.
3. At the welfare level, known as endpoints, such as the valuation of the impact of environmental pollution on human health, ecosystem services, capital goods, raw materials and well-being.

This Environmental Prices Handbook consists of an integrally coherent analytical framework that presents the relationship between emissions and welfare effects in physical and monetary terms. This framework is outlined in Figure 2 on the next page. This identifies all relationships between emissions, midpoints and endpoints and their valuation that are relevant to this Environmental Prices Handbook.

Figure 2 - Relationship between intervention, midpoints, endpoints and valuation in the Environmental Prices Handbook



Solid lines refer to relationships that have been investigated and partly quantified within the framework of this Handbook. The dashed lines represent relationships that are not directly quantified as relationships because a different approach was taken in this Handbook for quantifying the impact. Depletion includes land use. Disturbance also includes noise pollution. See Chapter 4 for a further explanation.

Results: environmental prices at pollutant level

Pollutant level is the most commonly used level in the analysis and provides information on the cost of environmental pollution per kilogram of emissions. This Environmental Prices Handbook presents environmental prices for more than 3,000 environmentally hazardous pollutants. Table 1 provides an overview of valuations of the most common pollutants for emissions to air.

Table 1 - Environmental prices for emissions of air pollutants in the EU27, in €$_{2021}$/kg

Pollutant	Pollutants name	Lower	Central	Upper
CO_2	Carbon dioxide	€ 0.050	€ 0.130	€ 0.160
$PM_{2.5}$	Particulate matter	€ 58.5	€ 95	€ 134
PM_{10}	Particulate matter	€ 31.3	€ 51.6	€ 73.3
NO_x	Nitrogen oxides	€ 13.5	€ 21.5	€ 31.8
SO_2	Sulphur dioxide	€ 17.8	€ 30.5	€ 45.3
NH_3	Ammonia	€ 18.2	€ 28.7	€ 39.5
NMVOC	Volatile organic compounds (non-methane)	€ 1.62	€ 2.49	€ 3.49
CH_4	Methane	€ 1.80	€ 4.68	€ 5.77

Chapter 2 presents environmental prices for many more pollutants, for emissions to air, water and soil. At the pollutant level, lower and upper values are recommended for use in SCBAs and central values are recommended for other uses.

Results: environmental prices at midpoint level

At the midpoint level, an environmental prices valuation is presented in terms of environmental themes. This midpoint level can be used as a weighing ratio in life cycle assessments (LCA) or to calculate the external costs of certain materials or products. Table 2 presents an overview of the calculated midpoint-level prices for the ReCiPe 2016 characterisation model. Chapter 6 contains an explanation of all calculations per midpoint.

Table 2 – Environmental prices for LCA: ReCiPe 2016 midpoints for the EU27, in €$_{2021}$ per unit

	Unit		Lower		Central		Upper
Climate change	€/kg CO_2-eq.	€	0.05	€	0.13	€	0.16
Ozone depletion	€/kg CFC-11-eq.	€	15.2	€	29.1	€	69.6
Ionising radiation	€/kBq Co-60-eq.	€	0.00275	€	0.00422	€	0.00594
Oxidant formation, human health	€/kg NO_x-eq.	€	1.38	€	2.17	€	2.98
Oxidant formation, terrestrial ecosystems	€/kg NO_x-eq.	€	0.416	€	0.416	€	0.526
Particulate matter formation	€/kg $PM_{2.5}$-eq.	€	61.7	€	99.2	€	138.1
Acidification	€/kg SO_2-eq.	€	2.66	€	5.27	€	9.30
Freshwater eutrophication	€/kg P-eq.	€	2.56	€	3.74	€	10.13
Marine eutrophication	€/kg N-eq.	€	7.64	€	14.25	€	27.60
Terrestrial ecotoxicity	€/kg 1.4-DCB-eq.	€	0.00045	€	0.00064	€	0.00083
Freshwater ecotoxicity	€/kg 1.4-DCB-eq.	€	0.0148	€	0.0209	€	0.0271
Marine ecotoxicity	€/kg 1.4-DCB-eq.	€	0.0022	€	0.0032	€	0.0041
Human toxicity, cancer-related	€/kg 1.4-DCB-eq.	€	2.70	€	3.99	€	6.01
Human toxicity, non-cancer-related	€/kg 1.4-DCB-eq.	€	0.048	€	0.071	€	0.106
Land use	€/m^2 a crop-eq.	€	0.070	€	0.099	€	0.128
Mineral extraction	€/kg Cu-eq.	€	0	€	0.0140	€	0.0826
Fossil extraction	€/kg oil-eq.	€	0	€	0.028	€	0.163
Water consumption	€/m^3	€	0	€	0.407	€	0.811
NO_2-mortality *	€/kg NO_x-eq.	€	4.31	€	6.37	€	9.62

* The NO_2 mortality is an extra calculation step that can be performed in addition to the LCA analysis to ensure that NO_2 is included in the external cost estimates properly.

Unlike the previous Environmental Prices Handbook, this Handbook assigns a value to all midpoints from ReCiPe.In addition to environmental prices for ReCiPe midpoints for the EU, the Environmental Prices Handbook also provides environmental prices for a portion (CAT I and II) of the PEF for the EU27.

These prices can be found in Paragraph 2.4 of this Handbook.

Dose-effect relationships, valuation and uncertainty

Environmental prices have been developed based on dose-effect relationships that have been determined for individual pollutants. For this purpose, we took a conservative assumption that mainly included dose-effect relationships recommended by the World Health Organization (WHO) or extensively documented in meta-analyses. For environmental modelling, we chose the most conservative assumption of dose-effect relationships in the low and central variant of the prices. It should be remembered that this may underestimate the actual damage: for many pollutants, individual studies are available showing that the range of damage to health or ecosystems could be much greater. Because these studies have not yet been sufficiently replicated in follow-up studies, however, no definitive statement can be made as to whether this provides robust scientific results.

Nevertheless, it is clear that science is advancing over time and is getting better at estimating the harmful effect of emissions. This also explains why the environmental prices in this Handbook have increased significantly for a number of pollutants from the Environmental Prices Handbook 2018 EU version. It is anticipated that a subsequent version of the Handbook will again arrive at higher values as more is known about the adverse effects of environmental pollutants. It is therefore most appropriate to think of current environmental prices as conservative estimates about the actual social costs of environmental pollution.

This Handbook is far from complete, despite presenting environmental prices for more than 3,000 pollutants. There are very many chemical substances whose effects on human health or ecosystem services are unknown. If no environmental prices for a pollutant is included in this Handbook, it does not mean that the damage is zero. This applies in particular to bioaccumulative substances, such as PFAS. The Handbook does not provide sufficient guidance on how to value pollutants that do not degrade in the environment. In such cases, it is better to perform a dedicated study to determine the dispersion of the toxic substances in the environment, their uptake in humans, plants and animals and the effects of that uptake on human health or ecosystem services, including a risk analysis of the fact that these substances no longer disappear from the environment. This Handbook therefore cannot be used in SCBAs on bioaccumulative substances.

Using environmental prices

Environmental prices can be used as a calculation tool in studies and practical applications by both the government and industry. Three main areas of application can be distinguished:
1. **Social cost-benefit analyses** (SCBA) in which the welfare effects of a policy measure or investment are calculated. Environmental prices are used to value the environmental impact in an SCBA.
2. **Life Cycle Analysis** (LCA) in which the environmental impact of a product or process is determined from cradle to grave. Environmental prices allow LCA researchers to weigh the environmental impact of an LCA to produce a ‘single score’. Environmental prices have been included as a weighing ratio in popular LCA software, such as SimaPro. Over 100 scientific publications have been published that use environmental prices as a weighing ratio to determine the external costs of a product or process.
3. **Corporate Social Responsibility** (CSR). Environmental prices can be used in annual environmental reports, social business cases or to prepare environmental profit and loss accounts. Environmental prices are used by many companies, such as Philips, Samsung, Knauf, Vodafone and Repsol, to optimise operations with respect to environmental impact and to report transparently on the progress of sustainability policies.

Environmental prices are shown for lower, central and upper variants. For use in SCBAs, we recommend the lower and upper values so as to reflect uncertainties, such as in valuations, in the SCBA balances. For use in LCAs and CSR, we recommend the central value because it contains the most likely outcome in light of all uncertainties.

Environmental prices are calculated as the average of emissions at an average location in the EU. For some of the commonly occurring pollutants, the handbook provides a specification to the source of the emission (see Paragraph 6.4.11). Environmental prices are less suitable for site-specific studies, such as studies into emissions from a specific factory, because the emission situation may differ from the average for the EU.

Environmental Prices Handbook overview

This Handbook consists of three sections:

Section 1, Chapters 1 and 2 are focused on the user. Chapter 1 provides accountability for the research process and discusses the basic assumptions, and Chapter 2 presents environmental prices for the main pollutants and explains their use in specific situations.

Section 2 is the methodological part and includes Chapters 3 to 7.
Chapter 3 includes the methodological framework and Chapter 4 describes the approach taken to arrive at the environmental prices. Chapter 5 describes the valuation framework and Chapter 6 describes the treatment of the Impact Pathway approach adopted for each environmental theme. Finally, Chapter 7 compares the current environmental prices with the prices in the previous Handbook and with other studies and includes recommendations for the use of environmental prices in the future.

Section 3 includes the annexes which provide further elaboration and insight into the methodological part. For example, Annex H contains environmental prices for emissions of more than 250 environmentally hazardous pollutants to air, soil and water.

List of abbreviations

Abbreviation	Meaning
AGF	Age Group Functions
AOT 40 value	Accumulated Ozone Concentration above a Threshold of 40 ppbV
Bq	Becquerel
CAFE-CBA	Cost-benefit Analysis for Clean Air for Europe (EU research programme)
CASES	Cost Assessment for Sustainable Energy Systems
CBA	Cost-Benefit Analysis
CFK	Chlorinated fluorinated hydrocarbons
COI	Cost of Illness
CPI	Consumer Price Index
CRF	Concentration Response Functions
CVM	Contingent Valuation Method
DALY	Disability-Adjusted Life Year
dB	Decibel
EEA	European Environment Agency (Europees Milieuagentschap)
EDP	Ecosystem Damage Potential
EMEP	European Monitoring and Evaluation Programme
GDP	Gross Domestic Product
GHG	Greenhouse gases
GWP	Global Warming Potential
HICP	Harmonised Index of Consumer Prices
IPA	Impact Pathway Approach
CBA	Cost-benefit analysis
LCA	Life cycle analysis
LCIA	Life Cycle Impact Assessment
LRS	Lower Respiratory Symptoms
LYL	Life Years Lost
SME	Small and medium-sized enterprises
SCBA	Social cost-benefit analysis
MRAD	Minor Restricted Activity Days
CSR	Corporate Social Responsibility
NEEDS	New Energy Externalities Developments for Sustainability
NMVOC	Non-Methane Volatile Organic Compounds
NPV	Net Present Value
ODS	Ozone-Depleting Substances
PAF	Potential Affected Fraction of Species
PBL	PBL Netherlands Environmental Assessment Agency (Planbureau voor de Leefomgeving)
PBq	PetaBequerel
PDF	Potential Disappeared Fraction of species
PEF	Product Environmental Footprint
PM_{10}	Particulate matter less than 10 micrometres in diameter
$PM_{2.5}$	Particulate matter less than 2.5 micrometres in diameter
$PM_{0.1}$	Particulate matter less than 0.1 micrometres in diameter
PPP	Purchasing Power Parity
QALY	Quality-Adjusted Life Year
RAD	Restricted Activity Days

Abbreviation	Meaning
REACH	Regulation, Evaluation and Authorisation of CHemicals
RWS	Public Works and Water Management (Rijkswaterstaat)
SCC	Social Cost of Carbon
SIA	Secondary Inorganic Aerosols
SOA	Secondary Organic Aerosols
SP	Stated Preference
SRM	Source-receptor matrices
VEDP	Value of Ecological Damage Potential
VOLY	Value of Life Year
VOC	Volatile Organic Compounds
VPF	Value of Prevented Fatality
VSL	Value of Statistical Life
WHO	World Health Organisation
WLD	Work Loss Days
WTA	Willingness-To-Accept
WTP	Willingness-To-Pay
WVOW	Pollution of Surface Waters Act (Wet verontreiniging oppervlaktewateren)
YLD	Years Lived with Disability
YOLL	Years Of Life Lost

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SECTION 1: USER GUIDE

1 Introduction

1.1 Background

Environmental prices refer to key indicators that calculate the social damage of environmental pollution and express it in euros per kilogram of pollutants. Environmental prices reflect the loss of economic welfare that occurs when one additional kilo of the pollutant is released into the environment. Moreover, environmental prices can also be applied to non-material pollution, such as noise.

There is no market for environmental quality, which means that environmental prices cannot be directly observed empirically. Instead, they must be calculated using the results of studies on human preferences for avoiding the impact of pollution. In many cases, this means that environmental prices correspond to the external costs.

Since 1997, CE Delft has published studies on the valuation of environmental pollution in the Netherlands. Since 2010, this has taken the form of Handbooks: in the Shadow Prices Handbook 2010 (CE Delft, 2010), the Environmental Prices Handbook 2017 for the Netherlands (CE Delft, 2017a),and the Environmental Prices Handbook EU version from 2018 (CE Delft, 2018b). This EU version of the Dutch Environmental Prices Handbook 2023 provides key indicators for the valuation of emissions to air, water and soil for more than 2,500 environmentally hazardous pollutants, plus prices for land use and noise. In addition, this Handbook includes a valuation of the outcomes of life cycle analyses.

The Environmental Prices Handbook is frequently used in cost-benefit analyses, in life cycle analyses and in reports by companies and institutions on their social impact. The valuations in the Environmental Prices Handbook should be periodically updated to reflect new scientific understanding of the relationship between emissions and welfare, as well as current research on valuations and prices. The Environmental Prices Handbook 2023 incorporated these new insights and assigned a completely new valuation to all pollutants. The Environmental Prices Handbook 2023 therefore replaces the Environmental Prices Handbook 2017. The EU version of this Environmental Prices Handbook presents the EU prices according to these new insights. It therefore replaces the previous Environmental Prices Handbook EU version from 2018.

1.2 Why environmental prices?

A clean environment provides all kinds of important services for society: clean air and clean drinking water are important conditions for human health; nature contributes to an attractive living environment and to people’s well-being, and nature provides products (food crops but also genetic diversity) that form the basis of human life on earth.

These services therefore have obvious value to society but it is difficult to express this value in a price: for example, there is no shop where one can buy clean air. This is why economists speak of ‘missing markets’ or ‘market failure’. One consequence of this market failure is that environmental value is not sufficiently considered in all kinds of decisions.

Prices play an important role in modern societies. When we enter a shop, we see a lot of products with prices. These prices help us decide whether to buy product A or product B, or both, or not to buy anything in a particular shop but to buy a product at another shop down the road. At stock exchanges, prices are used to trade in companies, goods, physical products and financial products such as derivatives. Online, billions of prices are available at any given moment. Based on these prices, traders, investors, corporations, consumers and producers decide whether to buy or sell.

Market prices are thus a key variable steering the economic process. They reflect what consumers are prepared to pay for a given product or service. In general, if the price goes up fewer consumers will want to buy the product. For the marginal consumer, the price indicates the precise amount of income he or she is willing to spend on that product or service. Prices basically reflect the value that society, at the margin, is prepared to pay for a given product or service.

Not all products or services are traded in the marketplace. These are things that are currently classified as welfare beyond GDP. These include products or services, such as dykes, safety on the streets, social norms, amount of leisure time, beautiful nature or a clean environment. All of these things are not traded in the marketplace. But although these things do not have a direct 'price', everyone will agree they are important for the welfare of a country's citizens. An unsafe country, with no standards of decency, where no one has any leisure time, where floods occur in heavily polluted areas and where there is no nature left, begins to approximate Dante's inferno.

There are various ways to ensure that the value of non-priced goods and services are factored into business, consumer and government decisions. One of these ways is to value these goods so that they can be included in economic analysis tools, such as investment, return or social cost-benefit analyses or socio-financial reporting.[1] This creates an integrated decision-making framework that incorporates the value for a clean environment using environmental prices. Such environmental prices can also serve as a basis for taxes or voluntary contributions that ensure that consumers pay for the external costs of their consumption. Finally, environmental prices can also be used in business processes where an attempt is made to include environmental values in decision-making using internal transfer prices.

Environmental prices are not empirically observable: the price for 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)). Most of the studies assume the *damage* caused by environmental pollution. 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.

[1] There are also other ways: the government can also ban the use of certain substances, for example, to eliminate external costs.

1.3 Use of environmental prices

Environmental prices are frequently used in studies and for practical applications by the government, industry and NGOs. In general, three user goals can be described:

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. The 'Environmental Prices Handbook' of (CE Delft, 2017a) has been recommended by the Dutch Parliament for use in cost-benefit analyses that have a significant environmental impact (Ministerie van I&M, 2017). In Europe, a separate European Handbook on External Costs of Transport (Europees Handboek over Externe kosten van Transport) has been prepared on behalf of DG Move, applicable to transport emissions (CE Delft et al., 2019).

2. **Corporate Social Responsibility (CSR) and benchmarking:** companies and other organisations 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 environmental prices are a useful tool for this purpose. Environmental prices have been used in environmental annual reports by companies such as Dutch Railroads (Nederlandse Spoorwegen) (NS, 2014); (Philips, 2018) and (Vodafone, 2015) to prepare social or environmental profit and loss statements. In addition, companies such as Repsol and Knauf Insulation use environmental prices in their internal business tools. Environmental prices can also be used to compare the environmental performance of a company or organisation with other companies or organisations. This occurs, for example, with the Environmental Barometer for Small and Medium-sized Enterprises developed by the Stimular Foundation (Stimular, 2022).

3. **Weighting in life cycle assessment (LCA)**: in environmental analyses (such as a life cycle analysis or environmental impact assessment), the effects of a product are represented in scores on environmental themes. Environmental prices allow all of these effects to be added together. This creates a 'single score' based on a product's external costs. The Environmental Prices Handbook is used in more than 100 scientific publications as a method for weighting and monetising environmental impact.[2] The valuations in the Environmental Prices Handbook are also used as a weighting method in popular LCA software, such as SimaPro.

In Chapter 2, we provide specific points to be considered for use in these three applications.

[2] Information based on Google Scholar, 23 September 2022.

1.4 Purpose and scope of this study

1.4.1 Research objective and explanation

The objective of this study is fourfold:

1. To develop a set of scientifically robust and consistent environmental prices for the EU28 for pollutant emissions and environmental impacts at midpoint and endpoint level, based on the earlier handbook for the Netherlands.
2. To make this set of valuations as comprehensive as possible in terms of types of environmental impact and the number of pollutants included.
3. To make this set of valuations 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 valuations widely available by means of an interactive online interface guaranteeing consistent use of the Environmental Prices Handbook across all types of users.

The Environmental Prices Handbook frequently uses the terms ‘midpoints’ and ‘endpoints’. These terms are derived from life cycle analysis and have been adopted by us. They mean the following:

Midpoints indicate the contribution of an emission to a specific environmental impact. Examples of midpoints are climate change and acidification. In this process, several pollutants create a similar environmental impact: this similar environmental impact is referred to as ‘midpoints’. Another older word for midpoints is ‘environmental themes’.

Endpoints are defined as the ultimate damage caused by the environmental impact on people and nature. These include a wide range of damage to human health, ecosystem services, human-produced capital, resource extraction and general well-being. It is these endpoints that matter for prosperity.

For example, climate change at the midpoint level describes the increase in temperatures relative to pre-industrial levels. At the endpoint level, climate change then includes the damage caused by that rise in temperature. The purpose of this Environmental Prices Handbook is therefore to develop environmental prices at the level of individual pollutants (emissions), midpoints (environmental themes) and endpoints (things important to welfare).

1.4.2 Scope

Environmental prices are based on damage costs. A valuation of *additional* damage caused by a kilogram of extra emissions can be determined by assessing and valuing the damage caused by environmental pollutants at various endpoints.

The environmental prices reported in this study refer to the *average* prices for the year 2021 per kilogram of emissions from an *average* source at an *average* location (with average population density and average income, for example). Environmental prices are therefore rough-and-ready estimates that are not necessarily valid in specific cases. For particulate matter and noise, traffic-specific valuations have also been reported in this Handbook. For particulate matter and NO_x, a distinction has also been made according to the location of the emissions (Paragraph 7.4).

Basically, environmental prices reflect the social value of environmental pollution for emissions in 2021. These prices can also be used for situations in 2022, 2023 or 2024 (except for major societal disruptions). Guidelines for adjusting environmental prices for inflation are given in Chapter 7, Paragraph 7.4. These include the recommendation to redetermine environmental prices after 5-7 years as the values are then in danger of becoming outdated and inconsistent with underlying societal preferences and scientific knowledge on dose-effect relationships.

1.4.3 Range

Three sets of environmental prices are reported in this Handbook:

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 Dutch SCBAs, as laid down in the new Dutch General SCBA Guidelines (Algemene Leidraad) (CPB & PBL, 2013).[3]

C): A central value derived according to economic principles that can be used by companies in their CSR efforts and as a weighing ratio in LCAs.

The range presented here addresses common uncertainties surrounding the valuation and impact assessment of environmental assets. However, this range does not exactly coincide with a margin of uncertainty. The uncertainty margin is likely to be larger than the range presented here (see also discussion in Annex E).

1.5 Limitations

This Handbook presents sets of environmental prices and weighing ratios for use as key indicators in economic and environmental analyses. These prices are the average values for emissions from an average source in the EU in 2019. CE Delft takes responsibility for calculating environmental prices. Responsibility for the application of environmental prices lies with the user. However, guidelines are being developed for this project regarding *which* sets of environmental prices or weighing ratios should be used, depending on user demand.

The user application tools identified in this Handbook are:
- external cost estimates and social cost-benefit analyses;
- life cycle analyses;
- tools for corporate social responsibility, such as benchmarking.

However, the Environmental Prices Handbook does not include a user manual on how to set up these tools. Therefore, typical issues involved in these analyses, such as system boundaries, sensitivity analyses, distributional effects, allocation issues, etc., are not addressed here.

The objective of the research in the Environmental Prices Handbook is to develop concrete and consistent sets of environmental prices and weighing ratios that can be used in practice. These estimates have been made by CE Delft based on the latest scientific insight.
These estimates were submitted and discussed with the Guidance Committee that included representatives of the planning agencies and scientific experts (see Paragraph 1.8).
The estimates have been adjusted in line with the comments received. When choosing methods, we focused on what is currently considered *mainstream* in the science surrounding valuation, characterisation and weighting, with a slight preference for what is *recent*.
This means there are *alternative* valuation and weighting methods available. While these are mentioned here (along with references), they are discussed only briefly, and with the purpose of explaining how they compare to the methodology adopted in this Handbook.

[3] For SCBAs in other countries, different rules may apply.

Given the very extensive literature on valuation and weighting, it is not feasible to summarise all the methods currently in use. Those using environmental prices or weighing ratios developed in this Handbook must therefore judge for themselves whether the figures presented here are preferable to those cited in other publications (see also Chapter 7).

Unless otherwise stated, the environmental prices presented here are expressed in €/kg emission.[4] Environmental prices have been determined as average values for emissions or other impact, such as noise nuisance, in the EU27. Users of environmental prices should make their own judgement as to whether these average values can be used in a specific application, such as cost-benefit or life cycle analyses. Given that the justification for such choice will always depend on the specific issue for which environmental prices are used, the question of whether the use of national averages is justified cannot be answered by us in this study. Local conditions, such as population density, pre-existing pollution and locally applicable limit values, may mean that the figures presented here are not readily applicable at the local level (e.g., municipality or province). Nor can additional effects in other countries, including developing nations, be determined using environmental prices. It is possible, however, to set up a benefit transfer of the values presented here with those of other countries.[5] Finally, the use of environmental prices is also highly contingent on the pollution source: transport emissions are far more damaging to human health than average emissions, for instance, because they occur closer to the ground, which means that a higher proportion of emissions enter the human body. For some pollutants (e.g. particulate matter, NO_x), we differentiated prices to location sources in this Handbook, but we have not done this for all emissions (e.g. benzene). Users must therefore make their own assessment of whether environmental prices used for their specific purpose corresponds to averages for the EU.

All the environmental prices and weighing factors presented here are always 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 a multitude of uncertain factors. The formal treatment of uncertainty in this study (see Annex E) shows that the variations can be very significant. It is important to note here that this variation applies not only to environmental prices, but in fact to all studies that rely on valuations of environmental goods although these usually do not carry out a formal treatment of uncertainty. The fact that the uncertainties in CE Delft's environmental prices are mentioned should certainly not lead to the conclusion that CE Delft's environmental prices are therefore more uncertain than other valuation methods that do not mention the uncertainties. For the user of environmental prices, it is a question of choosing the lesser evil though: either one refrains from using environmental prices with the consequence that financial data cannot be compared with environmental impact and those impacts cannot be mutually compared, or one does use environmental prices, but recognises that the results have a degree of uncertainty. This choice will depend in part on the purpose for which environmental prices are used and how certain one wants the final result to be. In some cases, sensitivity analyses can help make the uncertainties more transparent.

4 For noise pollution, radiation, land use and resource depletion, other units apply.

5 Benefit transfers are calculations that allow the value of one region to be applied to another. CE Delft developed the Benefito model for this purpose (CE Delft, 2011).

1.6 Differences from the previous Environmental Prices Handbook

The current Environmental Prices Handbook is an update of the previous Environmental Prices Handbook. **Environmental Prices Handbook EU version 2024 therefore replaces Environmental Prices Handbook EU version 2018**.
The methodology for determining environmental prices has remained essentially the same (see Chapter 4), but almost everything in its internal workings has changed. The most notable changes are the following:

- The NEEDS model for determining the effects of air pollution on human health from 2008 has been replaced by the results of EEA research from 2021 (EEA, 2021b). The 2021 EEA study has much more sophisticated atmospheric modelling and is more in line with the current state of science in terms of impact assessment than the NEEDS study.

- The ReCiPe model for determining the relative environmental impact and life cycle analyses is based on the 2016 version rather than the 2009 version. This is more than a simple update, since ReCiPe 2016 has a different determination method for most midpoints than the 2009 version.

- All demographic and epidemiological data on mortality and morbidity are now adjusted to the year 2019 (the year before the corona pandemic).

- All prices have been adjusted to the 2021 price level. In addition, a Paragraph 7.4 is included on how environmental prices can be adjusted for future inflation.

- The valuation for human toxicity has been adjusted using EEA estimates and in accordance with the methodology described in CE Delft (2022b).

- Valuations for human health and biodiversity have been updated to reflect the latest scientific insight and incomes for the year 2021.

- The valuation for CO_2 emissions has been updated to reflect the latest scientific literature.

- The valuations for noise emissions and damage to buildings have been updated to reflect the latest scientific findings and now include, for example, a valuation for noise below 50 dB.

- A separate LCA midpoint category has been formulated for nitrogen-related impact, such as eutrophication and damage caused by NO_2. This should be considered separately in the environmental impact determination by researchers using environmental prices in an LCA.

- For life cycle analysis results, valuations at the EU27 level have been developed and, in addition to midpoint prices from ReCiPe 2016, a start has been made on developing midpoint prices according to the European PEF.

In Chapter 2 we elaborate on the differences with other studies that have developed valuations for environmental goods.

1.7 Presentation

1.7.1 Units and relationships environmental prices

All environmental prices presented in this Handbook relate to emissions of environmentally harmful pollutants anno 2019 in the territory of the EU27. The reason for choosing 2019 and not later years is that the emissions situation in 2020 and 2021 was fundamentally different from 2019 due to the corona pandemic and lockdowns. As a result, the years 2020 and 2021 are more likely to be seen as outliers and not representative of the current situation regarding emissions.

Environmental prices are shown in €/kg emissions at the 2021 price level (often abbreviated as $€_{2021}$). Unless otherwise indicated, the environmental prices can be considered to include the average VAT.[6]

Some of the emissions occurring on EU territory will drift across the border and have an impact on neighbouring countries. The impact on residents of these neighbouring regions has been valued at the same level as the impact on residents of the EU. This is because environmental pollution in most cases is considered to have a public good nature. If EU citizens only cared about the impact on its own residents and our neighbouring countries did the same, the total air pollution would be greater than if all countries also considered the impact on their residents. Therefore, when valuing the environmental impact, it is common practice to also take into account the damage caused by EU emissions in other countries.[7]

Some of the impact will not manifest itself now, but only in the longer term. For instance, it can take a very long time to recover from a loss of biodiversity. The future impact of today's emissions has been implicitly and explicitly discounted in our calculations, with a 2.25% discount rate being employed for the explicit discounting, in line with the recommendations of the Discount Rate Working Group (Werkgroep Discontovoet) in the Netherlands (Ministerie van Financiën, 2020).

1.7.2 Rounding of values

The environmental prices reported in this Handbook have been rounded to three decimal places when expressed in a floating-point number.[8] Such a degree of precision obviously provides a false sense of certainty in current reporting. As environmental prices are also used in e.g. cost-benefit analyses where, for example, 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 leave it to the user's discretion to determine the level at which to round the results of the calculations.

6 This is because prices are based on consumers' Willingness-To-Pay and consumers express their Willingness-To-Pay in prices including VAT. This does not mean, however, that environmental prices can be calculated by deducting a VAT percentage to arrive at prices excluding VAT.

7 Conversely, the damage caused by air pollution in other countries that ends up in the Netherlands is not included in determining environmental prices. Environmental prices concern the valuation of Dutch emissions.

8 The floating point number of three decimal places: 145; 14.5; 1.45; 0.145 then indicates the same degree of precision. If environmental prices are represented in lower, central and upper values, the central value determines how the comma is placed. Therefore, a number such as 14.52 may appear in the upper value if the central value is below 10. Prices exceeding 1,000 are not further rounded by us in tens, but are shown as the actual calculated cost of damage.

We feel this is more appropriate than us recommending a preferred degree of rounding. As such, we do not prescribe a level of rounding here.

1.7.3 Report overview

This Handbook consists of three sections:

Section 1 is the user part and includes Chapters 1 to 2. Chapter 1 provides accountability for the research process and discusses the basic assumptions. Chapter 2 presents environmental prices for major pollutants. In addition, Chapter 2 provides concrete guidance on when and how environmental prices can be used for specific groups of users.

Section 2 is the methodological accountability report of this study and includes Chapters 3-7. Chapter 3 discusses the methodological background of environmental prices and Chapter 4 describes the approach taken to arrive at environmental prices. Chapter 5 describes the valuation framework and Chapter 6 describes the treatment of the Impact Pathway approach adopted for each environmental theme. Finally, Chapter 7 compares the current environmental prices with the prices in the previous Handbook and with other studies, and includes recommendations for the use of environmental prices in the future.

Section 3 includes the annexes which provide further elaboration and insight into the methodological section. For example, Annex H contains environmental prices for emissions of more than 250 environmentally hazardous pollutants to air, soil and water.
In total, this study determined environmental prices for more than 3,000 environmentally hazardous pollutants.

1.8 Accountability

1.8.1 Guidance and timeframe

Guidance for this Handbook was provided by the following persons at the Ministry of Infrastructure and Water Management (Ministerie van Infrastructuur en Waterstaat) in the Netherlands: Mark Overman. In the Dutch version of 2023, guidance was provided by Bernard Cino, Marije Slump and Robin Hamerlinck of the Ministry of Infrastructure and Water Management.

In developing this Handbook, information was taken from databases and the literature available to us up to September 2022. Information created or released thereafter has no longer been considered by us in the price determination.

1.8.2 Expertise

In developing the EU version of this Handbook, the following people have served as an advisory group:
- Sander de Bruyn, PBL Netherlands Environmental Assessment Agency (Planbureau voor de Leefomgeving);
- Alexander Verkerk, Ministry of Agriculture, Fisheries, Food Security and Nature (Ministerie van Landbouw, Visserij, Voedselzekerheid en Natuur);
- Andries Hof, National Institute for Public Health and the Environment (Rijksinstituut voor Volksgezondheid en Milieu);
- Rob van der Veeren, Directorate-General for Public Works and Water Management (Rijkswaterstaat);
- Karel van Hussen, Ministry of Finance (Ministerie van Financiën).

In the Dutch version in 2023, substantive guidance was provided by an advisory group that assisted us in the design of the Handbook. This advisory group was comprised of the following people:
- Bert Hof, PBL Netherlands Environmental Assessment Agency (Planbureau voor de Leefomgeving);
- Rob Maas, National Institute for Public Health and the Environment (Rijksinstituut voor Volksgezondheid en Milieu);
- Joep Tijm, CPB Netherlands Bureau for Economic Policy Analysis (Centraal Planbureau);
- Rob van der Veeren, Directorate-General for Public Works and Water Management (Rijkswaterstaat);
- Herman Vollebergh, PBL Netherlands Environmental Assessment Agency (Planbureau voor de Leefomgeving);
- Bob Vermeent, Ministry of Infrastructure and Water Management (Ministerie van Infrastructuur en Waterstaat);
- Martijn Badir, Ministry of Finance (Ministerie van Financiën);
- Niels Broekman, Ministry of Agriculture, Fisheries, Food Security and Nature (Ministerie van Landbouw, Visserij, Voedselzekerheid en Natuur);
- Saeda Moorman, Netherlands Institute for Transport Policy Analysis (Kennisinstituut voor Mobiliteitsbeleid).

Moreover, a scientific expert committee consisting of:
- Prof. dr. Mark Huijbregts, Radboud University;
- Prof. dr. Carl Koopmans, SEO Amsterdam Economics;
- Dr. Onno Kuik, Institute for Environmental Studies.

In addition, the following people provided input for the Handbook:
- Simone Schucht, INERIS;
- Mike Holland, EMRC;
- Stale Navrud, Norwegian University of Life sciences;
- Paul Koutstaal, PBL Netherlands Environmental Assessment Agency (Planbureau voor de Leefomgeving);
- Hans van Grinsven, PBL Netherlands Environmental Assessment Agency (Planbureau voor de Leefomgeving);
- Anna Krabbe-Lugnér, Directorate-General for Public Works and Water Management (Rijkswaterstaat);
- Joost Enzing, National Health Care Institute (Zorginstituut Nederland).

Finally, the following people from CE Delft made contributions to the preparation of this Handbook: Geert Warringa, Arno Schroten and Nicole Imholz.

We are very grateful to all the people mentioned above for their assistance in the preparation of this Handbook. This assistance has enabled us to significantly improve the quality of this Handbook and bring it in line with the latest scientific findings. It goes without saying, though, that we alone (and not the experts or the expert committee) bear ultimate responsibility for the ideas and results presented here.

1.8.3 Version management and errata with previous versions

The current Handbook concerns Version 1.0. Although this Handbook has been compiled with the utmost care, errors in calculations or argumentation may occur. If any errors are discovered, they will be corrected by us in a new version.

At this time, there are no errata to be listed, but readers are encouraged to check www.ce.nl/method/milieuprijzen/ to ensure that Version 1.0 is in fact the latest version. Paragraph 1.8.3 and following will include any errata.

2 Results: new environmental prices

2.1 Introduction

In this chapter, we discuss the use of environmental prices and present environmental prices at the pollutant and midpoint levels. All environmental prices are expressed in price levels of the year 2021.

First, in Paragraph 2.2 we provide an introduction to the different perspectives that can be applied when using environmental prices. This knowledge can help in applying the correct environmental prices. Next, we provide environmental prices for some common environmental pollutants to air, water and soil in Paragraph 2.3 and midpoint-level environmental prices for environmental themes in Paragraph 2.4. We then explain the use of environmental prices using three user profiles:

1. Researchers conducting a social cost-benefit analysis (SCBA) (Paragraph 2.5).
2. Researchers who want to weigh the environmental impact of an LCA to produce a single score (Paragraph 2.6).
3. Companies seeking a breakdown of their impact on the environment (Paragraph 2.7).

For each of these target groups, this chapter will provide concrete guidance on how to apply environmental prices and what specific aspects should be considered that apply to environmental prices for that target group.
Finally, in Paragraph 2.8 we consider the limitations of using environmental prices: in what situations would it be better not to use environmental prices?

This chapter does **not** provide information on how environmental prices were determined: that information can be found in the comprehensive second section of this report (Chapters 3-7). Also, this chapter does not provide information on all environmental prices calculated by us. In particular, our focus is on environmental prices that can be used in the most common situations. Annex F provides environmental prices for many more pollutants that are of relevance to policies.

2.2 Perspectives and use

2.2.1 User perspectives: valuation and weighting as an application

In practice, environmental prices can contribute to a decision-making process in two ways[9]:

1. **Valuation**. The objective is then to include the environmental impact alongside the financial impact so that they can be compared with the financial quantity during the weighting. This can play a role in **investment decisions** made by companies that have significant environmental consequences. Or when the government commissions a **social cost-benefit analysis** into the effects of policies. In these cases, the primary purpose of

[9] This distinction between the two ways focuses on the use of environmental prices rather than on the pollutant. In principle, valuation is actually also a form of weighting: prices indicate how the social benefit of one good compares to the benefit of another good.

environmental prices is ***valuation***: a way to compare the environmental impact with other financial quantities in order to achieve an integrated consideration of all the effects involved in a decision, including an investment decision.
2. **Weighting.** In environmental analyses, the various environmental impact identified can be weighted using environmental prices. This involves life cycle analyses (LCAs) or benchmarks. Environmental **weighting** is the main objective: a way of comparing the scores on various environmental themes.

Valuation is likely to be the most common use of environmental prices. Environmental prices have, for instance, been recommended by the Dutch Parliament for use in cost-benefit analyses in the Netherlands that have a significant environmental impact (Ministerie van I&M, 2017). Since then, it has been standard practice to value the environmental impact in social cost-benefit or social cost-effectiveness analyses using environmental prices. Environmental prices are also used in 'true price' calculations that calculate the price of a product including all its external costs (see, for example (CE Delft, 2018a))). In this way, environmental prices can form the basis for experiments in which consumers are asked to pay the actual price of a product with the intention of encouraging consumers to choose less harmful alternatives.

Weighting of environmental impacts sometimes takes place as the final step in an LCA to summarise the results in a uniform figure, known as the *Single Score* (see text box). Following the methodology of the Environmental Prices Handbook, the welfare effects of emissions are monetised in a framework commonly used in welfare economics. More monetary valuation methods exist (see (Amadei et al., 2021) for an overview). The earliest of these is (Steen, 1999), which uses the EPS system (Environmental Priority Strategies in product design) for monetary weighting, but the principles here are based on a monetisation of a hierarchy of principles rather than welfare economics.[10] True Price (True Price, 2020) also has a valuation system based on ethical principles rather than welfare economics. Such an approach contrasts with that of this Handbook, which develops environmental prices using damage costs based on welfare theory (see Chapter 3).

Environmental prices as a single score when weighting midpoint scores
There are over 10,000 known substances that potentially adversely affect our environment. For a long time, environmentalists have been looking for a way to capture the large flow of data that environmental analysis can produce in a single indicator. This compression of data can be achieved in two ways: via characterisation and via weighting.

Characterisation is a process in which an index, known as a characterisation factor, is used to express how much a standard amount of a given pollutants contributes to a particular environmental impact. The higher the characterisation factor, the greater the contribution. The gas methane has a higher characterisation 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 characterisation factors, emissions can be grouped into a series of aggregated environmental themes such as climate change, acidification and human toxicity, which are known as 'midpoints' (see also Paragraph 3.3.3). However, these effects on the various environmental themes cannot then be compared to each other. Take the example of a comparison of a product made from recycled materials and a product made from virgin materials using LCA. All a researcher can conclude is that a recycled product will impact positively on climate,

[10] The EPS system is closer to the concept of 'unpaid costs', where the author derives WTP valuations via a hierarchy of 'principles' or 'assumptions'. Future effects are not taken into account. In particular, the method sets relatively high valuations for resource extraction based on 'recovery costs'.

but negatively on eutrophication. The question is then: Is this policy option good or bad for the environment? In other words: Which environmental theme is more important? To answer this question the various environmental impacts should 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 therefore a process in which midpoint scores are combined to yield a single, uniform indicator. For weighting the environmental theme at midpoint level, various methods have been proposed in the literature, including methods based on 'distance to target' (Ministerie van VROM, 1993) or expert panels (Huppes et al., 2007). In this context, environmental prices can be seen as another method for mutually weighting environmental themes and combining the total environmental impact 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'.

Monetary valuation as a weighting method is frequently used in various LCAs and in specific calculation tools, such as the Environmental Barometer (Milieubarometer) (used for SMEs), DuboCalc (used in construction) and in GreenCalc (used for comparing buildings in terms of environmental performance). These methods use adaptations of environmental prices or the older shadow prices.[11]

2.2.2 Lower, central and upper values

This Handbook presents environmental prices at the pollutant, midpoint and endpoint levels. All environmental prices are presented as a lower value, a central value and an 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), such as to calculate the effect of government policy. Since publication of the General SCBA Guidelines (CPB & PBL, 2013) for the Netherlands, 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, 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 lower and upper values are developed at the endpoint valuation level and interact through the system of relationships between pollutant and endpoints to produce lower and upper values at the pollutant level.

Central values are recommended for other users. The central value provides the best possible estimate given the uncertainty in valuing the environmental impact. For some themes, where uncertainty is very high, the lower and upper estimates also show differences in dose-effect relationships. This applies to both human toxicity and ecotoxicity, in order to properly interpret the greater degree of uncertainty for these themes. In general, toxic substances therefore have a greater degree of variation between the lower and upper values than substances that are not primarily toxic.

Central values are also recommended for companies that want to use environmental prices in business cases or environmental annual reports. Corporate financial annual reports do not generally give ranges in values and use of our central values. This is thus in line with standard practice.

[11] For example, the MKI (Milieu Kosten Indicator) scores in DuboCalc are based on previous environmental prices, known as shadow prices, determined by CE Delft in the year 2000 (CE Delft, 2002).

Central values are also used in LCAs. As a sensitivity variant, the LCA can also be calculated with lower and upper values.

2.3 Results: environmental prices at pollutant level

In this paragraph, we provide the results for several common pollutants that can be used in environmental prices valuation. All environmental prices are expressed in euros per kg of pollutant, in 2021 prices.

2.3.1 Environmental prices for emissions to air

Emissions to air represent significant harm, especially for human health, because these emissions can be inhaled. Air pollutants are also the best researched compartment to describe and quantify the impact of environmental pollutants.

The following table shows environmental prices per kg of pollutant for emissions to air.

Table 3 - Environmental prices for emissions of air pollutants, in €$_{2021}$/kg

Pollutant	Pollutant name		Lower		Central		Upper
CO_2	Carbon dioxide	€	0.050	€	0.130	€	0.160
CFC-11	Chlorofluorocarbons	€	283	€	725	€	926
$PM_{2.5}$	Particulate matter *	€	58.5	€	95	€	134
PM_{10}	Particulate matter *	€	31.3	€	51.6	€	73.3
NO_x	Nitrogen oxides	€	13.5	€	21.5	€	31.8
SO_2	Sulphur dioxide	€	17.8	€	30.5	€	45.3
NH_3	Ammonia	€	18.2	€	28.7	€	39.5
NMVOC	Volatile organic compounds (non-methane)	€	1.62	€	2.49	€	3.49
CH_4	Methane	€	1.80	€	4.68	€	5.77
As	Arsenic	€	6,271	€	9,275	€	13,980
Cd	Cadmium	€	105,034	€	155,294	€	233,924
Cr-VI	Chromium VI	€	1,815	€	2,703	€	4,121
Pb	Lead	€	18,455	€	27,287	€	41,106
Hg	Mercury	€	9,983	€	14,951	€	23,019
Ni	Nickel	€	68	€	126	€	257
	1.3 Butadiene	€	1.40	€	2.01	€	2.88
	Benzene	€	0.278	€	0.405	€	0.593
	Benzo(a)pyrene	€	3,859	€	5,704	€	8,590
	Dioxins	€	34,071,638	€	50,367,195	€	75,846,980
	Formaldehyde	€	0.491	€	0.694	€	0.967

* The analysis should include either $PM_{2.5}$ or PM_{10} and not both at the same time.

As indicated in Paragraph 1.4.2, the environmental prices reported in this chapter are average values for the EU27. The damage costs of most of these pollutants can vary due to local conditions (such as population density) and the nature of emissions (e.g. tall chimneys or exhaust). This is especially true for classic air pollutants that enter the human body through inhalation, such as particulate matter, NO_x, SO_2, NH_3 and NMVOC. As averages for the EU27, these environmental prices cannot readily be applied to specific local environmental pollution, environmental pollution in other countries and/or environmental pollution from non-average emission sources. In Chapter 6 these issues are considered in more detail, as well as the background to the calculations. The chapter also gives values for particulate matter and NO_x that depend on the source of emissions.

2.3.2 Environmental prices for emissions to water

To determine environmental prices to water, in addition to eutrophying pollutants, we also looked at pollutants describing water quality in accordance with the Water Framework Directive Monitoring Regulations and priority pollutants identified in the 2009 Water Quality Requirements and Monitoring Directive. The following table gives the corresponding environmental prices for some of these pollutants. A more comprehensive list of pollutants can be found in Annex H.[12]

Table 4 - Environmental prices to water for some commonly used substances in water policy, in €$_{2021}$/kg

Pollutant name	Emissions to freshwater			Emissions to saltwater		
	Lower	Central	Upper	Lower	Central	Upper
Arsenic	€ 171	€ 2,411	€ 11,361	€ 0.061	€ 188	€ 958
Barium	€ 2.07	€ 6.02	€ 20	€ 0.0043	€ 0.161	€ 0.79
Benzo(a)antracene	€ 0	€ 33.9	€ 43.9	€ 0	€ 5.44	€ 7.0
Benzo(a)pyrene	€ 100	€ 148	€ 223	€ 7.7	€ 11.4	€ 17.0
Cadmium	€ 3.06	€ 31.5	€ 144	€ 0.119	€ 9.0	€ 44.8
Carbendazim	€ 0.77	€ 1.10	€ 1.45	€ 0.0138	€ 0.0195	€ 0.0252
Cypermethrin	€ 957	€ 1,360	€ 1,770	€ 122	€ 174	€ 230
Deltamethrin	€ 98	€ 139	€ 180	€ 9.0	€ 12.7	€ 16.5
Esfenvalerate	€ 1,241	€ 1,760	€ 2,280	€ 34	€ 48.0	€ 62.1
Fluoranthene	€ 10.3	€ 14.6	€ 19.2	€ 0.89	€ 1.27	€ 1.65
Phosphate	€ 0.845	€ 1.23	€ 3.34	€ 0	€ 0	€ 0
Imidacloprid	€ 0.267	€ 0.386	€ 0.541	€ 0.00325	€ 0.00461	€ 0.00598
Irgarol/Cybutryne	€ 0	€ 2,141	€ 2,772	€ 0	€ 58.1	€ 75
Cobalt	€ 0.087	€ 0.225	€ 0.747	€ 0.0294	€ 0.176	€ 0.781
Copper	€ 2.21	€ 3.46	€ 5.56	€ 0.79	€ 2.28	€ 6.60
Mercury	€ 9.1	€ 1,346	€ 6,802	€ 0.40	€ 554	€ 2,819
Lambda-cyhalothrine	€ 734	€ 1,045	€ 1,364	€ 140	€ 202	€ 270
Methylpirimiphos	€ 21.0	€ 30.4	€ 42.2	€ 0.318	€ 0.453	€ 0.597
Nickel	€ 9.9	€ 37.7	€ 140	€ 0.219	€ 4.70	€ 23
Nitrate (N total).	€ 2.27	€ 4.23	€ 8.19	€ 7.64	€ 14.3	€ 27.6
Selenium	€ 0.215	€ 0.513	€ 1.59	€ 0.073	€ 0.358	€ 1.53
Thallium	€ 7.3	€ 149	€ 733	€ 0.106	€ 25.0	€ 140

[12] Incidentally, environmental prices could not be established for all substances involved in water policy.

The table shows that the range in environmental prices between the lower and upper values can be enormous. This is due to the fact that these are mainly pollutants that have an effect on human toxicity and ecotoxicity, where different characterisation models from ReCiPe 2016 are used to reflect the uncertainty in the impact and time horizon for the lower and upper values. The central value gives the value we consider most plausible – this value is also recommended for most applications. When uncertainty needs to be explicitly reflected, as in Dutch SCBAs, one does need to work with these lower and upper values.

2.3.3 Environmental prices for emissions to soil

Emissions to the soil can occur via waste dumping or leakage or eutrophication, potentially impacting ecosystems and/or human health. The following table gives the environmental prices of a number of soil contaminating pollutants.[13]

Table 5 – Environmental prices for emissions to soil, in €$_{2021}$/kg

	Lower	Central	Upper
Antimony (Sb)	€ 5.15	€ 19.1	€ 70.0
Anthracene	€ 0.119	€ 0.168	€ 0.218
Arsenic (As)	€ 19.5	€ 168	€ 884
Barium (Ba)	€ 15.4	€ 25.5	€ 48.1
Benzo(a)antracene	€ 0	€ 0.219	€ 0.284
Benzo(a)pyrene	€ 1.57	€ 2.31	€ 3.49
Cadmium (Cd)	€ 9.3	€ 2,224	€ 11,320
Chromium (Cr) (III)	€ 0.00256	€ 0.0092	€ 0.0298
Phenanthrene	€ 0.028	€ 0.0403	€ 0.0520
Fluoranthene	€ 0.13	€ 0.18	€ 0.243
Cobalt (Co)	€ 0.000557	€ 0.094	€ 0.551
Copper (Cu)	€ 0.0166	€ 0.431	€ 2.15
Mercury (Hg)	€ 1.69	€ 280	€ 1,425
Lead (Pb)	€ 0.97	€ 23.0	€ 118
Molybdenum (Mo)	€ 4.57	€ 28.0	€ 143
Naphthalene	€ 0.038	€ 3.11	€ 4.68
Nickel (Ni)	€ 5.80	€ 45.1	€ 287
Selenium (Se)	€ 0.00369	€ 0.201	€ 1.15
Tin (Sn)	€ 0.000143	€ 0.0708	€ 0.418
Vanadium (V)	€ 0.226	€ 1.60	€ 7.5
Zinc (Zn)	€ 0.0510	€ 787	€ 4,003

Emissions to soil only give the impact on human health and ecosystems. The impact on human health does not include an estimation of the effects on IQ.

This table again shows that the environmental prices of pollutants to soil can vary enormously between the lower and upper values. This is due to the uncertainty regarding the distribution and dose-effect relationships of toxic substances that has been explicitly identified for human toxicity and ecotoxicity. For most applications, the central value provides a good measure of the most likely valuation of the pollutant.

[13] In selecting substances, we were guided by an exploratory analysis of environmental damage from waste, see CE Delft (2022a). No IQ effects were monetised in environmental prices to soil.

2.3.4 Environmental prices for other impacts

For land use, we calculated the external costs resulting from the loss of biodiversity due to average land use in the EU27. These costs were calculated by determining the loss of biodiversity relative to the 'natural state': the state in which nature would be in the absence of economic activities (see further Paragraph 5.4) and valuing it over a 50-year period (see Paragraph 0). This results in an average valuation for the loss of biodiversity due to land-use occupation. This value can be used to determine the external costs of land-use occupation, such as is used by companies in natural capital accounting.

Table 6 - Environmental prices for land use in the EU27 for effects on biodiversity relative to a 'natural state', in €2021/m^2 per year

$€_{2021}$/m2/year	Lower	Central	Upper
Land use occupation	€ 0.037	€ 0.053	€ 0.069

We recommend using this valuation with caution. It is preferable to measure the precise loss of biodiversity by land-use occupation than to use these generalised key indicators. For example, an SCBA will more often involve land-use change where both the baseline and the policy alternatives would already involve loss of biodiversity compared to the 'natural state'. Land-use change cannot be valued using the table above - Paragraph 0 provides general key indicators by land-use type that can be used to estimate losses of biodiversity due to land-use change in a very rudimentary way.

2.4 Results: prices at midpoint and endpoint level

Environmental prices can also be represented as the weighted results at midpoint level. This involves assigning emissions of individual pollutants to the various midpoints and weighting them by emissions. Chapters 4 and 6 indicate the methods used to determine the midpoint price.

2.4.1 Results at midpoint level: ReCiPe 2016 for the EU27

The following table summarises the calculated midpoint-level prices that can be used in life cycle analyses. The midpoints and their units are from ReCiPe (2016). If the results of these midpoints are used in an LCA, the researcher should also determine the environmental impact according to ReCiPe's LCA characterisation method. More information on usage can be found in Paragraph 2.6. Explanations of all calculations for each midpoint can be found in Chapter 6.

Table 7 - Environmental prices for LCA: ReCiPe 2016 midpoints, in $€_{2021}$ per unit for EU27

	Lower	Central	Upper	Unit	Cat.
Climate change	€ 0.05	€ 0.13	€ 0.16	€/kg CO_2-eq.	A
Ozone depletion	€ 15.2	€ 29.1	€ 69.6	€/kg CFC-11-eq.	A
Ionising radiation	€ 0.00275	€ 0.00422	€ 0.00594	€/kBq Co-60-eq.	A
Oxidant formation, human health	€ 1.38	€ 2.17	€ 2.98	€/kg NO_x-eq.	A
Oxidant formation, terrestrial ecosystems	€ 0.416	€ 0.416	€ 0.526	€/kg NO_x-eq.	A
Particulate matter formation	€ 61.7	€ 99.2	€ 138.1	€/kg $PM_{2.5}$-eq.	A
Acidification	€ 2.66	€ 5.27	€ 9.30	€/kg SO_2-eq.	A
Freshwater eutrophication	€ 2.56	€ 3.74	€ 10.13	€/kg P-eq.	A

	Lower		Central		Upper		Unit	Cat.
Marine eutrophication	€	7.64	€	14.25	€	27.60	€/kg N-eq.	A
Terrestrial ecotoxicity	€	0.00045	€	0.00064	€	0.00083	€/kg 1.4-DCB-eq.	A
Freshwater ecotoxicity	€	0.0148	€	0.0209	€	0.0271	€/kg 1.4-DCB-eq.	A
Marine ecotoxicity	€	0.0022	€	0.0032	€	0.0041	€/kg 1.4-DCB-eq.	A
Human toxicity, cancer-related	€	2.70	€	3.99	€	6.01	€/kg 1.4-DCB-eq.	A
Human toxicity, non-cancer-related	€	0.048	€	0.071	€	0.106	€/kg 1.4-DCB-eq.	A
Land use	€	0.070	€	0.099	€	0.128	€/m^2 a crop-eq.	B
Mineral extraction	€	0.0000	€	0.0140	€	0.0826	€/kg Cu-eq.	B
Fossil extraction	€	0.000	€	0.028	€	0.163	€/kg oil-eq.	B
Water consumption	€	0.000	€	0.407	€	0.811	€/m^3	B
NO_2 addition	€	4.31	€	6.37	€	9.62	NO_x-eq.	C

The category classifications in the last column give an indication of use: Category A is a ‘certain’ category, while Category B has a higher degree of uncertainty. Category C is a new category that falls outside ReCiPe’s existing characterisation methodology but is nevertheless an important one to include if the purpose of the LCA is to determine damage costs. Paragraph 2.6 gives more guidance on how to use them.

When comparing the table above with the table from Paragraph 2.3.1, the midpoint price is not the same as the pollutant price. For example, the midpoint price for 1 kg of SO_2-eq. is almost nine times lower than the price for SO_2. This is partly because SO_2 also has other impacts (such as particulate matter formation) and partly because other pollutants also have an impact on the acidification theme (such as NO_x and NH_3). The resulting midpoint price is the emission-weighted average of the pollutants that have an effect on that theme.

The latter also explains why the midpoint price for $PM_{2.5}$, for instance, has become much higher: emissions of other pollutants also have an impact on particulate matter formation. In fact, the central value of €168 per kg $PM_{2.5}$-eq. indicates the monetary damage of the pollutants that are characteristic of the weighted theme by the probability of them appearing in an LCA score. This probability is determined by emissions. All this is explained in more detail in Paragraph 6.4.

2.4.2 Results at midpoint level: PEF for the EU27

The Product Environmental Footprint (PEF) is a European method, co-developed by the Joint Research Centre (JRC) that is recommended by the European Commission for assessing the environmental impact of products and organisations. The PEF partly uses other environmental models and therefore has a different characterisation than ReCiPe (see also Annex B). The PEF is standard in LCA assessment methods such as European EN15804-A2 and will gradually be more widely used.
In this Environmental Prices Handbook, we have converted environmental prices at pollutant level to midpoint prices according to the PEF. The PEF assigns three uncertainty categories to the various midpoints. In determining environmental prices, we limited ourselves to the categories rated as ‘recommended and satisfactory’ (CAT I) and ‘recommended but in need of some improvement’ (CAT II). Category III impacts could not be determined within the timeframe of this Handbook and could be added in future.

Table 8 – Midpoint prices for PEF impact categories, CAT I and II, in €$_{2021}$ per unit

Name of Environmental theme PEF	Unit	Environmental prices* per unit, EU27	Environmental prices* per unit, NL
Climate change	kg CO_2-eq.	€ 0.130	€ 0.130
Ozone depletion	kg CFC-11-eq.	€ 29.1	€ 29.1
Ionising radiation	kBq U235-eq.	€ 0.00071	€ 0.00071
Oxidant formation, human health	kg NCSRC-eq.	€ 1.48	€ 1.40
Particulate matter formation	Disease incidence	€ 890,182	€ 1,937,047
Acidification	mol H+-eq.	€ 2.04	€ 2.01
Freshwater eutrophication	kg P-eq.	€ 3.74	€ 5.53
Marine eutrophication	kg N-eq.	€ 14.25	€ 14.25
Terrestrial eutrophication	mol N-eq.	€ 0.331	€ 0.344

* This refers to the environmental prices per unit for the central variant.

Annex B explains more about the PEF impact categories and how they differ from ReCiPe. Chapters 6 and 7 include further information on how these values have been calculated.

2.5 Use of environmental prices in an SCBA

Pollutant prices (Paragraph 2.3) will generally be used in an SCBA. In this paragraph, we provide specific points that are considered necessary for using environmental prices in an SCBA. We adhere to the general guidelines in the Netherlands for conducting SCBA.

2.5.1 General framework

A social cost-benefit analysis (SCBA) is a decision-making tool that can be used to clarify public policy considerations. Most policy alternatives have a range of effects. By expressing these effects in monetary terms, we can compare them. This provides valuable information on the pros and cons of each alternative.

For the Netherlands specifically, General Guidelines for SCBAs were published in 2013 (CPB & PBL, 2013), prescribing how such analyses should be carried out. Further elaborations on the General Guidelines are given per policy area in Working Guides. CE Delft wrote and published the Environmental Working Guide in 2017 (CE Delft, 2017b). The rules, recommendations and key indicators in this Working Guide can be applied to environmental policies and to other policies with significant side effects on the environment. In addition, the MIRT (RWS, 2018), Nature (CE Delft & Arcadis, 2018) and Aviation (SEO et al., 2021) Working Guides also describe effects that can be quantified as environmental prices. These guidelines are specific to the Netherlands but can be seen as suggestive practice for SCBAs outside the Netherlands as well.

An SCBA can be used for different types of policy considerations, such as:
- Specific government investments, such as motorway construction or introduction of separated household waste collection. In this case, there are public investment costs that have social benefits in the form of reduced environmental pollution.
- The introduction of environmental policy instruments, such as a waste levy or a renewable energy subsidy. In this case, the government sets the frameworks within which companies and consumers can be forced or enticed to make investments or modify behaviour. In such cases, besides policy costs, there are mainly private costs to companies and/or consumers and societal benefits in the form of reduced environmental pollution.

— Exploratory policy options, such as whether air quality policy should be further tightened or whether higher recycling targets, are socially desirable. In this case, the SCBA supports the problem analysis and examines, in an exploratory role, whether additional environmental policies are desirable from a welfare perspective.

2.5.2 How are environmental prices used in an SCBA?

In the SCBA, the environmental impact is identified as changes in the volume of emissions of environmental pollutants to soil, air and water, as much as it can possibly be quantified.[14] Emissions are dispersed via the environment, leading to an impact on endpoints: human health (morbidity and death), ecosystem services, buildings and materials, availability of raw materials and nuisance. Environmental prices establish a link between emissions and endpoint effects and assigns a value to those effects.

Environmental prices are especially recommended in situations where it is not known where in the country or region the environmental effects occur, or if the environmental effects are minor side effects of an SCBA. If the SCBA involves a measure with a marked regional or local impact, the use of environmental prices is inadvisable and an environmental impact analysis, such as the Impact Pathway approach, would be recommended to quantify the environmental damage. In such case, the endpoint-level valuations (see Chapter 5) developed in the Environmental Prices Handbook can be used.

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. This differs with regard to effects on climate. For valuation of effects on climate, prices are based on costs of measures and these are generally determined excluding VAT. In the previous Handbook we chose to raise the prescribed prices for CO_2 valuation in an SCBA by an average VAT rate of 18% (SEO, 2016b). We will continue to do this until a new set of prices for SCBA becomes available.

2.6 Use of environmental prices as a midpoint weighing ratio in an LCA

2.6.1 General framework

Environmental prices can also be used for weighting the environmental impact in an LCA (Life Cycle Assessment). These prices signify the relative value of emissions compared with one another and with other goods circulating in the marketplace. When emissions are valued, such as in an SCBA, their value is usually considered relative to other financial parameters. However, emission weighting primarily concerns the relationship between emissions. These weighing factors can then be regarded as the socio-economic weight attributed to the various environmental impact.

Environmental prices have been developed at the midpoint level in this Handbook for use as a weighing factor. These are described in Paragraph 2.4. It is important to note that environmental prices depend on the method of characterisation. Various life cycle assessment tools exist, such as ReCiPe 2008 (Goedkoop et al., 2009); ReCiPe, 2016 (Huijbregts et al., 2016); CML (Guinée et al., 2002); ILCD (JRC, 2012) and the Product Environmental Footprint (PEF)(EC, 2021).

[14] In addition, there are also disruptive interventions, such as noise or visual nuisance, which are not identified in emissions but are part of an SCBA and can also be valued with environmental prices.

Because environmental prices are composed of estimates for individual pollutants and characterisation factors, midpoint-level environmental prices are always available only for one particular LCIA. In this Handbook we present results based on ReCiPe 2016 and some categories from the PEF. No conversion factors exist to make these environmental prices available for other characterisation methods. This means that these environmental prices cannot be used in LCAs where the ReCiPe 2008 method has been used. Nor is it the case that the environmental prices from the previous Handbook (which applied to ReCiPe 2008) can then simply be used. Therefore, at the moment there is no current estimate of environmental prices using the ReCiPe 2008 characterisation method. Such environmental prices can be developed, but this should be undertaken in future research as it is beyond the scope of the current study.

In addition to use in life cycle analyses, midpoint prices can also be used in other valuation research or to compare pollutants. An LCA can also be carried out in some SCBAs to investigate upstream and downstream environmental impact.

2.6.2 Using ReCiPe 2016 results

In the previous Handbook, two sets were presented: one set for a single-score weighting method and one set for external cost estimates. The difference between the two sets was that the single-score weighting method was determined exclusively for the hierarchical worldview, while the external cost estimates corresponded to the concept of environmental prices. In practice, this led to much confusion among users. In the new Handbook, we therefore present only one set of midpoint prices that are based solely on external costs (environmental prices). Of course, these prices can also be used to derive a single score.

In addition, we decided to explicitly address uncertainty for use in LCAs by:
- Using a range of lower, central and upper values.
- Creating a subdivision by type of effect, where Type A effects are reasonably certain and Type B effects are tentative. Tentative values are preferably not included in cost-benefit analyses but can be included in LCAs to allow for comparison of results.

In addition, we also distinguish a Category C: Effects of NO_2 on human health. This is a separate category, which is not standard in LCA and includes health effects exclusively related to emissions of NO_2, as determined by (COMEAP, 2018). Therefore, in our allocation of NO_2 damage costs across the various ReCiPe midpoints, a residual category remains. This residual category can also be valued in an LCA, but researchers have to add this value themselves. This is explained in Paragraph 6.5 of this Handbook.

We recommend that researchers conducting LCAs create a separate analysis of the NO_x (or NO_2) emissions generated by the product and value them with the value for NO_2. This value can then be added to the value on the other midpoints.

When testing the environmental prices, it further emerged that models that used country-specific impact factors could result in very high values for land use and particulate matter formation. This suggests that this partly depends on the way of environmental modelling, where there may be double counting if the environmental prices for e.g. the Netherlands are used in impact assessments that also use country-specific factors, such as for the Netherlands. Therefore, the recommendation is to use average (global or Western European) impact factors when using environmental prices and also not to specify them by country.

2.6.3 Example of the use of environmental prices from ReCiPe 2016

In this paragraph, we discuss a fictitious example of how environmental prices can be applied in the case of an insulation material manufacturer. The manufacturer is considering replacing the composition of polystyrene in the insulation material with polyethylene and wants to investigate the resulting environmental impact.

A life cycle analysis can be a good tool when it comes to choosing the base material. The following table compares polystyrene with polyethylene through an LCA and values the effects with environmental prices.

Without environmental prices, the results would be the same as those in the second and third columns and the manufacturer would see that, while polystyrene causes more climate change, it does score better on terrestrial ecotoxicity and human toxicity. The manufacturer would be unable to make much sense out of this jumble of numbers. However, weighing the results of the LCA with environmental prices shows that polystyrene has about 30% more damage costs over the product life cycle than polyethylene.

Table 9 – Comparison of scores between polystyrene and polyethylene per tonne of material and associated environmental costs

LCA midpoint	Score PS (polystyrene)	Score PE	Unit	Environmental prices/unit	€/tPS	€/tPE
Climate change	3812	2361	kg CO_2-eq.	€ 0.13	€ 495.56	€ 306.93
Ozone depletion	0.0001	0.0003	kg CFC11-eq.	€ 29.15	€ 0.00	€ 0.01
Ionising radiation	0.7	4.6	kBq Co-60-eq.	€ 0.0042	€ 0.00	€ 0.02
Oxidant formation, human health	6.4	4.8	kg NO_x-eq.	€ 2.17	€ 13.89	€ 10.41
Oxidant formation, terrestrial ecosystems	7	5.1	kg NO_x-eq.	€ 0.416	€ 2.91	€ 2.12
Particulate matter formation	3	2.4	kg $PM_{2.5}$-eq.	€ 99.2	€ 297.59	€ 238.07
Acidification	9.2	6.4	kg SO_2-eq.	€ 5.27	€ 48.50	€ 33.74
Freshwater eutrophication	0.1	0.1	kg P-eq.	€ 3.74	€ 0.37	€ 0.37
Marine eutrophication	0.02	0.01	kg N-eq.	€ 14.25	€ 0.29	€ 0.14
Terrestrial ecotoxicity	2457	4761	kg 1.4-DCB	€ 0.0006	€ 1.58	€ 3.07
Freshwater ecotoxicity	1.7	1.2	kg 1.4-DCB	€ 0.0209	€ 0.04	€ 0.03
Marine ecotoxicity	3.5	4.2	kg 1.4-DCB	€ 0.00316	€ 0.01	€ 0.01
Human toxicity, cancer-related	19.9	19.2	kg 1.4-DCB	€ 3.99	€ 79.48	€ 76.68
Human toxicity, non-cancer related	78.7	351.8	kg 1.4-DCB	€ 0.0705	€ 5.55	€ 24.81
Land use	6	21.4	m2a crop-eq.	€ 0.099	€ 0.59	€ 2.12
Mineral extraction	0.7	4.6	kg Cu-eq.	€ 0.014	€ 0.01	€ 0.06
Fossil extraction	1897	1712	kg oil-eq.	€ 0.0276	€ 52.34	€ 47.23
Water consumption	65	20.7	m^3	€ 0.407	€ 26.43	€ 8.42
NO_2 addition	5.4	4.5	kg NO_2-eq.	€ 6.37	€ 34.39	€ 28.66
Total					**€ 1,060**	**€ 783**

In addition, an analysis using environmental prices also provides tools that allow companies to see where the most damage occurs. In both production processes, the largest damage is caused by emissions of particulate matter formation, followed by climate change. Human toxicity is a third major cost. Matters such as ozone depletion and radiation cause almost no damage costs in this life cycle. This can give manufacturers an indication of where they can improve their production processes if they want to reduce their environmental impact (see also below).

2.7 Use of environmental prices by companies

Sustainability is an important prerequisite for production by companies these days. The developments and challenges in the field of sustainability are accelerating. Companies are constantly being challenged to make the right choices in terms of making their production processes more sustainable and to report transparently on their environmental performance.

A growing number of companies currently view sustainability as an opportunity to be grasped rather than as merely a prerequisite. By saving energy and raw materials and reusing them, they add economic value to their production processes while simultaneously contributing to a sustainable world. There is a growing understanding that innovations can be implemented not only in production processes, but also across the entire chain. This is where financial value (price) plays a key role and, within that role, environmental prices can fulfil an essential function.

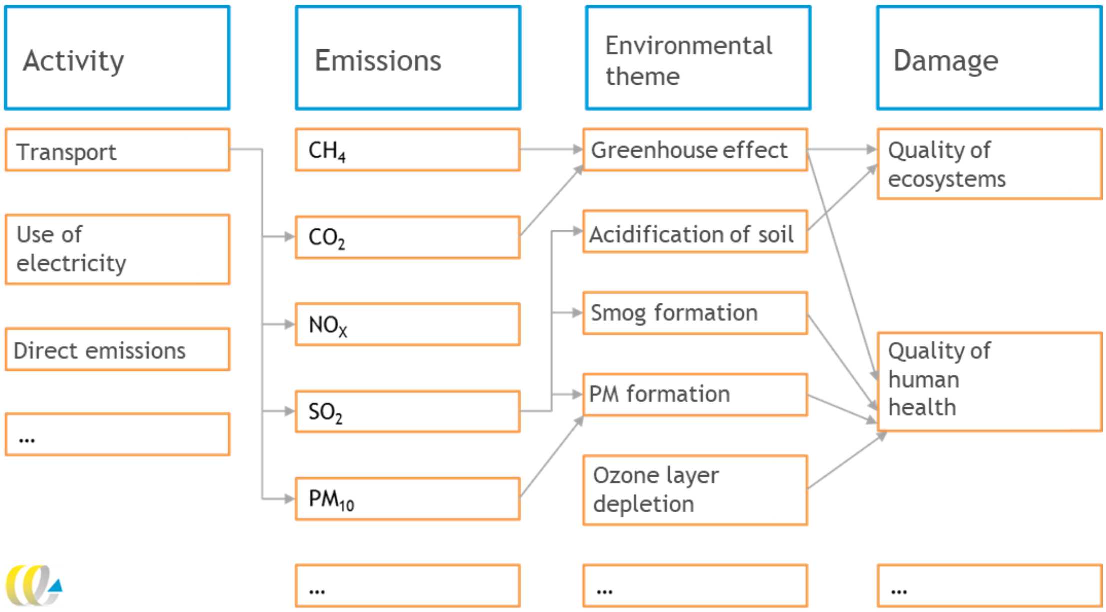
In this paragraph, we describe some common applications of environmental prices for companies:
- communication of social impact to shareholders and society (Paragraph 2.7.2);
- ranking and scoring a company's performance (Paragraph 2.7.3);
- considerations regarding optimisation of production processes, including procurement and the development of a social business case (Paragraph 2.7.4).

Hereafter, we first explain how environmental prices can be applied to companies.

2.7.1 How do environmental prices work for companies?

Business activities such as transport, electricity and gas consumption or the production of materials and raw materials cause emissions of environmentally damaging pollutants. These pollutants have different effects on the environment. Some pollutants contribute to global warming. Others cause soil eutrophication, deplete the ozone layer or are toxic to humans or animals. Sometimes emissions of a particular pollutant even have multiple environmental effects. SO_2, for instance, causes particulate matter formation, photochemical oxidant formation and soil acidification. Figure 3 provides an illustrative example of how business activities cause various environmental impacts and ultimately harm human health and ecosystems.

Figure 3 - Relationship between industrial activities, emissions and the environment



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

Environmental prices calculate the price of the life cycle from emissions to ultimate damage. Environmental prices are not helpful in translating business activities into emissions. There are dedicated tools available for this purpose, such as the Environmental Barometer for Small and Medium-sized Enterprises developed by the Stimular Foundation (in the Netherlands). Companies can also carry out their own analyses. Frameworks available for this purpose include existing reporting obligations to the competent authorities, emissions registration under EU ETS and environmental or emission reporting to the European Pollutant Release and Transfer Register (E-PRTR).

If quantitative emissions are known, environmental prices can be used to calculate the environmental damage of the business activity concerned or the environmental benefits of an envisaged investment. This involves multiplying the physical emissions (kg of pollutant) by the environmental prices (in €/kg of pollutant) to express the aggregate resultant impacts in euros. The environmental prices thereby reflect all effects of the particular environmentally harmful pollutant on the environment. In the case of SO_2 emissions, for example, it accounts for soil acidification, oxidant formation and particulate matter formation. This allows all the environmental impact resulting from the various business activities to be expressed in euros.

2.7.2 Environmental prices as part of ESG Reporting

Such use of environmental prices may have new applications within the European Union's Corporate Sustainability Responsibility Directive (CSRD). The CSRD is a legal framework that requires companies to report on their sustainability efforts in an annual report (see the text box below). To satisfy the requirements of this directive and achieve greater transparency, companies can monetise their environmental impact. Environmental prices offer companies a standardised and transparent way to calculate the environmental impact of their activities. This information is then used to inform stakeholders about the company's sustainability efforts and to identify areas for improvement.

The CSRD Directive and the Non-financial Reporting Directive

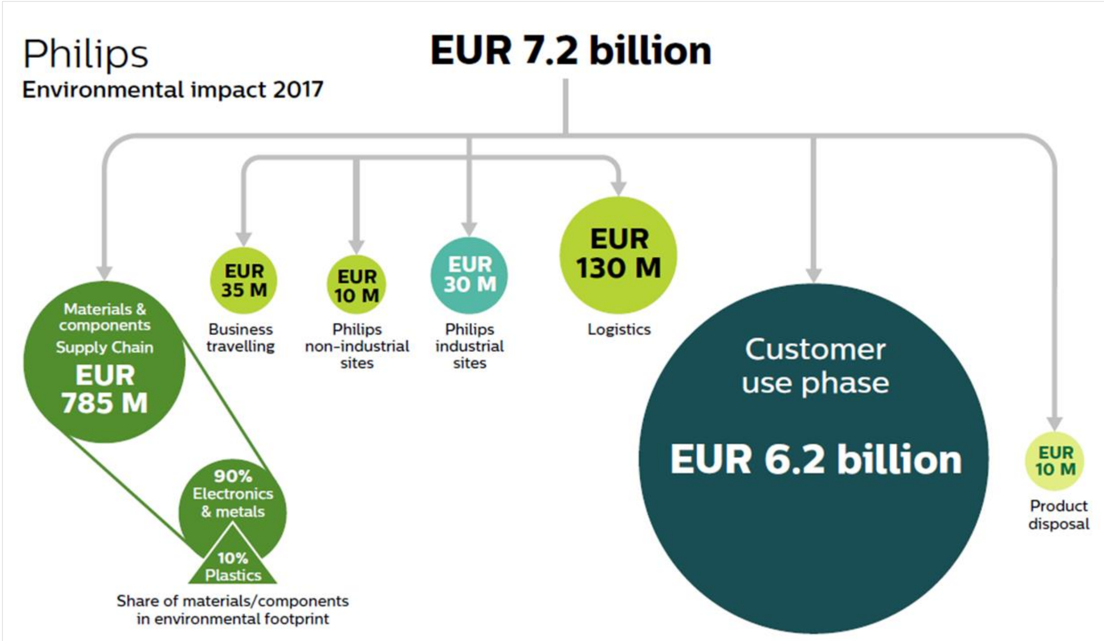
The Corporate Sustainability Reporting Directive (CSRD) is a European Union directive requiring companies to report on their environmental, social and governance (ESG) performance that will succeed the Non-financial Reporting Directive (NFRD). The NFRD provides guidelines on how to report on non-financial aspects in the annual report for companies with more than 500 employees. Starting in 2023, these companies are required to report on biodiversity and pollution control, among other things. By 2024, it will become the CSRD Directive and additional requirements will be added to the reporting and the companies that are required to report. Ultimately, this leads to all companies being required to report by 2028.

Under the CSRD, companies are required to publish an annual sustainability report, which must include information on the company's ESG performance, as well as its policies, risks and future plans with regard to sustainability. The report must be based on internationally recognised sustainability reporting standards, such as the Global Reporting Initiative (GRI) standards.

An important application of environmental prices is to report on the effectiveness of a company's policies. Environmental prices allow for the development of an integrated indicator that shows how much the environmental damage caused by a company has decreased.

The following image shows how Philips reported on its environmental impact using environmental prices in 2018. This graph clearly shows that energy use in the consumption phase accounts for the largest environmental impact of products sold by far.
Such information is helpful when prioritising a company's sustainability strategy.

Figure 4 – Application of environmental prices at Philips in its sustainability report



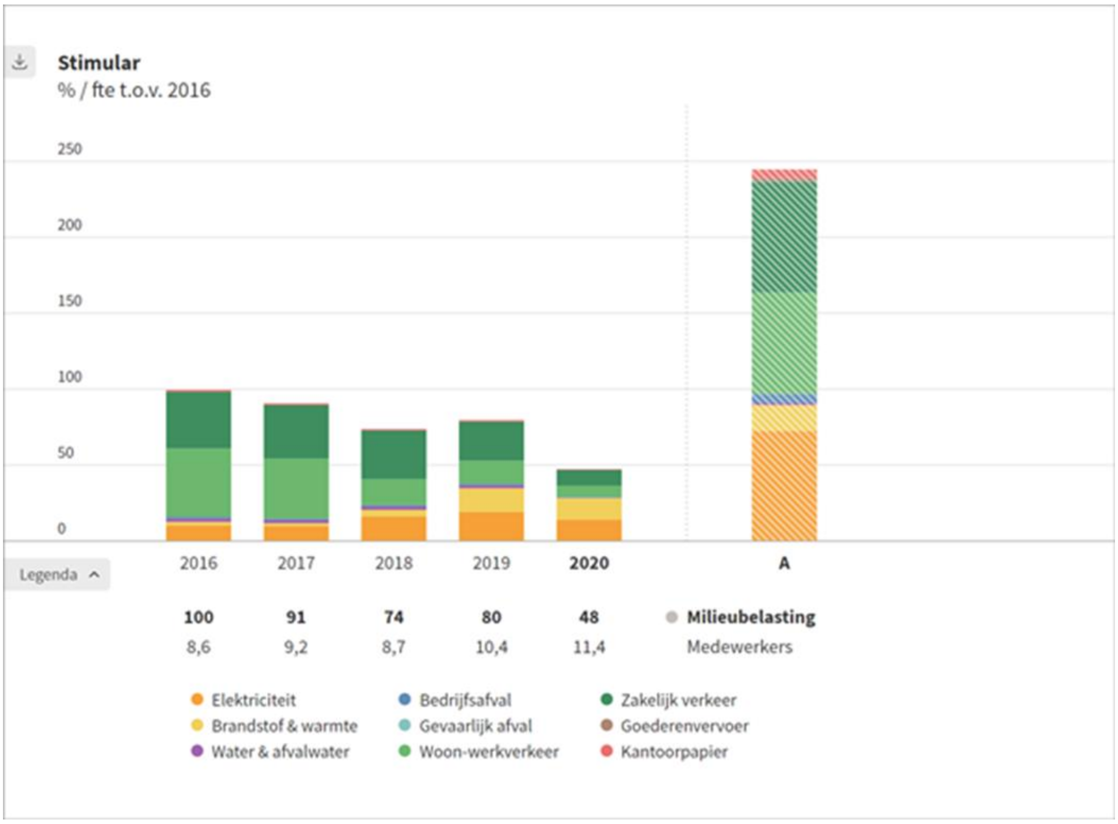
Source: (Philips, 2018). Used with permission from Philips.

2.7.3 Ranking and score

Environmental prices can be used as a ranking mechanism, comparing a company's results with those of other companies in the same industry. Tools such as the Environmental Barometer from the Stimular Foundation or the CO_2 Performance Ladder from SKAO (Stichting Klimaatvriendelijk Aanbesteden en Ondernemen) in the Netherlands calculate a company's emissions using questionnaire surveys. If these emissions are weighted by environmental prices, they can provide an integral score. Because Stichting Stimular and SKAO perform calculations for many companies, industry averages can also be calculated allowing companies to compare among themselves.

The following figure shows an example of an industry average (A) and the annual turnover of a company where the environmental burden is expressed as a relative quantity to the number of full-time equivalent jobs (FTE).

Figure 5 - Example of a company that can compare its environmental performance over time with environmental prices in the Stimular Foundation's Environmental Barometer and also compare its environmental performance to the industry average (A)



Source: Stichting Stimular, used with permission. Left to right the categories included are: electricity; company waste; business transport; fuel and heat; hazardous waste; freight transport; water and waste water; commuting traffic; use of paper.

2.7.4 Optimisation of business operations with respect to environmental impact

Environmental prices can be helpful for companies for various activities in which the company tries to optimise its operations by taking into account its environmental impact. In these situations, a company will apply either life cycle analyses (see Paragraph 2.6) or cost-benefit analyses (see Paragraph 2.5).

Combined with life cycle analyses, a company can:

- gain insight into where the most environmental gains can be made in the company's value chain;
- calculate the sustainability gains that can be achieved through improved procurement policies;
- assess whether the additional energy input of recycling outweighs the reduced primary resource consumption;
- calculate a real price for products which integrates all environmental impact.

Previously in Paragraph 2.6.3, we gave an example of how a company, through life cycle analysis, can make decisions in the design phase of a product. In Paragraph 2.7.2, we also showed how Philips uses life cycle analyses, with environmental prices, to gain insight into which step of the chain offers the most sustainability gains.

Combined with cost-benefit analyses, a company can create a social business case. A social business case is broader than the traditional business case, which involves only financial values for the company being quantified in a cost/budget balance. Instead, a social business case highlights the social costs and benefits from the perspective of different stakeholders. Environmental prices can be helpful in this case to put a price on the environmental impact.

A company that would have chosen investment A on the basis of a traditional business case can use a social business case to gain the insight that while investment A may be more financially profitable than alternative B, on the basis of a social business case alternative B would yield a higher return. By contrasting the social business case with the financial business case, the company can streamline the discussion of social impacts of its operations and provide it with hard data. This can be a particularly powerful tool for structuring the discussion of the social impact of business choices.

2.8 Limitations in the use of environmental prices

Although environmental prices are frequently used to value the impact of environmental pollution, there are also objections to or limitations in the use of environmental prices. In this paragraph, we discuss the limitations that occur with regard to:
- ethical objections to valuing environmental pollution;
- the use of environmental prices for established policy objectives;
- the use of environmental prices for substances that do not degrade in the environment;
- the use of environmental prices in non-standard situations.

2.8.1 Objections to the economic valuation of environmental pollution

Objections have sometimes been raised in the scientific or popular literature about economically valuing environmental goods for their impact on welfare. It can be argued that it is undesirable, wrong or morally reprehensible to put a price tag on health or nature.

Environmental prices as developed by CE Delft are strongly grounded in welfare economics which seeks to maximise the well-being of the average person. The economic valuation of environmental pollution is anthropocentric: valuation of the environment is undertaken by people. From a philosophical point of view, there could also be biocentric (Singer, 1983) or ecocentric values). The environmental prices in this handbook do not address such values.

It is important to note here that economic valuation also need not affect such values. Economic valuation merely facilitates and rationalises choices between alternative ways of allocating scarce resources (time, money). Money spent on Alternative A cannot be spent on Alternative B. When weighing these choices, recognition of intrinsic values may very well be taken into account by limiting the scope for economic choices. The same is implied in nature policy, for example, where the *Habitats Directive* mandates the protection of vulnerable species regardless of people’s preferences for such protection.

Economic choices can therefore be made while respecting intrinsic values and the boundaries within which choices can be made. When we decide what portion of our money to spend on development cooperation, we are also not denying the intrinsic value of people living in the developing world. Economists look at how much people are willing to pay for various goods and objectives, and from that information they 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 occurring in society at large.

A related criticism is that environmental prices would legitimise the exploitation of nature. As long as a 'proper price' is paid, nature can be endlessly depleted. However, environmental prices can only be used to factor in the influences that decisions have on the environment and on nature. Illegal pollution of the environment should always be dealt with within the legal frameworks. Environmental prices should not replace norms and values that determine what we want to preserve in this world.

When it comes to valuing human health, there are sometimes misconceptions. In putting a value on health, it may seem as if a 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). But more importantly, economic valuation does not value lives, but rather values risk of death expressed in *statistical* lives. For instance, if a certain risk is reduced from fifteen in a million to fourteen in a million for a population of one million, one statistical life is saved. In everyday life, such comparative assessments are unconsciously made all the time, such as when deciding whether or not to get into a car or plane, or pursue a certain lifestyle with associated risks of premature death. In other words, while life as such is priceless, safety in the sense of statistical risk reduction is not. A problem therefore arises in economic terms in deciding which risks are acceptable and which are not. Environmental prices make this weighing up of choices explicit and involves it into other forms of decision-making.

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 argue that issues such as environmental protection should be assessed based on public interest, i.e. on what is best for society as a whole. 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 emphasise that environmental prices based on Willingness-To-Pay that can be used for cost-benefit analyses are not a substitute for a political process; they only 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 from this.

2.8.2 Limitations on the use of environmental prices in policies that place restrictions on emissions

Besides ethical concerns, there are also limitations on the use of environmental prices. One example is a situation where an increase in emissions is legally capped. To give an example, this applies regarding the nitrogen emissions issue in the Netherlands or the Water Framework Directive.

As an example, in the Netherlands, the relatively high population density with a relatively large agricultural and primary industrial sector ensures high nitrogen emissions, both in the form of NO_x (traffic and industry) and NH_3 (agriculture). The issues involved have been known for a long time and were identified in the past, such as regarding manure surpluses (Dietz, 1989).

In accordance with the European Habitats Directive, the Nature Conservation Act (Wet natuurbescherming) stipulates that when an economic activity emits nitrogen, its effect on Natura 2000 areas must be assessed. Nitrogen creates nutrient-rich soil, causing the disappearance of protected plant and animal species. This issue has been regulated via the Integrated Approach to Nitrogen (Programma Aanpak Stikstof, PAS) since 2015. The PAS took the effects of measures taken later into account, making it possible for companies to obtain a permit for nitrogen emissions. The administrative courts put an end to that practice in 2019 and since then, all nature permits can be challenged if nitrogen emissions have not been taken into account.

This has resulted in a situation where environmental prices cannot be used for nitrogen emissions in certain situations. This is because a limit has been set and any additional nitrogen emissions can be contested in court. As a result, the cost of nitrogen emissions for the Netherlands is not determined by damage costs, but rather by abatement costs. This means that the abatement costs for nitrogen emissions possible under the Nature Conservation Act are much higher than the damage costs. If we value the additional nitrogen emissions from a road expansion in a SCBA against damage costs, for instance, we underestimate the cost of the project. Therefore, in such a SCBA, a researcher should value the additional nitrogen emissions on the basis of the costs to offset them rather than on damage costs. However, environmental prices for nitrogen can obviously be used in situations where offsetting is not necessary.[15]

A similar situation could arise when EU countries are obliged to meet the targets set by the Water Framework Directive (WFD, 2000/60/EC) by 2027.[16] The objective of the WFD is to improve the quality of water systems, such as groundwater and surface water. It aims to reduce and prevent the pollution of water bodies, promote sustainable water use and reduce the effects of floods and droughts. The WFD sets specific goals for each surface and groundwater body and for specifically protected areas such as Natura 2000 sites. Some of the water bodies are not expected to meet the required quality standards and this may hamper the issuing of permits. Similarly, the policy context may impose constraints on activities so that valuation with environmental prices do not measure the correct price or shadow price of those constraints.

2.8.3 Limitations in the use of environmental prices for bioaccumulative substances

Bioaccumulative substances are those that persist in the environment because they do not degrade. Traditionally, heavy metals have been classified bioaccumulative because they are very poorly degradable. However, heavy metals also occur naturally in the biosphere, which is why organisms have found ways to interact with them. There is also a long tradition of research into the impact they have on the environment.

[15] Moreover, a researcher does have the ability to analyse the social costs and benefits of such strict environmental policy standards. It must be recognised, however, that there may be non-anthropocentric reasons for setting such standards and that they cannot be properly assessed through welfare economics.

[16] In a similar policy dossier, such as the European Marine Strategy Framework Directive, targets must officially be met even before 2020.

In contrast, many chemical compounds developed in the last century are also not degradable in the environment, but they do not occur naturally. One example is PFAS: a group of polyfluoroalkyl and perfluoroalkyl substances consisting of thousands of individual substances, which are poorly degradable or do not degrade in the environment. As a result, every emission increases the concentration in the environment. The effects of these substances are often poorly known or unknown. Despite the fact that European regulations on the authorisation of chemicals have greatly improved, we know little to nothing about their accumulative and long-term effects.

This Environmental Prices Handbook does not provide valuations for many of the bioaccumulative substances with unknown effects. This is due to the fact that the uncertainty is too large, and we cannot currently assess potential impact in the future. Instead of environmental prices, such substances should be addressed through risk analysis (see also Annex E).

2.8.4 Limitations in the use of environmental prices in non-average situations

Environmental prices are shown for an average emission at an average location in the EU. In addition, Paragraph 6.4.11 provides another breakdown by emission location for particulate matter to air.

As such, environmental prices cannot properly be used to estimate the damage of situations that are very different from this average situation, such as emissions on the open sea, emissions from a specific point source located in a city, etc. In such cases, specific environmental prices will need to be developed (see also Chapter 7).

SECTION 2: METHODOLOGICAL SECTION

3 Methodological framework

3.1 Introduction

Environmental prices are a key indicator that calculates the Willingness-To-Pay for reducing environmental pollution expressed in euros per kilogram of pollutant. Environmental prices therefore reflect the welfare losses that occur if one additional kilogram of the pollutant is released into the environment. In general, environmental prices are equivalent to external costs.

In this chapter, we outline the methodological framework for environmental prices. First, we explain the concept of external costs in Paragraph 3.2. Next, we present the framework for valuing external costs in Paragraph 3.3. This chapter provides a general introduction to environmental prices but does not specifically discuss how we arrived at an environmental prices valuation. Information on all the considerations involved in environmental prices is listed in Chapters 4, 5 and 6.

3.2 Economic significance of environmental prices

3.2.1 Background of 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 established directly – via market prices, for example – it must be calculated.

Research into the financial valuation of environmental impact 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, the valuation of environmental impact 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 and much has been undertaken in terms of both methodological development and numerical valuation (Hoevenagel & De Bruyn, 2008).

From the perspective of welfare economics, a clean environment can be understood as a 'public good'. The central characteristic of a public good is that it is 'non-rivalrous' and 'non-excludable' when consumed. Non-rivalry occurs when one person's consumption does not lead to reduced consumption opportunities for others. Non-excludability means that it is not possible to bar people from consumption at an acceptable cost if they do not pay for it. A clean environment is a textbook example of a public good: (i) consumption of a clean environment (e.g., by breathing clean air) does not prevent others from also enjoying a clean environment; (ii) it is not possible to delineate the availability of a clean environment, such that only people who pay for it can enjoy clean air. As a result, no individual and tradable property rights can be assigned to a clean environment.

Besides being a public good, a clean environment is also scarce since the availability of environmental services is limited and our consumption and production processes affect their availability (Hueting, 1980). In economic terms we can speak of the existence of negative externalities: side effects of production and consumption that affect the well-being of others without them receiving financial compensation for their loss of well-being. Lack of property rights is considered in neoclassical economics as the main reason for the existence of externalities (Coase, 1960); (Buchanan, 1985).

Internalising external costs so that they are included in policy considerations leads, *ceteris paribus*, to improved welfare. External costs are thus also an important component in the concept of 'welfare beyond GDP'. Welfare beyond GDP means everything that people consider of value (CPB & PBL, 2022). In addition to material welfare (income), it also concerns issues such as health, education, environment and living conditions, social cohesion, personal fulfilment, insecurity and safety.

3.2.2 Environmental prices as damage costs, including external damage costs

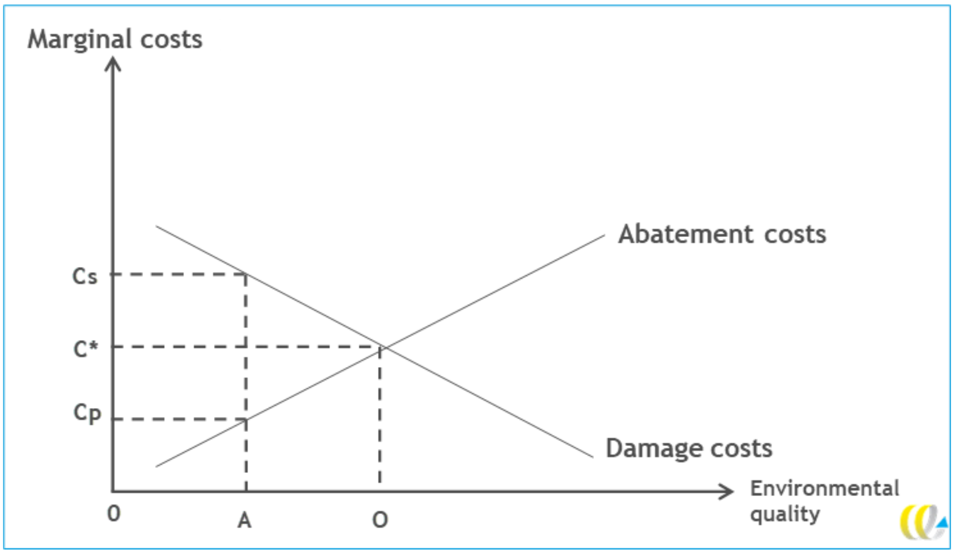
Environmental prices reflect the Willingness-To-Pay for preventing environmental pollution and are thus in principle equivalent to damage costs: where damage includes both damage to capital goods (natural and man-made), health and intangible damage. However, it is usually not optimal to reduce the damage to zero, as this can lead to very high costs. The optimal pollutant level is where the benefits of an additional unit of pollutant reduction outweigh the costs. This is illustrated in Figure 6 where supply and demand curves for environmental quality are shown. The demand curve is represented by the damage cost function: as the environment becomes cleaner, the Willingness-To-Pay for additional reductions in environmental pollution decreases.[17] The supply curve is determined by abatement costs. These increase as more pollution is reduced due to decreasing excess returns from pollutant reduction.

As an example, suppose that current environmental quality is *A* due to environmental policies with marginal abatement costs at level *Cp*. The current level of environmental quality (interpreted here as the inverse of pollution) is below the optimal level of *O* where damage costs and abatement costs intersect. The damage costs associated with the current situation are therefore represented at level *Cs*. The damage cost *Cs* in this case represents the value to be given to a small change in environmental quality. *Cs* represents the marginal cost of damage as the infinitesimal increase or decrease in damage due to an infinitesimal decrease or increase in environmental quality. This is referred to as environmental prices.[18]

[17] In practice however, marginal damage costs do not always decrease as the environment becomes cleaner due to the role played by a whole range of processes, such as atmospheric chemistry, which means that decreasing concentrations could even lead to higher damage costs. Similarly, to some extent people will often value damage to their health independently of the level of environmental pollution, which could flatten the damage cost function.

[18] In the Handbook on Shadow Prices 2010, this was referred to as Shadow Prices. Formally, shadow prices are the value of a constraint (the Lagrange factor) for the optimal solution, meaning it is the infinitesimal change in the objective function caused by an infinitesimal change in the constraint. Shadow prices are therefore the correct name for abatement costs. However, for the damage cost function, these are derived shadow prices of the limited presence of environmental quality due to policies. To avoid a semantic discussion, we chose to use the more neutral term 'Environmental Prices' in this study.

Figure 6 - Environmental prices in relation to damage costs and optimal pollution levels



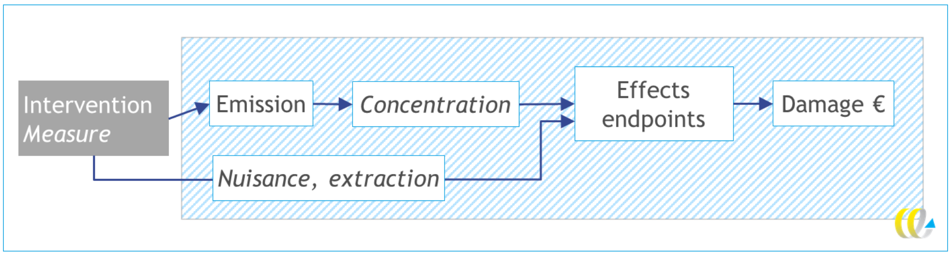
Environmental prices therefore indicate the relative value of emissions to each other and to other goods in society.

3.3 General framework for key indicators Environmental Prices Handbook

3.3.1 General framework

Environmental prices are determined based on a cause-effect relationship between emissions, the environmental impact and damage. The cause-effect relationship is represented in Figure 7.

Figure 7 - Relevant cause-effect relationships in this study



Any activity leads to a certain intervention in the environment. This could be emissions, nuisance or extraction, such as the consumption of water or the use of land or raw materials. In the case of emissions, these are transported via air, soil or water to other areas, where they contribute to a change in existing emission concentrations. This altered concentration then leads to changes in aspects relevant to human welfare, such as health or biodiversity. These 'aspects' are called endpoints in environmental science, and we also use this term in the Environmental Prices Handbook (see below for further explanation).
The damage caused at these endpoints is the starting point for the financial valuation of emissions. The whole relationship between emissions, nuisance or depletion and damage is the subject of this Environmental Prices Handbook. The effectiveness of interventions or policy measures is beyond the scope of the Handbook.

3.3.2 Relevant endpoints

The Environmental Prices Handbook refers to 'endpoints' as the ultimate consequences of environmental pollution that are important for human welfare. The endpoint level is that at which there are no longer any 'feedback' effects. This level thus forms the basis for valuation. 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 agricultural crop yields).
3. Buildings and materials (man-made capital).
4. Raw materials (stocks of raw materials).
5. Well-being (aesthetic and ethical values).

This categorisation is generally more comprehensive than what can be found in environmental literature. ReCiPe, for example, distinguishes three endpoints: human health, ecosystem services and scarcity of raw materials (Huijbregts et al., 2016). In the Environmental Prices Handbook, we try to include as many relevant welfare effects as possible and therefore take a broader perspective than environmental analyses. In Chapter 5, we examine people's Willingness-To-Pay for an improvement by reducing environmental pollution across these five endpoints. In Chapter 5 the endpoints are also described in more detail.

3.3.3 Relevant midpoints as a link between emissions and endpoints

There are more than 10,000 potentially environmentally hazardous substances and it would not be feasible to establish the relationship between emissions, concentrations and endpoints for all of them. This is why there is a step between this cause-effect relationship, the midpoints. Midpoints can be translated as environmental themes and refer to categories of environmental effects where various pollutants have similar physical effects.
For instance, greenhouse gases: both carbon dioxide and methane affect climate change, and the interrelationship between the warming potential of CO_2 and CH_4 can be expressed in terms of a number. This *Global Warming Potential* is usually represented in CO_2 equivalents where all greenhouse gases are given a score similar to CO_2 emissions for their warming potential.

Midpoints are frequently used in life cycle analyses and there is no agreement on the number of midpoints used. For the purposes of this update of the Environmental Prices Handbook, we follow the midpoints used in ReCiPe 2016 and distinguish the following eleven midpoints:
1. Ozone depletion.
2. Climate change.
3. Particulate matter formation.
4. Photochemical oxidant formation (damage to human health and ecosystems).
5. Acidification.
6. Ecotoxicity (freshwater and saltwater).
7. Human toxicity (carcinogenic and non-carcinogenic).
8. Ecotoxicity (terrestrial, freshwater and saltwater).
9. Ionising radiation.
10. Nuisance (noise and visual nuisance).
11. Extraction (land use, water consumption and fossil and mineral resource extraction).

These midpoints are described in detail in Chapter 6 and largely correspond to what is used in the literature for characterisation at the midpoint level, see (Guinée et al., 2002); (Goedkoop et al., 2013); (JRC, 2012). Relative to ReCiPe 2016, this means we are adding one midpoint: nuisance (noise pollution). In addition, there is a recommendation to include a separate midpoint to properly quantify the health effects of air pollution: nitrogen dioxide (see Paragraph 2.4).

Environmental prices at all midpoints can also play a role in SCBAs, but it is not recommended for the midpoint extraction: these environmental prices are so uncertain that we recommend specific valuation, should these aspects play a role in policy decisions. For use in LCA, these values can be included as indicative values where we recommend also performing analyses without them to gain more insight into the influence of these uncertain values on outcomes in in LCA (see Chapter 2).

A number of midpoints mentioned in literature (Guinée et al., 2002) have not been considered in this Environmental Prices Handbook. These relate primarily to interventions at the interface between nature and the environment:
- erosion of farmland soils;
- salinisation of farmland soils;
- light pollution;
- stench;
- visual impact (‘horizon pollution’);
- spread of invasive species.

These all impact primarily on the endpoints ‘ecosystems’ and ‘well-being’. In many cases there is no directly observable relationship between emissions and these midpoints. In addition, often no EU average can be calculated for these kinds of environmental impacts, 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. In case one can establish relationships between the endpoint-level variables and the interventions, then a valuation according to the Environmental Prices Handbook can be added.

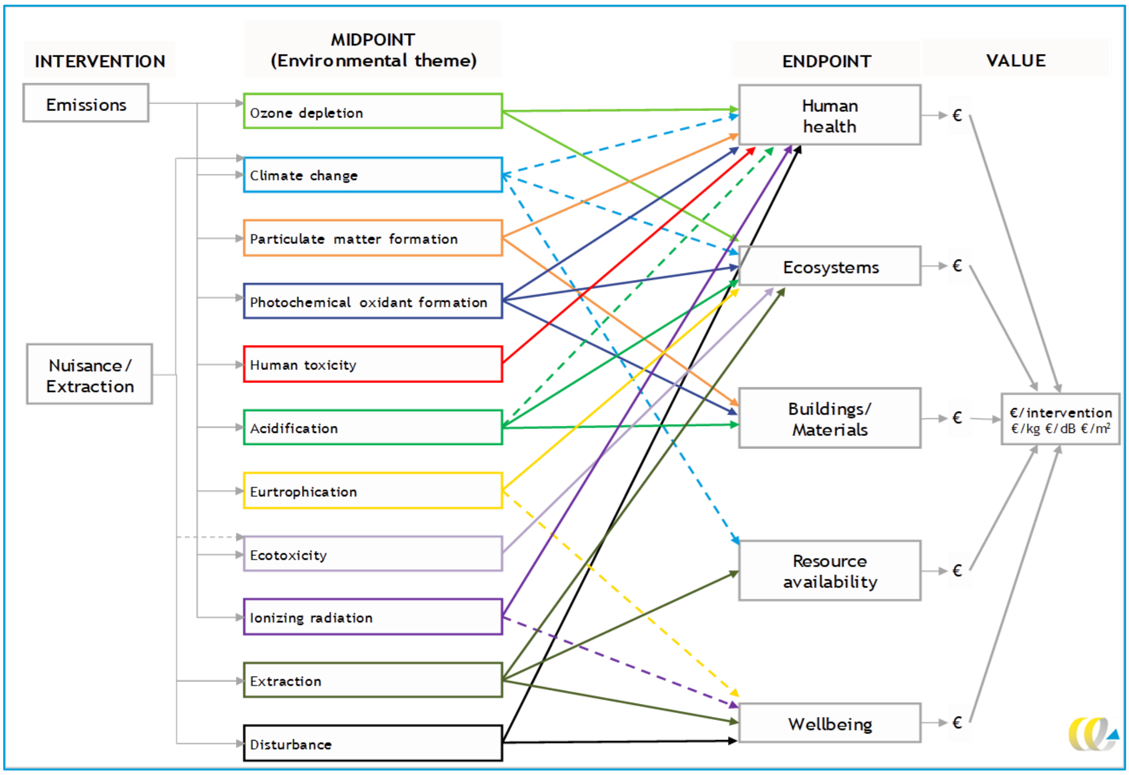
3.3.4 Relationship between pollutant, midpoint and endpoint levels

At the core of this Environmental Prices Handbook are two steps:
1. Establishing the relationship between environmentally hazardous substances (emissions) or causes of ‘disturbance’ (noise, land-use change) and their impact on midpoints and endpoints.
2. Valuing these endpoints and translating them back to damage per intervention.

This framework is outlined in Figure 8. This identifies all relationships between emissions, midpoints and endpoints and their valuation that are relevant to the Environmental Prices Handbook.[19]

[19] This is not to say that all relationships were actually determined quantitatively.

Figure 8 - Relationship between intervention, midpoints, endpoints and valuation in the Environmental Prices Handbook



Solid lines refer to relationships that have been investigated and partly quantified within the framework of this Handbook. Dashed lines represent relationships that are not directly quantified as relationships because a different approach was taken in this Handbook for quantifying the impact. Depletion includes land use. Disturbance also includes noise pollution. See Chapter 6 for further explanation.

Air pollution involving SO_2 can be used as an example. Emissions of SO_2 lead to an altered concentration of SO_2 in the air. This has an impact on the environmental themes of particulate matter formation and acidification. Each of these themes produces an impact on ecosystems (soil degradation and consequent loss of biodiversity), human health (inhalation of particulate matter caused by SO_2) and buildings (pollution and damage to cultural heritage). This impact is valued financially. Subsequently translating that financial valuation back into emissions of SO_2 and adding them together gives a valuation for a kilogram of SO_2.

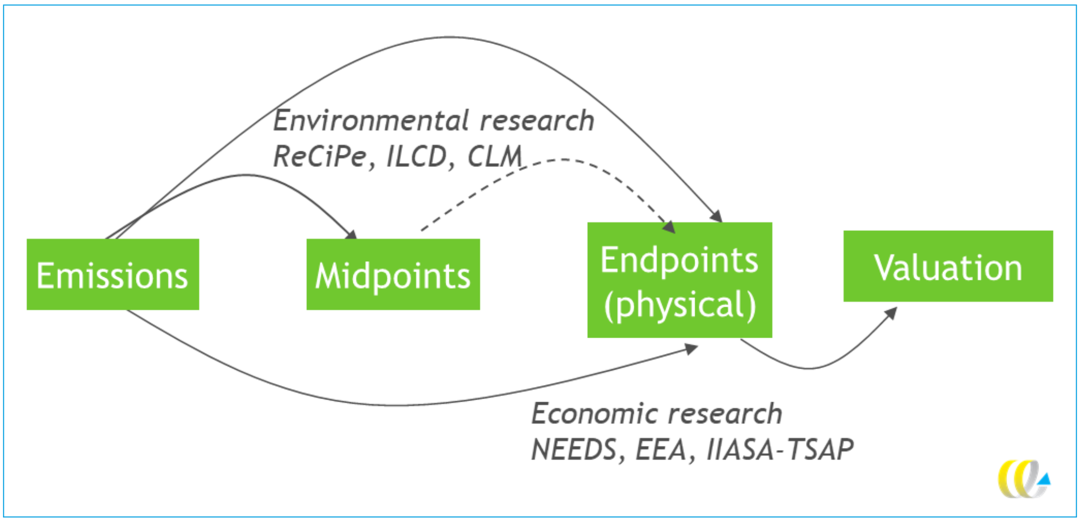
The relationships between emissions and impact at the endpoint level are laid out in Impact Pathway models. An Impact Pathway model describes the causal link between emissions through a change in concentrations in endpoint-level impact. The Impact Pathway approach has played an important role in two types of research used in the Environmental Prices Handbook:

1. Environmental research, as in life cycle impact analysis (LCIA) where physical-chemical models and Impact Pathway models are used to determine relationships between pollutants and midpoints on the one hand and pollutants and endpoints on the other. This produces numbers representing the interrelationships of pollutants at midpoints and endpoints. This information is used in LCA software packages such as SimaPro. In particular, environmental research is very much focused on describing the physicochemical effects of emissions and their relationships with the endpoints as precisely as possible.

2. Economic valuation studies, such as (NEEDS, 2008b), (Holland, 2014) or (EEA, 2021a), in which dispersion models and concentration-response functions are used to establish a relationship between pollutant level (emissions) and valuation at endpoint level. Economic research combines Impact Pathway modelling with economic valuation techniques to calculate the social damage of an emission. The focus here is on valuation and use in social cost-benefit analyses.

   Both forms of research model part of this chain of relationships, as described in Figure 9.

Figure 9 - Relationships between emissions, midpoints, endpoints, valuation and relevant research fields



The dotted line indicates that these steps were used in the study but not frequently applied.

The main differences between the two modelling approaches are listed in Table 10. These show that the environmental models are generally comprehensive with regard to the number of pollutants considered, but the economic models are more sophisticated in distinguishing between countries/regions and in distinguishing and characterising of endpoints. In principle, economic models transparently align with epidemiology in terms of describing dose-effect relationships. Such information is often also included in environmental models, but less transparently because other existing models are used.

Table 10 - Differences between economic and environmental models

	Environmental models	Economic models
Purpose of the model	Characterisation: expressing the impact of emissions on environmental themes	Valuation: damage cost calculation of pollutants
Number of pollutants included in the analysis	As many pollutants as possible (> 3,000)	Focus on various pollutants with the largest damage costs, especially with regard to air pollution
Scale of the analysis	European or global, in recent years there has also been a trend towards national analysis	Countries and regions
Number of endpoints	Especially human health and ecosystems	Human health, ecosystems, buildings and materials

	Environmental models	Economic models
Refinement within endpoints	Limited or not directly transparent	High degree of refinement and transparency; for example, human health is broken down into twelve endpoints (see Annex A)

Both studies therefore outline a relationship between emissions and endpoints, but with different emphases in terms of assumptions and detail. The great advantage of ReCiPe, for example, is its attempt at consistency between the impact at midpoints and endpoints (Goedkoop et al., 2009); (Huijbregts et al., 2016). However, the drawback of ReCiPe for monetary valuation is that the relationship between an emission and endpoints is primarily represented as an average global value. Economic studies, on the other hand, also establish a relationship between emissions and their impact, but do so in a way that paints with a broad brush in environmental terms. In addition, the main drawback of the economic studies is that the relationship between emissions and damage at the endpoint level has only been established for a limited number of environmentally damaging pollutants. This means that for the thousands of other environmentally damaging pollutants, these studies provide no useful information.

3.3.5 Combining modelling approaches

The methodology of environmental prices involves combining these environmental and economic models to arrive at a consistent estimate of the welfare costs associated with emissions at the pollutant, midpoint and endpoint levels.
This involves four steps:

1. **Establishing an endpoint-level valuation framework.** First, a consistent set of endpoint-level valuations is established for the EU for the impact on human health, ecosystems, buildings/materials, raw materials and well-being.
2. **Economic models for the relationship between emissions and valuation endpoints.** Economic models are then used to estimate the relationship between emissions and the impact at endpoints for about forty primary pollutants. These endpoints are valued, leading to damage estimates per pollutant consistent with the valuation framework.
3. **Environmental models for allocating damage estimates from emissions to midpoints.** These damage estimates are then distributed among the various midpoints using data from environmental models and the contributions per pollutant are weighted by emissions in the EU. This produces the 'mid-point price': the damage estimates for the environmental theme.
4. **Environmental models for the relationship between midpoint and pollutants.** As a final step, the scores from the ReCiPe characterisation model for each pollutant are multiplied by the midpoint price and added together. In this way, a damage estimate is constructed for each pollutant from the underlying impact at midpoint-level.

Chapter 4 explains these steps in more detail. The question is sometimes asked why we are complicating things in the Environmental Prices Handbook, since every pollutant in ReCiPe also directly quantifies the physical damage at the endpoints. Would it not be better to value the endpoint characterisation directly? This is of course possible, but it results in a less accurate estimate because some of the refinement found in economic/epidemiological models is then missing. In the case of ecotoxicity or ozone depletion, however, there are insufficient economic studies available, which is why we do value them directly through the endpoints from ReCiPe.

In Chapter 4 we look in more detail at how the various modelling approaches have been harmonised in this Environmental Prices Handbook. In Chapter 5 an extensive review is provided of the valuation methods adopted.

4 Environmental prices methodology

4.1 Introduction

In this chapter, we discuss the manner in which we technically implemented environmental prices. Paragraph 4.2 describes the general methodology based on harmonising assumptions of valuation methods, Impact Pathway analyses and environmental characterisation models. This is essentially the same methodology that underpinned the Environmental Prices Handbook 2018 and the Shadow Prices Handbook 2010. However, this methodology will be explained in more detail because questions often arose in the past about calculation steps. We then outline how changes have been made to the methodology compared to the Environmental Prices Handbook 2018.

Paragraph 0 then presents the valuation framework developed in this Handbook (more details can be found in Chapter 5), Paragraph 0 covers the Impact Pathway approach to some 40 primary pollutants (more detail can be found in Chapter 6 and Annex A), and in Paragraph 4.5 we discuss the characterisation models chosen (more detail can be found in Chapter 6 and Annex B). Finally, in Paragraph 4.6 we outline the process by which we determined the environmental prices for 3,000 pollutants.

4.2 General methodology

The general methodology of the Environmental Prices Handbook consists of four steps. These steps are shown graphically in Figure 10.

Step 1: Establishing a valuation framework

First, a consistent framework of valuations is determined at the endpoint level for premature death, various diseases caused by environmental pollution, genetic effects (IQ loss), biodiversity loss, ecosystem productivity, damage to buildings and materials, scarcity of raw materials and well-being. In the first step, valuations are established for each of the five selected endpoints, which are in line with international literature. This yields values for human health, biodiversity, agricultural crops and material restoration costs, all at 2021 prices. These valuations are discussed in outline in Paragraph 0. More details can be found in Chapter 5, Paragraphs 6.3 and 6.11 and in the Annexes.

Step 2: Establish environmental prices for primary pollutants

Subsequently, for some 40 *‘primary pollutants’*, largely using Impact Pathway models, damage costs are calculated for the emission of 1 kg of that pollutant from Dutch territory.[20] These primary pollutants include emissions to air of$PM_{2,5}$, PM_{10}, elemental

[20] We have qualified ‘primary’ to indicate that an accurate environmental price is determined for these pollutants using Impact Pathway analysis. This accurate determination is then used at a later stage to calculate environmental prices for all pollutants.

carbon, SO_2, NO_x, NH_3, NMVOC, arsenic (inorganic), cadmium, chromium VI, lead mercury, nickel, benzene, butadiene, benzo(a)pyrene, dioxin, formaldehyde, CO_2 and CFCs and to water from N and PO plus several radionuclides (radioactive substances) for their emissions to air and water.[21] It is important to note here that the IPA approach is applied using adjustments of results from existing models (EEA, NEEDS, literature). For elemental carbon and CO_2, the valuation is not determined via an IPA approach but is based on literature.

Step 3: Allocation of midpoint environmental prices

In a third step, the damage estimates of these individual pollutants are imputed to the various environmental themes using familiar environmental models from life cycle assessment (LCA). We assign the components of the pollutant-level damage estimates to the LCA themes and then determine how much a given pollutant contributes to the midpoint level based on emissions in the EU. This way, an emission-weighted midpoint price is created. This is the same methodology as used in the Shadow Prices Handbook (CE Delft, 2010) and the Environmental Prices Handbook 2018 EU version. In this update, we apply new impact assessment models for use in LCAs (ReCiPe 2016 and PEF).

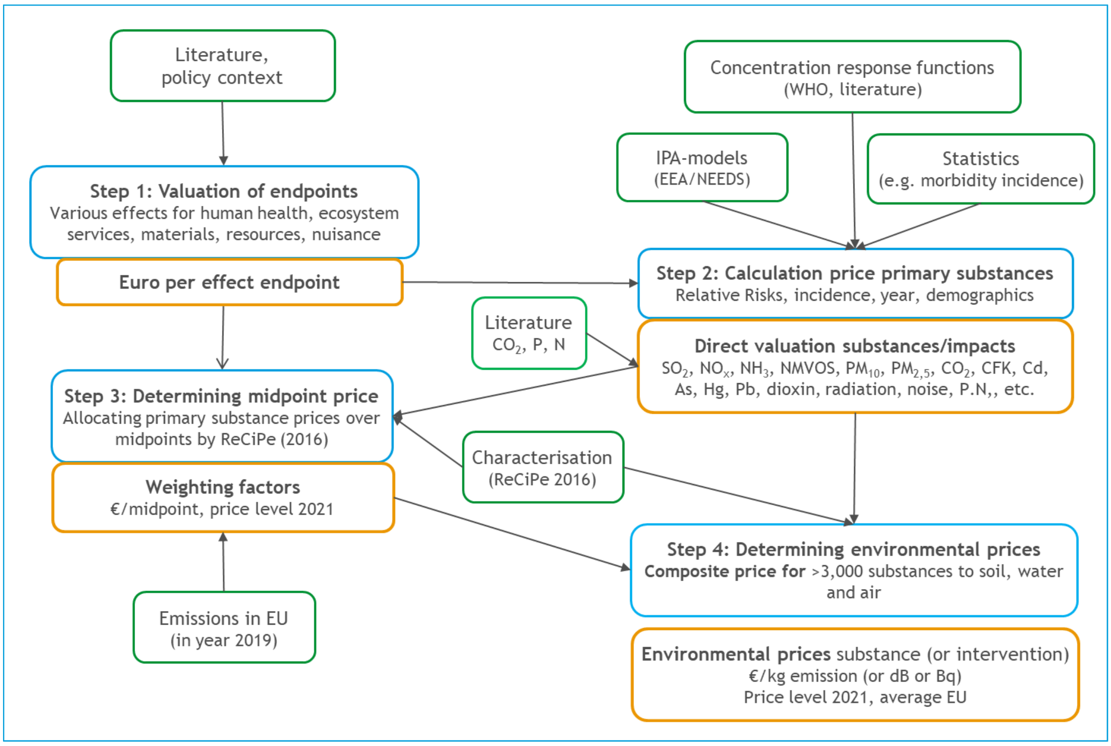
Step 4: Differentiation across all emissions

In the final step, the midpoint prices are differentiated across individual pollutants by adding together all damage costs per midpoint of the pollutant. To give one example, we explain this step using emissions from pentane (C_5H_{12}), a chemical gas frequently used as a solvent, blowing agent or refrigerant. In ReCiPe 2016, an emission of pentane to air has an impact on human damage from oxidant formation, ecosystem damage from smog formation, terrestrial ecotoxicity, marine ecotoxicity, freshwater ecotoxicity and human toxicity. By multiplying the characterisation factor for all these midpoints by the midpoint environmental prices from Step 3, a composite environmental price for pentane emerges that is constructed from all underlying valuations.

The following figure outlines the methodology. The various data sources and models used in the Environmental Prices Handbook are shown in green. Blue describes the four steps on the basis of which environmental prices are calculated and yellow shows the outputs of each step. As can be seen, the Handbook's environmental prices are the result of all the steps. Below, we elaborate on the four steps and indicate how these steps are completed differently compared to the Environmental Prices Handbook 2018 EU version.

[21] These include Carbon-14, Cesium-137; Hydrogen-3; Iodine-129,131 and 133; Krypton-85; Radon-222; Thorium-230; Uranium-234, 235 and 238; Lead-210; Polonium-210, Radium-226, Strontium-90 and Rubidium-106.

Figure 10 - Methodology overview of Environmental Prices Handbook



Green-circled fields show the inputs for the respective step and yellow-circled fields show the outputs of that step flowing through to subsequent steps.
IPA = Impact Pathway Approach.

4.3 Step 1: Valuation of endpoints

The first step is to develop a valuation framework for the various endpoints related to human health, ecosystem services, impact on buildings/materials and nuisance/well-being. This valuation framework determines how these endpoints are valued: a valuation that also plays a role in Step 2 and 3.

First of all, all endpoints that are distinguished in the models created in Step 2 and 3 are identified. Subsequently, each of these endpoints is assigned a value in the valuation framework. This step is described in detail in Chapter 5 of the Handbook with regard to the valuation of:
1. Human health (Paragraph 5.3).
2. Ecosystem services (Paragraph 5.4).
3. Damage to buildings and materials (Paragraph 5.5).
4. Availability of raw materials (Paragraph 5.6).
5. Nuisance/well-being (Paragraph 5.7).

Multiple effects are valued within each of these endpoints (see Table 11).

Table 11 – Endpoints and impact valued in the Environmental Prices Handbook

Endpoint	Impact valued for this endpoint
Human health: mortality	Premature death (VOLY), post-neonatal infant mortality
Human health: morbidity	Hospital admissions, chronic bronchitis (COPD), lost working days, restricted activity days, childhood asthma and bronchitis, diabetes, medication use, cancer
Ecosystem services	Agricultural and forestry yields, biodiversity (species.year)
Materials/buildings	Repair costs (including cultural heritage) and clean-up costs
Raw material scarcity	Cost of strategic stocks, economic damage price fluctuation, cost of recycling, diminishing capital gains from mining
Well-being	Noise pollution from various sources and levels, visual nuisance

4.3.1 Situation of the previous and new Handbook

The valuation framework presented in the previous Environmental Prices Handbook 2018 was based on:

- an analysis of preferred approaches for valuing the impact of environmental pollution;
- literature analysis of the most plausible valuations for all impact distinguished in Table 12;
- routine of converting all values found in the literature to a uniform price level for the year 2015.

This approach has not changed in this new Environmental Prices Handbook. However, new research has been conducted into the most plausible valuations as several new studies have appeared in the past seven years that have valued the impact of environmental pollutants. Chapter 5 gives a detailed account of this. In broad terms, the following changes have been made in this new Handbook:

- all prices have been adjusted to the 2021 price level;
- valuation of mortality is the same as in the Environmental Prices Handbook 2018, but has been adjusted to an income elasticity greater than 0;
- valuation of morbidity has been redefined using information from the literature and our own calculations;
- valuation of ecosystem services has been updated to reflect new literature and the calculations have been revalidated;
- valuation of the scarcity of raw materials has been added in this Handbook, incorporating a breakdown by valuation for metal depletion, fossil fuels and water supply.

Chapter 5 explains the choices made in more detail. Table 12 provides an outline of the adjustments made in this Handbook.

Table 12 - Valuation structure adjustments

	Environmental Prices Handbook 2018	Environmental Prices Handbook 2024
Price level	Prices 2015	Prices 2021
Income elasticity	0%. There is also no adjustment of prices to income elasticity between 2005 and 2015.	0.3-1%
Discount rate	3%	2.25%
Valuation of mortality	The range for mortality at 2015 prices is €50,000 as a lower value, €70,000 as a central value and €100,000 as an upper value.	A range of €57,500 for the lower value, €85,000 for the central value and €129,000 for the upper value.
Valuation of morbidity	Adjusted based on NEEDS (2008a).	Redetermined using publicly available literature and data.
Valuation of crop damage	Based on NEEDS (2008a).	Based on (EEA, 2021a)
Valuation of ecosystems	Based on (Kuik et al., 2008).	Based on (Kuik et al., 2008) and (Costanza et al., 2014).
Valuation of buildings/ materials	Low values based mainly on repair costs of physical damage.	Higher values because clean-up costs have also been added.
Valuation of the scarcity of raw materials	No valuation added.	A valuation range based on damage costs and abatement costs has been added, including recommendations for inclusion in LCA and SCBAs.

4.4 Step 2: Determining environmental prices of primary pollutants

Based on the valuation system in Step 1, environmental prices are determined for about forty individual pollutants using the Impact Pathway approach. These pollutants are also referred to as 'primary pollutants' and include CO_2, $PM_{2,5}$, PM_{10}, elemental Carbon, SO_2, NO_x, NH_3, NMVOC, arsenic (inorganic), cadmium, chromium VI, lead, mercury, nickel, benzene, butadiene, benzo(a)pyrene, dioxin, formaldehyde, N and P via surface water and several radionuclides.

The relationship between an emission in the EU and the impact on the different endpoints is estimated for each of these pollutants using Impact Pathway Approach (IPA) models. This means taking a model-based approach to cause-effect relationships involving environmental pollution. In this process, a relationship is established between emission of a pollutant at a particular location and its effect at endpoints. An IPA therefore follows a pollutant from the moment it is emitted somewhere, via transport through the various environmental compartments (water, air, soil) to its effects on people, ecosystem services and human capital goods.

IPA models (see also Annex A) describe the relationship between emissions and the impact at endpoint level. By valuing this impact using the valuations from Step 1, a valuation is created for an emission of a primary pollutant and the subsequent damage caused by it. Several IPA models are distinguished in the literature. The Environmental Prices Handbook is always based on European IPA models developed in European Framework Projects, which are also known as NEEDS models (NEEDS, 2008b); CAFE-CBA, Gains (IIASA, 2014) and EEA (EEA, 2021a). Although these models differ in outcome and assumptions, they broadly follow the same approaches.

In fact, these European IPA models consist of four linked models/ databases:
1. Emission databases (and/or emissions forecasts).
2. Dispersion models that translate emissions into concentrations and combine meteorological models with models describing effects in atmospheric chemistry.
3. Concentration Response Functions (CRFs) that translate concentrations of emissions into a physical impact on endpoints such as health, ecosystem services and buildings.
4. Monetary valuation of this physical impact.

4.4.1 Status of the Environmental Prices Handbook 2018 and updated Handbook 2024

The previous Environmental Prices Handbook 2018 used the Impact Pathway approach from the NEEDS project with regard to air pollution and heavy metals. NEEDS results (NEEDS, 2008b) were corrected for changes in population size and population composition, for changes in atmospheric chemistry (background concentrations) and for changes in concentration response functions (see text box below).

Concentration response functions in the Environmental Prices Handbook 2017 and European Handbooks
Concentration response functions show the relationship between a change in concentration and a change in various ‘endpoints’, such as premature death or hospitalisation. The Environmental Prices Handbook 2017 and the European versions ((CE Delft, 2017a) and (CE Delft, 2018b)) contain different concentration response functions because they have dealt with the HRAPIE guidelines (WHO, 2013) to different degrees. In the Environmental Prices Handbook 2017 for the Netherlands, the CRFs for NO_2 and O_3 have been adjusted to WHO HRAPIE guidelines but the CRF for PM_{10} and $PM_{2,5}$ have remained the same. In the European Handbooks, all CRFs have been adjusted to the (WHO, 2013) guidelines.

For various reasons, the NEEDS approach is not entirely satisfactory nowadays:
1. NEEDS modelling assumes a grid cell approach of a 50 x 50 km resolution. Nowadays, an impact is calculated on a much finer spatial scale.
2. The concentration-response functions are obsolete because they generally assume an incidence of disease burden in the year 2000 or even earlier (1995). Anno 2019, for example, cancer survival rates have improved dramatically, which should also lead to a reduction in mortality rates due to air pollution (see also Annex A).
3. The NEEDS project assumes the year 2000 situation for atmospheric modelling. This is very outdated. On the one hand, the concentration of pollutants in the atmosphere today is fundamentally different from the year 2000. In fact, all pollutants have decreased significantly, but the decrease was smallest for nitrogenous pollutants (especially NH_3). As a result, emissions of SO_2 are much more harmful today due to more SO_2 reacting into secondary particulate matter. Therefore, the methodology from the year 2000 no longer applies well to the situation in the EU.

Partly for these reasons, we decided to use the modelling from the (EEA, 2021a) in the Environmental Prices Handbook 2024 as a starting point for determining the impact of air pollution - especially on human health. We also rely on (EEA, 2021a) for the valuation of toxic substances, in the same manner as we did in (CE Delft, 2022b) around determining the environmental prices of waste. For phosphates and nitrates in surface waters, we base ourselves partly on the Impact Pathway approach taken in (IEEP et al., 2021), but we also check this with other approaches. This revises the entire empirical basis at pollutant level in the Environmental Prices Handbook.

4.4.2 Impact on human health

(EEA, 2021a) provided an update to their 2014 study on the effects of air pollution from industrial emissions. We rely on the results of this study in the Environmental Prices Handbook and indicate below in what ways (EEA, 2021a) differs from the modelling in the (NEEDS, 2008b) project used as a starting point in the previous Environmental Prices Handbook 2018.

The EEA 2021 methodology made the following improvements compared to the NEEDS project:

1. The modelling in the NEEDS project used a spatial resolution of 50 x 50 km and was based on meteorological conditions in the year 2000. EEA uses a spatial resolution of 0.2°x 0.3° (about 22 x 33 km) and for NO_2 an even finer resolution of 7.5 x 7.5 km. The meteorological conditions are based on the situation in the year 2017.
2. Concentration response functions in EEA are based on (WHO, 2013). In some scenarios, additional sources are added. The concentration response functions in the NEEDS project were based on older WHO figures (WHO, 2003);(WHO, 2006).
3. Atmospheric chemical modelling in the EEA project is based on more recent literature than in the NEEDS project.
4. In addition, a damage cost is allocated for Secondary Organic Aerosols (SOA), while the NEEDS project did not include this (see also the discussion in Paragraph 6.4).

In this Handbook, we therefore use the EEA 2021 results and modelling instead of the NEEDS model. We apply the relative risks (RR), in a similar way to (EEA, 2021a), as recommended in the report by (WHO, 2013). The relative risks show what the increased probability is of a given health endpoint as a result of an increased concentration of environmental pollutants. Their advantage over the CRFs from NEEDS is that they are not tied to absolute numbers (such as for deaths) in the past but can be applied to disease burden and mortality rates in the measurement year. This approach therefore makes the calculation more accurate and flexible to apply. Annex A describes the method in more detail.

Several things are modelled differently in this Handbook than in (EEA, 2021a), which changes the valuations. Specifically, this Handbook contains the following changes compared to EEA 2021 methodology:

1. The relative risk of premature death from $PM_{2,5}$ has been set at 1.08 per 10 $\mu g/m^3$ in line with (Chen & Hoek, 2020). See also Paragraph 6.4 and Annex A.
2. The relative risk of premature death from NO_2 has been set at 1.01 per 10 $\mu g/m^3$ in line with (COMEAP, 2018). See also Paragraph 6.5 and Annex A.
3. The incidence of disease burden has been determined as much as possible on a statistical basis rather than calculated using the European guidelines in (WHO, 2013). See also Annex A.
4. The number of life years lost has been determined with lifetables modelling the effects of air pollution for each 1-year age cohort. See also Annex A.
5. The valuations include a lower, central and upper variant (see Paragraph 5.3).
6. Separate valuations have been added for black carbon (see Paragraph 6.4).

In the case of toxic substances with human health effects, we also based ourselves on the European prices as stated in (EEA, 2021a). Paragraph 6.8 contains more detail on how we dealt with these toxic substances.

4.4.3 Impact on nature

For the impact on nature, we rely on existing models:
- The impact on agricultural crops due to oxidant formation is derived from (EEA, 2021a).
- The impact on biodiversity is based on (NEEDS, 2008b).

Although (EEA, 2021a) also calculated valuations for eutrophication on biodiversity, we have not adopted these because the results are difficult to reduce to physical effects, such as those on the indicator PDF (see Paragraph 5.4). Moreover, the results from the EEA study relate only to biodiversity loss in Natura 2000 areas. The size of Natura 2000 areas is an important measure of the relative damage burden of, for example, NH_3. It does not take into account that countries with a low proportion of Natura 2000 sites would have more available for protection due to diminishing marginal utility. For these reasons, we focused on the framework provided by (NEEDS, 2008b) when valuing biodiversity losses.

We scaled the impact determined in (NEEDS, 2008b) to actual emission levels compared to the relationship that could be derived between impact on PDF and emissions in the different scenarios using the EcoSense model. This routine is similar to that in the Environmental Prices Handbook 2018.

4.4.4 Impact on materials and buildings

The impact on materials and buildings involves the effects on cultural heritage, clean-up costs and weathering of paint. In the Environmental Prices Handbook 2018, clean-up costs are only quantified in the upper estimate of environmental prices.

For the current Handbook, we conducted new source research on specific clean-up costs (window lapping, façade cleaning) because it emerged from foreign literature that these can account for a substantial share of total clean-up costs in a country. By relating these to the total clean-up costs spent on glass and façade cleaning, we arrived at an estimate of the damage costs of air pollution on building maintenance.

4.4.5 Greenhouse gases

The valuation of greenhouse gases is not based on an Impact Pathway Approach (IPA) but on a literature analysis of the costs of policy targets. Paragraph 6.2 explains how we valued greenhouse gases.

4.5 Step 3: Midpoint prices using allocation via the characterisation model

In Step 3, the individual prices of various pollutants are allocated across the various midpoints determined in LCA models through characterisation. Characterisation is a quantification of emissions based on their contribution to a specific environmental theme. An example is the contribution of CO_2 and CH_4 emissions to climate change. In characterisation models, these emissions are quantified using a common unit for specific environmental themes, allowing aggregation and expression in a specific score. For climate change, this common unit is kg CO_2 equivalents (kg CO_2-eq.). CO_2 has a characterisation factor of 1 kg CO_2-eq./kg CO_2 and for methane it is 34 kg CO_2-eq./kg CH_4

(ReCiPe 2016 - hierarchical perspective).[22] Thus, the characterisation factor describes the load per environmental theme per amount of a specific emission.

In addition to the choice of a characterisation model, Step 3 also involves a decision on the allocation of individual environmental prices at the pollutant level (Step 1) across various environmental impacts and the subsequent calculation of the midpoint price.

4.5.1 Characterisation and allocation in the Environmental Prices Handbook 2018

In the Environmental Prices Handbook 2018, the characterisation was based on ReCiPe 2008 (update 2013) (Goedkoop et al., 2013). In that version, the individualistic perspective was adopted as a general starting point. For some environmental themes (metal toxicity, land use), a hierarchical approach was preferred. ReCiPe 2008 characterised the various pollutants based on a European average.

The pollutants were then allocated to the selected environmental themes in the Handbook, partly based on NEEDS modelling results and partly on characterisation factors. Annex G in the 2018 Handbook provides an example of how this division was applied for the theme of photochemical oxidant formation.

4.5.2 New developments relevant to Environmental Prices Handbook 2024

Since 2018, two major developments have taken place in the field of characterisation of environmental themes:
1. A new version of ReCiPe was released in 2016 (Huijbregts et al., 2016). ReCiPe 2016 includes characterisation factors for more environmental themes than ReCiPe 2008. Moreover, these factors have been updated to the latest scientific standard. The exact differences between ReCiPe 2008 and ReCiPe 2016 will be discussed for each environmental theme in Chapter 6.
2. The development of the Product Environmental Footprint and Organisation Environment Footprint (PEF/OEF) frameworks (EC, 2021). The PEF partially uses different environmental models than ReCiPe 2016. Annex B provides an overview of the differences between ReCiPe 2008, ReCiPe 2016 and PEF.

4.5.3 Choices in the Environmental Prices Handbook 2024 EU version

The choice of characterisation in the Environmental Prices Handbook 2024 is based on the ReCiPe 2016 characterisation factors (version 1.1 update January 2018). This means that the environmental modelling for several themes follows the framework of ReCiPe 2016. This Handbook will also include a valuation of all midpoints in the ReCiPe framework. Table 133 provides an overview of the various midpoints monetised in this Handbook and to which endpoint these midpoints relate.

[22] Annex B describes in more detail exactly which characterisation factors were chosen. For greenhouse gases, this is based on the latest IPCC report (IPCC, 2022). These factors are also used in other sources such as the Environmental Accounts (Milieurekeningen).

Table 13 - Relationships between midpoints and endpoints in determining environmental prices

	Midpoint	Endpoint	Determination^
1	Global warming	Human health, ecosystems	Weighted pollutant price
2	Ozone depletion	Human health, ecosystems	Valuation EP
3	Ionising radiation	Human health	Weighted pollutant price
4	Particulate matter formation	Human health, buildings/materials	Weighted pollutant price
5	Photochemical oxidant formation	Human health, ecosystems, buildings/materials	Weighted pollutant price
6	Toxicity (cancer)	Human health	Weighted pollutant price
7	Toxicity (non-cancer)	Human health	Weighted pollutant price
8	Water consumption	Human health, ecosystems	Valuation EP
9	Acidification	Ecosystems, buildings/materials	Weighted pollutant price
10	Ecotoxicity - Terrestrial ecosystems	Ecosystems	Valuation EP
11	Land use - occupation and transformation	Ecosystems	Weighted pollutant price
12	Eutrophication - Freshwater ecosystems	Ecosystems	Combination
13	Ecotoxicity - Freshwater ecosystems	Ecosystems	Valuation EP
14	Ecotoxicity - Marine ecosystems	Ecosystems	Valuation EP
15	Eutrophication - Marine ecosystems	Ecosystems	Combination
16	Scarcity of mineral resources **	Raw material scarcity	Weighted pollutant price
17	Scarcity of fossil resources**	Raw material scarcity	Weighted pollutant price

* This midpoint also characterises ecosystems. In the valuation, we used a prevention cost approach that implicitly includes ecosystem valuation.

** Indicative prices are included for this midpoint and not the final prices in this Handbook.

^EP = endpoint, see the text below for an explanation.

These midpoint prices were determined in two ways:

1. Using the **weighted pollutant price method.**
   This involves allocating price components per main pollutant to the various midpoints and weighting them by emissions. This was made possible by using disaggregated results made available to us by the researchers responsible for the EEA 2021 study. These allowed a direct allocation to the various environmental themes based on the damaging nature of the pollutants for those environmental themes. The midpoint price is then calculated as an emission-weighted average price of the pollutants characterising on that theme. This is done by using the following formula:

$$Midpoint\ price\ Theme, j \ = \frac{\sum_i DC_{i,j} * \ E_i}{\sum_i CF_i * \ E_i}$$

   where CF = characterisation factor for pollutant i on theme j, E are the emissions from pollutant i and DC are the damage costs from pollutant i on theme j. This method involves determining the total environmental prices per theme by dividing the total damage due to emissions on the environmental theme by the total score on the unit of the characterisation factor. Thus, the damage costs of various pollutants are weighted by their emissions (and thus their occurrence).

2. Via the midpoint to endpoint characterisation factors and a direct **valuation of the endpoint.** For some themes (ozone depletion, ecotoxicity and water consumption), individual pollutant prices were not available, and a different route was required. Here, the midpoint to endpoint factors were used from the ReCiPe 2016 study. These are multiplied by the price per endpoint, as mentioned in Chapter 5.

4.5.4 Result: midpoint prices

The result of Step 3 is a set with environmental prices for each midpoint. This Handbook presents midpoint prices for emissions in the EU according to the ReCiPe 2016 characterisation model.

In addition, the methodology of Chapter 7 of the Environmental Prices Handbook has been applied to other emissions and other characterisation models resulting in two more types of midpoint prices:
- for emissions in the EU27 according to ReCiPe 2016;
- for emissions in the EU27 according to the PEF characterisation.

For the latter, only some midpoint prices are given that correspond to the more certain impact determinations (CAT I and II, see Annex B).

4.6 Step 4: Calculation of environmental prices

Based on these environmental prices by midpoint, as a final step, a comprehensive list of implicit environmental prices can be created for all pollutants appearing in ReCiPe (in addition to those for which an individual environmental price has been calculated). This is done by using the environmental ratio of pollutants contributing to the same environmental theme as determined in ReCiPe. By adding together all damage costs determined at each endpoint/midpoint combination from Table 13 for each pollutant, a composite damage cost per pollutant is obtained. As an example, we will use emissions from pentane (C_5H_{12}), a chemical gas frequently used as a solvent, blowing agent or refrigerant. In ReCiPe 2016, an emission of pentane to air characterises on human damage from oxidant formation, ecosystem damage from smog formation, terrestrial ecotoxicity, marine ecotoxicity, freshwater ecotoxicity and human toxicity. By multiplying the characterisation factor for all these midpoints by the midpoint environmental prices from Step 3, a composite environmental price for pentane emerges that is derived from all underlying valuations.

Annex F provides the main valuations to air, soil and water for more than 250 pollutants. The selection of these pollutants is based on those listed in the Emission Inventory supplemented by a number of Substances of Very High Concern for which emission inventory data are not available.

5 Valuation of damage at endpoint level

5.1 Introduction

In this chapter, we discuss the endpoint valuations used in the construction of environmental prices. The valuation of damage at the endpoints is the most important variable in establishing environmental prices.

These valuations are based on literature review. First, we give a general overview of valuation methods in Paragraph 5.2.
We then go into more detail on valuation of the various specific endpoints:
1. Human health (Paragraph 5.3).
2. Ecosystem services (Paragraph 5.4).
3. Buildings and materials (Paragraph 5.5).
4. Availability of raw materials (Paragraph 5.6).
5. Well-being and other environmental qualities, such as noise (Paragraph 5.7).

In each of these paragraphs, we will provide a justification for the choices made in the previous 2018 Environmental Prices Handbook and discuss to what extent these choices need adjustment in the Environmental Prices Handbook 2024 EU-version. For each endpoint, we then calculate values used in the valuation of emissions and midpoints.

5.2 General methodology valuation

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: i.e. their Willingness-To-Accept (WTA). The concepts of WTP and WTA are therefore both defined in terms of individual preferences.

In traditional economic valuation literature, if market prices are not available, Willingness-To-Pay can be estimated in two different ways:
1. **Revealed preferences.** These preferences are reflected in the actual choices people make in other areas than the one studied. For instance, recreational travel costs may implicitly indicate something about the appreciation of a particular nature conservation area.
2. **Stated preferences**. These preferences are derived from surveys that measure people's WTP for maintaining or improving environmental quality.

It is difficult to ascertain Willingness-To-Pay for many environmental issues, either through revealed or stated preferences, because most people have no real understanding of what environmental quality means for their lives. Surveys with questions such as ‘How much would you be willing to pay for a reduction of emissions of NO_x by 1 kilotonne?’ will not yield meaningful results because 1 kilotonne (kt) of NO_x emissions is too abstract a notion. In addition, many people also do not understand that NO_x emissions, which are not immediately observable, can harm their health. Questions therefore need to be carefully constructed in such a way that respondents can express their views on specific issues they can understand.

In terms of the Environmental Prices Handbook, this means that the Willingness-To-Pay for a clean environment is mainly determined by looking at valuations at the endpoint level, such as human health, ecosystem resilience or impact on crops, fish and biodiversity. The entire relationship from emission to that endpoint level is then modelled in dispersion and epidemiological models, among others, so that a relationship can be established between an emission of a particular pollutant and a physical change at the endpoint (see Chapter 4). This is not simply about ‘human health’ or ‘ecosystem resilience’ but rather about subdividing those generic endpoints into lots of types of endpoints because, for example, valuing premature death from a heart attack need not be the same as valuing premature death from cancer.

Although revealed preferences and stated preferences are the right ways to determine WTP/WTA from economic theory, in practice there will still be too little information available to fully value all endpoints, and their subdimensions, based on revealed and stated preferences. Therefore, the previous Environmental Prices Handbook 2018 identified two additional categories that are also sometimes used: valuation based on *restoration costs* and valuation based on *abatement costs*. Valuation based on restoration costs is sometimes applied in the valuation of nature, where it refers to the costs to be incurred to undo the degradation of nature. One example is injecting lime into soils affected by acidification. Valuation based on abatement costs involves the costs incurred by the polluter to prevent the emission from occurring. This can be done using ‘marginal abatement cost curves’: the costs to be incurred to reduce environmental pollution within socially acceptable norms. A valuation based on a levy, such as a levy on wastewater discharge, is also a valuation based on abatement costs.

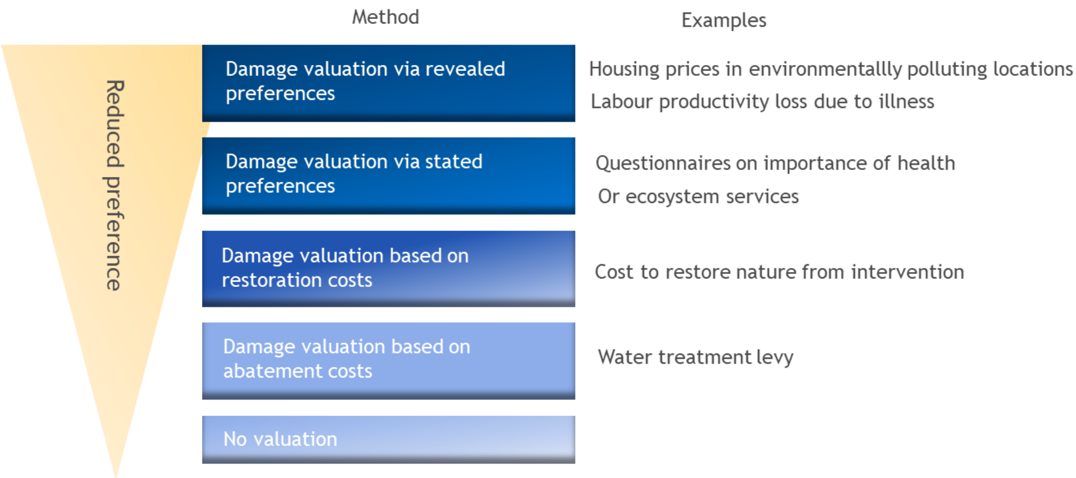
5.2.2 Sequential preference for valuation methods

This Environmental Prices Handbook therefore uses four methods to determine the Willingness-To-Pay for damage prevention (at the five endpoints):
1. Damage valuation via revealed preferences.
2. Damage valuation via stated preferences.
3. Damage valuation based on restoration costs.
4. Damage valuation based on abatement costs.

In general, there is a decreasing preference for use in economic valuation studies of the above methods: direct valuation of damage (WTP) via revealed or stated preferences is preferable to the other methods, and valuation based on abatement costs is the least recommended method (see also Figure 11). Nevertheless, in the methodology of the Environmental Prices Handbook, a valuation based on abatement costs is preferable to no valuation, because with ‘no valuation’ there is a risk that an assessment framework will not adequately include the environmental impact (see also the discussion in Paragraph 2.8.1).[23]

There may be exceptions to this general rule, though. In the case of climate change therefore, 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 (see also the discussion in Paragraph 6.2).

Figure 11 - Preferred order of valuation methods in the Environmental Prices Handbook



In some cases, none of the above valuation methods are truly satisfactory. A different method may then be explored: damage valuation based on loss of income models, such as Gross Domestic Product. In this Handbook this valuation base is explored with regard to the scarcity of raw materials endpoint, among others (see Paragraph 5.6).

Below, the four main methods are discussed, and it is explained which method has been adopted for which environmental theme.

5.2.3 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.

Revealed preference studies generally use econometric methods. In the Netherlands, this method is usually used for house price analysis (hedonic prices).[24] This is how noise nuisance has been valued in the past (Theebe, 2004).[25] By comparing house prices at locations exposed to noise nuisance with prices in quieter locations, an implicit value for the damage due to noise nuisance can be derived, provided this is properly corrected for other effects.

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23 The only reason for 'not valuing' is if it can be shown that the damage is not an 'external effect' but is already priced into existing market goods: in that case, the volume of the external effect equals zero and therefore nothing needs to be priced.

24 In addition, valuation based on travel times is also an option. This looks at how far people are willing to travel for recreation, such as to a nature conservation area.

25 Also see Section 6.11.

The main advantage of this method is that it assumes actual choice behaviour by people (in adjacent markets) facing budget constraints. One 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 overestimates or underestimates.[26] 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.

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. With environmental pollution in particular, it appears that people are poorly informed about the effects of environmental pollution on, for example, health. For instance, there is growing evidence that noise pollution causes not only nuisance but also health damage. This kind of damage is not always fully included when people put a value on nuisance.[27] For this reason, many researchers (see for example (Schoeters et al., 2021)) currently prefer valuation based on stated preferences.

In this update of the Environmental Prices Handbook, the valuation of the effects of air pollution on buildings has been determined partly based on revealed preferences.

5.2.4 Valuation based on stated preferences

Willingness-To-Pay can also be derived on the basis of stated preferences via surveys, interviews or other methods. One such method is the Contingent Valuation Method (CVM), in which survey respondents are asked directly about their Willingness-To-Pay for a particular good, carefully described in the survey scenario. This can also be done indirectly through Discrete Choice models where respondents have to make choices between various options. Based on consumers' response to questions about 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. Well-known issues include the absence of budgetary constraints, which leads to people reporting a higher value than they would realistically be prepared to pay. In addition, the results are highly sensitive to the research design, the questions set (see the next text box) and the participants' perception of how the results will be used. A typical 'bias' arises when people have the opportunity to provide socially desirable or strategic answers.

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[26] A negative correlation leads to underestimation and a positive correlation to overestimation.

[27] In part, this is also because the cost of health damage does not fall entirely on the homeowner.

Difference between WTP and WTA in CVM methodology

In the Contingent Valuation Method (CVM) respondents are, for example, asked to report their Willingness-to-Pay (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. 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 revealed by econometric analysis.

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 equivalent, but empirical and experimental studies have shown that people, on average, assign 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, 1979). This is due in part to people attaching more value to material assets and being risk averse.

For example, research by (Kahneman et al., 1990) shows that the price people ask (WTA) for an item they have just received is higher than the price they would want to pay for that item (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 a SCBA this would mean there is an implicit preference for the 'status quo'. Other reasons mentioned in the literature that may explain the difference between WTP and WTA have to do with the irreplaceability of nature combined with income effects (Hanemann, 1991) or strategic behaviour when answering questions (Bateman & Turner, 1993).

In this Environmental Prices Handbook, the valuation of the impact on health is based primarily on literature using stated preferences, both for air pollution and noise. This is explained in more detail in Paragraph 5.3.

5.2.5 Valuation based on (potential) restoration costs

A third approach to valuing the impact of environmental pollution is by using the (potential) restoration costs. Valuation with restoration costs examines how much it costs to undo the damage caused by environmental pollution. In literature (NEEDS, 2008a) it is generally recognised 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 3 we saw that the welfare-optimal pollutant level is higher than 0 if the demand and supply functions of environmental quality are price-sensitive (elastic). 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'. It can also lead to underestimation because property rights are not well defined or there is a 'split incentive dilemma'. For example, landlords of flats may choose not to clean flats in polluted locations as often because the welfare loss will fall on the tenants rather than on themselves.

The objection of overestimation can be alleviated by using actual expenses incurred by, for example, homeowners, as a point of departure, rather than the hypothetical restoration costs. In that case, the restoration costs are used to derive a revealed preference value. However, this will likely lead to an underestimate, because not all homeowners will opt to repair the damage due to the split incentive dilemma.

For these reasons the restoration cost method is less accurate than the revealed preference and stated preference methods. Nevertheless, valuation using restoration costs has been applied in this Environmental Prices Handbook to assess the impact of air pollution on buildings and materials. This is because insufficient research is available for these themes to allow for valuation based on stated or revealed preferences.

5.2.6 Valuation based on abatement costs

The final valuation method is based on abatement costs. The abatement cost method takes environmental policy as its starting point and is based on the marginal costs incurred to meet environmental policy targets. Many environmental policies have a policy target, such as 55% reduction in carbon emissions compared to 1990. The abatement cost method is based, more specifically, on the costliest abatement measure. This corresponds to the cost of the least cost-effective measure needed to achieve that policy target in the most cost-effective way. These costs are referred to as the marginal abatement costs.
If the prevention cost methodology is used, it is important that it is based on 'efficient prices'. Efficient prices reflect the minimum prices to achieve the policy objective. If we assume that the government is fully informed and economically rational, it will shape policy objectives to achieve the optimal level of pollution. To achieve this optimal level of pollution, welfare economics introduce the Pigouvian tax on the polluting activity that internalises externalities at minimal cost. A Pigouvian tax is an efficient use of policy to optimise welfare.

5.2.7 Summary table of methods used

In the Environmental Prices Handbook, the endpoints were valued based on a review of the literature. The final chosen values are based on a range of methods. Table 14 outlines the type of valuation research used in determining the endpoints.

Table 14 - Valuation methods based on literature review that is used at the endpoints

Endpoint	Methods
Human health Mortality	Stated preferences
Human health Morbidity	Stated preferences, revealed preferences
Ecosystem services	Stated preferences
Buildings and materials	Restoration costs
Raw materials	Abatement costs, damage costs
Nuisance	Stated preferences, revealed preferences

The valuations for the various endpoints affect the determination of environmental prices at pollutant and midpoint levels for all themes (see Paragraph 6.1) except climate change. For climate change, a valuation based on abatement costs was taken as the starting point (see Paragraph 6.4).

5.2.8 Limitations in valuing environmental quality

Ascribing a value to environmental quality has several serious limitations. Despite the thousands of publications over the last two decades, there are still major uncertainties about the reliability of the valuation methods employed. This is primarily because the value for environmental quality found in research is difficult to verify with people's actual preferences (see (Carson, 2000); (Bateman et al., 2002)). A strong bias in the research methodology plays an important role in this.

The principal limitations are as follows:
1. **Completeness**: there appear to be no methods that can represent the full spectrum of human appreciation of environmental quality. In particular, optional and non-use values are poorly covered in valuation studies.
2. **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 (Delucchi et al., 2002).
3. **Research bias**: CVM methods in particular expose a wide variety of outcomes depending on the research design. (Carson et al., 1997) show that the order of questioning is important for valuation, a fact that has also been empirically demonstrated (Payne et al., 2000). Economically, this is also explicable, but it is often not taken into account in the case of valuation in cost-benefit analyses. This criticism is recognised in science, and recently more and more valuation surveys have been conducted using Choice-Experiments, where the order of questions can vary and can be corrected for (see also the discussion above).

We make no claims that the values presented here in this Environmental Prices Handbook are complete and certain. Rather, we emphasise in this Handbook the high degree of uncertainty associated with valuations of environmental goods. One way we do this is by working with ranges on valuations and environmental impact.

The only alternative to the lack of scientific certainty about the level of valuations is *not* to value environmental goods. Although such a course may initially seem to solve the problem of scientific uncertainty, it stands in stark contradiction to the fact that each and everyday consumers, industries and governments make decisions involving *implicit* weighting of financial data and effects that cannot be expressed in financial terms. Therefore, in the decision-making of governments, consumers and producers, not valuing is often equivalent to valuing at a price of zero. Since these are externalities, effects that affect the welfare of others, we can assert with certainty that a zero price is incorrect. Therefore, in many situations, valuing with uncertain environmental prices is a better alternative than valuing with certain erroneous prices.

5.3 Human health

Human health effects are classified into mortality and morbidity. Mortality refers to premature death. Morbidity refers to disease. Mortality is typically distinguished into acute mortality and chronic mortality. In addition, environmental pollution leads to health problems (morbidity).

Health effects due to environmental pollution can thus be divided into three types of effects:
1. **Chronic mortality:** expressed as a reduction in life expectancy. Epidemiologically, it has been shown that people in polluted areas live shorter lives than those in cleaner areas, a relationship that persists even at lower concentrations of environmental pollutants in the air (OECD, 2012). Health risks in this category primarily include respiratory and cardiovascular diseases.
2. **Acute mortality**: expressed as an increase in mortality risk. Certain types of environmental pollution, such as smog, are also associated with acute heart failure.
   A positive link has also been shown between air pollution and sudden infant death syndrome.

3. **Morbidity:** expressed as an increase in the disease burden. Environmental pollution leads to a higher 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 relationship

The following midpoints have an impact on the human health endpoint:
1. Particulate matter formation.
2. Photochemical oxidant formation
3. (Ionising) radiation.
4. Human toxicity.
5. Nuisance (noise nuisance).
6. Ozone depletion.
7. Climate change*.

With the exception of climate change, all impacts are included in this study. In this Handbook, the impact of climate change has been determined on the basis of abatement costs. This means the health impact of climate change is not treated separately, but integrally included (as a proxy) in the valuation of climate change policy (see Paragraph 6.3).

A good number of health effects are explicitly valued in the Environmental Prices Handbook (see Table 15). This mainly concerns health effects caused by air pollution. In addition, many health effects, such as IQ losses, osteoporosis or cataract, are implicitly valued through the models we used to arrive at a valuation of environmental damage.

5.3.2 Indicators of health effects

Effects on health can be expressed in actual quantities such as: ‘number of people who died’ or ‘number of hospital admissions’ or ‘number of days not worked’.
The effects of a change in the concentration of pollution on these quantities at *pollutant level* (see Chapter 4) is shown (see also Annex A). The valuation of these quantities for mortality is discussed in Paragraph 5.3.3 and for morbidity in Paragraph 5.3.4.

In addition, overarching physical indicators are often used that express the number of years ‘lost’ in terms of life (mortality) or a certain quality of life (morbidity). Common indicators for the impact on human health are: YOLL, DALY and QALY.[28] Table 15 provides an overview with a brief explanation for each indicator.

[28] For abbreviations, see Table 15. YOLL is also sometimes expressed as LYL (Life Years Lost).

Table 15 - Different physical indicators for an assessment of the impact of environmental pollutants on health

Indicator	Meaning	Explanation	Used for the environmental impact in:
YOLL	Years of Life Lost	The number of life years lost due to premature mortality	EEA, 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	In a number of individual studies such as (Hubbell, 2006)

In these indicators, mortality is expressed as 'years of life lost'. Morbidity (illness) is normally also expressed in these indicators using a conversion table in which illness and disability are expressed as partial mortality, such as (Hubbell, 2006) for the QALY framework, for example. Generally speaking, morbidity is more usually expressed in QALYs rather than DALYs or YOLL.

The three physical indicators are also valued in euros. As the unit of the above indicators is set in 'years', a valuation based on (lost) life years, such as the VOLY (Value of Life Years), is usually used. But in principle, the YOLL, DALY and QALYs can also be converted and used for a valuation based on lives lost, such as the VSL (Value of Statistical life).

5.3.3 Valuation of health effects: mortality

An important discussion point in monetising the health effects of environmental pollution concerns the valuation of premature death (mortality). There are roughly two schools of thought here: studies that value the number of lives lost (using the VSL) or studies that value the number of life years lost (using the VOLY). Both concepts are explained below.

The Value of a Statistical Life (VSL) is a widely used measure for the valuation of measures in the field of traffic and transport but is also used for healthcare or the environment. The Value of a Statistical Life gives a value for a human life lost. Such valuation can be established through questionnaires or revealed preferences, for example by looking at wage premiums for high-risk occupations. (Schoeters et al., 2021) argue that on methodological grounds, a valuation based on questionnaires is preferable to a valuation based on stated preferences.

The VSL has been estimated by the OECD, in a meta-analysis, at $ 3.6 million per life lost for EU27 countries at 2005 price levels (median value, (OECD, 2012)). A recent pan-European study (Schoeters et al., 2021) estimates this value (by means of questionnaires on hypothetical situations for variants that varied in travel costs, time and probability of accidents among 8,003 respondents) to be even higher: at €6.2 million in 2021 price level.[29] An alternative measure used is the VOLY. The VOLY gives a value to a lost year of life. The VOLY can be determined either via revealed or stated preferences, but in practice the VOLY is almost always determined in surveys that ask respondents implicitly via choice experiments or explicitly to give a valuation for 'a 1-year reduction in life expectancy'. The VOLY is therefore strongly related to life expectancy.

[29] The survey included 2,005 Belgian respondents, 2,000 French, 2,000 from Germany and 1,998 from the Netherlands.

Extensive research has been conducted on the valuation of the VOLY. For example, in the NEEDS project (NEEDS, 2008b), the VOLY was valued via CPM (Certified Preferences Method) by asking people in face-to-face surveys about their Willingness-To-Pay for three or six months longer life due to improved air quality.[30] 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 in the NEEDS project than in previous studies in which people were asked (in Discrete Choice Experiments) about their risk of dying prematurely. 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 (price level 2005).[31] This figure is for chronic mortality, which is the shortening of life expectancy. The NEEDS researchers considered it plausible to use a higher value of €60,000 (2005 price level) for health risks of acute death, based on literature analysis.

The previous Handbook of External Costs of Transport (CE Delft et al., 2019) presents a comprehensive review of studies that, when converted to 2016 price levels, give results of between €53,000 and €250,000, with the upper limit being determined by studies that did not assume an individual valuation for life, but included a valuation for compassion. These studies are grouped around three focal points:
1. Studies with results around €55,000.
2. Studies with results around €70,000-80,000.
3. Studies with results over €110,000.

A similar distribution was also observed in the previous Environmental Prices Handbook, which then resulted in a lower value of €50,000, a middle value of €70,000 and an upper value of €110,000.[32]

There is a relationship between the VSL and the VOLY. For respondents taking part in a WTP survey, the VSL can be understood as a number of discounted VOLYs at the time of the survey. Based on this assumption, and a discount rate of 2.25%, the VSL should be about a factor 15-30 higher than the VOLY if questionnaire respondents are not older than 65 on average. However, this result is also highly dependent on people's time preferences. For example, if people have a 10%-time preference, then the average VSL should be only a factor of 10 higher than the VOLY and vary much less with age.

With these experience figures, there is a significant discrepancy between relatively high VSL valuations of around €5-6 million with the VOLY valuations. Assuming factors of 10 to 30, a VSL of €6 million should relate to VOLY valuations of between €200,000 and €600,000 per year. We note that the lower end of this range is in the VOLY estimates that use valuations of survivors for the deceased (based on the study of (Chanel & Luchini, 2014)). We must therefore conclude that valuations based on VSL leads to considerably higher damage estimates than valuations based on the VOLY.

The question is then which valuation basis is preferable on argumentative or methodological grounds. First, we note that it would make no sense from a policy perspective to value deaths differently in different domains (Fourcade, 2009). The VSL, especially in the US,

30 Besides direct queries, researchers also worked with a system of payment cards, including payments for alternatives.

31 The NEEDS project proposed a single uniform value for VOLY for all EU member states. Optionally, the VOLY could be varied between EU15 plus Switzerland for € 41,000 and for the new member states € 33,000 (NEEDS, 2006).

32 The lower and upper values were then also based on the Social Domain Working Guide (SEO, 2016) for the QALY, in which a relationship of 1 QALY = 1 VOLY was assumed for the lower value and a value of 1 QALY = 1,086 VOLY was assumed for the upper value.

is a widely accepted measure to compare different risks and it would therefore be obvious to apply it to premature deaths due to environmental pollution as well. In their meta-analysis, (OECD, 2012) also did not see a statistically significant difference between VSL studies that focused on ‘environment’ versus studies focused on ‘health’.

Nevertheless, the methodological literature often suggests that valuation of premature death due to environmental pollution with a VOLY would be more ‘accurate’ than valuation with a VSL for the following reasons:

1. Environmental pollutants cannot usually be identified as the primary cause of an individual’s death, only as a contributing factor to premature death. In epidemiological studies, the effects of air pollution are also measured as a shortening of lifespan (WHO, 2013). It is therefore an obvious choice to value these effects with a VOLY rather than a VSL.
2. A VSL does not account for the fact that the loss of life expectancy due to associated with air pollution is much shorter (a few years) than for typical (traffic) 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 just as likely to happen at an early age.
3. Research shows that respondents understand the concept of life expectancy better than the concept of ‘risk of dying’ (Desaigues et al., 2007). This fact is further supported by econometric research showing that surveys assuming a reduction in life expectancy have a better ‘fit’ than surveys that assume a reduction in risk (Grisolía et al., 2018). Therefore, surveys based on gains in life expectancy may be more in line with people’s Willingness-To-Pay than questionnaires on risk reduction.

5.3.4 Treatment in the Environmental Prices Handbook 2018 and updates

The previous Environmental Prices Handbook 2018 assumed a valuation of a VOLY between €50,000, €70,000 and €110,000 for valuing the probability of premature death, based on the literature review and the QALY valuation from the Social Domain Working Guide (SEO, 2016a). The upper limit of valuation for a VOLY was higher than that for a QALY and based on an analysis that a DALY was higher than the inverse of a QALY due to age weighting, among other things. No age weighting was applied to the central and lower values.

Annex B again addresses the question of whether a conversion factor can be found between a VOLY and a QALY. Based on more recent data and a new analysis of the available literature, we have concluded that, unlike in the previous Handbook, there is no reason to assume that a QALY should be valued differently from a VOLY or a DALY. With this, the valuation for VOLY at the upper limit this time exactly matches that of the Social Domain Working Guide.[33]

Since, with the exception of the VSL, there have been no major new studies that value the VOLY, we have decided to continue to assume the range of valuation of a VOLY of €50,000 to 100,000 with a central value of €70,000 in the Environmental Prices Handbook 2024 EU version. However, we did explore ways to adapt it to the situation in 2021, looking at the underlying factors of changes in the value of VOLY over time. This is explained in more detail in Paragraph 5.3.5.

[33] The annexes to the Social Domain Working Guide contain a discussion on whether a QALY for preventive health care is lower than a QALY for curative healthcare. Although there are studies that claim this, (SEO, 2016a) argue that there is no theoretical justification per se for using a lower QALY for preventive healthcare than for curative care. They therefore recommend not valuing it separately. This then also implies that there would be no reasons to value environmental pollution differently from healthcare interventions.

5.3.5 Update: VOLY valuations over time

There are three reasons why the VOLY should be updated over time:
1. Price level adjustments.
2. Adjustments of health valuations due to higher income.
3. Adjustments to health valuations due to higher educational attainment.

These three issues are discussed below:

1. Price level adjustments
It is common practice to adjust valuations in line with price levels. Consumer prices, measured in Eurostat's Harmonised Index of Consumer Prices, rose by almost 9% in the EU27 between 2015 and 2021. This would mean that valuation for a VOLY should also have risen proportionately. The effects on morbidity should also increase by the same percentage.

2. Income related adjustments
The previous Environmental Prices Handbook 2018 states that, in principle, it is better to periodically reassess the valuation of the VOLY with stated/revealed preference research than to use discount rates to reflect a positive income elasticity. Then the question of whether to convert past values to the present with income elasticity need not be answered. But since there is no new research on the VOLY, the question remains whether we should increase the VOLY from the previous Handbook by a factor for income elasticity.

The discussion on the valuation of the QALY seems very much inspired by the 'healthcare market', where the government has to make decisions on which care is and is not reimbursed. In that case, it makes sense that both the supply and demand sides in the healthcare market should be reviewed, so that increased demand is offset by increased supply and the relative price of a healthy life decreases. In the case of premature death due to environmental pollution, however, demand is key because the increased supply is already accounted for by a decrease in mortality due to improved treatments. Using the relative risk approach (see Paragraph 4.4.1), an improvement in the treatment of heart failure and cancer cases is already accounted for by a reduction in mortality rates. Therefore, international analyses (see, for example, (CE Delft, 2020)) also show that environmental pollutants per µg/m^3 are much more lethal in countries with poor healthcare than in countries with good healthcare. The increased demand for healthcare therefore does need to be included in analyses based on epidemiological studies, such as environmental pollutants. For this reason, we use a positive income elasticity on the value for premature death in the update of the Environmental Prices Handbook. This is without prejudice to our view that a revision of the VOLY valuation based on new research is preferable.

Based on a literature analysis, we conclude that there is a considerable range between the income elasticities found in research. Meta-analyses of (Masterman & Viscusi, 2018) and (OECD, 2012) show income elasticities for developed countries of 0.55 and 0.8, respectively. The range of outcomes runs roughly from 0.3 to above 1.0. In the update of the Environmental Prices Handbook, we propose the following income elasticities for the VOLY.

Table 16 – Income elasticities and VOLY used in the Environmental Prices Handbook 2024

	Lower	Central	Upper
Income elasticities	0.3	0.65	1
VOLY 2015	€ 50,000	€ 70,000	€ 110,000
VOLY 2021	€ 57,500	€ 85,000	€ 128,000

This involves adjusting the VOLY relative to the 2015 values for inflation and the income elasticity belonging to the value path.

3. Adjustments to the level of education
Finally, research shows that, apart from income, the level of education also affects the valuation of health. As people become more educated, the appreciation of a VOLY becomes higher. For example, see (OECD, 2012) in general and (Istamto et al., 2014) for the appreciation of the occurrence of environmental pollution in five European countries. Because the level of education in the EU in 2021 is higher than in the period on which the NEEDS (2005) survey is based, one would expect that the valuation for a VOLY could be higher if one were to conduct a newer survey. Since it is not clear how much higher the valuation for the VOLY could be, we have not adjusted it in this study.

5.3.6 Valuation of health effects: morbidity

There are two ways to value morbidity effects in the literature:
1. Through a valuation using the QALY framework.
2. Through a direct valuation of the various disease burdens caused by environmental pollution.

For the valuation of morbidity effects, the previous Environmental Prices Handbook relied on the NEEDS 2008 project, which estimated healthcare costs for various endpoints, partly using a QALY framework. These healthcare costs were then adjusted to 2015 price levels and values specific to the EU were calculated for several endpoints, such as loss of working days due to illness.

The valuations from the NEEDS project were declared applicable to the central value. Lower and upper values were artificially created by varying them with the variation in the QALY, so that the lower value always ended up at 5/7 of the central value and the upper value was set equal to 10/7 of the central value.

As part of the update of the Environmental Prices Handbook, we reviewed the literature regarding the valuation of morbidity effects. Based on this, we established a new valuation, where we relied on the range found in the literature concerning the cost of morbidity effects for both the lower, central and upper values. The following table shows the values used in this update of the Handbook. Annex A.3.5 provides more information on the reasoning behind these values.

Table 17 - Overview valuation morbidity effects, €2021 per day or case

Endpoint	Lower value	Central value	Upper value	Source
VOLY/DALY/QALY	57,500	85,000	128,000	(CE Delft, 2017a)
Post neonatal infant mortality	408,2431	6,208,720	8,364,679	(OECD, 2016)
Prevalence of bronchitis in children	285	407	407	(OECD, 2016)
Asthma symptoms in asthmatic children	40	57	81	(EEA, 2021a); (CE Delft, 2017a)
COPD in adults	50,717	72,452	350,498	(EEA, 2021a); (OECD, 2016)
Hospital admissions, CVDs (excl. stroke)	4,731	6,759	6,759	(EEA, 2021a)
Hospital admissions, respiratory diseases	3,785	5,407	5,407	(EEA, 2021a)
Lost working days	176	211	266	(CE Delft, 2017a); Own calculation based on Eurostat data; National Health Care Institute (2016)
RADs (days of restricted activity)	104	148	190	(EEA, 2021a); (CE Delft, 2017a)
MRADs (days of small, restricted activity)	57	81	81	(EEA, 2021a); (OECD, 2016)

5.4 Valuation of ecosystem services and nature

Ecosystems, i.e. collections of organisms in a given environment, combine the abiotic environment with biological communities (plants, animals, fungi, and micro-organisms) to form self-organising, regenerative functional units. By this we mean combinations of life forms that control fluxes, such as those of energy (e.g., photosynthesis), nutrients (e.g., nitrogen fixation) and organic matter (e.g., decomposition of organic waste).

Ecosystems contribute to human well-being by providing ecosystem services. These services are classified by the Common International Classification of Ecosystem Services (CICES). CICES (EEA, 2011) distinguishes three classes of services:
1. **Supply services (**e.g., food from agricultural crops, biomass as fuel, fisheries, forestry, freshwater).
2. **Cultural services** (e.g., recreation, aesthetic value of environment, spiritual values).
3. **Control and maintenance services** (e.g., climate control, soil formation, biological pest control, water treatment).

Each of these services contributes directly or indirectly to human well-being (Dasgupta, 2021). Economically, ecosystems are considered a form of ‘natural capital’ that provides a flow of services similar to man-made capital goods (e.g., roads and buildings). As with man-made capital, natural capital diminishes in value when it is misused or overused. However, natural capital differs from human-made capital (Dasgupta, 2021) in three ways:
— a decline in value is irreversible in many cases;
— it is not possible to replicate a depleted or degraded ecosystem;
— there are tipping points that allow ecosystems to collapse abruptly, without prior warning.

Biodiversity can be defined as the diversity, number and quality of species, populations and ecosystems. Biodiversity plays a crucial role in providing ecosystem services because it supports fundamental processes, such as soil formation and the hydrological cycle, that are crucial to providing ecosystem services. Biodiversity is thus an essential factor influencing the productivity of natural capital and is vital to ecosystem health. In addition, biodiversity has an intrinsic value: people value the preservation of the world's rich diversity of natural species, both for themselves and to preserve it for future generations.

Recent studies have shown that the unprecedented loss of biodiversity may threaten the provision of ecosystem services and the future well-being of humans (Dasgupta, 2021).
The vast majority of ecosystem and biodiversity values are declining rapidly. The average abundance of native species in most large terrestrial biomes has declined by at least 20%, affecting many essential ecosystem processes (Hill et al., 2018); (IPBES, 2019). Species extinction rates are estimated to be 100 to 1,000 times higher than their background rates over the past tens of millions of years (0.1-1 per million species per year), and these rates continue to rise. According to WWF, since 1970 there has been a global decline in biodiversity of 68% of vertebrate populations (WWF, 2020). Currently, more than 40% of the total land area is under agricultural or urban influence, intended to meet human needs. Certain biomes have been altered to the point that they eventually become anthropogenic biomes, also called 'anthromes' (IPBES, 2019).
One of the main reasons for biodiversity reduction is fragmentation and alteration of natural habitats.[34] Species extinction will irreparably damage the biosphere, with unknown numbers of tipping points that, if exceeded, will result in persistent and irreversible changes in ecosystem structures, functions and services. This, in turn, will result in future economic prospects being much bleaker than we imagine today (OECD, 2019); (Dasgupta, 2021).
The costs of biodiversity loss are increasingly recognised and quantified by economists (see also Paragraph 5.4.5).[35]

5.4.1 Characteristics of biodiversity and its relationship to ecosystem services

A fundamental problem with the term 'biodiversity' is that there is no single, clear, agreed-upon definition (Koricheva & Siipi, 2004). Biodiversity is a very abstract, complex concept, which has led to a kind of terminological chaos. According to (Stirling, 2010), 'diversity' in general is a combination of three properties of systems: diversity, balance and inequality. For biodiversity, these three components would correspond to wealth, abundance and phylogenetic distance, respectively. Similarly, (Dasgupta, 2021) identifies the three relevant characteristics of biodiversity: richness, evenness and heterogeneity.

Species diversity is therefore an important component of biodiversity. It can be defined as the diversity of life in all its forms, which can be interpreted as the total number of species of organisms inhabiting the earth (Raven et al., 2020). About 8 to 20 million and possibly more species of eukaryotic organisms are believed to be found on the planet: of these, only about 2 million have been recognised and named (Raven et al., 2020).[36]

[34] Fragmentation reduces biodiversity by up to 75% and exposes species to harsh environmental conditions (Haddad et al., 2015). A major cause of fragmentation in terrestrial ecosystems are fenced areas that prevent wildlife migration and fragmentation of forests into smaller plots. In freshwater ecosystems, dams are considered the main contributors to biodiversity loss.

[35] A decrease in the number of species does not necessarily mean a deterioration of the ecosystem; sometimes a particular niche in the ecosystem is simply filled by another species.

[36] In addition, there may be a much larger number of prokaryotes, consisting of archaea and bacteria, which have not yet been described (Locey & Lennon, 2016);(Larsen et al., 2017).

In this Handbook, we put the valuation focus on biodiversity rather than ecosystem services. However, biodiversity and ecosystem services are not always congruent. Biodiversity is seen by many people as a precursor to ecosystem services, but the relationship is not that simple. (Science for Environment Policy, 2015) concludes from the available literature that, even after 20 years of research, the exact relationship between biodiversity and ecosystem services is still not entirely clear. The relationship between biodiversity and the various ecosystem functions is nonlinear and can vary from service to service. In general, control and maintenance services benefit from greater biodiversity. However, supply functions such as agriculture and forestry on average have the highest yields at relatively low biodiversity. In general, cultural functions benefit from increased biodiversity, but very high levels of biodiversity can have negative impacts on recreational functions.

In conclusion, there is justification for taking biodiversity as a proxy for the intrinsic and extrinsic value of ecosystems (i.e. nature), given the crucial role of biodiversity in the quality of ecosystem services.

5.4.2 Indicators of biodiversity

There are numerous indicators for measuring biodiversity, which was also shown at the CBD meeting on Aichi targets, where more than 100 different indicators were presented (Pereira et al., 2012). To select appropriate indicators, the Essential Biodiversity Variables framework helps prioritise indicators that can reflect the essential dimension of biodiversity change. Six main areas were identified that are essential to measure biodiversity: ecosystem structure, ecosystem functions, community composition, species populations, species traits and genetic composition (Dasgupta, 2021). (Mace et al., 2018) proposed three indicators already developed that jointly address the essential dimensions of biodiversity:
1. The IUCN Red List Index, which measures vulnerability and extinction risks.
2. The Living Planet index, which measures species abundance.
3. Biodiversity Intactness Index (BII) measurement composition.

The complexity of biodiversity is currently not fully reflected in LCA analyses. Many LCA models include biodiversity loss as an endpoint indicator, but scientific consensus on indicator choices is still lacking. Optimally, a single indicator would reflect all six main areas together, including characteristics such as rarity, endemism, irreplaceability and vulnerability on multiple taxonomic groups with high spatial differentiation. However, an indicator reflecting all that complexity of biodiversity can hardly be determined (Lindner et al., 2019).

Nevertheless, most LCA models have chosen one comprehensive indicator for biodiversity. The development of a single measure encompassing the major components of ecosystems and biodiversity would be very useful to increase the relevance and accuracy of results in LCA, but it is a very challenging exercise due to the complexity and dynamics of ecosystems (De Souza et al., 2015); (Van Zelm & Huijbregts, 2013); (Lindner et al., 2019). An aggregate indicator covering the various dimensions of biodiversity is desirable, although aggregation of an indicator across impact categories carries the risk of double counting (Woods & Damiani, 2018). No such indicator currently exists.[37]

[37] Compared to the indicators currently used, many improvements would be needed, such as the development of species-based measures and feature-based measures, including species. Vulnerability levels based on occurrence and dispersion on both local and global scales, improved modelling of ecosystem complexity for species loss, indicators including aspects such as biogeography, and applying more comprehensive aspects of ecosystem damage.

The impact of land use on biodiversity is a very complex matter and a single measure cannot accurately reflect the relationship between land use and biodiversity loss. Species richness (SR) is the most commonly used indicator to quantify the impact of land use. Species richness is an indicator that counts the number of species in a given area. The SR indicator is often used as a relative measure and referred to as relative species richness (S_{rel}): it measures species richness relative to the species richness of a reference area (S_{ref}). An area reflecting the natural state of the area is chosen as the reference area. All land occupation activities are then related to the species number in this 'nature conservation area'. This means that in a given region j the relative species richness is defined as:

$$S_{rel,j} = S_j - S_{ref}$$

Several indicators used in LCA and valuation studies are derived from the SR indicator. Annex C describes these in more detail. The three indicators that are most important for analysis in the Environmental Prices Handbook are discussed below.

Potentially Disappearing Fraction (PDF) is defined as the degree of species loss within a given area during a given time, due to human intervention. This includes as many species on land as at sea. The PDF for an area j is defined as a comparison of the species loss in that area with the baseline condition (Rabl et al., 2014):

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

PDFs are often compiled for a specific time and space. A PDF * m^2 * yr of 1 implies that in 1 m^2 all species became extinct during 1 year. It is comparable to 10% of species going extinct in an area of 10 m^2 during 1 year, or 10% of species going extinct in 10 years. The determination of the PDF depends on ecological models but is often based on the diversity of vascular plant species associated with a particular area of land cover, such as deciduous forest, grassland, etc. (Köllner, 2001). The PDF is sometimes considered comparable to the extinction rate of a particular biome.

Potentially Affected Fraction (PAF) is an indicator more commonly used in ecotoxicological models. It describes the fraction of species affected by a pollutant. The No Observed Effect Concentration (NOEC) is regularly used as a threshold concentration to determine PAF (Klepper & van de Meet, 1997). PAF uses concentration-effect relationships not related to species loss but defines the damage that occurs to species. Ecotoxicity effects are estimated based on laboratory-derived concentration-response curves related to the fraction of the affected test group. The effect can refer to many different health states, such as mortality or morbidity. In most cases, the EC50 factor (affecting the 50% of the population above the background concentration), or the LC50 factor (killing 50% of the population) is used. Based on these factors, models can be constructed that describe the response of the entire ecosystem to a specific stressor.

The PAF can be converted to a PDF at midpoint level. However, because of the underlying differences, conversion is not a simple step, as PDF indicates the loss of species, while PAF refers to the fraction of species affected to some extent by a stressor. The conversion factors also depend on the type of ecosystem, the type of stressor and the Impact Pathway approach used (De Souza et al., 2013). Studies suggest conversion factors from PDF to PAF_{EC50} between 1 and 10. (Goedkoop & Spriensma, 2001) recommended a factor of 10 for conversion, while Impact 2002+ suggested a factor of 2 (Jolliet et al., 2003). Among the newer LCA models, ReCiPe models (2008, 2016) assume equality between PAF_{EC50} and PDF

(Goedkoop et al., 2009), while LC-Impacts uses a factor of 2 for conversion (Verones et al., 2020). In our study, we follow ReCiPe and equate PAF and PDF. This means that the impact of ecotoxicity on biodiversity is valued in the same manner as the impact of land use on biodiversity.

Ecological and Biodiversity Damage Potentials (EDP and BDP). The Ecological and Biodiversity Damage Potential (EDP) was developed by (Köllner & Scholz, 2007) and is similar in format to PDF, but uses hectares instead of m^2. Each land-use type was assigned a specific EDP value, based on the Corine Plus land-use classification system. Based on the EDP indicator, (De Baan et al., 2013b) developed a new characterisation factor (CF) called Biodiversity Damage Potential (BDP), in line with the UNEP-SETAC land-use assessment framework by (Köllner et al., 2013). This CF quantifies the effects of land occupation on terrestrial ecosystems, differentiating the effects for nine major biomes. An important addition, compared to the EDP, was that the authors used spatially differentiated land-use type effects for different taxonomic groups. To obtain sufficient data, the GLOBIO3 database (Alkemade et al., 2009) was used, supplemented by national monitoring data from Switzerland (BDM, 2004). The derived BDP indicator measures the relative changes in species composition (relative species richness - S_{rel}) compared to a semi-natural reference habitat. The characterisation factor $CF_{occ}LU_{ij}$ compares the difference between a reference situation and a land use type for a given region. The late-successional habitat stage, usually applied in restoration ecology, was used as a reference.

The median S_{rel} is subtracted from 1 to obtain the CF:

$$CF_{Occ,LU,i,j} = 1 - S_{rel,LU,i,j}$$

The value is usually between 0 (indicates no change) and 1 (indicates complete change), but with improvement in the country's species richness, a negative value can also be achieved.[38]

The indicators described here are used in our valuation framework. In addition, other indicators are used such as the nature points methodology (PBL, 2014). As no dose-effect relationships quantifying the effects of emissions on nature points are known, this route is not further elaborated here. An overview of several other indicators is given in Annex C.

5.4.3 Value of biodiversity in the Environmental Prices Handbook 2018 and updates

The previous Environmental Prices Handbook 2018 uses (Kuik et al., 2008) to derive a value for biodiversity loss. Through a meta-analysis, (Kuik et al., 2008) derived a value for the EDP indicator, which is used as a proxy for the valuation of PDF.[39] For the upper value, the *average* value of €0.47/PDF/m²/yr for the EU28, as reported in (Kuik et al., 2008), was used as the basis for the calculations, while the lower value was based on the *median* value (€0.06/PDF/m²/yr at 2004 prices), as reported in (Kuik et al., 2008). Finally, a central value was derived from estimates of restoration costs by (NEEDS, 2006). This central value was found to be in the middle of the median and average values from (Kuik et al., 2008).

[38] Pastures in the 'Desert and xeric scrub' biome, for example, showed a slightly positive median land-use impact.

[39] As mentioned in the previous section, (Kuik et al., 2008) suggest that EDP and PDF should be considered equivalent in practical applications.

The advantage of using the results of the (Kuik et al., 2008) study, is that it provides not only an average value for biodiversity loss, but also a meta-analysis that provides key factors for regional variation in valuation. This model was used in the Environmental Prices Handbook 2018 to provide a value for the EU27. The values from (Kuik et al., 2008) were thereby converted to the price level in the year 2015 by adjusting for inflation, plus an autonomous price increase of 1% per year for nature that cannot be replaced, as described in the Discount Rate Working Group, and set out in the guidelines for the SCBA in the field of environment (CE Delft, 2017b) and nature (CE Delft & Arcadis, 2018). The rationale for an autonomous price increase of 1% per year is based on the fact that irreplaceable nature is limited in supply and becomes scarcer over time, thus increasing its value. Because income elasticities in nature valuation are not always conclusive (see (Kuik et al., 2008), among others), including an annual price-autonomous price increase in nature valuation seems logical. This practice was later updated and formalised in (PBL, 2018).

To prepare the current update, we thoroughly reviewed the available literature on valuation and worked out whether we could replace the existing framework with a new one that would better reflect more recent estimates on the overall value of biodiversity. However, we concluded that although many recent studies have appeared that value biodiversity and have used a wider range of studies in their meta-analysis (see Paragraph 5.4.5), the results of these studies cannot be easily linked to emissions because there is no new information available for the emission pathway approach we followed in the previous Handbook. Moreover, even recent valuation literature still often uses species number indicators, such as PDF. In addition, we found that the new ReCiPe (Huijbregts et al., 2016) still uses the 'species.year' indicator from the previous ReCiPe (Goedkoop et al., 2013).

Matching the figures to other LCA frameworks has proved to be impossible at the moment, as many of these frameworks are still under development. Therefore, we have concluded that it would be best if we *scaled* the results of (Kuik et al., 2008) to newer literature on biodiversity loss cost estimates. To this end, we have made three adjustments:
1. Update of characterisation factors (see Paragraph 5.4.4).
2. Review of the literature on valuation (see Paragraph 5.4.5) and propose adjustment factors to adjust the EU28 value of (Kuik et al., 2008).
3. Critical examination of the use of regression analysis in (Kuik et al., 2008) to derive specific values for the EU (see Paragraph 5.4.6).

In Paragraph 5.4.7, we present new monetary estimates for the value of biodiversity that will be included in this Handbook, and outline what other endpoints we have valued.

5.4.4 Update: central characterisation factors for land use and valuation

In this study, we use the characterisation factors (CF) established in ReCiPe 2016 and have updated the 2008 version to the 2016 version. This implies that this method still bases the characterisation factors on relative species richness data (S_{rel}), which are compared to a reference natural habitat. To calculate the total impact of land occupation, the CF is multiplied by area (A) and time (t).

ReCiPe 2016 contains a number of differences from the 2009 version used in the previous Handbook. It offers CFs on a global rather than a European scale. It also provides more specific data on certain species groups to increase model accuracy. Only local land use effects are included in the model, as regional land-use methods were found to be too arbitrary. Moreover, land transformation is not treated separately by the ReCiPe model, but estimated together with land use and expressed in PDF/annual crop equivalent. $S_{rel(ac)}$ is relative species loss for annual crops. This value was identified as 0.6 (De Baan et al., 2013b).

When land occupation stops, an 'easing period' follows, describing the process after the land occupation ends and returns to the natural state. This period still has some negative impact on species richness until it reaches a new equilibrium as a natural or semi-natural habitat. CFs are therefore calculated separately for land occupation and relaxation at the midpoint level, and they are also expressed in different units. CF_{occ} is expressed in PDF/year crop equivalent, while CF_{rel} is expressed in PDF*year/year crop equivalent. Essentially, the CF_{rel} is also calculated from the CF_{occ}. The land-relaxation CF is based on the (Köllner & Scholz, 2007) model. To calculate the actual damage, the CFs are multiplied by the LCI data on area (A) and duration (t). CFs are calculated from the relative species richness comparing a reference situation to the species richness during a land use type i in region j.

$$S_{rel,i} = 1 - \frac{S_{LU,i,j}}{S_{ref,j}}$$

This is similar to PDF (see Paragraph 5.4.3). The CF for centre-level occupancy is obtained by dividing $S_{rel,x}$ by S_{rel} for the year-weighted equivalent.

$$CF_{m_{occ,i}} = \frac{S_{rel,i}}{S_{rel(ac)}}$$

The CF_{rel} is calculated by multiplying the recovery time by half (T_{rel}). The model uses (Curran et al., 2014)-based recovery times, which categorised ecosystems into two main types: non-forested and forested ecosystems. The study also showed that recovery times did not depend on land-use type (Curran et al., 2014). For example, the ReCiPe model applied recovery times of 73.5 years for forested and 7.5 years for non-forested ecosystems, with a global average of 33.9 years (Curran et al., 2014). During the active recovery process, species richness grows about 80% faster and the CFs change accordingly.

In ReCiPe 2016, species richness data were defined on the basis of several taxa, including mammals, birds, plants and invertebrates (mainly arthropods), the collection of which excludes some very sensitive groups, such as amphibians (De Baan et al., 2013a); (Elshout et al., 2014). Species richness is based on the assumption that all species are equally important. Therefore, their loss also has the same effect. For instance, the same PDF values are considered more important in a species-rich ecosystem than in an ecosystem with low species richness (Huijbregts et al., 2016). The model used potential natural vegetation (PNV) as the reference habitat based on monitoring data (De Baan et al., 2013a); (Elshout et al., 2014). Building the model on ecoregion-specific or biome-specific data would be a preferred approach, but coverage of different biomes was too sparse to adopt a higher level of spatial resolution. The ReCiPe 2016 model therefore does not take spatial differentiation into account (Huijbregts et al., 2016). Instead, the methodology followed the approach of using two types of vegetation from forested and non-forested ecosystems and applied them as reference habitats for biomes and ecoregions as a most likely reference (Curran et al., 2014). The conversion factor from midpoint to endpoint is calculated based on terrestrial species density, in accordance with the methodology of (Goedkoop et al., 2013) in ReCiPe 2008, and the relative species loss of annual crops, as identified by (De Baan et al., 2013a).

It should be noted that the ReCiPe land-use characterisation framework is still being developed. There are also other characterisation factors. Specifically, the work of (Chaudhary & Brooks, 2018) who developed a specific set of land-use characterisation factors should be mentioned here. These updated CFs give a projection of potential species losses for five different taxa (mammals, birds, amphibians, reptiles, plants) in five broad land-use types (managed forests, plantations, grassland, cropland, urban) under three intensity levels (minimal, light and intense use) in each of the 804 terrestrial ecoregions, providing a highly regional and specialised estimate of species loss due to land-use occupancy.

5.4.5 Update literature on biodiversity valuation

The question relevant to this Handbook is how the values from the previous Handbook, which were again based on the approach of the Shadow Prices Handbook 2010, can be updated in light of more recent literature. Since 2009, a number of studies have been published that have estimated the total value of biodiversity on this planet after an initial approximation by (Constanza et al., 1997). These studies include: (IEEP, 2009); (De Groot et al., 2012) and finally (Costanza et al., 2014) who updated their 1997 study.[40] In addition, regionalised studies have been conducted, including (FEMA, 2022) for the United States and (Vysna et al., 2021) the EU. These studies are described in detail in Annex A.1. All studies except INCA (Vysna, et al., 2021) used a meta-analysis of several studies that were harmonised and from which a valuation was calculated.

In this handbook, we compare the average and median values of (Kuik et al., 2008) with these other studies and recalculate all the values of the original studies to the main types of ecosystems (terrestrial, freshwater, marine and if possible urban).
For a proper comparison, we have adjusted the original values by:
- a relative price increase in the value of biodiversity of 1% per year according to (PBL, 2018);
- the exchange rates of prices in the year of the survey;
- a deflator based on the HCPI in the eurozone.

The following table shows the results in values for global or regional biodiversity in €/ha/yr.

Table 18 - Valuation of ecosystem services according to different studies, converted to values in €$_{2021}$/ha for terrestrial, freshwater, marine, urban and total areas

	(Costanza et al., 2014)	(Costanza et al., 2014)	(FEMA, 2022)	(IEEP, 2009)	(Kuik et al., 2008)	(Kuik et al., 2008)	(Vysna et al., 2021)
Unit: original research	\$$_{2007}$/ha	\$$_{2007}$/ha	\$$_{2021}$/ha	€$_{2007}$/ha	€$_{2004}$/ha	€$_{2004}$/ha	€$_{2019}$/ha
Unit: recalculation	€$_{2021}$/ha	€$_{2021}$/ha	€$_{2021}$/ha	€$_{2021}$/ha	€$_{2021}$/ha	€$_{2021}$/ha	€$_{2021}$/ha
Area	Global	Global	USA	Global	Global	Global	EU
Type of meta-analysis	Average	Median	Average	Average	Average	Median	Average
Terrestrial	5,571	2,275	24,981	2,192	7,239	929	586
Freshwater	14,536	5,701	34,007				752

[40] In addition, there are many more studies that address regional valuations for nature, such as nature in the EU (Vysna et al., 2021) and the United States (FEMA, 2022). The first-mentioned study is interesting because of its European scale, but the values found are consistent with the median value in (Kuik et al., 2008). The last-mentioned study is based on a valuation method that arrives at a 40 times higher value per hectare than in (Kuik et al., 2008).

	(Costanza et al., 2014)	(Costanza et al., 2014)	(FEMA, 2022)	(IEEP, 2009)	(Kuik et al., 2008)	(Kuik et al., 2008)	(Vysna et al., 2021)
Maritime with open ocean	1,743	663	1,596				
Urban	6,805		32,470				69
Overall average without open ocean and without agricultural crops	7,540	4,106	19,793	2,192	7,239	929	556
Overall average with open ocean with agricultural crops	2,621	1,021	12,063				243

The 'Overall average without open ocean' was used by us for comparison because it matches the values used by Kuik, et al. (2008). These figures show that the recalculated value of €7,239/ha for the average (Kuik et al., 2008), is still in line with Constanza's recalculated value of €7,540/ha.

The median values of Kuik, et al., however, are lower than other results except those of the INCA study. However, the latter study examines only part of the total ecosystem services (see Annex A.1).

For the Handbook update, we have made the following adjustments to the (Kuik et al., 2008) valuation:
1. The values of the euro (2004) are adjusted for inflation and an autonomous price increase of 1% per year.
2. Kuik's median and average values are adjusted by a factor so that these values are comparable to Constanza's newer results. Based on Table 1818, this factor is 4.4 and 1.04 for median and average values, respectively.
3. The central value that is in the middle of the median and upper values, in line with the mean value of the Environmental Prices Handbook 2018.

5.4.6 Updating the valuation framework specific to the EU

The valuation framework for the EU is based on the regression analysis in (Kuik et al., 2008), in which further research was conducted on the Willingness-To-Pay for biodiversity. The study followed 24 studies with a total of 42 observations on the value of land-use change and biodiversity. The average Willingness-To-Pay from (Kuik et al., 2008) in Europe is on average €0.47/PDF/m^2 (at the 2004 price level), while the median value was €0.06/PDF/m^2. The following PDF/ha Willingness-To-Pay function was estimated from a meta-analysis:[41]

$$\ln(\text{Value EDP}) = 8.740 + 0.441 \ln(\text{PD}) + 1.070\ \text{FOR} - 0.023\ \text{RIV} + 0.485\ \text{COA} - 2.010\ d(\text{EDP}) - 0.312 \ln(\text{AREA})$$

- EDP = Ecological Damage Potential, which is equal to PDF/ha;
- PD = population density;
- FOR = dummy variable for forest ecosystems;
- RIV = dummy variable for river ecosystems;

41 The comparison estimates the indicator 'Ecosystem Damage Potential', an indicator equivalent to a PDF per hectare. All variables are in 2004 prices.

- COA = dummy variable for coastal ecosystems;
- d(PDF/ha) = the change in species richness per hectare due to an intervention;
- AREA = size of ecosystem in hectares.

This comparison shows that the valuation for a PDF becomes higher if the population density is higher, which mainly gives a valuation for recreational values. In addition, the valuation for an ecosystem is higher for forest and coastal systems and, conversely, lower for freshwater ecosystems.[42] The valuation is also higher for smaller ecosystems and for ecosystems that already have lower species richness: both variables are indicative of decreasing boundary utility with larger nature conservation areas and/or greater species richness. Intuitively, such a result is also plausible for recreational values in particular: urban parks, for example, are used much more intensively than large nature conservation areas, while species richness is lower.

An estimate of nature in the EU was made on the basis of this comparison. This involved studying Natura 2000 sites. Based on European Natura 2000 reports, it has been estimated that 12% of Natura 2000 sites in the EU are comprised of a river or marsh landscape, 38% are primarily forest landscapes and 2% are coastal landscapes. The population density is set at 109 persons per km^2 based on Eurostat data (2019).

The variable d(EDP) represents the change in PDF due to a measure. At the margin, this is obviously very small and was approximated in the 2018 Handbook by assuming emissions of 1 kg SO_2, NO_x and NH_3. In this update, which is based on ReCiPe 2016 and has 'land use' as its theme, we do not take such a marginal approach for granted. Here we calculate the value of a PDF by assuming the average valuation for species richness in the EU. This valuation depends on the degree of 'naturalness' of the ecosystems.

The naturalness of ecosystems is determined by starting from the information contained in ReCiPe 2016 (Huijbregts et al., 2016). It provides characterisation factors for relative species richness compared to agricultural land. Species richness per m^2 for agricultural land was set at 0.6 (De Baan et al., 2013a). The d(EDP) estimation thus used the ReCiPe characterisation factor which represents the change in PDF by type of area compared to the biodiversity of a m^2 with annual agricultural crops (grain, etc.) weighted by land use. Based on this, we see that the average PDF of 1 m^2 of land use in the EU is 0.321 (see next table). This value was used for the d(SPD) variable in the regression equation of (Kuik et al., 2008).

Table 19 – Characterisation factors from ReCiPe 2016 (CFocc), conversion to PDF and average land area in the EU

	CFocc	PDF (Srel)	Area EU
Forest	0.3	0.18	1,695,726
Grasslands	0.55	0.33	952,668
Agriculture, annual crops	1	0.6	735,574
Agriculture, perennial crops	0.7	0.42	113,536
Mixed farming	0.33	0.198	149,394
Other (urban, park landscape)	0.73	0.438	275,619
Total area allocated in ReCiPe		0.321	3,922,517
Non-divided land (e.g. wetlands)			202,590
Total land area			4,125,107

[42] So the negative value for rivers in the equation does not mean that it is negative for rivers, but that it is less important than for other ecosystems.

Lastly, (Kuik et al., 2008) showed that the valuation of PDF is unaffected by income levels (or GDP per capita).

5.4.7 Valuation of nature chosen in this Handbook

Biodiversity

The valuation for PDF is based on the regression equation of (Kuik et al., 2008) (see Paragraph 5.4.8) and the scaling factor chosen to make the results comparable with Costanza, et al. (Paragraph 5.4.7), adjusted for inflation and a 1% annual price increase in the value of biodiversity. Endpoint valuation in ReCiPe 2016 is based on the indicator species.year. This has not changed from ReCiPe 2008. This indicator of species richness has a fixed relationship with the PDF. The relationship between PDF and species.year is derived from ReCiPe 2008 (Goedkoop, et al., 2009) where the PDF.m^2 is determined by looking at the total land area and then dividing that by the number of species in the world.
The method has not changed from the previous Environmental Prices Handbook.

The following table shows the chosen valuations for PDF and the indicator species.year used in this Handbook.

Table 20 - Chosen valuations for nature indicators in the Environmental Prices Handbook, in €$_{2021}$ per unit

	Unit	Lower	Central	Upper
Potentially Disappeared Fraction	€/PDF.m^2.yr	0.229	0.325	0.421
Species.year terrestrial	€mln/species.year	21.2	30.1	39.0
Species.year freshwater	€mln/species.year	15.5	22.0	28.5
Species.year marine	€mln/species.year	15.5	22.0	28.5

The value for biodiversity plays a role in determining the impact of air polluting emissions on biodiversity on the acidification theme (Paragraph 6.7), in determining the impact of pollutants on ecotoxicity (Paragraph 0) and in determining the impact on land use (Paragraph 0). In addition, these values play a role in determining the ranges for the eutrophication theme (see Paragraph 6.6).

5.4.8 Valuation for agricultural crop yields and forest management

In addition, for emissions of pollutants that have an impact on smog formation, estimates of the impact on agricultural crop yields and forest management were made. This damage, which is relatively very small, is added to the pollutant-level damage estimates. In the Environmental Prices Handbook 2018, values for these pollutants were derived from the NEEDS project. For the current project, for emissions giving effects on oxidant formation, we based the damage costs on (EEA, 2021a). These damage costs have been adjusted to the 2021 price level. Otherwise, no other adjustments were made.

To avoid double counting, we have reduced the quantification for biodiversity loss on the oxidant formation theme with the crop losses from Costanza, et al. (2014).

We have based the damage costs to agriculture and forestry for ozone-depleting pollutants on (Hayashi et al., 2006) (see also Paragraph 6.2).

5.4.9 Effects on biodiversity not included in this Handbook

This Handbook primarily provides a framework for the impact of pollutant emissions on ecosystem services. This means, among other things, that a large part of the impact on nature and biodiversity is not valued through this Handbook because it is caused by intangible effects. Some examples of such effects are given here:

Noise nuisance and reproduction

Animals can be harmed by noise from humans on land or underwater. The influence of noise nuisance on biodiversity has received considerable attention recently because noise nuisance can have negative effects on reproduction as many species attract each other's attention with sound signals. This is true to an amplified extent for sound underwater, because sound can travel much further distances underwater. Underwater noise consists of impulse noise (e.g. pile driving during wind farm construction) and continuous noise (ship engines but perhaps also wind turbines). Both are expected to increase in the coming years, for example due to the expansion of offshore wind farms in North and West-European countries and could potentially lead to ecological impact. Nevertheless, monetary valuation of this is still in its infancy (see also Paragraph 6.11.2) and therefore is not included in this Handbook.

Litter and plastic soup

Litter consists largely of plastic waste that often looks attractive. Birds, fish and other animals can starve if they swallow plastic waste. This happens when the animals' stomachs become clogged with waste. In the oceans, this problem is called plastic soup, which refers to all plastic pollution in the ocean, including pollution with microparticles and nanoparticles.

The Environmental Prices Handbook does not currently provide a valuation of these effects on biodiversity.

Light nuisance

Artificial light at night can cause physiological and behavioural changes in animals and plants. Light nuisance can lead to changes in reproduction and problems with orientation. This could reduce biodiversity. The Environmental Prices Handbook does not provide a valuation for these effects on biodiversity.

5.5 Valuation of buildings and materials

5.5.1 Description of the endpoint

Environmental 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)[43], iron and steel (reinforced concrete)

[43] Cement and concrete react with carbon dioxide from the air to form calcium carbonate. Acidifying substances wash out this calcium carbonate. Also, with this calcium carbonate and the NO_x present in the air, cement forms calcium nitrate, which quickly leaches out.

and zinc gutters (VVM, 2013b). This shortens the useful life of these materials and leads to additional maintenance costs, as well as potentially causing permanent damage to cultural heritage (Watt et al., 2009). Moreover, particulate matter leads to dirty windows and visual deterioration of buildings. This can lead to the weathering of buildings, as well as aesthetic effects that impose costs on society, including the costs for a more frequent cleaning of buildings.

Acidification and ozone (photochemical oxidant formation) also corrode rubber and paint, again pushing up maintenance costs. Effects are also expected from the discharge of toxic and corrosive materials to surface waters and sewers, burdening operators of water treatment and sewage plants with extra costs.

The damage to buildings, materials and machinery is normally minor compared to the other endpoints, but recent research shows that these damage costs can nevertheless become very large in urban environments with a lot of cultural heritage and a certain set of materials.

5.5.2 Impact of environmental pollution on the endpoint

Damage to buildings and materials is primarily caused by air and water pollution at the following midpoints:
- Acidification.
- Particulate matter formation.
- Photochemical oxidant formation.
- The other midpoints have no direct impact on this endpoint.

5.5.3 Status of the 2018 Handbook and updates

In the Environmental Prices Handbook 2018, damage costs are quantified using four identified impact categories in line with the British Defra (Watkiss et al., 2006):
1. Damage due to acid corrosion of metals, paint and stone in utilitarian buildings. The values here were based on research by (NEEDS, 2008b).
2. Damage due to acid corrosion of calcareous building stone in historic buildings. The damage from this was quantified on the basis of (Rabl, 1999) and (VVM, 2013c).
3. Damage to paint and rubber due to ground-level ozone. The damage from this was quantified based on (Watkiss et al., 2006).
4. Damage from particulate matter pollution of buildings: weathering and clean-up costs. The damage costs from this were based on (Rabl, 1999).

The Handbook assumes restoration costs, as does most of the literature available on the subject (see (Rabl, 1999); (Holland & al, 1998); (Bal et al., 2002); (Watkiss et al., 2006); (Grontoft, 2020)). This determines the damage costs per unit of emissions based on the additional expenditure on building maintenance.[44]
However, using restoration costs is a less accurate measure of damage costs because of the following points:
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.
2. If valuation based on restoration costs assumes actual expenditures on building repairs by homeowners, this objection is overcome (after all, the homeowner decides whether the welfare loss from weathering outweighs the cost of repair). In principle, one can then speak of a ‘revealed preference’ and this has also been applied to (Rabl, 1999).

[44] Although pollution by particulate matter formation would also lend itself perfectly to CVM studies of the visual nuisance of sooty buildings, this has hardly been carried out in practice (see also(Rabl, 1999)).

However, this leads to an *underestimation* for rented housing in a rental market that is not fully efficient due to scarcity and regulation. In this case, the party renting a soot-soiled building may suffer a loss of welfare, but the owner may be unwilling to clean it as they can still rent it for the set price. (Rabl, 1999), without providing any supporting evidence, states that expenditure on restoration costs amounts to approximately half the total loss of welfare.

3. Finally, not all damage can be restored: besides damage, there is also a potential loss of cultural heritage, making restoration costs an *underestimation*. According to (VVM, 2013c), case studies show that aesthetic effects on such objects are of the same order of magnitude as restoration costs.[45]

These damage costs are included for acidification for the central value and for particulate matter and photochemical oxidant formation only for the upper value. For direct emissions to water, it did not prove possible to arrive at a damage estimate.

The valuation in the 2018 Handbook was done based on literature review distinguishing four types of endpoints: corrosion due to acidification (valuation based on (NEEDS, 2008b), degradation of cultural heritage (valuation based on (Rabl, 1999) and (VVM, 2013c), damage to paint and rubber (valuation based on (Watkiss et al., 2006)) and damage due to buildings becoming dirtier (valuation based on (Rabl, 1999)). These resulted in relatively low amounts for damage to buildings and materials: €0.6/kg SO_2-eq. for acidification in the central value and €1.2/kg SO_2-eq. for the upper value. For PM_{10} the damage was €0.8/kg PM_{10} in the upper value and for NMVOC €0.1/kg.

In this update of the Environmental Prices Handbook, we examined two issues:
1. New studies have emerged that have estimated the damage costs of air pollution on capital assets.
2. Whether all relevant capital assets have been included with buildings or whether there are other effects from air pollution, such as on machines.

5.5.4 Update damage costs for buildings

Damage costs for buildings and materials have been provided by a number of new studies, notably from Sweden and Norway (for an English overview, see (Grontoft, 2020)).
These studies show that damage costs are much higher than assumed in our Handbook, especially in cities with a high proportion of cultural heritage. (Grontoft, 2020) shows, based on the Impact Pathway approach, that damage costs for Norway as a whole are about €5/kg PM_{10} and in Oslo reach €47 for PM_{10} and €51 per kg SO_2 is emitted. These significantly higher values relative to previous studies are primarily caused by the fact that Grontoft has established concentration response functions for a wide range of materials for both clean-up costs and repair costs. In doing so, Grontoft also notes that the damage costs from (Rabl, 1999), which were also used in the previous Handbook, primarily include repair costs rather than clean-up costs. The (Grontoft, 2020) analysis shows that clean-up costs are much greater than repair costs.

To some extent, these higher damage costs are also consistent with Switzerland’s external cost estimates (INFRAS et al., 2019) showing that damage costs to buildings and materials can be as much as 20% of the damage costs of human health.

[45] Because case studies are not cited, this claim is difficult to verify.

Nevertheless, such high values are unlikely EU-wide due, among other things, to the fact that in Oslo and Switzerland most houses are plastered and painted something that occurs less in other European countries. The clean-up costs of painted plaster layers (façades) of buildings is by far the largest category in the (Grontoft, 2020) study which can reach almost €40/kg PM_{10} in Oslo. Looking only at the cost of window cleaning, in Oslo it is €6.3; €3.8 and €2.8 per kg SO_2, PM_{10} and NO_2, respectively. For sparsely populated Norway, values arise that are about a factor 3-4 lower. If we take the value for Norway and scale up the difference in population density between Norway and Oslo as an indicator of the impact of population on the damage costs, we obtain a valuation of €1.43 for PM_{10}, €2.61 for SO_2 and €0.81 for NO_x.

We validated these population density-scaled clean-up costs for glass cleaning from Grontoft by looking at spending on glass and façade cleaning in the Netherlands. It shows that these damage costs are a factor of 2.5 below a conservative assumption of the calculated expenditure on glass and façade cleaning in the Netherlands (see text box). Therefore, we propose using the population density-scaled values from Grontoft for the central value, and 2.5 times these values for the upper value, which would correspond to a conservative assumption of spending on glass and façade cleaning in the EU. For the lower value, we propose using the values calculated for PM_{10}, SO_2 and NO_x from the previous Handbook (for the upper value), adjusted for inflation.
For the impact of NMVOC on the durability of painted surfaces, we have decided to use the value from the previous Handbook adjusted for inflation. This gives an estimate of €0.10 per kg of NMVOC for the upper value alone.

Spending on glass and facade cleaning, comparing values

(Grontoft, 2020) properly identified the cleaning costs of air pollution in Norway for the first time. We converted these to values adjusted for population size. To get a feel for whether these are plausible, we looked at spending on glass and façade cleaning in the Netherlands. Based on CBS data, it can be seen that between 2002 and 2005, the average turnover of glass and façade cleaning activities was about 36% of the total turnover of sector 'SBI 93: 74701 Building and industrial cleaning' excluding interior cleaning.[46] By using this 36% on the turnover trend of the sector 'SBI 08 81,220: Other building cleaning and industrial cleaning' we obtain an estimate of turnover in this industry. In 2019, the estimated turnover of glass and façade cleaning in the Netherlands was €770 million. Emissions of PM_{10}, SO_2 and NO_x in the Netherlands in 2019 were (approximately) 60%, 31% and 46% of emissions, respectively (analysis based on EMEP's Source Receptor Matrixes for the year 2019 for primary and secondary aerosols). By multiplying the emissions in the Netherlands by the immission factor and the population-based scaled damage costs of PM_{10}, SO_2 and NO_x (see text above), we arrive at the insight that the damage costs paid by environmental pollution would be around €250 million, which is equal to one-third of the total spending on glass and façade cleaning in 2019.

On the one hand, the figure of €250 million in cleaning costs due to particulate matter pollution is an overestimate because coarse dust particles (such as sand and sea salt) are also polluting windows (PM_{100}). There is little data on the proportion of primary and secondary PM_{10} in PM_{100} but the rule of experience is that it could be at most 50% (see, for example,(Chardon & Hoek, 2002)). On the other hand, this amount is an underestimate because there are also windows washed by individuals that do not result in clean-up costs but do result in welfare losses. In a market survey (Regioplan, 2009), the figure emerges that window cleaners who have a monopoly in one municipality could hold about 25% market share of the total number of windows. If we were to stretch this figure to 50%, the overestimate due to pollution by sand would be exactly corrected by the underestimate due to dirty windows also being cleaned by individuals. On the other hand, if we were to stick with 25% market share and assume that 75% of window pollution is caused by natural sources and coarser particles (Sahara sand), both effects would also cancel out against each other.

[46] The remaining 64% is converted into activities such as chimney sweeping, tramway cleaning, industrial cleaning, post-fire cleaning, etc.

5.5.5 Damage costs for modern electronics

Not included in the Environmental Prices Handbook 2018 are the damage costs due to air pollution on materials used in modern electronics, in particular by SO_2 and NO_x. Due to these acidifying pollutants, electronic components of equipment weather faster (see, for example, (Salas et al., 2013) and (Badilla et al., 2013)). These are electronic contact points on circuit boards, for example, that break or otherwise cause failure. Although these effects have been known for a very long time, no quantification of damages is known to us, so far.

5.5.6 Valuations for this 2024 Handbook

The following table shows the valuation for emissions in euro per kg in 2021 prices for emissions in the EU. For the lower value, this is based on the inflation-adjusted upper value from the Environmental Prices Handbook 2018 for SO_2 and PM_{10} emissions.[47] These values include costs for cultural heritage and clean-up costs of façades. For the central value, we add the population density-adjusted valuation from (Grontoft, 2020) for glass cleaning. For the upper value, the damage costs are based on the damage costs from (Grontoft, 2020) for glass cleaning multiplied by a factor so that the damage costs correspond to expenditure on façade and glass cleaning in the EU. For NMVOC, we only use the damage costs from the previous Environmental Prices Handbook in the upper value, adjusted for inflation. In summary, this yields the following table:

Table 21 – Environmental prices for effects on buildings and materials, in €$_{2021}$ per kg emissions in the EU

	Lower	Central	Upper
PM_{10}	€ 0.77	€ 2.20	€ 4.46
SO_2	€ 1.17	€ 3.78	€ 8.15
NO_x	€ 0.09	€ 0.90	€ 2.54
NMVOC	€ 0.00	€ 0.00	€ 0.10

It is not known whether $PM_{2.5}$ leads to visual pollution to the same extent as PM_{10}. For the sake of consistency in the Handbook, where the user has to make a choice between either PM_{10} or $PM_{2.5}$ and due to the high correlation between emissions of PM_{10} and $PM_{2.5}$, we decided to base values for $PM_{2.5}$ on those of PM_{10} corrected for the difference in concentration in the EU between the two particulate matter categories.

In the allocation to midpoints, the damage estimate for PM_{10} was added to the particulate matter formation theme, NMVOC to the photochemical oxidant formation theme and SO_2 and NO_x to the acidification theme.

[47] For NO_x, we add the value reported in NEEDS adjusted for inflation.

5.6 Valuation of the availability of raw materials

5.6.1 Introduction and defining the scope

Security of supply of raw materials is generally considered to be an important social value. (Barnett & Morse 1963) indicated more than 50 years ago that security of supply of raw materials had been the focus of US politicians and researchers since the late 19th century. Since then, attention has not waned, from the release of the Club of Rome report in 1972 (Meadows et al., 1972) to EC policy papers on 'sustainable use of natural resources' (EC, 2005) or 'critical materials' (EC, 2011) or the 'circular economy' (EC, 2014). Such policy documents emphasise the importance of saving raw materials, especially priority, crucial or critical raw materials, from the view that raw materials are fundamental to our prosperity. This is followed by the notion that most of our raw materials are now imported which may pose a risk to our prosperity. It also stresses that closing carbon cycles is desirable from a sustainability point of view.

The question, however, is whether this reasoning has any relevance not only politically but also from a welfare perspective. The crucial question here is whether the *use* of raw materials, in addition to the price of raw materials, generates an external effect that could be taken into account in a SCBA or, for companies, could be included when calculating the company's social impact. In short, the question is whether, by saving on the use of raw materials (including water and energy), there will be an overall societal benefit larger than the monetised savings of raw materials based on the market price.

This question is not easy to answer and depends partly on the perspective adopted. In this paragraph, we first cover the different perspectives in Paragraph 5.6.2, and then explain in Paragraph 5.6.3 what we did in Environmental Prices Handbook 2018 and what research directions we have explored in this Environmental Prices Handbook 2024. Paragraphs 5.6.4 to 5.6.7 then contain three different approaches for determining environmental prices, leading to a conclusion in Paragraph 5.6.8. This work was initially undertaken specifically for the Dutch context, marking a pioneering effort. The text below therefore serves as an inspiration for other initiatives. If an approach for determining an environmental price is based on the Dutch context, we will specify this in the text. Where feasible, we provide some recommendations for an approach that can be applied at the European or country-specific level. This is summarised in Paragraph 5.6.9.

In the Environmental Prices Handbook, we only consider the availability of abiotic raw materials. Some of the reasoning that applies to abiotic raw materials can also apply to biotic raw materials. However, with biotic raw materials, such as fish, additional issues come into play. These are self-renewing public goods being jointly exploited. As known from the parable of the Tragedy of the Commons (Hardin, 1968), this leads to overexploitation. The valuation of this is more in the realm of nature appreciation (see Paragraph 5.4).

5.6.2 Perspectives

The availability of raw materials can be considered from different perspectives. Depletion of raw materials can lead to loss of value due to:
1. Environmental and social issues in the extraction phase.
2. Security of supply.
3. Rent-seeking behaviour and lack of intertemporal efficiency.
4. Intrinsic or ethical perspectives.

These perspectives are described below.

1. Environmental and social issues in the extraction phase
The extraction of non-renewable resources creates a significant amount of environmental pollution with damage to nature and health. These externalities are not currently factored into the price of raw materials. Saving on the use of raw materials can also save on the externalities of extraction. This argument was first raised by (Cleveland, 1991), among others, and is also a key driver behind the desire to move towards a circular economy. If an SCBA does not explicitly include the effects of raw material extraction, a social value could be calculated for the external effects of raw materials use that, expressed per kilogram, could be equivalent to the reduction in damage due to reduced environmental pressure.

Another problem concerns the social conditions under which raw materials are extracted. Many of our raw materials come from countries where work is carried out in appalling conditions and the environment is widely polluted. Because this information is not available to consumers, it is not included in purchasing decisions. We do know from several case studies that when such conditions were made known to the wider public (Apple, Nike, Shell), companies suffered market damage as a result. WTP research shows that there is a Willingness-To-Pay in the market for food crops with better social conditions of between 10% (De Pelsmacker et al., 2005) to more than 40% of the price of those products (Arnoldussen et al., 2022).

However, it remains complicated to derive general quantities that can be used as a valuation for raw material savings. The preferred approach, if important for a particular study, is to identify environmental and social impacts during the extraction phase and to quantify and value the environmental impacts with environmental prices. This seems to us a better alternative than a generic value for all raw materials.

2. Rent-seeking behaviour
Empirical economic research has often found the correlation that countries with high availability of abiotic raw materials tend to grow more slowly and have weaker political institutions than countries with few raw materials at their disposal (Lane & Tornell, 1996); (Gylfason, 2001; Van der Ploeg, 2011). In such countries, rent-seeking behaviour is more common where mine owners try to monetise their stocks of raw materials as quickly as possible. This leads to more raw materials being extracted from the mines in the short term than on an optimal price path, as outlined by (Hotelling, 1931). Thus, the path by which raw materials are consumed by private parties is steeper than socially optimal and more raw materials are now being exploited at the expense of use by future generations. Future generations will then bear the cost by having less availability of raw materials than is socially desirable. As a result, the current price of raw materials is de facto too low, and an external cost premium could be charged on the price of raw materials as if those raw materials were extracted in the intertemporally most efficient manner.

While this argument is, from a welfare economics point of view, the most appealing example of the existence of external costs, deriving a valuation for it is very complicated and leads to damage estimates that are highly dependent on the difference between the internal interest rate used by the extraction company and the socially optimal discount rate (see (CE Delft, 2017a)) – both of which are not easy to observe empirically.

3. Security of supply and price volatility
Another argument for why saving on raw material use represents a social value higher than the market price of that raw material is related to the impact that security of supply and price volatility can have on broad welfare. Anno 2022, this is a highly relevant issue due to political-geographical developments in Europe, which have greatly reduced the security of gas supply (gas can be considered a fossil raw material). This situation provides a potential

case to analyse the cost of alleviating this scarcity or increasing security of supply. In the context of the Netherlands, a subsidy scheme to replenish the Netherlands' gas reserves was launched in 2022. These can be considered abatement costs to reduce the scarcity of energy supplies. This approach could be applied to other European countries facing similar challenges, whether through comparable subsidies or alternative policies designed to secure resource supply amidst security concerns.

The external costs of lack of security of supply can be determined in two ways: through the damage cost method and the prevention cost method. Under the *damage cost method*, costs are based on the damage to society if security of supply is compromised. This could include damage from industrial production outages, damage to equipment or, at worst, health damage if gas cannot be delivered to hospitals and households. In the *prevention cost method*, external costs are based on the costs incurred by society to achieve a government objective. In the case of gas in the Netherlands, these are the costs incurred by society to ensure that gas reserves are filled. These costs therefore reflect the importance society attaches to ensuring that gas can continue to be supplied.

There can, of course, be a debate as to whether the lack of security of supply is an external cost. It has often been argued in the literature that price volatility in fossil fuel markets in particular can lead to economic damage. Individual users do not consider the contribution of their consumption to global price volatility, which means there could be an external effect. In a formal sense, however, this is classified in economic science as so-called pecuniary externalities, which result from effects of market forces but only lead to a distribution of wealth. In a world of perfect markets, this would indeed be true and additional consumption of a raw material would only lead to a shift along the demand curve. But in a world with significant market frictions, a transfer of income from productive, open economies to less open and more inefficient ones may well result in damage costs. Here, we point out that raw material producing countries are often less productive (see above).

4. Ethical principles

Precautionary principles or stewardship may result in citizens showing a Willingness-To-Pay for saving raw materials and natural resources on top of the market value of the saved raw materials because they feel that future generations will be better off as a result and their choices between natural and economic capital will remain unaffected.

This principle has played an important role in the attempts made so far to derive a monetary value for raw material savings. The literature here looks at the potential increase in extraction costs caused by consumption now, as consumption now results in the need to extract less economically attractive supplies in the future (Goedkoop et al., 2009).

It should be emphasised that this would be a Willingness-To-Pay on top of the prevailing market price. Indeed, the expected increase in extraction costs should normally just be reflected in the price of raw materials. However, if companies are guilty of rent-seeking behaviour (see above), this could be a reason why they are not, or not sufficiently, reflecting this increase in prices.

5.6.3 Treatment in the Environmental Prices Handbook 2018 and update in the Environmental Prices Handbook 2024

The Environmental Prices Handbook 2018 also distinguished these four perspectives. Literature research on the valuation of these principles yielded limited results. Based on the Hotelling rule, a model was developed (in Annex D of the previous Handbook) quantifying the welfare loss due to overly fast resource extraction. The model showed that these welfare losses can increase very rapidly as the distance between the social discount rate and the mine owner's discount rate increases. It was eventually decided not to value this endpoint in the Handbook because the two discount rates cannot be determined through independent empirical research.

For the Environmental Prices Handbook 2024, we decided to re-examine these four perspectives. In the process, four types of valuations were derived:
1. Valuation via abatement costs security of supply.
2. Valuation via abatement costs circular economy measures.
3. Valuation via damage cost price volatility.
4. Valuation via higher extraction costs and ethical perspective.

For the current European Environmental Prices Handbook 2024, we searched the literature for valuations for the four perspectives at the European level but found no significant results. The following studies may however offer an alternative approach to valuation of raw material availability. The first, by (Arendt R., 2022) investigates the environmental costs associated with the demand for abiotic resources needed to achieve the EU's low-carbon development goals. The study focuses on materials required for renewable energy technologies and infrastructure, such as metals and minerals. Abiotic depletion costs are calculated based on the extraction rates of resources above the social optimum (Perspective 2 in Paragraph 5.6.2), considering the GDP contributions of sectors using these materials. The total damage costs for abiotic resource depletion based on annual material demand in Europe in 2050 are €17.5 billion.
The study by (Yokoi, 2024) calculates the external cost of abiotic resource use using the user cost model. The user cost refers to the cost that future miners would incur due to the use of current capital assets (abiotic resources). They provide country-specific characterisation factors and calculated the global external costs of abiotic resource use in 2020 at $1.9 trillion.

5.6.4 Determination via abatement costs security of supply

Following concerns about the security of gas supply in 2021 and 2022, a subsidy scheme was set up by the Dutch government. This was intended to give companies an incentive to increase gas stocks further to 68%, and later to 80%. There was no incentive to do so without a subsidy, given that prices were estimated to be lower in winter than in summer, so buying in summer and selling in winter would result in losses. In this situation, it is unfavourable to replenish gas reserves in summer. The aim of the subsidy is to compensate for this price difference so that increasing the gas supply does not lead to losses and the security of gas supply is increased in the winter of 2022/2023. These subsidy costs can be viewed as proxy abatement costs for supply security, but they are specific to the Dutch context and may not be directly applicable to other European countries, which can physically not rely on gas infrastructure and gas storage facilities for their energy security. Nevertheless, we outline the calculation steps below as a potential approach for other European countries with similar subsidy schemes.

There are gas storage facilities in the Netherlands, including in Groningen, Epe and in Bergermeer. The latter was selected for the subsidy scheme. The subsidy scheme consisted of two parts, of €623 million in total: €406 million for the open subsidy scheme and €216 million as compensation for EBN to increase gas stocks in Bergermeer.[48]

The subsidy scheme for filling the Bergermeer gas storage facility initially totalled €623 million. Of this, €200 million remained in the spring of 2022, which was supplemented by €10 million and expanded again to replenish the gas supply. Bergermeer has a capacity of 4.1 bcm. The fill rate for the subsidy was about 38%. The subsidy aimed to increase the fill rate to 68%, equivalent to a subsidised amount of 1.23 bcm (billion cubic meters) of gas. To determine the cost of security of supply, we can make an average or marginal estimate. In the average calculation, we determine over a longer period of time what the average cost is to ensure the necessary security of supply. To do this, we distribute the cost of security of supply in a given period over the total demand for gas in the same period. The result then shows what we spend per m^3 of gas in external costs due to increasing security of supply outside the free market. In the marginal estimate, we only determine the price for the gas that is stored additionally using government support. This therefore concerns only the useful increased security of supply, in other words, as long as the additional supply does not exceed demand. Below we explain how external costs were calculated using both methods.

Average method

In the average method, we allocate supply security costs to demand over the same period. Selecting a period over which we calculate this is subject to a degree of randomness. The number of times that security of supply is compromised is highly irregular and highly dependent on, as it turns out, economic and mainly geopolitical developments. Therefore, as far as is known, it has not previously been necessary in the past 20 years to secure the security of gas supply in the Netherlands through government support. Therefore, the choice remains arbitrary. For the purposes of this Handbook, we propose a view period that corresponds to the update period of the Handbook. That is on average once every seven years. We then look back at developments since the last update, with the latest developments included.

During the period 2015-2021, the total gas demand was 284.5 bcm (CBS, 2022). With a total subsidy of €623 million over that period for security of supply; the average price for security of supply comes to €0.00219/m^3. This corresponds to 0.24% of the price of gas during that period.

Marginal method

In the marginal method, we consider only the actual increase in security of supply due to government support. The 2022 subsidy of €623 million was intended for 1.23 bcm of additional gas storage. Per m^3 of gas, that comes to €0.51. Again, we take a seven-year view period. We then assume that this marginal case occurs only once every seven years, given that no other similar situations have arisen during these seven years. The price for scarcity of raw materials then comes to €0.072/m^3, equivalent to 7.8% of the price of gas during that period.

[48] www.zoek.officielebekendmakingen.nl

Discussion and conclusion

Within the context of the Netherlands, the above values could very conservatively be used as environmental prices for security of supply of gas. However, a complicating factor in the 2022 subsidy scheme is the government's intention to recoup this subsidy through an additional surcharge on transport tariffs for gas transported over the national gas transmission grid (from Gasunie Transport Services). Proportionate storage for all users would mean that ultimately the user would repay the subsidy in the price of gas, already internalising the cost of security of supply in the price at a later date. As a result, there would no longer be any external costs: they would still be paid, albeit at a later date.

Another point of discussion is the application of these prices as external costs. The outlined situation is largely driven by high gas prices. We therefore relate external costs to gas prices at the time the subsidy was published. The subsidy was initially made available in May 2022. The average gas price in that month on the wholesale market is about €0.92/m^3.[49] We present the calculated external costs as a percentage of this price, and then relate this percentage to the expected price of gas from the KEV for the year 2030 (PBL, 2021).

In summary

Based on the foregoing discussion, the lower value for externalities is €0/m^3. We base the central and upper value on the average and marginal method as explained in previous paragraphs. The results are summarised in the following table.

Table 22 – Calculated environmental prices based on the Netherlands' state aid security of supply

Environmental price of scarcity of raw materials (€/m^3)	Lower value	Central value	Upper value
Based on security of supply of gas	€0	€0.00219	€0.072
As % of gas price in May 2022	0%	0.24%	7.8%
External costs (€/MJ)	€0	€0.000014	€0.0004919

We stress again that this is an estimate based on the abatement cost method. From a damage costs perspective, the external costs could potentially be much higher.

5.6.5 Determination via abatement costs: plastic recycling

Under the abatement cost approach, the price is based on the cost of the measures that society has to incur to achieve security of supply targets. The more exacting the target, the more measures we have to take to achieve it and the higher the cost. The costs therefore reflect the importance we attach to security of supply. These are the costs of most expensive measure to be taken to achieve the security of supply target (known as marginal cost). After all, these are the costs of measures we have to incur if we consume one extra kg of primary materials. For example, suppose we buy a computer with one kg of extra plastic in it. We will then become more dependent on oil imports (raw material of plastics) and have to take additional measures to become less dependent, such as recycling an extra 1 kg of plastics.

[49] Based on historical data of TTF prices for gas in the Netherlands: www.nl.investing.com/commodities/dutch-ttf-gas-c1-futures-historical-data

In this domain of abatement cost, the following approach may serve as an example for deriving a price for security of supply of oil imports for producing plastics. The Dutch government aims to use 50% fewer primary materials by 2030 and achieve a fully circular economy by 2050. In particular, there is no target for security of supply, although security of supply is, alongside environmental gains, the reason why circular targets have been formulated. The abatement costs to achieve the circular targets should therefore be allocated partly to environmental gains and partly to security of supply. Similarly, the EU's circular Economy Action Plan, which includes the Critical Raw Materials Act (CRMA), aims to increase the use of recycled materials and reduce dependency on primary raw materials. In contrast to the Dutch target, the EU's focus is specifically on strategic raw materials, which are defined as having both a high economic importance for the EU and a high risk of supply disruption. The EU has set several targets for these strategic materials by 2030, including recycling at least 25% of its annual consumption.

For the Netherlands, we estimate the abatement costs of the circular economy based on the cost of recycling plastics. In fact, of all the materials in household residual waste, plastics are one of the most expensive streams to recycle (CE Delft, 2013). This is because plastics are light and bulky and therefore collection costs per kg are high. The costs incurred by municipalities for separate collection and sorting of plastic packaging are reimbursed by the Packaging Waste Fund (Stichting Afvalfonds Verpakkingen); in 2019 this was €656 per tonne of plastics, in 2022 it is €218 per tonne of plastics, metals and drink cartons. Without this reimbursement, it would not be possible to establish a profitable business case for plastic recycling across the chain. In the absence of more recent values, we use the €656 per tonne as the marginal abatement costs in circular economy policy.

The abatement costs therefore reflect the cost of both achieving environmental gains and increasing security of supply. Plastics are less dependent on oil as a raw material. Per tonne of waste plastics collected separately, an average of 2.5 tonnes of CO_2 is avoided across the chain. At an environmental price of €130 per tonne, this equates to an environmental gain of €325 per tonne of waste plastics (2.5 * 130 = 325). If we assume that €325 of the price accounts for CO_2 gains, the abatement costs for security of supply amount to €330 per tonne of waste plastics collected separately (656 - 325 = rounded 330).

Table 23 - Environmental price based on abatement costs, plastic recycling for the Netherlands

Environmental price based on plastic recycling costs	Results	Unit
Cost of recycling plastic	€330	€/tonne
Petroleum saved	85,400	MJ petroleum
Environmental price	€0.0039	€/MJ petroleum

5.6.6 Determination via damage costs price volatility

A second method of determining a price for scarcity of raw materials is to analyse price shocks of raw materials and their macroeconomic impact. Well-known shocks include the 1973 and 1979 oil crises, and the very recent increases in the price of gas since 2021 following the Russian invasion of Ukraine.

(Awerbuch & Sauter, 2006) examine the relationship between oil price shocks and global GDP. This reveals that oil supply shocks in the 1970s have shown that oil price increases and volatility lead to macroeconomic losses through rising inflation, unemployment and by depressing the value of financial and other assets. It is estimated that a $10 increase in oil prices leads to a 0.5% decline in world GDP. In absolute terms, this concerns hundreds of billions of dollars of productivity losses.

(Oladosu et al., 2018) conducted a meta-analysis on oil price elasticity of GDP in the US. This reveals an elasticity of -0.020%, with a confidence interval of -0.035-0.006. That means that from this analysis, a -0.02% effect of oil price on GDP was identified; in other words, for a $10 increase in oil price, GDP falls by about 0.2%.

Based on this data, we can estimate the impact of various energy price shocks in the past. Then, in a similar way to the government support method, we can marginally and averagely distribute the GDP cost over the combined demand for oil and gas in recent decades. The calculation outlined below is specific to the Netherlands, but by modelling the GDP loss for other European countries, one could derive country-specific values using the price volatility method.

Calculation

Using the two elasticities mentioned (-0.02 and -0.05), we estimate a range of GDP loss for the Netherlands over the period 1970-2021 due to the two oil price shocks in 1973 and 1979, and the gas price increase in 2021. During this period, these were the three major energy crises. We look at what the price rise over that time has meant for the GDP level over the same period. We then relate these losses to energy consumption over the entire period: including the years when no shocks to security of supply occurred. In this way, we determine the average monetary losses per MJ over a longer period (50 years).

In 1973, the price of a barrel of crude rose from $2.74 to $11.65 in a week. Then OPEC raised the price of oil from $12 to $33 a barrel in the year 1979. In one year between 2021 and 2022, gas prices rose from around €0.95 to €3.77 per m^3. We apply the GDP elasticity to these increases. In this context, we assume that the price elasticity for oil also applies to the price of gas. To do so, we first convert the gas price increase to $/BOE (barrels of oil equivalent). Furthermore, we use historical data on GDP in the Netherlands, converted to dollars:
1. 1973: GDP in the Netherlands was €84.4 billion, equivalent to €349.3 billion at 2021 prices.
2. 1979: GDP in the Netherlands was €151.5 billion, equivalent to €397.7 billion in 2021 prices.
3. 2021: GDP in the Netherlands was €856.4 billion.

Oil price increases were equal to $8.91 and $21 per barrel in 1973 and 1979, respectively. The gas price hike was about $567/BOE. With a price elasticity of -0.02, these increases result in a total GDP loss of €99.4 billion. With an elasticity of -0.05, it leads to a total loss of €248.5 billion.

We allocate these losses to total energy use since 1970 (the earliest available data). Because energy prices are interrelated, we allocate the costs to total energy use, including use of energy other than oil and gas. The overall energy consumption for the period 1970-2021 totalled more than 3.6 billion kilotonnes of oil equivalent (KTOE). This leads to an average external cost of €0.00061-0.00152 per MJ (calculated with price elasticities of -0.02 and -0.05).

Table 24 - Calculated environmental prices based on the Netherlands' GDP elasticity oil price

Environmental prices of resource scarcity (€/MJ)	Lower value	Central value	Upper value
Total GDP loss (€ billion)	€0	€99.4	€248.5
Energy consumption 1970-2021 (KTOE)	3,640,656	3,640,656	3,640,656
Environmental price: based on GDP elasticity oil price (€/MJ)	€0	€0.00061	€0.00152

5.6.7 Determination via change in extraction costs (ReCiPe 2016)

Depletion of abiotic raw materials has long been included as a relevant endpoint of environmental intervention in LCAs (Goedkoop & Spriensma, 2000). Quantification concerns the risk of future generations of humanity running out of resources. Given the importance of the 'precautionary principle' and 'stewardship' in the LCA perspective, there is logic in putting a value on this forgotten item. In ReCiPe (Goedkoop et al., 2013), this endpoint quantifies effects that assume that current use leads to higher extraction costs over time.

ReCiPe distinguishes scarcity of raw materials in two domains: mineral raw material scarcity and fossil raw material scarcity and expresses it in dollars as the extra cost of extracting future mineral and fossil raw materials. Behind this is the idea that the cheapest raw materials are extracted first: if these are consumed, future generations will be left with only raw materials that are more expensive to extract. The change in extraction costs due to consumption of a raw material are taken as a characterisation factor in ReCiPe. Of particular interest to our analysis is the cost of extraction of fossil fuels.

End point characterisation of crude oil, natural gas and coal is expressed in surplus cost potential (SCP) and based on cumulative cost-tonnage relationships for these three fossil raw materials. Because there is no complete understanding of the full cause-effect, no constant mid-to-end factor can be given (see(Vieira et al., 2016a)). The following table shows the characterisation factors converted in $€_{2021}$/MJ where the values from ReCiPe, in dollars from the year 2013, have been converted to euros from 2021 and the 'Fossil fuel potential' which in ReCiPe represents the ratio between the various energy carriers has been taken into account. Table 25 summarises this.

Table 25 - Endpoint characterisation factors for fossil fuels, expressed in $\$_{2013}$/unit of raw material and $€_{2015}$/MJ, hierarchical worldview

Fossil raw materials	ReCiPe 2016	Unit	Converted	Unit
Crude oil	0.457	$\$_{2013}$/kg	0.00898	$€_{2021}$/MJ
Coal	0.034	$\$_{2013}$/kg	0.00028	$€_{2021}$/MJ
Natural gas	0.301	$\$_{2013}$/Nm^3	0.00498	$€_{2021}$/MJ

The above hierarchical values can be weighted with country-specific energy consumption data from the year 2019. In the case of the EU, energy consumption data is measured using the Energy Balance from Eurostat data, with which we obtain a weighted value of €0.0063/MJ.

5.6.8 Summary overview and choice Handbook 2024

Most of the above research has been performed for the Dutch context. As such, the conclusions in this chapter are mostly relevant to the Dutch context. However, as a second-best option, we advise that in an EU context, these prices are applied as well. We argue that valuation with these options is preferable over not valuing this topic at all.

Resource scarcity is a major theme within the Dutch policy context and has gained major concern after the Russian invasion of Ukraine and the resulting gas crisis. We presented several methods to derive a price for resource depletion withing the geopolitical context of dependency on imports. However, it is possible to debate to what extent scarcity of raw materials is not already factored into the current price as economic actors' factor in market expectations on the raw material scarcity into their decision making. Security of supply and rent-seeking behaviour may be the main arguments why there are still external costs associated with the consumption of raw materials. At this stage, we suggest not including scarcity of raw materials in general in economic analysis tools such as SCBAs, because of a lack of consensus on methodology and possible discussions on pros and cons of these methods. If effects on security of supply from the use of raw materials are to be expected, our recommendation is to analyse and value them specifically within a SCBA and not to rely on average prices.

For use in LCAs, we did try to arrive at a valuation, as scarcity of raw materials is a common item in social discussions, such as on circular economy. These values are, however, mostly specific to the Dutch context. It is clear from the analysis in this chapter that the different methods lead to a huge variety of estimates. To do justice to this variety, we maintain a range that reflects the different methods. At the lower end of the range, we choose to value scarcity of raw materials at zero. It can be argued from an economic principle that the cost of increasing scarcity is already discounted into the price. At the upper end are the estimates regarding the abatement costs of plastic recycling and the extraction costs from ReCiPe 2016 (valid in EU context) (Huijbregts et al., 2016)of €0.0039/MJ and €0.0063/MJ, respectively. We have chosen to set the valuation conservatively at €0.0039/MJ. For the central value, we then have four remaining valuations: two via the GDP loss estimates and two via the cost of holding stocks ranging from €0.000014/MJ for the average cost of security of supply to €0.00152/MJ for the high GDP loss estimate. We suggest taking the average of these four estimates as the indicative central value. This yields a valuation of €0.000658975/MJ. In summary, this gives us the following range in the table below:

Table 26 – Environmental prices of scarcity of raw materials, in €/MJ, for the Netherlands

Environmental price (€/MJ)	Lower value	Central value	Upper value
Raw material scarcity	0	0.000659	0.00390

In Paragraph 6.12, these valuations are applied to both fossil fuel scarcity and metals.

5.6.9 Summary overview and choice Handbook 2024 for the EU

In this chapter, we provided an overview of potential valuation methods for the availability of abiotic resources within the Dutch context. Here, we suggest how these methods could potentially be applied in the European context, based on the four types of valuations presented in the Handbook 2023. Note that the discussion of alternative perspectives on calculating resource availability is beyond the scope of this Handbook.

Valuation Methods and Approaches:
1. Valuation via abatement costs security of supply.
2. Valuation via abatement costs circular economy measures.
3. Valuation via damage cost price volatility.
4. Valuation via higher extraction costs and ethical perspective.

Table 27 - Recommended approach for the EU on valuing the availability of raw materials

Valuation method	Approach for the Netherlands	Recommended approach for the EU
Abatement costs for security of supply	Subsidy scheme for filling gas storage facilities	For gas-reliant countries, a similar calculation could be applied to comparable country-specific subsidy schemes. For countries not relying on gas, alternative critical resources should be considered.
Abatement costs for circular economy measures	Cost of recycling plastics	For countries with similar raw material reduction targets, apply a similar calculation for the costs of recycling plastics. For an EU-wide approach, focus on the costs of recycling critical raw materials (not necessarily plastics).
Damage cost from price volatility	Relationship between oil price shocks and global GDP.	Apply the same method for other European countries. Gather country-specific GDP and energy consumption data for the price shocks used in the Netherlands' calculations.
Higher extraction costs and ethical perspective	ReCiPe end-point characterisation of crude oil, natural gas and coal	This value is already specified for the EU.

Depending on the outcomes, the results can be used to express the environmental price within a range, similar to Table 28 in this Handbook. For the time being, we suggest the international user is using the Dutch values because there are currently no EU-level alternatives.

5.7 Valuing the effects on well-being

In the literature, there are a number of categories of 'nuisance' that can play a role in valuation in cost-benefit analyses, for example, and which are usually grouped together in analyses of the living environment. This concerns effects on people's well-being that do not directly lead to health effects but are perceived as 'irritating' or 'annoying'. There is therefore a Willingness-To-Pay to prevent this nuisance, although it is not easy to value this:
- valuation of noise nuisance;
- valuation of stench nuisance;
- visual nuisance;
- other nuisance.

These are briefly explained below. Since the valuation of noise nuisance is the most important endpoint here, it is only discussed briefly here and otherwise dealt with in Paragraph 6.11.

5.7.1 Noise nuisance

Noise nuisance causes inconvenience and health damage. Health damage concerns issues such as heart problems due to long-term exposure to ambient noise, which we discuss in detail in Paragraph 6.11.

In addition, noise nuisance causes inconvenience. Inconvenience can be regarded as that part of noise that is perceived as a nuisance but for which no direct health effects are known. Inconvenience is classified (WHO, 2018) as annoyance and is often measured in academic studies by charting the percentage of respondents who report being '(highly) annoyed'. As argued in Paragraph 6.11, both health harm and nuisance can be distilled from stated preference research. Since that is where the damage is discussed further, we will not go into it further here.

5.7.2 Valuation of stench nuisance

In a number of cases, environmental pollution comes with stench nuisance. This is particularly the case for industrial processes and agriculture where manure can lead to stench nuisance.

Stench nuisance occurs particularly in agricultural processes that lead to ammonia (NH_3) or hydrogen sulphide (H_2S) emissions. High concentrations of ammonia are one of the causes of the stench associated with manure. In addition, stench nuisance occurs during industrial processes.

Stench nuisance is complicated to measure and calculate back to a valuation for NH_3 or H_2S because there is a threshold of emissions below which stench nuisance does not occur. A valuation will therefore have to be determined on a case-by-case basis. For example, there are case studies from case law surrounding valuations for the Purposes of the Valuation of Immovable Property Act (WOZ-waarde) in the Netherlands. In a 2019 judgment at the Arnhem - Leeuwarden Court of Appeal on the WOZ valuation of a house located near a sewage pumping station, the court ruled that a 10% discount on the land tier used by the municipality when determining the WOZ value of a housing, adequately compensated for stench nuisance (several times a year) that varies in intensity and is not always detectable in the house (Arnhem-Leeuwarden, 2019).

5.7.3 Visual nuisance

Visual nuisance can be a relevant welfare effect if a project reduces the quality of the living environment. This involves blocking the view or changing the character of the landscape so that the view is disturbed. Examples include the construction of a wind turbine, solar farm or high-voltage pylon.

The effect of visual nuisance can be measured in the decrease in the value of houses near the object in question. In terms of welfare effects, this drop in value can be seen as a reflection of the actual loss of welfare for local residents. The decrease in value due to construction of a particular object can be measured by comparing house prices in the vicinity of this object before and after construction. This refers to the actual prices at which housing is sold on the market. In a data set, these data can be enriched with additional data, including information about the distance to and height of the object in question. Using this data, a model can be estimated that can distil the effect of the presence of the object in question. The estimated price drop can then be expressed as a percentage of the value of the housing.

Several studies have been conducted on the effect of wind turbines on the value of a house. These show that there is a sharper drop in value when houses are closer to a wind turbine. The height of a wind turbine also affects the degree of perceived annoyance: the higher the turbine, the greater the effect. (Dröes & Koster, 2021) estimate a 5.4% drop in value of housing within two kilometres of a wind turbine with a minimum tip height of 150 metres. They also show that it is especially the first turbine near the housing that has an effect on the value of the house. The order of magnitude of these effects is confirmed in other research on the impact of wind turbines on housing prices in the Netherlands (TNO, 2022). They show that wind turbines with a tip height of more than 150 metres cause an average drop in value of 8% within a 1-kilometre radius, while within 1 to 2.5 kilometres it is 4.5%.

The impact on solar parks has also been studied with regard to housing values. (Dröes & Koster, 2021) estimate the effect of solar parks within a 1-kilometre radius at a 2.6% decrease in the value of housing.
In order to determine the total welfare loss based on these key indicators, the number of houses around the relevant objects must be counted or estimated. This should take into account the principle that the first wind turbine has the strongest effect on the value of the housing. For example, the drop in value of housing near a wind farm with six turbines will not be six times higher than if one wind turbine had been installed. The size of a solar farm should also be taken into account when determining its impact.

Moreover, the concentration of particulate matter in the atmosphere can lead to visual nuisance. (Rabl et al., 2014) cite some studies from the United States that show these costs can be significant. For example, in a widely cited publication, (Muller & Mendelsohn, 2007) calculate that the damage costs for reduced visibility of PM_{10} in the atmosphere in 2002 was about $1.3 billion. This is a significant harm: after mortality and morbidity, reduced visibility is the third largest damage in this study contributing to about 3.7% of total damage.[50]

5.7.4 Other impact on well-being

In addition, there are other economic activities that can cause damage that affects people's well-being, such as earthquakes from gas extraction, vibrations from rail and road transport, damage to quays by inland navigation, etc. Although these damages can cause a lot of inconvenience, they have not been investigated and quantified in this Handbook.

[50] Given that US emissions of PM_{10} in 2002 were about 18.4 kt, and most of the US emissions also end up in the US itself, one can calculate that the damage costs of PM_{10} in this study are about €70/kg PM_{10}. However, the damage amounts of all pollutants in Mendelsohn and Mueller's study are much higher than those in Europe.

6 Average midpoint level

6.1 Introduction and general methodological framework

In this chapter, we present the determination of environmental prices at midpoint level with regard to environmental themes. We have identified twelve midpoints in this Environmental Prices Handbook:[51]

1. Ozone depletion (Paragraph 6.2).
2. Climate change (Paragraph 6.3).
3. Particulate matter formation (Paragraph 6.4).
4. Photochemical oxidant formation (Paragraph 6.5).
5. Eutrophication (Paragraph 6.6).
6. Acidification (Paragraph 6.7).
7. Human toxicity (Paragraph 6.8).
8. Ecotoxicity (Paragraph 0).
9. Ionising radiation (Paragraph 6.10).
10. Noise (Paragraph 6.11).
11. Extraction (resources/water) (resource scarcity) (Paragraph 6.12).
12. Extraction (land use) (Paragraph 0).

These midpoints are described in Paragraphs 6.2 to 0, along with the methods used to arrive at impact estimates. The following is a general overview of the relationships between midpoints and endpoints in Paragraph 6.1.1.

6.1.1 Relationship between midpoints and endpoints

The relationship between midpoints and endpoints varies depending on each specific midpoint. Table 28 provides an overview of how the selected midpoints affect the five endpoints.

The number of endpoints with valuations remains the same compared to the 2018 Environmental Prices Handbook. However, additional effects were found at the various midpoints, such as visual nuisance from particulate matter formation or degradation of materials by UVB radiation caused by ozone depletion. These effects were not mentioned in the 2018 Handbook but are included now, though we have been unable to assign monetary values to them in this Handbook.

[51] These midpoints do not necessarily follow the classification of characterisation models, such as ReCiPe. Ecotoxicity is subdivided in ReCiPe, for example, into ecotoxicity to freshwater, saltwater and land. Noise pollution is not a midpoint in ReCiPe. In total, these twelve midpoints correspond to all nineteen midpoints identified in ReCiPe (2016).

Table 28 - Overview of relationships between midpoints and endpoints in the Environmental Prices Handbook

Midpoint / Endpoint	Human health	Ecosystem services	Buildings & materials	Availability of raw materials	Well-being
Ozone depletion	Yes	Dl	X		
Climate change	Other	Other	Other	Other	Other
Particulate matter formation	Yes		Yes		X
Photochemical oxidant formation	Yes	Yes	Dl		
Ionising radiation	Yes	X			X
Acidification		Yes	Yes		
Human toxicity	Yes				
Ecotoxicity		Yes			
Eutrophication		Yes			X
Nuisance (noise)	Yes	X			Yes
Extraction (land use)		Yes			X
Extraction (resources/water)		Dl		Dl	

- Yes (green) indicates that this effect has largely been taken into account and monetised.
- Dl (yellow) indicates that this effect has been partially monetised (incomplete).[52]

  An X (red) indicates that this midpoint does characterise the endpoint but is not included in the Environmental Prices Handbook.
- 'Other' indicates that the effects were determined in a different way. In the case of climate change, these are abatement costs (blue).
- An empty cell means the theme was not characterised with respect to the endpoint or that impacts are negligible.

6.2 Ozone depletion

6.2.1 Description of theme

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 some of the incoming ultraviolet radiation (UVB), 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 chemicals containing chlorine and bromine. These compounds react with stratospheric ozone, reducing the 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 the 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.

[52] Whether a square is green or yellow is based on our estimation of known effects. It should be emphasised that we start from what is considered 'mainstream' about dose-effect relationships, for example by the WHO. A green box therefore does not mean that all effects reported in the scientific literature are included, as there are many effects that, while plausible, are not yet subject to scientific consensus.

Global emissions of ozone-depleting substances (ODS) peaked in the mid-1990s and have been 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 (VVM, 2013a), it is only recently that the ozone layer has begun 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 primarily 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 fumigation (methyl bromide). Global production of ODS has declined substantially since the mid-'90s due 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 Impact

Ozone depletion impacts humans, plants and animals. UV-radiation can damage DNA and proteins in the skin and eyes, leading to skin cancer and cataract over time. It also affects the physiological functioning of wild plants and agricultural crops and can cause radiation damage (VVM, 2013a). Ozone depletion also damages phytoplankton reproduction, reducing food availability in oceans (Smith et al., 1992). Finally, increased UVB radiation may accelerate ageing of some materials, such as (synthetic) polymers, although there are many uncertainties regarding the exact impact response relationships (Andrady et al., 1998). Ozone depletion thus impacts both human health and ecosystems, as well as potentially buildings/materials.

Most ozone-depleting substances (ODS) are also greenhouse gases, thus contributing to climate change. These impacts are characterised under the endpoint 'climate change', however, and are included there in this Handbook. Depletion of stratospheric ozone should not be confused with the increase in ground-level ozone due to smog. The latter effects are described in Paragraph 6.5.

6.2.4 Treatment in the Environmental Prices Handbook 2018

The valuation of pollutants with an impact on the theme 'ozone depletion' was based on the 2009 ReCiPe methodology for human health. The effect of changes in UVB radiation on human health is calculated using the AMOUR model (RIVM, 2007). The resulting damage factors are expressed in DALYs per unit change in the Effective Equivalent of Stratospheric Chlorine (EESC). These are then converted to characterisation factors, expressed as DALYs per tonne CFC-11-eq. for each class of ODS.

Only the negative impact on agricultural crops was included for the effects on ecosystem services.

6.2.5 Characterisation and indicator midpoints in Environmental Prices Handbook 2024

Pollutants with an impact on the theme 'ozone depletion' were characterised according to ReCiPe. ReCiPe is based on ozone depleting potentials (ODPs) calculated by the World Meteorological Organization in 2010 (WMO, 2011). Impacts on this midpoint are expressed in kg CFC-11-eq. CFC-11, a chlorinated fluorocarbon formerly used mainly as a refrigerant, has the highest ozone-depleting potential (ODP) of any compound in this family.[53]
By definition, it has an ozone-depleting potential (ODP) of 1.

For nitrous oxide (N_2O), which impacts ozone differently than chlorine- and bromine-containing pollutants, preliminary characterisation factors are included in ReCiPe 2016. ReCiPe 2008 did not include a characterisation factor for N_2O.These preliminary characterisation factors are provisional and may be subject to revision, though they are currently used for determining environmental prices.

We assume the individualistic perspective for the lower value and the hierarchical perspective for the upper value. For the central value, we assume an 'extended individualist perspective' (see Annex B.4 for an explanation).

6.2.6 Endpoint determination and environmental prices

To quantify the impacts of ODS on human health, ReCiPe 2016 is adopted (Hayashi et al., 2006). ReCiPe 2016 assumes that the impact of a change in ODPs leads to an increase in UV-B radiation, which in turn results in a higher disease burden.

The damage factor is based on the projected increase in the incidence of three types of skin cancer (malignant melanoma, basal cell carcinoma and squamous cell carcinoma) and is expressed in DALYs. The conversion from midpoint to endpoint factors expressed in DALY/kg CFC-11-eq. and varies across the three perspectives.

For human health impacts, a monetary valuation was obtained using a standard value for a DALY, under the assumption that 1 DALY = 1 VOLY. This is in line with the analysis in Paragraph 5.3. For impacts on ecosystem services, only the 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 Environmental Prices Handbook 2018. The following table gives the average damage costs for the ozone depletion midpoint, shown as the value for the midpoint characterisation factor for EU27.

Table 29 – Average damage costs for midpoints in terms of ozone depletion, in €$_{2021}$/kg

Midpoint	Unit	Lower	Central	Upper
CFC-11-eq.	€/kg CFC-11-eq.	€15.2	€29.1	€69.6

[53] In addition, CFC-11 is a major greenhouse gas.

6.3 Climate change

6.3.1 Description of theme

Climate change describes the gradual change in weather patterns, such as temperature and rainfall, over long time spans. Although climate change can occur due to natural variations in the position of the sun and volcanic activity, the term is usually used to refer to *anthropogenic climate change*. The climate is currently changing due to human activity leading to an increasing concentration of greenhouse gases (GHG) in our atmosphere. These gases, such as CO_2, CH_4 and N_2O allow incident solar rays to pass through, but block heat reflected from the earth. This phenomenon is known as the greenhouse effect and causes, among other things, global temperatures to rise.[54] Since the preindustrial era (1850-1900), the Earth's average temperature has increased by more than 1˚C due to human activity (IPCC, 2021). This increase will continue in the coming decades. The exact temperature rise we are heading for is uncertain and highly dependent on global climate policy. To understand the impact of emission reductions and climate change, the IPPC distinguishes several emission scenarios. In the emission scenario consistent with current climate policies (SSP2-4.5), global temperatures in 2100 will increase by 2.7˚C (2.1-3.5˚C) compared to average temperatures in 1850-1900 (IPCC, 2021). In the most extreme scenario (SSP5-8.5), where international climate policies fail, GHG emissions will not peak until 2090 and global temperatures will rise by 4.4˚C (3.3-5.8˚C) by 2100. Such a temperature rise will have a major impact on humans, animals and ecosystems, especially given that the temperature rise on land is on average 1.4 to 1.7 times greater than at sea (IPCC, 2021).

6.3.2 Sources

Fossil fuels are the largest source of GHG emissions. Fossil fuels are used in all sectors of the economy and their use has spiralled over the past 100 years. The combustion of fossil fuels mainly releases a large amount of CO_2, but also, for example, nitrous oxide (N_2O), which has a significant temperature-increasing effect. The extraction of fossil fuels also releases large amounts of methane. In addition, GHG emissions of methane (CH_4) and nitrous oxide (N_2O) occur mainly in agriculture and in landfills containing organic waste. GHG emissions are also released in industrial processes, such as in the cement and aluminium industries. Finally, refrigerants and propellants often contain GHG released in the production, use and waste phases. This applies to both traditional CFCs (which also have an effect on ozone depletion, see Paragraph 6.2) and their more modern replacements.

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). SO_2 has both a direct and an indirect cooling effect; the direct cooling is caused by SO_2 particles reflecting sunlight and the indirect cooling is due to SO_2 contributing to cloud formation, leading to a cooling effect (Fuglestvedt et al., 2010). Pollutants emitted by aircraft also have partly a cooling and partly a warming effect (CE Delft, 2014), although the warming effect dominates.

[54] Besides emissions that lead to an increase in global temperature, there are also pollutants that have a cooling effect. Human emissions of SO_2 and NO_x, among others, have therefore dampened the global temperature rise by about 0.4 ˚C (IPCC, 2021).

6.3.3 Impact

The impact of climate change is wide-ranging and not limited to certain countries or ecosystems. However, the impact on society will tend to be more severe in developing countries, which also have fewer opportunities to adapt (GHF, 2009)
The latest findings on the impact of climate change and associated probabilities on society are discussed in (IPCC, 2022).

Below, we briefly summarise the main effects:

- Climate change will lead to sea level rise due to melting glaciers and polar ice, as well as the expansion of water at higher temperatures. Sea level rising will lead to loss of land area, buildings and capital assets, especially in delta areas where by far the most people live worldwide.

- Climate change will lead to an increase in the number of tropically hot days and the frequency and duration of heat waves, resulting in higher mortality rates from heat stress. On the other hand, mortality due to cold will decrease.

- Rising temperatures increase the probability of the occurrence of certain parasitic diseases such as malaria and dengue fever. It also increases the probability of pests and diseases in plants and animals, such as the bark beetle that has affected large parts of woodlands in the Black Forest.

- An increase in heat waves and periods of drought will lead to a reduction in total food production as temperatures rise. In the event of more limited temperature increases, global food production will shift significantly, with reduced opportunities in warm countries offset by expanded opportunities in colder countries. These changes are expected to occur rapidly, which may lead to major socio-economic adaptation problems, where famines may occur more frequently, and migration flows may emerge.

- Climate change will lead to an increase in the number of forest fires and floods, as well as increased severity of storms, including tropical storms.

- Rainfall patterns are shifting as a result of climate change, which means some areas will face water shortages and others will experience water surpluses. This could also lead to flows of migrants.

- Isolated (water) ecosystems such as coral reefs are likely to disappear to a large extent due to climate change.

- Climate change can lead to feedback effects, which can manifest as hard-to-predict *tipping points*. For example, climate change could lead to the loss of the West Antarctic ice sheet, the melting of permafrost releasing large reserves of methane or the collapse of the Amazon rainforest. These events could in turn lead to further global warming, with potentially irreversible consequences.

The extent of the above effects depends on the global climate mitigation response.
In general, more ambitious climate policies lead to less warming and thus less damage to ecosystems, human health and the economy.
The harmful effects of climate change are unlikely to increase linearly: an additional degree of warming is more harmful at high baseline levels.

6.3.4 Characterisation and midpoint indicators

Because CO_2 is the main greenhouse gas, emissions of other GHGs are usually expressed as CO_2 equivalents. ReCiPe 2016 characterises the various GHGs by their Global Warming Potential (GWP), based on (IPCC, 2013), with the GWP of CO_2 set at 1. For the Environmental Prices Handbook, we therefore use (IPCC, 2013) for characterisation.

ReCiPe has characterisation factors for the 20-year (individualistic) and 100-year (hierarchical) perspectives. We note here that internationally, the 100-year perspective is leading:
- Reports to the UNFCC mandatorily use the 100-year perspective.
- All country pledges use the 100-year perspective.
- All data from, for example, emission factors or emissions data use the 100-year perspective.

For these reasons, we use the 100-year perspective for both the lower, central and upper values.

6.3.5 Damage costs, abatement costs and climate policy

The adverse effects of climate change can be quantified by means of a '*Social Cost of Carbon*' (SCC). This SCC indicates the damage costs of one tonne of CO_2 emissions. An SCC attempts to weigh all effects of climate change, such as economic damage from droughts, floods and forest fires, but also, for example, years of life lost due to heat stress, and express them in a monetary value. The SCC is usually calculated using climate economic models in which assumptions about impacts are combined with assumptions on global income trends and distribution (see for example (Bressler, 2021)). Average damage costs depend on global climate policies pursued: since the damage from climate change increases faster than in a linear path, the average SCC is higher in more extreme warming scenarios than in scenarios where ambitious climate policies manage to limit global warming.

When used in SCBAs, the damage cost methodology is usually preferred over the abatement cost methodology (see Paragraph 5.2). After all, in an SCBA, one ideally wants to assess how the cost of measures compares to the social damage avoided. If benefits are expressed as avoided costs of measures to achieve the government objective, the analysis takes on the character of a social cost-effectiveness analysis rather than an SCBA. Because estimates of the damage costs of CO_2 emissions were and still are very uncertain, the abatement cost methodology had been chosen in the Environmental Prices Handbook 2018.

In the abatement cost approach (see also Paragraph 5.2), the marginal cost of achieving a policy goal is taken as the starting point for valuation. The assumption here is that a Pigouvian levy will be able to precisely achieve policy objectives. The level of the levy is the same as the cost of the most expensive measure to be taken in the most cost-effective package of measures that achieves the objectives. The abatement costs of climate change depend on the costs of various mitigation techniques, but thus also on the climate target used. This is because the more ambitious the climate target, the more expensive the techniques have to be to achieve the intended CO_2 reduction. For the same reason, abatement costs generally increase over time: increasingly expensive techniques must be employed to reduce an additional tonne of CO_2.

6.3.6 Environmental prices in the previous Handbook

In the Environmental Prices Handbook 2018 GHG emissions were valued based on the abatement cost method. The abatement cost approach was chosen for two reasons and was furthermore advised by the Dutch government following an advise of a working group on discount rates and CO_2-valuation:

1. Literature showed that damage cost estimates became increasingly uncertain over time. There was no trend towards a reduction in uncertainty margins. Tol (2008) showed in a meta-analysis that the spread in outcomes was huge: from less than €1/tonne CO_2 to more than €500/tonne CO_2. Moreover, important cost components (such as health impacts, political instability, migration and losses in biodiversity) were often not included in SCC studies (Bergh & Botzen, 2015), there was much debate about the level of the discount rate to be applied, and many models did not seem to deal adequately with tail risks.[55] Taken together, damage estimates at that time seemed to underestimate the actual SCC.

2. The damage cost categories are quite diverse. In such a context, societal preferences for harm prevention can also be approached through politics that sets goals and makes trade-offs. This allows the abatement costs to be considered a crude proxy for the damage costs in the case where climate target is consistent with ambitious reduction pathways.

Because environmental prices for other pollutants are mainly based on Willingness-To-Pay, which is measured including VAT, the previous Handbook recommended that these prices need be increased by the average VAT rate when used together with other environmental prices, such as in cost-benefit analyses. In accordance with (SEO, 2016b), this could be calculated at an average rate of 18% (VAT and other indirect cost-increasing taxes).

6.3.7 New insights into damage costs

Since the publication of the Environmental Prices Handbook 2018, several new SCC estimates have appeared. Recent studies on the SCC primarily add new cost components to existing *Integrated Assessment Models* (IAMs), such as the impact of climate change on human mortality, *equity weighting* (in order to value inequality and/or the diminishing marginal utility of income) and risks of permanent damage to the growth rate of the economy.

Most of these studies find significantly higher values for SCC than the 2016 estimates. For example, (Bressler, 2021) extends the existing DICE model with a module that captures the impact of climate change on human mortality. This expansion increases the SCC from an average of 37 to 258 €/tonne CO_2 in a climate scenario where global temperatures rise by 4°C until 2100 compared to the period 1850-1900. In the associated economically optimal reduction pathway, CO_2 emissions fall to zero by 2050. Similar effects can be seen when existing IAMs are expanded with equity weights or permanent damage to the growth capacity of the economy. (Moore & Diaz, 2015) adapt the same DICE model so that temperature rise affects the average productivity of labour and capital. Under this assumption, SCC increases by almost 200 €/tonne CO_2. (Liu et al., 2022) show that the SCC as calculated by the PAGE model increases from 79 €/tonne to 291 €/tonne CO_2 when equity weights are added. A Nature study, conducted as part of the US government's SCC update, found an SCC of 185 €/tonne CO_2 (Rennert et al., 2022) using a new IAM (the GIVE model). The explanation for the high price, according to the researchers, is that the GIVE

[55] Tail risks refer to unlikely outcomes that cannot simply be neglected because of their large negative impact.

model incorporates all uncertainties in a systematic and consistent way. The (EPA, 2023) also employed the GIVE model and estimated a SCC of $120, $190, and $340/tonne CO_2, corresponding to near-term discount rates of 2.5%, 2.0%, and 1.5%, respectively.
The German Umweltbundesamt intends to adopt the GIVE model for evaluating the external costs of GHG emissions in the next update of its Method Convention, where equity weighting will be incorporated.

The inclusion of new cost components and improvements to existing IAMs typically lead to higher cost estimates, is confirmed in a meta-analysis by (Wang et al., 2019).
The researchers put 578 SCC estimates from 58 different studies side by side and find an average SCC of €200.57/tonne CO_2 (this is a simple average of studies using different discount rates; for studies applying a 3% discount rate, the researchers found an average value of €112.86). Richard Tol also found high average SCCs in a meta-analysis: €146 at a low discount rate and as much as €446 to 1,925 at lower discount rates (Tol, 2022). This is a huge increase since an earlier publication by Tol (2008), whose meta-analysis at the time arrived at an average estimate of about €5/tonne CO_2 at a 3% discount rate. A recent and expanded meta-analysis by (Tol, 2024) offers an overview of various methods and highlights emerging disparities.

The latest literature uses macro-econometric damage functions to estimate the impact of climate change by comparing GDP changes with temperature shifts (and sometimes other climate factors like precipitation). There is no consensus on the best method for these top-down approaches. Key debates include whether climate change affects economic growth rates or only levels, the exclusion of long-term adaptation, and the focus on GDP-related damages, ignoring non-GDP impacts like heat-related mortality (central to the GIVE model). Recent SCC estimates include (Ricke et al., 2018) at $417 per tonne CO_2 and (Bilal & Känzig, 2024) at $1,056 per tonne.

Although new studies arrive at higher SCC estimates on average, there is still a lot of variation between different study results. This shows (Yang et al., 2018) that the SCC is highly dependent on the chosen discount rate, global climate policy and the damage function included in the IAM. By varying these three parameters in the same DICE model, SCCs ranging from €0 to 1,200 can be obtained for the year 2100. Findings seem particularly sensitive to the shape of the damage function: when choosing damage functions that incorporate tipping points, a negative SCC (due to a mild temperature rise initially benefiting the global economy) can turn into a very high positive SCC within a relatively short period of time.

Some economists, such as (Wagner, 2021) see the remaining uncertainties regarding the model assumptions as a research agenda and advocate using the latest SCCs (with typical values of 100 to 200 €/t in 2030) in cost-benefit analyses. Others, such as (Stern & Stiglitz, 2021), argue that the SCC does more harm than good because IAMs fail to properly reflect crucial uncertainties even after including a wider variety of cost and aforementioned improvements. Instead of using an SCC, the researchers suggest calculating back how much it costs to achieve a given reduction target. This method is referred to as the abatement cost method and is also followed, for example, by (OECD, 2018) as an alternative to the SCC.

6.3.8 New insights into abatement costs

Several new estimates of climate change abatement costs have also been published since 2016. These studies generally assume more ambitious climate targets (-55% CO_2 emissions in 2030 and climate neutral in 2050) and therefore show higher abatement costs than used in the Environmental Prices Handbook 2018. The abatement costs found largely fall within the price range drawn up by PBL and CPB for the uncertainty forecast from the WLO (the two-degree scenario). In the following, we summarise the main new abatement cost estimates from the literature:

- (IPCC, 2018a) which show several reduction paths compatible with a 1.5 and 2˚C temperature rise by the end of this century. Some of these mitigation pathways assume a *temporary overshoot*: a period between 2050 and 2100 during which temperature rise temporarily exceeds 1.5˚C. Large-scale deployment of negative emission technologies then drops the temperature rise back below the target. The abatement costs found by the IPPC are highly dependent on the abatement path chosen, the extent to which CCS is applied and the extent to which a temporary overshoot is allowed. For example, undiscounted abatement costs under a 2˚C reduction pathway range from 15 to 220 $US\$_{2010}$ per tonne CO_2 in 2030 and from 45 to 1,050 $US\$_{2010}$ per tonne CO_2 in 2050. Estimates for a reduction pathway that is compatible with a 1.5˚C rise in 2100 range from 135 to 6,050 $US\$_{2010}$ in 2030 and 245 to 14,300 $US\$_{2010}$ in 2050. The abatement path that appears to best align with current European policy targets (1.5˚C with a limited overshoot) has a median realistic CO_2 price of 188 $€_{2021}$ in 2030 and 437 $€_{2021}$ in 2050.
- The UK government determined a new set of CO_2 prices in 2021 for use in policy research that are compatible with the UK's tightened climate targets (net-zero by 2050). To this end, the Department for Business, Energy & Industrial Strategy (BEIS) conducted a literature review and created its own abatement cost model (BEIS, 2021). This Global Carbon Finance Model (GloCaF) defines a global emissions trajectory based on a trading model with no transaction costs.
  Each of the 25 regions in the model are assumed to have the same marginal abatement costs. The model output is the most cost-effective CO_2 price at which the reduction target can be met. The efficient CO_2 prices calculated with the GloCaF model translate to 170 $€_{2021}$/tonne CO_2 in 2030 and 665 $€_{2021}$/tonne CO_2 in 2050. BEIS ultimately chose to adopt the aforementioned IPCC median prices from the limited overshoot scenario and not to use their own calculated prices. The reason cited is the independent, international nature of the IPCC and its larger evidence base.
- The French government also launched research in 2018 on new CO_2 prices for use in policy documents (France Stratégie, 2019). These prices are compatible with a net-zero target in 2050. An independent commission led by economist Alain Quinet, based on extensive modelling with the TIMES and POLES models, arrives at abatement costs of 264 $€_{2021}$/tonne CO_2 in 2030 and 819 $€_{2021}$/tonne CO_2 in 2050.
- Goldman Sachs draws up global cost curves for CO_2-reducing measures every year in its Carbonomics report. In recent years, the bank has consistently adjusted its cost curves downwards due to technological innovation (Sachs, 2021). Goldman Sachs expects this trend to continue so that technologies cheaper than 250 €/tonne CO_2 have a combined reduction potential of more than 40 Gigatonnes. Nevertheless, the researchers identify that 8 Gigatonnes of emissions cannot be avoided with current mitigation techniques. To this end, negative emissions as made possible by BECCS and DACS should provide a solution. Goldman Sachs assumes an average price of 290 €/tonne CO_2 for negative emissions. From these assumptions and cost curves, an efficient CO_2 price of around 100 €/tonne CO_2 in 230 and 290 €/tonne CO_2 in 2050.
- In 2021, as part of a study on a so-called Carbon Takeback Obligation, CE Delft again determined efficient CO_2 prices with the MERGE model (CE Delft, 2022a). This model was used by the Planning Agencies in 2015 to determine the efficient prices for the WLO

scenarios. MERGE has a very detailed energy module in which electricity prices can vary over time and across regions. This allows the model to properly incorporate the system effects of variable solar and wind power generation. In the new (CE Delft, 2022a) model runs, global GHG emissions are assumed to fall to net zero by 2050. CE Delft calculates two different scenarios. In the baseline scenario, a maximum contribution to CO_2 reduction by CCS is assumed to be 15%, in line with the contribution of CCS in the Climate Agreement (Rijksoverheid, 2019) and because of the lower-than-expected rate of CCS take-off in Europe. In the second scenario, this limitation is removed. The resulting efficient CO_2 prices in the base case are €90 in 2030, €110 in 2040 and €550 in 2050. Prices in the scenario with extra CCS are lower: €75 in 2030, €90 in 2040 and €200 in 2050.

Comparison of CO_2 prices from the MERGE model

In Table 30 the efficient CO_2 prices from the WLO (Aalbers et al., 2016) are compared with two different variants of (CE Delft, 2022a). All three price paths were calculated using the MERGE model.

Table 30 – CO_2 prices from CE Delft (2022a) compared to WLO prices

	2030	2040	2050
WLO - two-degree study (Aalbers et al., 2016)	€ 100-500		€ 200-1,000
(CE Delft, 2022a)- baseline	€ 90	€ 110	€ 550
(CE Delft, 2022a)- extra CCS	€ 75	€ 90	€ 200

The differences between the WLO values and those of CE Delft can be explained as follows:
- The WLO scenarios assumed 80-95% emission reductions in 2050, while CE Delft assumes zero emissions in 2050. A higher target leads to higher CO_2 prices.
- The WLO scenarios assumed costs for DACS of 1,000 €/tonne CO_2. (CE Delft, 2022a), based on new insights, (WRI, 2022) assumes substantially lower costs of 200 €/tonne CO_2 in 2050.
- In the WLO scenarios, DACS could be added without limitation. (CE Delft, 2022a), however, assumes binding expansion constraints on DACS, which means that the CO_2 price in 2050 could be greater than the marginal cost of DACS.
- In the WLO calculations, no restriction was assumed on carbon storage in Europe, while CE Delft assumes a maximum of 15% emission reduction by CCS in the baseline. This restriction makes the policy more expensive.
- The WLO scenarios assumed a binding annual carbon budget, while (CE Delft, 2022a) assumes the possibility of banking. This makes policies cheaper in 2030 but much more expensive in 2050.

It should be added that certain assumptions in MERGE on learning effects, nuclear power and hydrogen are still uncertain. In the future, it seems useful to make refinements to the model so that the baseline more closely matches the most recent World Energy Outlook of the International Energy Agency (IEA, 2022). It cannot be said in advance whether this leads to higher or lower efficient CO_2 prices.

6.3.9 New environmental prices

As in the previous edition of the Environmental Prices Handbook, we have chosen to base CO_2 prices on the abatement cost method. Although damage costs in the literature now seem to better reflect the actual costs of climate change, uncertainty ranges are still very large, there still appear to be missing effects, and certain crucial assumptions about damage functions and the discount rate used remain in question. In this regard, the difference compared to the abatement cost approach should also not be exaggerated. Most recent studies on the *social cost of carbon* show optimal reduction pathways where CO_2 emissions reach zero by 2050. This fits well with current European climate targets

(-55% in 2030 and climate neutral by 2050). Similarly, countries such as the UK and France use and update efficient CO_2 prices using the abatement cost methodology based on more ambitious climate targets. All in all, this reinforces our choice of the abatement cost method.

For most intended users (except for use in SCBAs where national advice is in force), we base the CO_2 prices on the abatement costs found in national and international literature. We recommend using the aforementioned median prices from (IPCC, 2018b) for the central values, based on the 1.5ºC scenario with a limited overshoot.
IPCC prices fall in the middle of observed values in literature and are supported by a broad base of technical research and modelling. For the lower price path, we recommend applying the efficient CO_2-price values of (CE Delft, 2022a) without restrictions on CCS. These prices seem to reflect a lower end of the possible range, where much of CCS can be applied and DACS can be realised relatively cheaply. Moreover, for the upper values, we recommend using the CO_2 prices of (Stratégie, 2019) because they represent the upper end of the range found. Annual growth rates are derived from 2030 and 2050 prices.[56]

Table 31 - New CO_2 prices, in $€_{2021}$ per tonne CO_2 excluding VAT

	2021	2030	2050	Growth factor per year
For all uses (except for SCBAs where national advices are in force)				
Lower	€ 50	€ 75	€ 200	5.0%
Central	€ 130	€ 188	€ 437	4.3%
Upper	€ 160	€ 264	€ 819	5.8%

6.3.10 Consistency environmental prices with European Green Deal targets

Since the CO_2 prices for the IPCC's 1.5ºC scenario with limited overshoot fall within the range of values observed in the literature and is supported by extensive technical research and modelling, this handbook adopts the CO_2 prices based on that IPCC scenario. To ensure that the CO_2 prices used here reflect Europe's climate objectives, the reduction targets of the European Green Deal and the IPCC's 1.5ºC scenario with limited overshoot must align. A comparison of the 2030 and 2050 reduction targets confirms this alignment: both the European Green Deal and the IPCC scenario aim for net-zero CO_2 emissions by 2050 (European Council, n.d.) (IPCC, 2023). For the 2030 intermediate target, the differences are minimal (see footnote for calculation).[57] Therefore, the CO_2 prices in this handbook are to be considered consistent with the European Green Deal objectives.

[56] For use in Dutch SCBAs, different CO_2-prices need to be used, consistent with the WLO (Aalbers et al., 2016).

[57] The IPCC 1.5ºC scenario (with limited overshoot) sets an intermediate target in 2030 of a 43% reduction in GHG-emissions compared to 2019 levels (IPCC, 2023). The European Green Deal sets an intermediate target in 2030 to reduce net GHG emissions by at least 55% from 1990 levels by 2030 (Council, n.d.).
GHG-emissions in EU27 equalled 4.915 million tonnes CO_2-eq. in 1990, and 3.713 million tonnes CO_2-eq. in 2019 (Commission, 2023).
Allowed GHG-emissions in 2030 according to Green Deal targets equal 4.915*45% = 2.212 tonnes CO_2-eq.
Allowed GHG-emissions in 2030 according to IPCC pathways equal 3.713*57% = 2.116 tonnes CO_2-eq.

6.3.11 Environmental prices and ETS

The EU Emissions Trading System (EU ETS) is one of the key instruments of the EU, which puts a price on carbon emissions and leads to emission reduction by limiting the number of allowances over time. However, this system and the environmental prices discussed above differ fundamentally in their scope. The EU ETS operates within a predefined market, where carbon prices are driven by supply and demand dynamics for emissions allowances. In contrast, environmental prices are intended to reflect the broader societal preference for reducing GHG emissions across *the entire economy*. While the EU ETS focuses on regulating emissions within specific sectors, environmental prices aim to capture the full societal cost of GHG reductions on a larger scale. Ideally, the EU ETS should reflect the environmental price to ensure highest welfare for all its citizens as much as possible.

The EU ETS, established in 2005, is a policy instrument for reducing GHG emissions within the EU (Commission, n.d.). It operates on a cap-and-trade principle, setting an overall limit (or cap) on total emissions from sectors like power generation and heavy industry. Companies under the EU ETS must hold enough emission allowances to cover their annual GHG emissions, and they can buy, sell, or trade these allowances on the ETS carbon market. Over time, the cap is gradually reduced, driving continuous emissions reductions and encouraging investment in climate technologies. Under the revised EU ETS, the emission reduction target has been raised, aiming for a 62% reduction in GHG emissions by 2030 compared to 2005 levels. Additionally, the system's linear reduction factor (LRF), which determines the rate at which the cap decreases, has been strengthened to between 4.3% and 4.4% annually. As a result, the emissions cap is set to reach zero by 2039, effectively phasing out emissions covered by the ETS.

In 2023, the average ETS auction price was €83 per tonne of CO_2-eq. (Partnership, n.d.). Despite a notable increase in the EU ETS price since 2021, it remains well below the established CO_2 price based on the median IPCC 1.5ºC scenario (with limited overshoot). This discrepancy is in part due to the fact that the EU ETS only covers a portion of the economy. Key sectors such as the built environment, road transport, and agriculture are currently not included in the ETS. The lower ETS auction price compared to the 'broader' CO_2 price could reflect that the marginal costs of reducing emissions are lower in sectors covered by the ETS (such as industry) than in those outside the system, where reductions are more costly.[58]

In addition, ETS prices are deemed too low relative to the environmental price because of the oversupply of allowances and decreased demand during the financial crisis and competing policies such as renewable and efficiency targets which are also reducing demand. In line with the strengthened ETS-cap it is anticipated that ETS prices and the environmental CO_2-prices will further converge in time.

6.3.12 Use of environmental prices in a national context

The application of CO_2 prices in national SCBAs or LCAs typically depends on the specific national policy context. Several European countries have proposed national CO_2 prices for policy research purposes. In 2021, the UK government introduced new CO_2 prices aligned with the UK's stricter climate targets (net-zero by 2050), resulting in prevention costs of £280 (€315) for 2030 based on median prices from the IPCC limited overshoot scenario

[58] The European Commission is developing a new system (ETS II) to incorporate more sectors into the ETS framework. This ETS II system is set to fully commence from 2028 onwards (Directive 2023/959).

(BEIS, 2021).[59] Similarly, in 2018, the French government conducted research on new CO_2 prices compatible with a net-zero target by 2050, arriving at prevention costs of €250 for 2030 using the TIMES and POLES models (Stratégie, 2019). In Germany, the national CO_2-price is determined using a damage cost approach, resulting in a CO_2 price of €215 for 2030 (Bünger & Matthey, 2020). The proposed CO_2 prices are calculated with a primary focus on the national context, considering the specific economic, environmental, and social conditions of each country.[60] In contrast, the CO_2-prices presented in this Handbook are average values for the EU27 and per definition not specific to the national context.
It is important to note that this Handbook does not supersede national perspectives. Users should evaluate whether these average values are suitable for specific applications, such as SCBA or LCA.

Table 32 - Examples of European countries (including UK) with a nationally determined CO_2-price

	Netherlands	Germany	France	United Kingdom
Study	CE Delft, Environmental Prices Handbook (2023) CPB and PBL (2015)	Umweltbundesamt (2020)	France Stratégie (2019)	Department for Business, Energy & Industrial Strategy (BEIS)
Method	Prevention costs method, IPCC 1.5°C scenario (limited overshoot) WLO scenarios	Damage costs approach using damage cost model FUND (Anthoff, et al., 2009)	Prevention costs method using TIMES en POLES-models	Prevention costs method, IPCC 1.5°C scenario (limited overshoot)
Publication year	2023	2020	2018	2021
National CO_2 price for 2030	22-550 €$_{2021}$	215-700 €$_{2020}$	250 €$_{2018}$	315 €$_{2020}$ (280 £$_{2020}$)[61]
Link	Handboek_Milieuprijzen, CE Delft, 2023	Methodological convention 3-1 value factors 2020, Umweltbundesamt, 2023	The Value for Climate Action, France Stratégie, 2019	Valuation of greenhouse gas emissions: for policy appraisal and evaluation, Gov.uk, 2021

6.4 Particulate matter formation

6.4.1 Description of theme

Particulate matter (PM) is a form of air pollution consisting of a mixture of individual particles (liquid or solid), with varying compositions and sizes. A gas containing suspended PM is known as an aerosol. All particles that remain suspended in the air are classified as particulate matter (PM). These particles can be classified according to several criteria, the most important of which are:

– **By origin (anthropogenic or natural):** anthropogenic emissions are caused by human activity and include soot and smoke formed in combustion emissions or dust from

[59] In this case, the year 2040 was used as the anchor point, and a discount rate of 1.5% was applied to calculate annual CO_2 prices. This approach differs from the method used in this handbook, leading to different CO_2 prices even though both methods assume the IPCC limited overshoot scenario.

[60] The CO_2 price for Germany and France take into account specific national economic, environmental and social factors. For the UK, the calculated CO_2 price is based solely on IPCC and does not take into account specific national factors.

[61] Used exchange rate British pond to euro: 1.1248. British Pound to Euro Spot Exchange Rates for 2020

building materials. In contrast 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.[62]
- **By size or diameter**: usually with a breakdown into PM_{10}, $PM_{2.5}$ and PM_1, referring to particles with a diameter smaller than 10, 2.5 and 1 µm, respectively. The smaller particles are more damaging. Ultrafine particulate matter refers to particles smaller than 0.1 µm.
- **By chemical composition:** PM comes in hundreds of forms. Although there are indications that PM depends not only on diameter but also on chemical composition, there is as yet insufficient solid evidence except in the case of 'elemental carbon', which appears more hazardous than other forms (see Paragraph 6.4.8).

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 fractions. PM also arises in certain mechanical processes, such as the milling of grain or car tyre wear. The material blown up in these processes as well as arising from construction sites or industrial sites belong primarily to the coarser fractions of PM. Deposited particles can be reintroduced into the air by wind or movement which is common along roads and highways. There are also natural sources of windblown coarse PM, such as wind erosion of soils and atmospheric dispersion of sea salt.

Secondary aerosols are another important source of PM. Emissions of SO_2, NH_3 and NO_x lead to the formation of sulphate and nitrate salts in the atmosphere. The concentration of these secondary aerosols is not necessarily proportional to emissions, because the rate at which they form depends on factors such as wind, temperature, relative humidity and nitrate availability. NH_3 plays a dual role in this process, binding to both SO_2 and NO_x.

6.4.3 Impact

Particulate matter has an impact on human health and damage to buildings and monuments. It also causes visual nuisance.

Health impact

Of all the environmental pollutants to which humans are exposed, it is primary and secondary particulates that cause the greatest health damage, because they transport a wide range of toxic pollutants directly into the 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 direct and indirect 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 anthropogenic particulate emissions than PM_{10} and thus more

[62] These gases are less volatile, so they form downwind aerosols by forming new particles (nucleation) or by attaching to pre-existing particles (coagulation).

amenable to policy action (RIVM, 2015). The literature (WHO, 2013) distinguishes several pathophysiological mechanisms to explain the effects of $PM_{2.5}$ on mortality and morbidity:
1. $PM_{2.5}$ aggravates the severity of COPD (chronic obstructive pulmonary disease) and asthma.[63]
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).
4. $PM_{2.5}$ can lead to lung cancer.

In addition, particulate matter can lead to DNA damage or allergic and/or inflammatory reactions (VVM, 2013c). Due to all these causes, particulate matter contributes to increased mortality and morbidity.

There are several indications that both the size and chemical composition of particulate matter affects toxicity. The ultrafine nature of particles increases the toxicity of particulate matter and partly explains the health effects, see (VVM, 2013c). Recent research has focused particularly on ultrafine particles ($PM_{0.1}$), which are considered more harmful due to their ability to penetrate deeper into the lungs and potentially breach the blood-brain barrier. There is also evidence that particulate matter particles containing certain heavy metals, and ‘black carbon’ cause additional toxicity effects.

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

Other non-health effects

Particulate matter accelerates the weathering of buildings and contributes to visual nuisance (soot deposits). Particulate matter pollutes streets and buildings and requires more frequent cleaning. Also, particulate matter can cause visual nuisance due to reduced visibility.

We are not aware of any literature that provides monetary estimates of the effects of particulate matter emissions on ecosystems or animal life. It is plausible that the effects that occur in humans also occur in animals and particulate matter could thus lead to lower life expectancy or increased infant mortality in animals as well.

6.4.4 Treatment in the Environmental Prices Handbook 2018 and updates

In the Environmental Prices Handbook 2018, the health effects of the theme of particulate matter formation were modelled by adapting the NEEDS Excel tool: incorporating new endpoint price factors, changing population compositions and new updated insights on the harmfulness of emissions. The EU28 version of the Handbook also ensured that the full impact assessment was equivalent to the WHO (2013) recommendations.[64] Additionally, both previous Handbooks included the damage of particulate matter on buildings and materials in the valuation.

[63] Strictly speaking, this was not proven in the study, but that may be due to the fact that COPD patients are usually diagnosed with pneumonia or flu at death.

[64] In the Dutch Handbook, this adaptation has not been fully implemented.

The Environmental Prices Handbook used ReCiPe 2009 (in the 2013 update) which characterises particulate matter in PM_{10} equivalents. Based on emissions and the damage estimate for PM_{coarse}, a characterisation was derived for $PM_{2.5}$ relative to PM_{10}.
The calculation led to the insight that $PM_{2.5}$ per kg is 1.79 times more harmful than PM_{10}.

In the new Environmental Prices Handbook 2024, the following has been updated compared to the 2018 version:
- The damage of particulate matter was estimated (EEA, 2021a) based on the reasoning in Paragraph 0. This has also led to a revised estimation of the harmful effects of secondary aerosols (Paragraph 6.4.5).
- The characterisation factors for calculating midpoint prices have changed (Paragraph 6.4.6).
- The relative risks of particulate matter pollution have been updated, and incidence rates of disease burden have been recalculated (Paragraph 6.4.7).
- An environmental price for black carbon has been added, along with guidelines on when it should be applied (Paragraph 6.4.8).
- The prices for PM_{10} and $PM_{2.5}$ have been adjusted based on new data about the impact of particulate matter emissions on the maintenance and repair of buildings (see Paragraph 5.5).

6.4.5 Update: formation of secondary aerosols

Secondary aerosols are particulate matter particles formed through reactions in the atmosphere, typically from NO_x, SO_2 and NH_3. These secondary particulates can transport over a greater distance than primary particulates because they are often lighter.
A distinction is made between secondary inorganic aerosols (SIA) and secondary organic aerosols (SOA).

Secondary inorganic aerosols (SIA)

There is much debate within the scientific community about whether SIA has the same toxicity as primary particles, with no consensus reached on this matter. The NEEDS project therefore assumed that damage caused by SIA is equal to that from primary aerosols, an assumption that was later also made by (WHO, 2013) and (EEA, 2021a).
We follow this assumption as well.

Secondary organic aerosols (SOA)

Another point of discussion is whether SOA (secondary organic aerosols) have the same toxicity as SIA. In ReCiPe 2016, the harmfulness of SOA is set at 0, which is why NMVOC are not characterised in $PM_{2.5}$ formation. However, (EEA, 2021a) does attribute a damage burden to NMVOC. In line with EEA 2021 methodology, we suggest including the damage costs of SOA due to NMVOC when determining the environmental prices for NMVOC. However, we suggest excluding this value from the calculation of the midpoint prices, because particulate matter formation in ReCiPe does not characterise on NMVOC.

6.4.6 Update: characterisation factor

In ReCiPe 2008, the impact on this theme was expressed in kg PM_{10} equivalents. In ReCiPe 2016 the characterisation factor shifted to $PM_{2.5}$ equivalents. ReCiPe adopts a cautious stance regarding the inclusion of secondary aerosols: in the individualistic worldview, these are not included, while in the hierarchical worldview, the SIA (secondary inorganic aerosols) from SO_2, NH_3 and NO_x are included. The characterisation factors in ReCiPe are (in general) lower than the values calculated by the EEA for secondary particulate matter formation from SO_2, NO_x and NH_3 (EEA, 2021a). Therefore, the *weighted* environmental price for the particulate matter formation midpoint theme will be *higher* than the price for $PM_{2.5}$.

6.4.7 Update: relative risks of particulate matter pollution and incidences of disease burden

In this update of the Environmental Prices Handbook, the entire modelling of particulate matter has been re-examined in line with the Impact Pathway approach. This involved establishing a relationship between the population exposed to particulate matter pollution, the effects it causes and the valuation of these effects. Our modelling is largely based on the results from (EEA, 2021a), but adds its own elements in some parts.

The starting point is the relative risks (RR) of particulate matter formation determined in literature. Below are the effects of particulate matter formation that we calculated in this study:

Table 33 - Relative Risks of impacts included in the determination of the Environmental Prices Handbook 2024

Endpoint (incidences)	Age group	RR per 10 µg (WHO)	Source
$PM_{2.5}$			
Mortality, all natural causes	30+	1.08	(Chen & Hoek, 2020)
Hospital admissions, cardiovascular disease	All	1.0091	(WHO, 2013a)*
Hospital admissions, respiratory organ diseases	All	1.019	(WHO, 2013a)*
Restricted Activity Days (RAD)	All	1.047	(WHO, 2013a)
Work Loss Days (WLD)	20-65	1.046	(WHO, 2013a)*
Days of asthma symptoms among children with asthma	5-19	1.028	(WHO, 2013a)*
PM_{10}			
Post neonatal infant mortality	0-12 months	1.04	(WHO, 2013a)*
Incidences of bronchitis in children	6-12	1.08	(WHO, 2013a)*
Incidences of chronic bronchitis in adults	18+	1.117	(WHO, 2013a)*

* The relative risks are from other studies and are recommended for use in (WHO, 2013a), see Annex A.

Compared to the EEA (2021) study, the following changes have been applied:

- A meta-analysis on the harmfulness of particulate matter was recently published by (Chen & Hoek, 2020). With advancing insights and increasing evidence of the harmful impact of particulate matter, this study presents a higher relative risk of 1.08, compared to the relative risk of 1.062 as reported by WHO in 2013. This analysis incorporated 107 studies examining the mortality associated with PM_{10} and $PM_{2.5}$. To reflect the robustness of this evidence, we have adopted the relative risk of 1.08. It is expected that this value will also be included in the future update of the WHO HRAPIE guidelines. As this RR value is the most decisive factor in determining environmental prices, we consider it prudent to include this effect in the calculations of environmental prices.
- Similar to EEA, we have converted all effects to the impact for $PM_{2.5}$. We relied on (RIVM, 2021) data regarding the ratio of PM_{10} and $PM_{2.5}$ concentrations in the EU. PM_{10} concentration contains a larger component of sea salt, which does not decrease over time. As anthropogenic sources of particulate matter emissions decrease over time, the $PM_{2.5}/PM_{10}$ ratio will decline.
- Incidences for each endpoint for the year 2019 were determined using the sources listed in Annex A. In EEA, the calculation of incidences is based on the WHO framework. In the Environmental Prices Handbook, we determined incidences of disease burden using statistical data as much as possible.

6.4.8 Update on the valuation of black carbon

Black carbon (BC, also referred to as elemental carbon (EC) or soot) impacts health and climate. The health impact was investigated through a systematic literature review (WHO, 2012). (WHO, 2012) indicates that there is sufficient evidence in epidemiological studies linking daily variations in soot concentrations to short-term changes in health. Studies on the short-term health effects also suggest that BC may serve as a more accurate indicator of harmful particulate matter from combustion sources than undifferentiated particles (PM_{10} or $PM_{2.5}$), especially for morbidity. A practical challenge is that it is difficult to distinguish the health effects of BC from those of $PM_{2.5}$. Therefore, one must decide whether to attribute health effects to BC specifically or to $PM_{2.5}$ as a whole. This choice will partly depend on the share of black carbon in $PM_{2.5}$ emissions.

Based on (WHO, 2012), a relative risk for all-cause mortality of 1.05 to 1.06 seems plausible per $\mu g/m^3$ concentration of black carbon. The applied RR for $PM_{2.5}$ emissions in this study is 1.006 per $\mu g/m^3$ – a factor of 10 lower.
In (CE Delft, 2018a), this factor is used as a dividing line:

1. If BC emissions are less than 10% of total $PM_{2.5}$ emissions, the emissions should be valued with $PM_{2.5}$.
2. If BC emissions are more than 10% of total $PM_{2.5}$ emissions, BC emissions should be taken as the starting point for valuation.

We suggest following this rule here as well because it is consistent with the (WHO, 2012) source material. To determine the additional damage from black carbon, we divide the relative risk from (WHO, 2012) by the relative risk used in this Handbook, which is based on (Chen & Hoek, 2020). This yields the insight that in terms of mortality, black carbon is a factor of 7.5 more harmful than $PM_{2.5.}$ It is plausible that the morbidity of black carbon is also higher than for ordinary $PM_{2.5}$, but there is not sufficient evidence for this. Therefore, in the Environmental Prices Handbook, only the additional mortality attributed to black carbon is considered, **for the portion of black carbon that exceeds the 10% threshold within $PM_{2.5}$** This results in a valuation of 603 €/kg black carbon for the central value, with €408/kg for the lower value and 906 €/kg black carbon for the upper value.
Black carbon also has a climate impact because it 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. Although this effect is important for determining the effects of climate change, IPCC believes that the scientific uncertainty is still too high to express these emissions in a Global Warming Potential. Therefore, the warming effect of black carbon is also not included in environmental prices.

6.4.9 Harmfulness of ultrafine particulate matter

Ultrafine particulate matter is formed by particulate matter particles smaller than 0.1 µm in diameter. Because these particles are even smaller than particulate matter, they can penetrate deeper into the human body and presumably cause more damage than larger particles. In this way ultrafine particulate matter can cross the blood-brain barrier (Gezondheidsraad, 2021). Although there is yet insufficient solid evidence of a higher mortality rate from $PM_{0.1}$ in addition to the mortality from $PM_{2.5}$, there are indications of several health effects caused by ultrafine particulate matter.

In 2022, the National Institute for Public Health and the Environment (Rijksinstituut voor Volksgezondheid en Milieu, RIVM) conducted a major study on the health effects of ultrafine particulate matter from air traffic (RIVM, 2022). This study identified both health effects and potential health effects of ultrafine particulate matter. Different health effects were analysed in four different studies. Each effect found was assessed for certainty and robustness. Relative risks are based on (RIVM, 2022). An overview of these risks in relation to age groups is given in Table 344. Only those effects that do not overlap with the health effects included in the price of $PM_{2.5}$ are given. In this handbook, we have not been able to include an environmental price for $PM_{0.1}$ at EU level. In Annex A.4 we describe how we calculated the value for the Netherlands specifically.

Table 34 - Relative Risks for health effects of ultrafine particulate matter

Health effect	Age group	Relative Risk (per 3,500 particles/cm^3)
High blood pressure (medication use)	19+	1.05 (1.00-1.11)
Diabetes (medication use)	19+	1.08 (1.00-1.17)
Diabetes (self-reported)	19+	1.16 (1.02-1.33)
Medication for dementia	40+	1.141 (1.013-1.286)

Source: (RIVM, 2022).

6.4.10 Environmental prices

The impact of particulate matter formation on the endpoints are based on an adjustment of the effects for $PM_{2.5}$, NO_x, NH_3 and SO_2 from the EEA project, with the adjustments mentioned above and the valuations discussed in Chapter 5.

Table 355 gives the average values for the EU27 for the pollutants characterising on this theme. For secondary aerosols, only the environmental prices related to health impacts have been considered, whereas for $PM_{2.5}$ and PM_{10}, the price also includes damage to buildings and materials.[65] No additional pollutants beyond those listed in the table characterise on this midpoint.

Table 35 - Average damage costs for emissions in EU27 from an average emission source in 2015, in €$_{2021}$/kg on the theme particulate matter formation

	Lower	Central	Upper
$PM_{2.5}$^	€ 58.1	€ 94.0	€ 131.2
PM_{10}^	€ 30.5	€ 49.3	€ 68.8
SO_2	€ 16.3	€ 26.3	€ 36.8
NO_x	€ 7.6	€ 12.3	€ 17.2
NH_3	€ 14.4	€ 23.3	€ 32.5
NVMOC^^	€ 1.3	€ 2.2	€ 3.0
Black carbon (> 10% $PM_{2.5}$)*	€ 58.1	€ 94.0	€ 131.2
Midpoint price $PM_{2.5}$-eq.	€ 61.7	€ 99.2	€ 138.2

* For the fraction of black carbon greater than 10% of $PM_{2.5}$.

^ The harmfulness of PM_{10} is factored into the harmfulness of $PM_{2.5}$ and vice versa. Both damage estimates can therefore never be included at the same time.

^^ Harmfulness of secondary organic aerosols from NCSRC are measured at the topic of oxidant formation.

[65] The valuation for secondary aerosols has been added to the acidification theme.

Several key points stand out:

- The damage costs are higher for $PM_{2.5}$ and SO_2 than in Environmental Prices Handbook 2018. The damage costs for $PM_{2.5}$ have increased mainly due to updated concentration-response functions and higher estimates of mortality caused by $PM_{2.5}$ compared to 2018. Similarly, the valuation of mortality for adults and especially infants has increased compared to 2018.

- The higher damage costs for SO_2 are due to the update in the modelling of secondary particulate matter formation from SO_2. NH_3, NO_x and SO_2 react to form particulate matter, where the formation is linear with SO_2 but quadratic with NH_3. As an example: for the Netherlands, emissions of these three pollutants decreased between 2010 and 2020, but this decrease is especially greater for SO_2 and to a lesser extent NO_x. Because NH_3 emissions declined to a lesser extent, there is proportionately more NH_3 in the atmosphere for SO_2 to react with. This is the primary reason why a decrease in NO_x and SO_2 emissions, without a corresponding greater reduction in NH_3, leads to higher damage costs per kg of emissions for these pollutants. Unless NH_3 emissions are controlled, this situation will persist in the Netherlands.

- The damage costs for NO_x and NVMOC are slightly higher and slightly lower, respectively, than those in the Environmental Prices Handbook 2018. In this Handbook, the harmful effects of these pollutants are separated (for the first time) into endpoints for particulate matter formation and oxidant formation, resulting in a slightly different allocation than the Handbook Environmental Prices 2018.

Based on the weighting with emissions on this theme, a midpoint price was determined for the characterisation factor $PM_{2.5}$ equivalent. This amounts to 99.2 €/kg $PM_{2.5}$ equivalent. The damage costs for the midpoint characterisation factor are substantially higher than in the previous Handbook. This increase is partly due to the harmfulness of secondary aerosols, which is now considered greater than in the previous Handbook and higher than the corresponding characterisation factor from ReCiPe. As a result, the midpoint price has increased. Additionally, $PM_{2.5}$ emissions have decreased, which means that in the weighted midpoint prices, emissions from $PM_{2.5}$ are less likely to contribute to the valuation of this theme. In the previous handbook, PM_{10} was also included in the calculation of the midpoint characterisation factor. Because $PM_{2.5}$ and PM_{10} partially overlap in that Handbook, this resulted in the midpoint price being determined more heavily by primary aerosols than by secondary aerosols. This is avoided in this Handbook by not assigning damage costs to the portion of PM_{10} that is not $PM_{2.5}$. As a consequence, either PM_{10} or $PM_{2.5}$ is chosen in this Handbook and the two pollutants are not considered simultaneously when determining the midpoint price.

6.4.11 Specific values for emission sources at lower altitudes and population density

The harmfulness of particulate matter emissions is highly dependent on how many particles enter people's lungs. This, in turn, depends heavily on the height of the emission source and the population density in the area where the emission takes place.

The Environmental Prices Handbook 2018 estimated the harmfulness of traffic emissions by using the differentiation between various sources described in the (HEATCO, 2006) project. This showed that the harmfulness of particulate matter from traffic in highly urbanised areas is 6-7 times higher than the national averages used in the Environmental Prices Handbook. In rural areas, this factor was 1.6.

For the Clean Air Agreement, (CE Delft, 2021a) re-examined the possibility of differentiating damage costs for particulate matter by emission level and population density. This study used the research of (Humbert, et al., 2011), which calculates average intake fractions for three emission heights (~100 m, ~25 m and at ground level) and three categories of population density (urban, rural and remote). These intake fractions indicate how many particles of an emission are, on average, absorbed by humans. By comparing the intake fraction at different emission sites with the average intake fraction, we arrive at a factor applied to the central value of the environmental price. This results in an average environmental price for each emission height and location.

We differentiate damage costs only for human health and to buildings, as these are linked to the presence of people and infrastructure. For population density, we use only the urban and rural categories, given that the thin population density in 'remote' areas rarely occurs in the EU. Damage to buildings is only differentiated by population density, as it is not likely that the emission height is related to the height of buildings in general. Other impacts are not differentiated because they are not related to the presence of people.

The results are shown in Table 366. For $PM_{2.5}$ and PM_{10}, prices are differentiated both by the stack height of the emission and area type. The price for the pollutants SO_2, NO_x and NH_3 are differentiated by area type only, as there is too much uncertainty about the dispersion at different altitudes in terms of the intake fractions.

Table 36 - Environmental prices differentiated by emission level (for particulate matter emissions) and population density, €$_{2021}$/kg for the central value

Pollutant	Stack height	Type of area (population density)		
		Urban	Rural	Average
$PM_{2.5}$	High	€ 59	€ 52	€ 49
	Low	€ 76	€ 62	€ 64
	Ambient	€ 217	€ 116	€ 182
	Average	€ 114	€ 75	€ 95
PM_{10}	High	€ 37	€ 26	€ 31
	Low	€ 52	€ 33	€ 41
	Ambient	€ 142	€ 80	€ 108
	Average	€ 66	€ 40	€ 52
SO_2	Average	€ 34	€ 28	€ 31
NO_x	Average	€ 23	€ 21	€ 22
NH_3	Average	€ 29	€ 29	€ 29

6.5 Smog formation (photochemical oxidant formation)

6.5.1 Description of theme

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

Ozone (O_3) is the most representative as well as the most important component of photochemical smog. O_3 is a strong oxidising 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 directly 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.[66] Ozone itself is 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 continually being disrupted, however: on balance, more NO is converted to NO_2, leading to rising O_3 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 a reduction in NO_x does not always necessarily mean that ozone levels fall. Especially if NO emissions are relatively high, an increase in NO_x emissions may even induce a drop in the O_3 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 plants 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 Impact

Elevated tropospheric ozone levels, and particularly the peak concentrations that 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 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.

[66] Emissions of CO and CH_4 are especially important for ozone background concentrations due to their longer transport distances.

In epidemiological studies, impacts have generally been quantified above an ozone threshold of 35 ppb or 70 µg/m^3 (known as SOM035). At higher concentrations there is considered to be a risk of heart failure during physical exercise. There is also a probability of worsening respiratory problems and hospitalisation.

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 antioxidants (vitamins C and E) and ethylene (a plant hormone). This disrupts normal cellular processes, which can cause crops to die or mature too early or lose their leaves too quickly (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 barely being positively affected.

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

6.5.4 Treatment in Environmental Prices Handbook 2018 and review of updates

The effects of pollutants causing photochemical smog have been calculated in the Environmental Prices Handbook using (NEEDS, 2008a) models. Both the impact on health and on crops were included. In addition, the upper value also includes the harmful effects of ozone formation on materials.

A separate point of discussion was the harmfulness of NO_2 on mortality and morbidity. In the Environmental Prices Handbook 2018, this was included for the first time under the theme of oxidant formation, but not factored into the midpoint characterisation factor. In practice, this approach proved ineffective.

For the update in the Environmental Prices Handbook 2024, the following changes were made:
1. The adverse effects of ground-level ozone formation have been determined for both human health and crops (EEA, 2021). For human health, this involved conversions by assuming European data on incidences and valuations (see Paragraph 6.5.5).
2. Mortality of NO_2 was redetermined and added to a new midpoint: nitrogen (see Paragraph 6.5.6).
3. Midpoint-level environmental prices are ultimately based on an adjustment of ReCiPe 2016 characterisation factors for the EU (see Paragraph 6.5.7).

6.5.5 Update: harmfulness O_3

(WHO, 2013a) has proposed new relative risks that were also adopted in the study of (EEA, 2021). We have made no changes to that, except a conversion into a functional unit.[67] Table 377 provides an overview of the relative risks used in this study.

[67] The relative risks have been translated into the indicator SOMO35. SOMO35 is defined as the annual sum of the daily maximum 8-hour running average of more than 35 ppb. For each day, the maximum of the running 8-hour average for O3 is chosen and values above 35 ppb are summed over the whole year.

Table 37 - Relative Risks included for pollution with O_3

Endpoint	Age group	RR per 10 µg
Mortality, all natural causes	All	1.0029
Hospital admissions, cardiovascular disease	65+	1.0089
Hospital admissions, respiratory organ diseases	65+	1.0044
Minor Restricted Activity Days (MRAD)	All	1.0154

In this Handbook, there is no longer a distinction between chronic and acute mortality due to O_3, as all mortality has been valued with an RR of 1.0029. Mortality due to O_3 involves elderly people relatively more often than from $PM_{2.5}$. Therefore, the methodology of lifetables cannot readily be applied to all-cause mortality due to O_3. In this study, we assume that the reduction in life expectancy due to mortality from O_3 is half that of $PM_{2.5}$.[68] Research on air pollution in European cities (CE Delft, 2020a) already showed that ozone formation generally has a small damage burden compared to $PM_{2.5}$ and NO_2.

Ozone also damages agricultural crops and forestry. (EEA, 2021) modelled this and provided damage estimates. We have taken the values from (EEA, 2021), adjusted them to the 2021 price and value level and applied them to the midpoint 'oxidant formation – ecosystems' (see Paragraph 6.5.8). It should be kept in mind that these prices may be an underestimate because they do not include other damage to ecosystems, such as biodiversity loss.

6.5.6 Update: mortality from NO_2

The previous Environmental Prices Handbook 2018 included an estimate for mortality from NO_2 for the first time. 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). Similarly, regarding the COVID pandemic, NO_2 was also suspected of worsening the mortality of infection morbidity (Copat, et al., 2020). Nitrogen dioxide exposure can also trigger irreversible effects on lung and respiratory functions, especially in those already suffering from COPD and similar disorders, and also contribute to cardiovascular disease, leading to premature mortality.

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, 2007). The WHO REVIHAAP project (WHO, 2013b) indicated that an increasing number of recent studies have identified both short-term and long-term associations between NO_2 exposure and mortality and morbidity, which are additional to the effects of NO_2 on particulate matter formation or NO_2 on mortality from photochemical oxidant formation (ozone). This points to a third category of adverse effects of NO_2 that operates separately from particulate matter formation or ozone formation.

The WHO HRAPIE projects >WHO, 2013 #4492< and (WHO, 2014) recommended including the long-term effects on mortality (all-cause and cardiovascular) of NO_2 and advise adopting a linear CRF of NO_2 for all-cause mortality of RR 1.055 per 10 µg/m^3 for concentrations above the annual limit value of 20 µg/m^3. In this context the WHO 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%. In addition, the WHO notes that when using this RR value in multi-emission studies, there

[68] This assumption is different from EEA, where the loss of life years due to premature mortality from O_3 is assumed to be one.

may be double counting with the effect of NO_2 on particulate matter formation. According to the WHO, this double counting may be as high as 33%.

Double counting for NO_2 with $PM_{2.5}$ is determined in differing ways in the Dutch Environmental Prices Handbook (CE Delft, 2017a) and the European Environmental Prices Handbook (CE Delft, 2018a).

Environmental Prices Handbook 2018

The COMEAP study, which looked in more detail at the mortality of NO_2 and the potential overlap with $PM_{2.5,}$ was published in 2018. In (2020b), CE Delft concluded that the majority approach proposed by COMEAP for NO_2 mortality without the confounding effect of $PM_{2.5}$, was in line with the European values calculated by (CE Delft, 2017a) and (CE Delft, 2018a). The calculated additional damage of NO_2 was added to the theme of photochemical oxidant formation in Environmental Prices Handbook 2018, because the effects are very similar to effects also counted as a result of the occurrence of O_3 at ground level.

Environmental Prices Handbook 2024

In this Environmental Prices Handbook 2024, we redetermined NO_2 mortality using the (COMEAP, 2018) study. We propose that this damage should no longer be added to the theme of photochemical oxidant formation, but that a new routine should be made for a new endpoint in LCA: Nitrogen. Both points are elaborated below.

The WHO proposed an RR of 1.055 for NO_2 for people older than 30 years for NO_2 concentrations with an annual mean above 20 µg/m^3, which overlaps with the effects for $PM_{2.5}$ in studies considering both the adverse effects of PM and NO_2. In (COMEAP, 2018) it was further investigated whether this overlap could be quantified. The authors write: *"We explored several approaches to account for possible confounding of the NO_2 mortality associations by associations of mortality with $PM_{2.5}$. However, we concluded that none of these potential approaches was appropriate and we have decided against formally deriving an NO_2 coefficient adjusted for effects associated with $PM_{2.5}$. Instead we have applied our judgement, informed by the available evidence, to propose a reduced coefficient which may be used to quantify the mortality benefits of reductions in concentrations of NO_2 alone, where this is necessary."*

The COMEAP study highlights that there could not be a uniform decision on how to quantify this confounding effect, but a majority of the study committee felt that one could consider using RR for all-cause mortality of 1.006 to 1.013 per 10 µg/m^3 NO_2 to estimate effects attributable to NO_2 alone, without thresholds or age groups. For the Environmental Prices Handbook, we assume the average in this range, yielding (rounded) an RR of 1.01 per 10 µg/m^3 for all age groups. This value is slightly higher than the one in (EEA, 2021), which used an RR of 1.008 to be on the safe side. In addition to mortality effects, NO_2 also affects the disease burden. These are the same in our study and that of (EEA, 2021). Table 388 gives the RRs recorded for NO_2.

Table 38 - Relative Risks included for pollution with NO_2 (source: WHO, 2013a)

Endpoint	Age group	RR per 10 µg
New cases of bronchitis symptoms in children with asthma	5-14	1.021
Mortality, all natural causes (short-term)	All	1.0027
Hospital admissions, respiratory organ diseases	All	1.018
Mortality, all natural causes (long-term)	All	1.01*

* Relative risk determined on the basis of (COMEAP, 2018).

6.5.7 Update: characterisation factors

ReCiPe 2016 distinguishes the impact of smog on human health and ecosystems (on land) and expresses impacts on this theme in kg NO_x equivalents. There is no difference in characterisation factors for the different perspectives because only short-lived pollutants affect ozone formation.

Compared to ReCiPe 2008, a number of changes have been made and the information is based on (Van Zelm, et al., 2016). For instance, the 2008 characterisation factors were based on European averages and have been replaced by global averages.

Table 39 - Characterisation factors of oxidant formation (hierarchical perspective), ozone formation potential in kg NO_x-eq./kg for human health and ecosystem damage

Pollutant	Human health	Ecosystems
	World	World
NO_x	1	1
NMVOC	0.18	0.29

6.5.8 New environmental prices

The effects of photochemical oxidant formation on the endpoints are based on the estimates from (EEA, 2021). For ecosystem effects, we adopted the results from (EEA, 2021) directly. For human health, we made an adjustment in line with the paragraphs described above. The valuation was also adjusted to the valuation framework, as described in Paragraph 5.3. Effects on materials, such as rubber, are only included in the upper value, as explained in Paragraph 5.4.

The environmental prices are only determined directly for NMVOC and NO_x. All other environmental prices are derived from characterisation from ReCiPe 2016. ReCiPe 2016 no longer characterises CH_4 and CO for their contribution to oxidant formation. Although these pollutants had a very low environmental price on oxidant formation in the previous Handbook, due to the large amount of emissions, this still played a role in analyses using environmental prices. Presumably, ReCiPe concluded that since these pollutants lead to very low characterisation factors, that this is not significantly different from 0. Since the harmfulness of both pollutants did play a role in the Handbook's use practice, we calculated their value by assuming the relationship between CO/CH_4 and NMVOC from the previous ReCiPe and multiplying it by the world characterisation factor for NMVOC from ReCiPe 2016 and the price for NMVOC in the new Handbook.[69] It should be noted that this conversion was

[69] When the valuation for CO and CH_4 are included in the determination of the midpoint price, this leads to an approximately 2% higher price for the EU27 than if CO and CH_4 are not included because they no longer characterise the oxidant formation theme.

only performed for the human health theme.[70] This gives environmental prices reflected in Table 40.

Table 40 - Environmental prices of emissions on the topic of oxidant formation including mortality, in €$_{2021}$/kg

Pollutant	Lower	Central	Upper
NO_x	€ 4.77	€ 6.85	€ 10.7
NMVOC	€ 1.62	€ 2.49	€ 3.53
CO	€ 0.0122	€ 0.0193	€ 0.0265
CH_4	€ 0.0027	€ 0.0043	€ 0.0059
Formaldehyde*	€ 0.34	€ 0.47	€ 0.64

* Including the harmful effects of nitrogen dioxides.

** Determined via valuation of the characterisation factor.

^ Only effects on human health.

Compared to the Environmental Prices Handbook 2018, the prices of most pollutants are lower with the exception of the price of NMVOC itself, which is about 25% higher. The lower price for NO_x is due to the fact that the Environmental Prices Handbook 2024 assumes the (COMEAP, 2018) study to avoid double-counting between NO_2 and PM effects and this leads to a lower estimate of NO_2 harmfulness.

For the price for the midpoint characterisation factors, we used the subdivision in ReCiPe 2016 into oxidant formation human health and oxidant formation ecosystems.[71] To this we add a separate midpoint: human health due to nitrogen dioxide. LCA users are encouraged to include the effects on NO_x (or better: NO_2) separately in their analyses and multiply these values by the midpoint prices used here. Since the other pollutants have a zero value at this new midpoint, the two midpoint prices cannot be added together.

Table 41 – Prices for midpoint characterisation factors on the topic of oxidant formation and nitrogen dioxides, in €$_{2021}$/unit

Midpoint characterisation factor	Area	Indicator	Lower	Central	Upper
Oxidant formation – health*	EU27	Kg NO_x-eq.	€ 1.38	€ 2.17	€ 2.98
Oxidant formation – health*	NL	Kg NO_x-eq.	€ 0.99	€ 1.70	€ 2.21
Oxidant formation - ecosystems	EU27	Kg NO_x-eq.	€ 0.416	€ 0.416	€ 0.526
Oxidant formation - ecosystems	NL	Kg NO_x-eq.	€ 0.043	€ 0.043	€ 0.153
Nitrogen dioxides - health	NL	Kg NO_x or NO_2	€ 4.31	€ 6.37	€ 9.62
Nitrogen dioxides - health	EU27	Kg NO_x or NO_2	€ 6.30	€ 9.32	€ 14.08

* Valuation for health in the upper variant includes valuation of the impact on buildings.

70 SO_2 also produced effects on oxidant formation in the previous ReCiPe. However, this is not included because the harmfulness of SO_2 on human health is already amply included in the midpoint particulate matter formation and there could potentially be double counting if we were to add a damage cost at oxidant formation here again.

71 The effects on buildings and materials are thereby attached to the ecosystem theme.

6.6 Eutrophication

6.6.1 Description of theme

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. Eutrophication is also known as overfertilisation. 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.

In the methodology of ReCiPe, there is no theme related to eutrophication through emissions of eutrophication pollutants into the air. Since the dose-effect relationships of airborne eutrophication pollutants emitted to soils are the same as for the acidification theme, we have included them there (see Paragraph 6.7). This chapter covers eutrophication of water and direct stress on soil.

6.6.2 Sources

In the EU, agriculture is the largest source of eutrophying emissions due to fertiliser application and livestock manure. In addition, wastewater discharges and sludge dumping cause eutrophying emissions to soil and water.

6.6.3 Impact

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

6.6.4 Characterisation indicator

Eutrophication is determined in ReCiPe 2016 for eutrophication of freshwater due to phosphorus and phosphate emissions and for eutrophication of saltwater due to nitrogen emissions. Characterisation factors are derived for emissions on agricultural land: emissions in freshwater and emissions in saltwater. For freshwater, emissions are expressed in kg P-eq. (phosphorus). For saltwater, the characterisation factor is expressed kg N-eq. (nitrogen). Both P and N are fertilisers.

6.6.5 Treatment in the Environmental Prices Handbook 2018 and updates

In the Environmental Prices Handbook 2018, environmental prices for N and P on the topic of eutrophication are based on the levy level for discharges of N and P to surface waters. A rate of €37.28 per pollution unit of oxygen-binding pollutants, or veO, applies in the Netherlands. One pollution unit represents the annual consumption of 54.8 kg of oxygen. For phosphorus, the discharge of 20 kg of phosphorus corresponds to 1 pollution unit. The shadow price for phosphorus was therefore set equal to €1.86 per kg phosphorus for emissions to water. This estimate matched well with the valuation that followed from an alternative method in which the adverse effect of eutrophication on species richness was directly quantified based on the endpoint-level valuation from ReCiPe 2008.

Based on the levy cost method mentioned above, the shadow price for 1 kg N was estimated at €3.11. This was used in the previous Handbook as an estimate for the environmental prices of nitrates on surface waters and corresponded to the ReCiPe midpoint characterisation factor of 1 kg Nitrogen total discharged to non-specific site. If the nitrogen was discharged directly into the ocean, a 43% higher environmental price was recommended.

In this version of the Handbook, the valuation of externalities of fertilisers has been redefined using a literature review of stated and revealed preference studies. We also show how the resulting prices compare with market prices of animal rights (phosphate rights, poultry rights and pig rights) and levy rates. We do this first for the environmental prices of phosphorus (Paragraph 6.6.6) and then for the environmental prices of nitrogen (Paragraph 6.6.7). When a valuation approach is based on the Dutch context, it will be clearly specified. For these valuations, we recommend how they may be adapted to country-specific valuations.

6.6.6 Study of new environmental prices of phosphates

Since the publication of the previous Handbook, levy levels per pollution unit in the Netherlands have remained unchanged. The current levy translates to a levy of 1.86 €/kg phosphorus for emissions to water. Because the levy level has not been adjusted for inflation and increasing scarcity of natural capital since the Water Act came into force in 2009 (see Paragraph 5.4), the current levy level probably underestimates the actual damage costs. We therefore adjust the levy rate for inflation and with a constant price increase of 1% per year. This gives a cost estimate of €2.56/kg phosphorus. This estimate is specific to the Dutch context. Other countries, such as Germany and Denmark, also have levy systems for nitrogen and phosphorus discharges, generally calculated based on the volume and type of pollutants discharged. These countries could determine a cost estimate based on national levy systems.

For the Netherlands, the costs based on levy rates, even after adjusting for inflation are lower than those found in (Hansen, et al., 2009) based on research on the relationship between house prices and water quality in southern Norway. Excess phosphorus emissions lead to freshwater turbidity. (Hansen, et al., 2009) find a negative effect of turbidity on the sales value of nearby houses. Based on a revealed preference study, the researchers find external costs of €4.19/kg P for phosphorus emissions to soil (2009 price level). (IEEP et al. , 2021) translate these costs to a European average of €0.90 (price level 2021).[72] This average is lower because Norway has more visible lakes than other European countries. The latter costs still need to be adjusted for the increasing scarcity of irreplaceable nature, at 1% per year. This leads to a European average price of €1.01/kg for phosphorus emissions to soil. Because this estimate reflects the cost of emissions to soil and because ReCiPe assumes that only a tenth of phosphorus leaches to freshwater, this method leads to an environmental price of €10.13/kg P to water.

Finally, we can estimate the damage costs of phosphorus emissions based on valuation of biodiversity loss. For this, we use the species.year valuation for freshwater from Paragraph 5.4. Using the midpoint to endpoint characterisation factor for P, this valuation can be converted to an environmental price of €3.74/kg P to water for the EU.

As none of these three data points has a clearly superior method, we present three environmental prices for emissions of phosphorus to freshwater: low, central and high. These three prices are based on the levy cost method, the biodiversity method and the

[72] Received via personal communication.

house price method, respectively. Table 42 summarises the results, both for P-total and phosphate (PO_4). Although the lower price is specific to the Dutch levy costs, we recommend these valuations in the EU context. The lower values may be adjusted by other countries based on their national context.

Table 42 – Environmental prices of emissions of phosphorus (total) and phosphate to freshwater, in €$_{2021}$/kg

	Lower	Central	Upper
P-total	€ 2.56	€ 3.74	€ 10.13
Phosphate (PO_4)	€ 0.84	€ 1.23	€ 3.34

6.6.7 Study of new environmental prices of nitrogen

We also considered three sources for the new environmental prices of nitrogen on the theme of eutrophication: the level of the environmental levy for nitrogen discharges and two academic studies using stated and revealed preference methods, respectively.

As with phosphorus, the Dutch levy levels for nitrogen discharges remained unchanged: 3.11 €/kg reactive nitrogen for emissions to water. Correcting these prices for inflation, and increasing scarcity of natural capital, we arrive at damage costs of €4.28/kg N. This damage cost is specific for the Netherlands. For the EU context, there are currently no sources available that provide specific costs for the EU. Therefore, we recommend applying the same price to the EU context.

Note that discharges of nitrogenous substances mainly occur to freshwater, while the environmental price for emissions of N is described in terms of emissions to seawater. Because not all N discharged to freshwater will also reach seawater, the damage costs are higher when discharging directly to seawater. Applying the mid-characterisation character to the concerning characterisation factor for emissions of N to freshwater from ReCiPe results in a cost estimate of 3.33 * €4.28 = €14.25 per kg N to seawater. These damage costs are reasonably close to the damage costs found in Annex 3 of (IEEP et al. , 2021). Based on the housing cost method discussed earlier, this time applied to coastal regions, the researchers found a European average valuation of 7.64 €/kg surplus N, for nitrogen emissions to saltwater (baseline year 2021). Lastly, (Grinsven, et al., 2013) assume damage costs of 5-20 €/kg N. The upper end of this range is based on studies by (Söderqvist, T & Hasselström, L, 2008) and (Gren et al., 2008).[73]
The lower limit was set by the researchers - somewhat arbitrarily - at 25% of the upper limit. We therefore choose to use only the upper limit and correct it for inflation and the increasing scarcity of irreplaceable nature. This leads to an estimate of 27.60 €/kg N to seawater. Table 43 provides an overview of the new low, central and high prices that can be applied in a European context.

Table 43 - New environmental prices of nitrogen emissions to freshwater and saltwater, in €$_{2021}$/kg

Pollutant	Discharge compartment	Lower	Central	Upper
N-total	Freshwater	€ 2.27	€ 4.23	€ 8.19
N-total	Saltwater (seawater)	€ 7.6	€ 14.3	€ 27.6

[73] A study re-examining the damage costs is currently taking place as part of the UNEP-UKCEH/GEF project ‘Towards the establishment of an international nitrogen management system’ (communication Hans van Grinsven). As the results of this study are not available yet, we cannot include them when determining environmental prices.

6.7 Acidification and eutrophication of soils

6.7.1 Description of theme

Acidification refers to the collective impact of airborne pollutants that are converted via the atmosphere into sulphuric and nitric acid and deposited on soil and vegetation by means of wet or dry deposition. The effects on soil determine whether a pollutant leads to acidification, rather than the chemical properties of the pollutant itself. For example, ammonia is alkaline rather than acidic. After deposition, it is converted by bacteria into nitrate (nitrification) or taken up by plants as ammonium. Both processes are acidifying. This is why ammonia is also included under the acidification theme.

Some of the nitrogen that is deposited on the soil is not converted into acidifying substances but remains available as fertiliser. Eutrophication of soil, similar to acidification, leads to a decrease in species richness and thus a reduction in biodiversity. In practice, both effects are difficult to distinguish from each other (Stevens, et al., 2010). This is why they are combined in this Handbook. Acidifying and fertilising substances have a long atmospheric residence time and can consequently be transported over long distances. This is particularly true for 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. NH_3, on the other hand, disappears from the atmosphere faster through dry deposition near the emission source or conversion to secondary aerosols (VMM, 2013a).

6.7.2 Sources

Anthropogenic activities such as agriculture (animal husbandry) and the use of fossil energy sources cause potentially acidifying emissions. There are also natural sources. Volcanic eruptions, for example, emit large quantities of sulphur dioxide that can be transported over long distances.

Fertilisation arises mainly from agricultural activities and to a lesser extent from burning biomass and fossil fuels (see also Paragraph 6.6).

6.7.3 Impact

Acidification impacts ecosystems and buildings. Fertilisation primarily has effects on ecosystems. In addition, NH_3 can cause odour nuisance (inconvenience) and can lead to health damage in very high concentrations. Moreover, all acidifying and acidifying pollutants also result in health damage through the formation of secondary aerosols. These are included in the theme of particulate matter formation.

Ecosystems

Soil starts to acidify when its acid-buffering capacity is exceeded. Soil acidification results from both anthropogenic and natural processes. Natural soil acidification can occur when there is a surplus of precipitation. This surplus drains away into the soil, carrying dissolved acid- buffering substances such as potassium, calcium and magnesium down into deeper layers. Emissions of 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. The disappearance of deep earthworm species also reduces humus mixing with the mineral soil and 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.

Emissions of NH_3 and NO_x also lead to eutrophication. On the one hand, eutrophication has a positive effect on ecosystems by increasing the productivity of food crops, for example. On the other hand, eutrophication leads to impoverishment of nature and loss of biodiversity. The damage from this second effect is considerably higher than the benefits from the first effect.

Buildings

Acidifying emissions can lead to accelerated erosion of buildings and monuments, particularly those made of limestone and other calcium-rich stone or concrete. This is described in Paragraph 5.5.

Health

Health effects arise primarily through the formation of secondary aerosols which have been addressed in Paragraph 6.4, and oxidant formation which has been addressed in Paragraph 6.5. There is some evidence that SO_2 and NH_3 in particular can lead to additional damage. Sulphur dioxide acts on the mucus membranes in the mouth, nose and lungs. Its main impact is on respiratory functions. This is because sulphur dioxide is converted into sulphuric acid in the airways in contact with water, causing narrowing of the airways, leading to bronchitis and even increased mortality with chronic exposure. However, given the low concentrations that still occur today, sulphur dioxide is unlikely to play a significant role (VMM, 2013a).

Ammonia can also cause respiratory effects, but this will only occur at relatively high concentrations that are likely to be confined to ‘workplace’situations at e.g. intensive livestock farms. Since the Environmental Prices Handbook gives prices for an average concentration in EU27, it cannot be used in such cases.

6.7.4 Treatment in the Environmental Prices Handbook 2018 and updates

Environmental prices for acidification/eutrophication in the Environmental Prices Handbook 2018 were estimated using the NEEDS model.

(EEA, 2021) also gave an assessment of biodiversity loss due to eutrophication. Acidification was not considered here. The valuation of biodiversity is based on a WTP study that looked at Willingness-To-Pay for nature conservation areas: therefore, (EEA, 2021) only modelled exceedance of critical deposition values in Natura 2000 sites.

In the NEEDS 2008 project, valuation was also primarily based on the valuation of Natura 2000 sites but was converted into a valuation for species richness that could then be declared applicable to any site. An appreciation of species richness loss can in principle also be obtained through ReCiPe 2016 via the endpoint characterisation factors. The following table shows the results via the different approaches for the pollutants involved in acidification for the EU.

Table 44 - Damage costs per kg of emissions from Dutch territory according to three studies, converted to €$_{2021}$/kg

	NO_x	NH_3	SO_2
(EEA, 2021b)*	€0.09	€ 0.26	€ 0
(NEEDS, 2008a)**	€1.43	€ 5.44	€ 0.32
ReCiPe 2016 endpoint valuation (Huijbregts, et al., 2016)***	€2.30	€ 12.51	€ 6.38

* This is a weighted average based on the values of the 27 EU member states, weighted with emission data.

** Based on the MetAv_Sall scenario - year 2020.

*** Based on global characterisation factors with an adjustment for biodiversity valuation excluding crop losses.

This shows that the spread in results is huge. The valuation used in NEEDS is most similar to our approach to valuing biodiversity through a value for the Potentially Disappeared Fraction (PDF). We therefore propose to start from the NEEDS model results, partly because they are also in the middle of the other estimates.

Further, we carried out the following further steps when updating environmental prices:
- The NEEDS model results have been adjusted for current lower emissions compared to 2008 and the higher valuation of biodiversity (Paragraph 6.7.5).
- The characterisation factors are now based on ReCiPe 2016 (see Paragraph 6.7.6).
- For the acidification theme, we added the damage costs of acidifying emissions (SO_2 and NO_x) to buildings (see Paragraph 5.5).

6.7.5 Update of the results from the NEEDS project

For the effects of acidification, we only considered the effects of deposition of N and S on biodiversity. In the NEEDS 2008 project, these were based on EcoSense model runs (see Annex C.3.8). We used these as the basis for the calculations. We adjusted the NEEDS model results in the following manner:
- The effects were calculated by scaling actual emissions on European territory in 2019 to those behind the NEEDS modelling results for emissions in the 2010 and 2020 scenarios. The actual emissions for SO_2, NO_x and NH_3 are in between these model results so that the scaling reflects the actual effects on soils in the EU.
- Prices and values for biodiversity have been adjusted to the 2021 price level, taking into account an autonomous growth in the value of biodiversity (irreversible nature) of 1%, in line with the assumptions in Paragraph 5.4.
- The valuation for the lower and upper values has been adjusted to reflect the variation in the valuation for species richness used in this Handbook to value biodiversity (see Paragraph 5.4).

6.7.6 Update: indicators for characterisation

In ReCiPe 2016, SO_2, NO_x and NH_3 are considered potentially acidifying pollutants. The potentially acidifying pollutants each possess different acid-forming capacity, also called potential acid equivalent (= pot. Z-eq.). One mole of H+ ions equals one acid equivalent. In ReCiPe 2016, the indicator SO_2 equivalents is used, where acid equivalents are converted into the amount of acid that can be caused by SO_2. For acidification, no distinction is made between different value perspectives.

Table 45 - Characterisation factors for acidification (global averages) in ReCiPe 2016, in kg SO_2-eq./kg

Pollutant	Kg SO_2-eq./kg
NO_x	0.36
NH_3	1.96
SO_2	1.00

* There is no difference in acid-forming ability between different perspectives.

6.7.7 Environmental prices

The resulting environmental prices on the acidification theme include:
- damage to biodiversity due to changes in soil conditions caused by acidifying and eutrophying emissions;
- damage to buildings (clean-up costs and restoration costs) not covered by the topic of particulate matter formation.

Table 46 - Environmental prices due to emissions to air on the acidification/eutrophication theme on ecosystems and damage to buildings, in €$_{2021}$/kg

Acidification/eutrophication	Unit	Lower	Central	Upper
SO_2	€/kg	€ 1.525	€ 4.221	€ 8.526
NO_x	€/kg	€ 0.304	€ 1.203	€ 2.929
NH_3	€/kg	€ 0.38	€ 0.54	€ 0.70
Midpoint characterisation factor	€/kg SO_2-eq.	€ 2.66	€ 5.27	€ 9.30

It should be noted that these values for eutrophication underestimate the Willingness-To-Pay that arises if the actual nitrogen emission allowance is taken as the starting point for monetisation. As explained in Paragraph 6.6.7, very high prices are currently paid in the market for nitrogen emission allowances, depending on the location of emissions.
This damage is not included in these prices above.

6.8 Human toxicity

6.8.1 Description of theme

Human toxicity covers all other pollutants that are potentially hazardous to human health, characterised 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. Their toxic impact falls into the following categories:
- acutely poisonous substances;
- pollutants that can cause cancer (carcinogenicity);
- pollutants that can cause genetic mutations (mutagenicity);
- pollutants that can impact reproduction (teratogenicity);
- pollutants that can irritate and damage skin, eyes or the respiratory tract;
- pollutants that affect the nervous system and lead, for example, to problems in the mental and physical development of children, including in unborn children.

6.8.2 Pollutants and sources

The main pollutants that have an impact on the theme of ‘human toxicity’ are heavy metals, chlorinated hydrocarbons, pesticides and other biocides and a wide range of specific chemical compounds used in products or semi-finished products.

Heavy metals in particular are major sources of human toxicity. Heavy metals are released during the production process as a result of mining activities and 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 fertilisers. 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. The main source of pesticides and biocides is agriculture.

Politically, these substances are classified as ‘Substances of Very High Concern’ (SVHCs) under the European Union’s REACH regulation. SVHCs are substances that are dangerous to humans and the environment because, for example, they impair reproduction, are carcinogenic or accumulate in the food chain. Many of the substances covered by human toxicity also have effects on ecotoxicity, although the knowledge base concerning effects of substances is smaller on the ecotoxicity theme than on the human toxicity theme.

In this Handbook, this group consists of almost 3,000 pollutants.

6.8.3 Impact

The toxic impact of heavy metals has been extensively studied. 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 can also infiltrate groundwater via seepage into the soil.

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. The damage many of these chemicals cause only manifests itself with the passage of time, particularly when it comes to non-acute health effects such as reproductive or bodily function damage. It was only in the 1970s, for example, that the toxic impact 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 effects, which have been unravelled by researchers 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.

Special attention needs to be paid to substances that are bioaccumulative. Some of the pollutants are not broken down much, if at all, when they are released into the environment. One such example is PFAS. Because these pollutants are likely to remain in the environment for a very long time, there is a risk that toxic effects attributable to these pollutants may be discovered later. Environmental prices are not appropriate for such pollutants because environmental prices do not take into account future damage that may be revealed due to progressive scientific insight. For such pollutants, valuation with environmental prices would always underestimate the actual damage. Substances that do not degrade in the environment after dispersion should be analysed through a risk analysis (see also Paragraph 2.8).

6.8.4 Treatment in the Environmental Prices Handbook 2018 and updates

The Environmental Prices Handbook 2018 found that valuations for human toxicity vary widely. Based on the results from NEEDS 2008, direct endpoint valuation via ReCiPe 2008 characterisation factors; direct endpoint valuation via ILCD characterisation factors and the calculations from (Nedellec & Rabl, 2016) in the AMESTIS project on the damage costs of toxic metals, an average of damage costs for four metals was calculated and a valuation for IQ effects was added.

The damage costs calculated in the Environmental Prices Handbook generally yielded a relatively low valuation for human toxicity. In recent research for the Human Environment and Transport Inspectorate (Inspectie Leefomgeving en Transport) and the Ministry of Infrastructure and Water Management in the Netherlands, (CE Delft, 2021b) reviewed scientific literature on the adverse effects of toxic substances on mortality and morbidity (see Annex F in that report for a total overview). This study concludes that the values listed in the Environmental Prices Handbook represent only a fraction of the harmful effects named in the scientific literature. Similarly, (EEA, 2021) has reviewed the evidence base of harmful pollutants again and concluded that it is much more extensive than assumed in previous research.

For this Handbook, we have therefore undertaken the following:
- The valuation for individual pollutants was adjusted by calibrating the results from (CE Delft, 2021b) to the results from (EEA, 2021) (see Paragraph 6.8.5). We went from four to ten pollutants for which damage costs were established.
- The characterisation factors are based on ReCiPe 2016 which characterises pollutants on two themes of human toxicity (carcinogenic and non-carcinogenic) (see Paragraph 6.8.6).

6.8.5 Update: damage cost estimates individual pollutants

(EEA, 2021) has made a new assessment of the harmfulness of toxic substances. Compared to older studies, more effects were distinguished here. Table 47 lists the pollutants and effects included in the study by EEA. In (CE Delft, 2021b), an additional analysis of the scientific literature on the effects of a number of pollutants considered by (EEA, 2021) was carried out. This shows that while the effects in (EEA, 2021) contain an extension of the original NEEDS results (looking primarily at cancer and IQ effects), they are not yet complete in terms of dose-effect relationships that have been identified in scientific literature.

Table 47 - Overview of the effects of four heavy metals

Pollutant		Effects taken into account in EEA (2021)	Other effects in literature	Sources
Arsenic (inorganic)	As	Non-cancer and Cancer mortality, Chronic bronchitis, IQ loss, diabetes	Ischemic heart disease, Cerebrovascular diseases	Medrano, et al, 2010 D'Ippoliti, et al., 2015
Cadmium	Cd	Cancer (fatal & non-fatal)	Chronic kidney disease, Myocardial infarction	Ginsberg, 2012 Tellez-Plaza, et al., 2013
Lead	Pb	Cardiovascular mortality, IQ loss	Anaemia, kidney damage	
Mercury	Hg	Cancer (fatal & non-fatal)	Myocardial infarction	Virtanen, et al., 2005 Wennberg, et al., 2012

We investigated whether supplementary literature could lead to the inclusion of the additional effects in the prices of the four individual pollutants. However, we conclude that there is currently too much uncertainty about the exact dose-effect relationships as used in (EEA, 2021). It was therefore decided to base environmental prices for the four metals mentioned in the EEA studies. As a result, the prices are likely to be an underestimate, given that there are effects found in the literature that are currently not included in the prices.

6.8.6 Environmental prices

Table 48 provides a valuation of the effects of various toxic pollutants released into the air on human health.

Table 48 - Environmental prices due to emissions to air of toxic pollutants at the human toxicity midpoint, in €$_{2021}$/kg

Pollutant		Lower		Central		Upper
Arsenic (inorganic)	€	6,261	€	9,255	€	13,938
Cadmium	€	104,979	€	155,186	€	233,692
Chromium (hexavalent, VI)	€	1,774	€	2,622	€	3,949
Lead	€	18,442	€	27,263	€	41,054
Mercury	€	9,583	€	14,165	€	21,332
Nickel	€	13,6	€	20,1	€	30,3
1,3-Butadiene	€	0.74	€	1.09	€	1.64
Benzene	€	0.204	€	0.302	€	0.454
Benzo(a)pyrene	€	3.858	€	5.704	€	8.589
Dioxins/Furans (TCDD equivalents)	€	34,071,638	€	50,366,770	€	75,846,430
Formaldehyde	€	0.142	€	0.210	€	0.316

Based on these damage estimates for individual pollutants, we made a damage estimate for the midpoint price for human toxicity. ReCiPe 2016 (Huijbregts, et al., 2016) distinguishes a midpoint characterisation factor for human toxicity from cancer cases from a midpoint characterisation factor for non-cancer cases. The endpoint to midpoint characterisation factors show that ReCiPe assumes that 94% of the total damage burden of both midpoints in DALYs can be explained by cancer cases and 6% by non-cancer cases. Using this estimate, we can divide the harm caused by individual pollutants between cancer and non-cancer cases. The resulting price per midpoint characterisation factor is shown in the following table.

Table 49 – Midpoint prices human toxicity, in €$_{2021}$/1.4-DCB-eq.

	Lower	Central	Upper	Unit
Human toxicity, cancer-related	€ 2.70	€ 3.99	€ 6.01	€/kg 1.4-DCB-eq.
Human toxicity, non-cancer related	€ 0.05	€ 0.07	€ 0.11	€/kg 1.4-DCB-eq.

6.9 Ecotoxicity

6.9.1 Description of theme

Ecotoxicity is the impact of toxic substances on non-human organisms in ecosystems. This occurs when organisms that are not the intended targets of these pollutants are exposed to them. Damage to ecosystems is primarily caused by pesticides used in agriculture, which are designed specifically to exterminate organisms deemed to pose a threat to crops and livestock. In addition, pesticides are also widely used by households as well as government agencies. Almost 80% of crop protection agents do not reach their intended target (VMM, 2013f), meaning significant pesticide emissions take place.

6.9.2 Sources (pollutants)

(VMM, 2013f) distinguishes two kinds of pesticides: crop protection agents and biocides. The first category concerns substances used in agriculture to protect crops from pests and kill unwanted plants or parts of plants. These substances are used mainly by farmers, in allotments and in public spaces. Crop protection agents can be subdivided into insecticides, herbicides, fungicides, bactericides, molluscicides, rodenticides, nematicides (to combat nematode worms) and acaricides (for ticks and mites).

Biocides are pesticides used in non-agricultural settings, except in applications similar to farm use. 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 such as 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 anti-fouling agent. 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’ such as 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 Impact

Crop protection agents damage ecosystems through their toxicity to non-target organisms, pollution of surface water, groundwater, aquatic sediments and soils, and bioaccumulation (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 the effects varies from a few days to several years. The longer a toxic substance remains active, the greater the risk of bioaccumulation. 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, 2013f), 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 gender 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 such as aldrin, atrazine, chlordane and dieldrin can cause anomalous gender 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, 2013f).

The main impact of the biocide TBT was its damaging effect on the endocrine system of shellfish (Tornero & Hanke, 2016). Copper is an essential trace element for many organisms, but it 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 dinoflagellate 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 such as sheep (copper) (VMM, 2013h).

In our treatment of ecotoxicity all these pollutants have been included. Via ReCiPe 2016 (Goedkoop, et al., 2013), the effects of more than 1,000 pollutants discharged to water, or as waste spreading in soil, and their effects on ecotoxicity can be included. These effects were updated in ReCiPe 2016.

6.9.4 Treatment in the Environmental Prices Handbook 2018 and updates

The valuation for ecotoxicity was determined in the Environmental Prices Handbook 2018 using the characterisation factors from ReCiPe 2008 by direct endpoint-level valuation (Goedkoop, et al., 2013). We assumed the individualistic worldview from ReCiPe.[74]

[74] For a limited number of heavy metals (cobalt, copper, manganese, molybdenum and zinc), the upper value was determined by the hierarchical worldview and the central value by the average of the hierarchical and individualistic value. The difference between the individualistic and hierarchical worldview consists of a combination of a difference in the studies that are included to determine ecotoxicity, which compartments are modelled and which background concentrations are included.

ReCiPe (Goedkoop, et al., 2013) expresses ecotoxicity in terms of the relative toxicity of benzene (more specifically, the pollutant 1.4-dichlorobenzene) discharged into the ocean. 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.

An updated version of ReCiPe was published in 2016. It includes various improvements. A detailed comparison between ReCiPe 2016 and ReCiPe 2008 is described in Annex D.2. To arrive at an environmental price for ecotoxicity, effects are converted from effects on biodiversity (in species per year) into kg 1.4-DB equivalents. In ReCiPe 2016, new characterisation factors are given for this purpose.

Table 50 presents the old and new midpoint to endpoint factors for ecotoxicity. The characterisation factors are the same in the individualistic and hierarchical perspectives. The largest relative difference is visible for ecotoxicity on land: it is a factor of 7,000 smaller than in ReCiPe 2008. For freshwater and saltwater ecotoxicity, the characterisation factors are a factor of 1.24 and 1.68 smaller, respectively, than in ReCiPe 2008.

Table 50 - Comparison midpoint to endpoint factors ReCiPe 2008 and ReCiPe 2016, species.yr/kg 1.4-DB-eq.

Midpoint	New factor	Old factor
Terrestrial ecotoxicity	2.14E-11	1.51E-07
Freshwater ecotoxicity	6.95E-10	8.61E-10
Marine ecotoxicity	1.05E-10	1.76E-10

In addition, the valuation of biodiversity, in €/species.yr, has been adjusted. A detailed description of the biodiversity valuation update can be found in Paragraph 5.4.

6.9.5 Environmental prices

The monetary valuation for this theme is based on ReCiPe endpoint characterisation. As explained in Paragraph 5.4 and Annex E, a relationship was established between the valuation of biodiversity from economic literature and the unit of the characterisation factor from ReCiPe 2016.

This leads to the following valuation for ecotoxicity expressed as the characteristic pollutant 1.4-dichlorobenzene to the various compartments.

Table 51 - Valuation for ecotoxicity Environmental Prices Handbook 2024, in €$_{2021}$ per unit

Midpoint	Lower	Central	Upper	Unit
Terrestrial ecotoxicity	€0.0005	€0.0006	€0.0008	€/kg 1.4-DCB-eq.
Freshwater ecotoxicity	€0.0148	€0.0209	€0.0271	€/kg 1.4-DCB-eq.
Marine ecotoxicity	€0.0022	€0.0032	€0.0041	€/kg 1.4-DCB-eq.

It should be emphasised that the valuation of ecotoxicity is more uncertain than the valuation of the other themes. If ecotoxicity is an explicit topic of the study, we advise against the use of environmental prices. In such cases, it is better to conduct a specific study on the effects of toxic substances on ecosystems with specific valuations for those ecosystems.

6.10 Ionising radiation

6.10.1 Description of theme

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 ionisation. Radionuclides (unstable atoms that emit ionising radiation during decay to their stable end product) are substances that occur naturally in the Earth's crust or can be made by humans. If living tissue is exposed to ionising radiation, this exposure 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 amount of ionising radiation resulting from radionuclide emissions is usually measured in Becquerel (Bq), named after French physicist Antoine Henri Becquerel, which expresses the decay of atomic nuclei per second. In addition, the unit 'Curie' sometimes still occurs, named after the Polish/French physics couple Marie and Pierre Curie. One curie corresponds to the activity of 1 g of radium and is as large as 37 billion becquerels.

6.10.2 Sources

We are all exposed to natural ionising radiation. The two main natural sources are cosmic radiation and radioactive minerals occurring naturally in the Earth's crust. A major contributor to human exposure to natural radiation is radon gas, which is found in the soil and can accumulate in the crawl spaces of homes. Besides radon gas, naturally occurring radionuclides include Uranium-238, Thorium-232, Potassium-40. Volcanoes can also be an important source of emissions of uranium and thorium.

In this Handbook, however, we are primarily interested in human-caused radiation as this forms the basis of environmental prices. The main source of radiation is caused by human activities related to the use of radiation for medical diagnostics (e.g., X-ray equipment). In addition, environmental pollution of radioactive waste from nuclear energy (including Uranium-235) and nuclear weapons testing (including Tritium, Cesium-137, Iodine-131 and Strontium-90) remains a major source of human radiation exposure worldwide. In some parts of the world, production of fissile material for military ends has left behind vast amounts of radioactive waste. The Chernobyl and Fukushima nuclear disasters have also caused long-distance dispersion of radioactive substances (especially Cesium-137 and Iodine-131). Finally, radioactive materials are emitted in minor amounts from fossil-fuel combustion and the use of certain materials in industry and agriculture.

6.10.3 Impact

The health effects of radiation absorption can manifest themselves in the form of fatal and non-fatal cancers and hereditary abnormalities. Human radiation exposure associated with emissions depends on the medium in which the radionuclide was generated (via water or via air), which radionuclide it is from and how much alpha, beta or gamma radiation is involved.

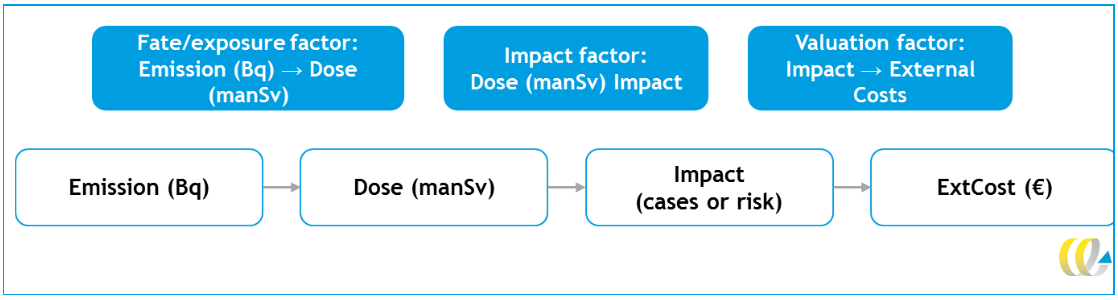
Although radiation is also lethal to ecosystems, literature has almost exclusively quantified the effects on human health. In this Handbook, we focus solely on the effects on human health.[75]

[75] The impact of radiation on ecosystems is also not included in ReCiPe 2016 because no impact assessment methods are available. In addition, it is also plausible to assume that effects on human health will give much larger welfare effects than effects on ecosystem services.

6.10.4 Approach in Environmental Prices Handbook 2018 and updates

The valuation of the effects of radionuclides in the Environmental Prices Handbook 2018 was based on (NEEDS, 2008a). This project proposed a simplified approach to calculating the external costs of radionuclide emissions, largely based on previous studies by the UN (UNSCEAR, 2000) (see Figure 12).

Figure 12 - Overview of valuation effects of radionuclides in Environmental Prices Handbook 2018



The associated damage costs per pollutant formed the basis for the valuation in the Environmental Prices Handbook 2018, where the damage costs for the pollutants for which both damage costs from NEEDS and characterisation factors from ReCiPe are available were used to calculate an average midpoint price where each pollutant in principle counts once towards that average (equal weighting).[76] The procedure in the Environmental Prices Handbook 2018 was the same as that in the Shadow Prices Handbook (CE Delft, 2010).

For the new Handbook, we have revised this procedure on the following points:
- update of the characterisation factors from ReCiPe 2008 to ReCiPe 2016;
- adjustment of the valuation chosen in (NEEDS, 2008a) in relation to the dose-effect relationships applicable there;
- adjustment of the equal weighting to a weighting based on how often these pollutants enter the environment through human action.

We explain these adjustments in the following paragraphs.

6.10.5 Update of characterisation factors

In both ReCiPe 2008 and ReCiPe 2016, the average probability of exposure was determined for each radionuclide (e.g. C14, Co60, KR85). This took place for three environments: air, freshwater and seawater. This distinction is made because exposure to radiation occurs via air on the one hand and by ingestion via water (followed by internal radiation), which is usually much more harmful. The ‘collective (effective) dose’ is the sum of individual doses to which the world population is exposed over a given time period, expressed for the population of the whole world in man.Sv/kBq.[77]

To determine fate and exposure, ReCiPe uses models of how a nuclide spreads (fate) and how much the impact actually affects people (exposure) (see ReCiPe (2008)). The midpoint ionising radiation potential (IRP) is expressed as the ratio of the collective exposure dose of emitted substance x divided by the collective exposure dose of Cobalt-60 in air.

[76] With the exception of Iodine-131 and radioactive noble gases, which were outliers in this procedure.

[77] Sievert is actually J/kg, the amount of harmful energy of a radiation, weighted for the type of radiation (e.g. alpha, beta, gamma because they are not all equally dangerous).

Characterisation factors are available for emissions to air, rivers and sea for all three perspectives.

For the individualistic and hierarchical perspective, ReCiPe 2016, after thorough examination of the source literature, arrives at a characterisation factor for fewer substances than ReCiPe 2008. For Uranium-235 and Uranium-238, ReCiPe does not assign a characterisation factor in these perspectives. For the egalitarian perspective (see Annex D), it follows the previous ReCiPe, which developed a characterisation factor for these substances.[78]

In the update, we calculated the midpoint price both for the restricted set from the hierarchical perspective and for the more comprehensive set in the egalitarian perspective. This comparison showed that hardly any difference arises between the two perspectives, implying that the narrower set from the hierarchical perspective can provide sufficient leads for the midpoint price. To determine the midpoint price, we linked pollutants, for which damage to water has been determined, to the characterisation factor for emissions to freshwater (which is the same as 'unspecified' in a software package like SimaPro).

6.10.6 Adjustment of the valuation

Dose-effect relations were established in (NEEDS, 2008a) for thirteen pollutants (for a number of pollutants in various isotope variants) for emissions to air and water: Carbon-14, Cesium-137; Hydrogen-3; Iodine-129,131 and 133; Krypton-85; Radon-222; Thorium-230; Uranium-234, 235 and 238; Lead-210; Polonium-210, Radium-226, Strontium-90 and Rubidium-106.

Three effects were determined for these pollutants based on a literature review:
1. **Fatalities due to cancer (mortality**). NEEDS hereby assumed 15.95 years of life lost (LYL) per cancer case - the Environmental Prices Handbook 2018 assumed a LYL of 13.
2. **'Cost of morbidity' per cancer case.** This refers to the welfare loss a person experiences when diagnosed with cancer and is a value for fatal and non-fatal cancer cases. Whereas (NEEDS, 2008a) assumed a valuation only for non-fatal cancer cases of €481,000 (2000 price level), the 2018 Environmental Prices Handbook assumed a valuation of €420,000 (2015 price level) in the lower value for both non-fatal and fatal cancer cases.
3. **Mortality due to hereditary (genetic) defects**. This affected 5% of cancer cases and was valued at a VSL of €1.5 million.

We took the same approach in this update and made the following adjustments:
- We assume a mortality rate of 43% in the Netherlands if cancer is diagnosed based on the calculations from (OECD, 2020). This is higher than assumed in NEEDS (2008a).
- When someone dies, we assume the same LYL of thirteen years, as in the previous Handbook. Work-related cancer cases have a slightly higher LYL of 15 (RIVM, 2016). For the population as a whole, however, this will be lower because on average work-related fatal infections involve younger people.
- The cost of morbidity breaks down into the hospital cost per treatment and nonhospital costs, also known as 'cancer premiums'. In the previous Handbook, these were based on (Rabl, et al., 2014). We are not aware of any new research on calculating cancer premiums in the European Union. However, we do see that hospital costs are similar to

[78] To get from the characterisation factor from ReCiPe 2008 (expressed in U235-eq.) to the characterisation factor in ReCiPe 2016 (expressed in Co60-eq.), divide by 0.786 (because 0.786 Co-60-eq. = 1 U235-eq.).

the values reported in (Rabl, et al., 2014). Therefore, we apply the previous valuation and multiply it by income elasticities and inflation.
- Heritable effects have been valued using the adjusted valuation of the VSL (see Paragraph 5.3).

The external costs per unit emission were calculated by multiplying the disease-specific valuations by the expected number of sick people due to radiation, which depends on the pollutant. Annex C.7 of (CE Delft, 2010) states more about the methodology NEEDS followed to arrive at an estimate of the burden of disease.

Table 52 provides information on the damage costs of the different radionuclides for the lower, central and upper variants.

Table 52 - Initial and adjusted valuation radionuclides, in €/kBq

Substance name (English)	NEEDS [€$_{2000}$/kBq]	Values €$_{2021}$/kBq	Substance name (English)	NEEDS [€$_{2000}$/kBq]	Values €$_{2021}$/kBq
Carbon-14, air	1.40E-03	1.90E-03	Uranium-238, air	9.01E-04	1.22E-03
Cesium-137, air	9.53E-04	1.52E-03	Lead-210, air	1.29E-04	2.06E-04
Hydrogen-3, Tritium, air	5.10E-07	8.15E-07	Polonium-210, air	1.29E-04	2.06E-04
Iodine-129, air	8.24E-03	1.32E-02	Radium-226, air	7.72E-05	1.23E-04
Iodine-131, air	2.61E-03	4.17E-03	Carbon-14, water	9.38E-06	1.50E-05
Iodine-133, air	3.76E-07	6.00E-07	Cesium-137, water	1.26E-05	2.01E-05
Krypton-85, air	2.75E-08	4.40E-08	Hydrogen-3, Tritium, water	1.09E-07	1.75E-07
Noble gases, radioactive, unspecified, air	5.53E-08	8.84E-08	Uranium-234, water	2.55E-05	4.07E-05
Radon-222, air	1.45E-08	2.31E-08	Uranium-235, water	9.20E-05	1.47E-04
Thorium-230, air	3.86E-03	6.17E-03	Uranium-238, water	2.53E-04	4.04E-04
Uranium-234, air	1.03E-03	1.64E-03	Strontium-90, water	6.05E-07	9.66E-07
Uranium-235, air	8.40E-04	1.34E-03	Ruthenium-106, water	4.25E-07	6.78E-07

Source: (NEEDS, 2008a).

6.10.7 Adjusting the weighting to arrive at a midpoint price

In the Shadow Prices Handbook 2010 and the Environmental Prices Handbook 2018, the pollutants for which dose-effect relationships of all pollutants for which characterisation factors were also available were added together and divided by the sum of the characterisation factors to obtain a weighted midpoint estimate. The problem here is that the equal weighting (each pollutant counted equally) does not accurately reflect the real-world situation regarding emissions of radionuclides, as some pollutants are more prevalent than others.

We are not aware of any information on radionuclide emissions in the European union. Monitoring programmes do exist, such as in the vicinity of the nuclear power station in Borssele. (Kwakman, 2018) reports on a monitoring programme by the operator of the nuclear power station, which shows that emissions of caesium-137 and iodine-131 to air are largely below the measurement detection values, while emissions of tritium in water (H-3) are between 1.3 and 6.7 Becquerels per litre - these emissions may also be caused by the more distant nuclear median in Doel, Belgium.

However, (UNSCEAR, 2000) did estimate emissions from nuclear power stations (including reprocessing plants) and radiation released from X-ray scanning equipment on a global scale as an average for the period 1995-1997. We used this information to weight the relative contribution of the various radionuclides. The emissions are listed in the following table.

Table 53 - Emissions in TBq radiation globally used to arrive at a midpoint price

Substance name (in English)	Emissions to	CF: kBq/kBq Co-60	Damage cost €/kBq, central**	Emissions TBq global^	Emissions expressed in PBq Co-60	Total damage €bn.
Carbon-14	Air	1.15	2.23E-03	607	698	1.36E+00
Hydrogen-3, Tritium	Air	0.000856	8.15E-07	9,923	8.5	8.08E-03
Iodine-129	Air	10.5	1.32E-02	29	305	3.82E-01
Iodine-131	Air	0.00909	4.17E-03	600	5.5	2.50E+00
Krypton-85	Air	0.00000848	4.40E-08	5,292,300	45	2.33E-01
Radon-222	Air	0.00000788	2.31E-08	12,960	0.1	3.00E-04
Hydrogen-3, Tritium	Water	0.0000412	1.75E-07	201,665	8.3	3.52E-02

* ReCiPe 2016, hierarchical

** Calculated for the central value

^ Emissions calculated by us using (UNSCEAR, 2000) Annex A concerning an average of the years 1995-1997.

Dividing the total damage by the emissions expressed in Co-60-eq. thus gives a weighted midpoint price for radionuclides.

6.10.8 New environmental prices for radiation

Based on the proposed approach, an environmental price was calculated from the individual substances that have both a characterisation factor in ReCiPe and as an estimation of emissions in (UNSCEAR, 2000) for the low, central and upper variants, with the range determined solely by the difference in valuation of mortality and morbidity (see Paragraph 5.3).

Table 54 - Environmental price for midpoint radiation, in €$_{2021}$/kBq Co-60-eq.

	Lower	Central	Upper
kBq Co-60	€0.00275	€0.00422	€0.00594

Comparing this environmental price with the previous Handbook, it is immediately noticeable that this price is much lower. The previous environmental price was 0.0473 €/kBq U-235-eq. Since 1 kBq U235-eq. is equal to 0.786 kBq Co-60-eq., the current environmental price corresponds to a price that is a factor 14 lower than the previous Environmental Prices Handbook. This shows that the weighting step with emissions has important implications for midpoint price estimation.

It should be mentioned that, even in the previous Environmental Prices Handbook, radiation was not an important category in environmental prices calculations in LCAs.

6.11 Noise

6.11.1 Description of theme and sources

Ambient noise is a major environmental problem with a range of impacts on human well-being, human health and nature. As traffic noise is the main source of ambient noise, most valuation studies are concerned with this type of noise (EY, 2016); (Navrud, 2002); (WHO, 2018), with only limited research on noise from other sources, such as building sites, public events, industry, wind farms and neighbours. Given this lack of data, this Handbook focuses solely on the valuation of traffic noise, making a distinction between road, rail and air traffic. In this EU version of the Environmental Prices Handbook, we have not included EU values for noise. Values are available for the Netherland, but they cannot generally be applied to the EU. To demonstrate the valuation of noise, we included the method and results for the Netherlands in Annex D. Below, we describe the impacts of noise.

6.11.2 Impact

We can distinguish five different potentially harmful effects of ambient noise (WHO, 2018):
1. Nuisance.
2. Health effects.
3. Productivity loss.
4. Disturbance of quiet areas.
5. Effects on ecosystems.

Below, we explain these different effects one by one.

Nuisance

Noise can lead to nuisance for people, for example by disturbing them when performing certain activities. This nuisance can lead to a wide range of negative feelings, such as irritability, disappointment, dissatisfaction, helplessness, depression, etc. (WHO, 2011). Some studies view these effects as health effects (for example, (WHO, 2018); (Defra, 2014); (IGCB, 2010)[79], while others explicitly distinguish between nuisance and health effects (for example, (Bristow, et al., 2015); (Nelson, 2008)). Inconvenience is classified as annoyance and is often measured in academic studies by charting the percentage of respondents who report being '(highly) annoyed'. In this Handbook, we treat nuisance as a separate impact on the endpoint well-being and not as a health effect (see next paragraph for an explanation of the additionality of nuisance and health effects).

Direct health effects

Exposure to ambient noise can lead to negative health effects in addition to nuisance. However, research on the various possible health effects of noise exposure is not unequivocal. WHO has therefore recently put out a series of reviews to substantiate its Environmental Noise Guidelines (WHO, 2018). The reviews thoroughly examined the evidence for a relationship between noise exposure and the following health effects:
- ischaemic heart disease;
- hypertension (elevated blood pressure);
- sleep disturbance;
- hearing damage and tinnitus;

[79] This is in line with the broad definition of health used by the WHO: 'a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity' (WHO, 2011).

— birth problems;
— metabolic diseases;
— learning achievements;
— mental health.

As we explain further below, WHO concludes that there is currently only strong evidence for a relationship with ischaemic heart disease and sleep disturbance. Moreover, there appear to be differences between the three traffic types (air, rail, and road).

Productivity loss

Noise can lead to lost productivity due to reduced employee performance (for example, due to concentration problems or fatigue from noise-related sleep problems), increased absenteeism due to noise-related health complaints, noise-related learning difficulties of children leading to lower educational attainment and work absences (TRL, 2011; WHO, 2018). These effects have only been studied to a limited extent in the literature. Moreover, it is quite conceivable that these themes overlap with some of the above health effects such as sleep disturbance. To avoid double counting and because monetisable dose-effect relationships are almost entirely absent on this topic, we do not include productivity losses separately.

Disturbance of quiet areas

(Anastasopoulos, et al., 2011) point out that ambient noise can lead to people being less able to experience the benefits of quiet areas (e.g. urban parks, forests), resulting in economic costs. However, research on these ambient noise costs is still very limited. We therefore omit this effect in this study.

Effects on ecosystems

There is a growing number of studies pointing to the harmful effects of noise on animals, for example by disrupting breeding periods (Dutilleux, 2012). Such research is still in its infancy, however, and reliable valuation numbers are therefore still lacking. The same applies to the effects of underwater noise, which could harm the lives of cetaceans (Marotte et al., 2022). We therefore do not include these effects in this study.

Effects taken into account

Based on the above overview, we conclude that there is sufficient scientific evidence and valuation knowledge to derive cost figures only for nuisance and health effect, as for the Netherlands. We discuss the economic valuation of these two effects in more detail in Annex D.

6.12 Use of raw materials and water consumption

6.12.1 Introduction

Use of raw materials and water consumption can affect ecosystems and people.
The environmental impact of both is usually modelled in life cycle analysis and included with other environmental impacts. In addition, use of raw materials or water consumption can also have other effects:

- The extraction of raw materials may reduce the availability of these raw materials for future generations. To the extent that these are not included in the price of raw materials, there may be an external effect.
- Water consumption can reduce the amount of water available for nature, causing it to dry up. This is generally not included in the price for water extraction and is therefore an external effect.
- Consumption of fossil fuels and rare metals in particular can create instability in global markets that can lead to economic damage. This is also an external effect.

Paragraph 5.6 elaborates on the valuation of the use of raw materials. In this paragraph, we describe how we arrived at a midpoint characterisation price, given the valuation framework in Paragraph 5.6.

6.12.2 Energy raw materials

For fossil raw material scarcity, a midpoint characterisation factor is developed in kg oil-eq., which is defined as the ratio of the higher heating value (HHV) of a fossil fuel to the energy content of crude oil.

The valuation of this midpoint can be directly linked to the valuation in MJ developed in Paragraph 5.6., given a factor of 41.9 MJ per kg of oil. This produces a valuation, as shown in Table 55. This value is specific for the Netherlands. The approach outlined in Paragraph 5.6 can be followed for other countries.

Table 55 - Valuations for the midpoint fossil scarcity of raw materials for the Netherlands, in €$_{2021}$/kg

	Lower	Central	High
Midpoint €/kg-oil-eq.	€0	€0.0276	€0.1633

It should be noted that this valuation is thus lower than the valuation that follows implicitly from ReCiPe due to increased extraction costs. As reasoned in Paragraph 5.6, it is plausible that the future increase in extraction costs as opportunity costs is already partly factored into the price of raw materials. If we place the valuation that follows from the increased extraction costs alongside other valuations, such as the abatement costs of raw materials stocks, the economic damage caused by oil price fluctuations or the shadow prices of circular measures, it can be seen that the increased extraction costs are indeed above the valuations from the other approaches. Therefore, the Environmental Prices Handbook has determined a *lower* valuation for the scarcity of raw fossil materials.

6.12.3 Mineral raw materials

The midpoint characterisation for mineral raw material scarcity is Surplus Ore Potential (SOP), which is expressed in kg Cu-eq. (Huijbregts, et al., 2016). Primary extraction of a mineral leads to a decrease in the available ore grade (ore grade). This decrease means that more and more ore has to be excavated over time to make the same amount of mineral available. In addition, the expected future extraction of an ore is considered and this is also reflected in the perspectives used. Reserves here are ‘resources that could economically be extracted or produced at the time of determination’ and ultimate recoverable resources as ‘the amount of a resource available in the upper crust of the earth that is ultimately recoverable’. Together these provide the SOP which, as it increases, creates an increased surplus cost potential.

For the factor to get from midpoint to endpoint, ReCiPe refers to the conversion of surplus ore to surplus costs of twelve different metals (see (Vieira, et al., 2016a). In the hierarchical perspective, this leads to a valuation of $0.23 (2013 price level). By comparing this value with the resource extraction valuation of petroleum (see Paragraph 5.6.7) in ReCiPe, a ratio can be developed whereby metal scarcity of raw materials can be compared with of the scarcity of fossil materials (i.e. 0.23/0.457 = 0.506). Table 56 gives the valuation of fossil scarcity of raw materials.

Table 56 - Valuations for the midpoint mineral scarcity of raw materials, in €$_{2021}$/kg for the Netherlands

	Lower	Central	High
Midpoint: €/kg CU-eq.	€0	€0.0140	€0.0826

6.12.4 Water consumption

The depletion of ground and surface water affects the availability of water for humans, animals and plants. When water scarcity occurs, it affects human health when people do not have enough water to drink or grow crops (Pfister , et al., 2009). In addition, water scarcity can lead to loss of grazing land and drinking water for animals, which can lead to biodiversity loss. Water use may also lead to a decline in fish species due to reduced river discharge (Hanafiah, et al., 2011). All these effects are normally not adequately accounted for in the price of water and can therefore be considered externalities.

Water scarcity was not valued in the Environmental Prices Handbook 2018.
The Environmental Prices Handbook 2024 does value water scarcity. This involves direct valuation of impacts on ecosystem services and human health at the endpoint level.
In doing so, these characterisation factors were entered country-specifically in ReCiPe.
This valuation can be used in LCAs.

Characterisation and indicators

The characterisation of the impact of water consumption on human health, terrestrial and aquatic ecosystems is based on ReCiPe 2016. ReCiPe considers the increase in water consumption in terms of Water Consumption Potential (WCP), expressed in m^3-water-eq. consumed.

The characterisation factor at midpoint is the ratio between the m^3 of water consumed and the m^3 extracted water. For agriculture and industry, ReCiPe (Huijbregts, et al., 2016) has estimated this based on literature.
By means of midpoint to endpoint factors, the impact on ecosystems (biodiversity) and human health is determined as follows:
- human health: malnutrition due to water shortage (Pfister , et al., 2009);
- ecosystems (terrestrial): decline in Net Primary Productivity (NPP) due to water shortage as a proxy for overall species loss (Pfister , et al., 2009);
- ecosystems (freshwater): loss of fish species due to reduced river discharge (Hanafiah, et al., 2011).

Table 57 shows the endpoint characterisation factors for the individualist and hierarchical worldview (see Annex B):

Table 57 - Characterisation at endpoint level for water consumption ReCiPe 2016 for the EU27

Midpoint to endpoint conversion factor	Unit	EU27*	
		Individualistic	Hierarchic
Water consumption - human health	Daly/m³ consumed	3.10E-08	2.22E-06
Water consumption - terrestrial ecosystems	species.yr/m³ consumed	0.00E+00	1.35E-08
Water consumption - aquatic ecosystems	species.yr/m³ consumed	6.04E-13	6.04E-13

* EU27 value based on SimaPro analysis.

Valuation

Based on Table 57, we calculated environmental prices. We chose not to include a valuation for damage costs to human health for water consumption in the lower and middle values because some of these effects may already be discounted in the price consumers pay for water. For ecosystem services, we assume the individualistic worldview in the lower value. For the central and upper value, we assume the hierarchical worldview.

Table 58 provides the corresponding environmental prices for water consumption, where the effects have been valued using the valuation framework in Chapter 5.

Table 58 - Environmental prices of water consumption at midpoint level, in $€_{2021}/m^3$

Midpoint	Unit	Lower	Central	Upper
Water consumption – EU27	€/m³	€0.00	€0.41	€0.81

6.13 Land use

6.13.1 Description of themes and effects

Large-scale agriculture, residential development and business park development all affect land use and land-use change. If this land use affects natural values, there is a welfare cost. By considering ecosystem services associated with land use, a value can also be allocated to land use.

Land use as a result of a particular economic activity should be compared with how the land would be used if that activity did not take place. Two options are available here:
- comparison with a reference value for average land use in a country;
- comparison with a reference value for nature in a country.

There is a preference in the literature for the second approach, specifically for applications in LCA and for companies. For example, the Draft Methodology for Standardised Natural Capital accounting for business states that land use should be valued according to the ‘pristine state of nature’, regardless of whether that land had a previous use occupation.[80]

[80] See (Transparant, 2021).

6.13.2 Treatment in the Environmental Prices Handbook 2018 and updates

For the value for land use in SCBAs, the Environmental Prices Handbook 2018 refers to the SCBA Nature Working Guide (CE Delft, 2017b). This Working Guide contains practical tools to arrive at a valuation for land-use change in the Netherlands. For use in LCAs, the valuation of species richness was considered using (Kuik, et al., 2008). This valuation was then discounted over 50 years with a discount rate of 3%.

Before the update, these steps were checked again. It was found that discounting the values from (Kuik, et al., 2008) is incorrect, because the values calculated therein are already in PDF.m^2.year and the discounting did not involve the cumulative effects but that the value was discounted in one year. As such, relatively low values were obtained.

In this Handbook, we have revised the land use values and based them on the following assumptions:
We assume land use for a period of 50 years. This is half the hierarchical period of 100 years and typical for the lifetime of a factory in SimaPro. We assume that the consumption of a material contributes to sustaining the production of the factory.
- The loss of species richness over that 50-year period is discounted for each year with an interest rate of 2.25%.
- No adjustment is made for future increases in value for biodiversity. The valuation of future biodiversity loss is thus done by using the 2021 valuation as a starting point.
- After those 50 years, we assume that nature recovers. We have assumed the recovery values as formulated in ReCiPe 2016: on average, this amounts to a recovery period of 33.9 years.

6.13.3 New land-use values

Based on this, and the characterisation factors from ReCiPe 2016, it is possible to estimate the annual cost per hectare for biodiversity loss from land occupation. These costs are shown in the following table.

Table 59 - Estimated costs of biodiversity loss for various land use types and the midpoint characterisation factor in €$_{2021}$/m^2 per year

Average for the EU27	Lower	Central	High
	€ 0.037	**€ 0.053**	**€ 0.069**
Value of forest	€ 0.021	€ 0.030	€ 0.038
Value of grasslands	€ 0.038	€ 0.054	€ 0.071
Value of agriculture, annual crops	€ 0.070	€ 0.099	€ 0.128
Value of agriculture, perennial crops	€ 0.049	€ 0.069	€ 0.090
Value of mixed farming	€ 0.023	€ 0.033	€ 0.042
Value of other (urban, park landscape)	€ 0.051	€ 0.072	€ 0.094
Midpoint characterisation factor: m^2-a crop-eq.	**0.070**	**0.099**	**0.128**

The midpoint characterisation factor was obtained by weighting the value by land-use type with area data from Eurostat.[81]

[81] The following statistics were used: Land cover overview by NUTS 2 regions [lan_lcv_ovw], data voor 2018, extracted 30.10.2022.

7 Interpretation and additional analysis of environmental prices

7.1 Introduction

In this concluding chapter, we compare current environmental prices with the previous Handbook, discuss the use of environmental prices in practice and address the durability of environmental prices over time. This chapter is less theoretical than the previous chapters and focuses primarily on how environmental prices should be interpreted in practise.

The format of the chapter is as follows: in Paragraph 7.2, we compare the environmental prices developed here with the previous Handbook of 2018. We also briefly discuss other Environmental Prices Handbooks that have provided a valuation. In Paragraph 0 we discuss an additional application we have developed in this Handbook for use in LCAs: prices using PEF characterisation.

In Paragraph 7.4, we discuss the use of environmental prices over time and in specific situations and indicate how environmental prices can be adjusted. Finally, Paragraph 7.5 contains recommendations for future research and addresses the uncertainties that exist when calculating environmental prices.

7.2 Comparison of environmental prices

7.2.1 Pollutant-level comparison of environmental prices with Environmental Prices Handbook 2018

Environmental prices for most pollutants are higher than in the previous Handbook. The following table provides a comparison of these new environmental prices with the previous Handbook 2018 for the most common environmentally hazardous pollutants.

Table 60 – Comparison of environmental prices at pollutant level between the central values of the previous and current Handbook for EU27

Pollutant name	Formula	EPH2018: €$_{2015}$	EPH2024: €$_{2021}$	% change
Carbon dioxide*	CO_2	€ 0.06	€ 0.130	128%
Chlorofluorocarbons*	CFC_{11}	€ 306	€ 725	137%
Particulate matter	$PM_{2.5}$	€ 38.7	€ 95	146%
Particulate matter	PM_{10}	€ 26.6	€ 51.6	94%
Nitrogen oxides	NO_x	€ 14.8	€ 21.5	45%
Sulphur dioxide	SO_2	€ 11.5	€ 30.5	165%
Ammonia	NH_3	€ 17.5	€ 28.7	64%
Volatile organic compounds	NMVOC	€ 1.15	€ 2.49	117%
Methane	CH_4	€ 1.74	€ 4.68	169%
Cadmium	Cd	€ 589	€ 155,294	26,266%
Lead	Pb	€ 5,367	€ 27,287	408%

This table shows that most pollutants now have higher values than in the previous Handbook. Part of this increase can be attributed to inflation (around 10%) and a higher valuation because the average income in EU27 has increased. A positive income elasticity for human health (see Paragraph 5.3) has led to a higher valuation, implying around 10% additional price increase.

For particulate matter, the remaining price increase is mainly due to improving our knowledge of dose-response relationships (see Paragraph 6.4). The same applies to cadmium and lead, where more health endpoints are now included (see Paragraph 6.8). For ammonia and sulphur dioxide in particular, a higher proportion of these emissions are now converted into secondary aerosols: the updated modelling used in EEA (2021) has allowed a more accurate relationship between emissions and concentrations to be included in the new Handbook. For CO_2, the reduction targets are now stricter: climate neutrality in 2050 versus an assumed 65% reduction in 2050 from the previous Handbook. Stricter climate targets mean an increase of marginal cost of prevention to achieve these targets.

7.2.2 Environmental Prices Handbooks in other countries

Several European countries also have a similar handbook of Environmental Prices. In most cases, such handbooks also provide valuations for damage costs of air pollutant emissions that can be included in cost-benefit analyses. Of these countries, only the Netherlands and Germany have a longer tradition of regular updates of the Handbook's methodology.[82]

Table 61 shows that the valuations in the new Environmental Prices Handbook are generally higher for most pollutants compared to those from other countries. While the previous Handbook still generated values in line with those of other countries, this is no longer the case today. However, it is expected that if those other countries start adjusting their Handbooks reflecting new insights on dispersion of pollutants and valuation of environmental quality, these will also lead to higher values. This is, on the one hand, due to stronger environmental impacts (dose-effect relations) than a few years ago and, on the other hand, valuations are also higher arising from inflation and increased incomes.

The Dutch Handbook is the only Handbook that also makes the application suitable for LCA by developing a midpoint price.

[82] A revised version (version 4.0) of the Methodenkonvention (the German Environmental Prices Handbook) is expected in 2024.

Table 61 – Overview of environmental prices published in other countries for estimating damage costs of air pollution in that country

	Belgium (Flanders)	Ireland	EU27	Germany	Denmark
Study	>VITO, 2010 #28414<*	>EnvEcon, 2015 #28415<**	CE Delft (2024)	>Umweltbundesamt, 2019 #26416<	>Andersen, 2019 #2870<
Method	IPA (Needs adjusted)*	Econometrics	IPA (EEA adjusted)*	IPA (Needs)	IPA
Background to calculating damage costs air pollutant emissions					
Concentration of PM	Yes	Yes	Yes	Yes	Yes
Concentration of O_3	Yes	Yes	Yes	Yes	Yes
Concentration of NO_2	No	No	Yes	Yes	Yes
Concentration of toxic substances	Yes	No	Yes		
WIE (2013) CRFs?	No	No	Yes^^	Yes^	Yes^
Sectoral aggregation	Transport, other (services, industry, electricity, households)			Transport, built environment	Transport, industry, other (including households, agriculture, shipping)
Spatial differentiation	High/low chimneys Transport (rural, urban, highways),	Average, rural and different city sizes	Range (low, central, high) For $PM_{2.5}$ also differentiation by level of emissions and population		Regions and population density
Mortality included?	Yes	Yes	Yes	Yes	Yes
Morbidity included?	Yes	Yes	Yes	Yes	Yes
Biodiversity included?	Yes	Yes	Yes	Yes	No
Ecosystem services	Crops	Unclear	Yes (full)	Crops	No
Buildings/materials	Yes	No	Yes	Yes	No
Price level	2009	Unclear	2021	2016	2016
$PM_{2.5}$ (€/kg)	22-141	7.5	95	58.4	76.8
PM_{10} (€/kg)	17-125	NA	51.6	41.2	NA
SO_2 (€/kg)	10	4.8	30.5	15.04	40.9
NO_x (€/kg)	6.3	1	21.50	17.93	34.1
NH_3 (€/kg)	NA	0.8	28.70	32	20.1
NMVOC (€/kg)	7.5	0.9	2.49	0.205	NA
CO_2 (€/t)	20	NA	50-160	180-640	NA
Value of years of life lost (VOLY)	€44,379	Unclear	€85,000	€70,000^^	€149,637

* Adjusted means that an original study was used as a baseline, but these results were modified.
^^ Not mentioned in the report but based on personal communication with the authors.

7.2.3 Comparison of midpoint prices in previous and new Handbook

The midpoint prices in the previous and new Handbook cannot be properly compared because they are set in different units: the environmental prices in the Environmental Prices Handbook 2018 were set according to the characterisation (and units) from ReCiPe 2008 and the 2024 environmental prices were set according to the characterisation (and units) from ReCiPe 2016. What is possible, however, is to show the differences between the two environmental prices in an LCA analysis. Based on Ecoinvent data we compare a product score following ReCiPe 2008 with the previous environmental prices and the other score following ReCiPe 2016 with the new environmental prices.

We take the example of PVC. From Ecoinvent, we extracted the results of 1 tonne of PVC for both ReCiPe 2008 and ReCiPe 2016. We then valued impacts using the previous EU Handbook and the new EU Handbook, respectively. The following table shows the results in external costs in € per tonne PVC.

Table 62 – External cost of producing 1 tonne of PVC

ReCiPe 2016 Impact category	Damage costs (€)	ReCiPe 2008 Impact category	Damage costs (€)
Global warming	€ 392	Climate change	€ 164
Stratospheric ozone depletion	€ 0	Ozone depletion	€ 0
Ionising radiation	€ 0	Terrestrial acidification	€ 83
Ozone formation, Human health	€ 13	Freshwater eutrophication	€ 0
Ozone formation, Terrestrial ecosystems	€ 3	Marine eutrophication	€ 1
Fine particulate matter formation	€ 431	Human toxicity	€ 31
Terrestrial acidification	€ 51	Photochemical oxidant formation	€ 10
Freshwater eutrophication	€ 1	Particulate matter formation	€ 199
Marine eutrophication	€ 1	Terrestrial ecotoxicity	€ 3
Terrestrial ecotoxicity	€ 8	Freshwater ecotoxicity	€ 0
Freshwater ecotoxicity	€ 0	Marine ecotoxicity	€ 0
Marine ecotoxicity	€ 0	Ionising radiation	€ 4
Human carcinogenic toxicity	€ 164	Agricultural land occupation	€ 3
Human non-carcinogenic toxicity	€ 58	Urban land occupation	€ 1
Land use	€ 5	Natural land transformation	€ 0
Mineral resource scarcity	€ 0	Water depletion	€ 0
Fossil resource scarcity	€ 37	Metal depletion	€ 0
Water consumption	€ 18	Fossil depletion	€ 0
Total	**€ 1,181**		**€ 500**

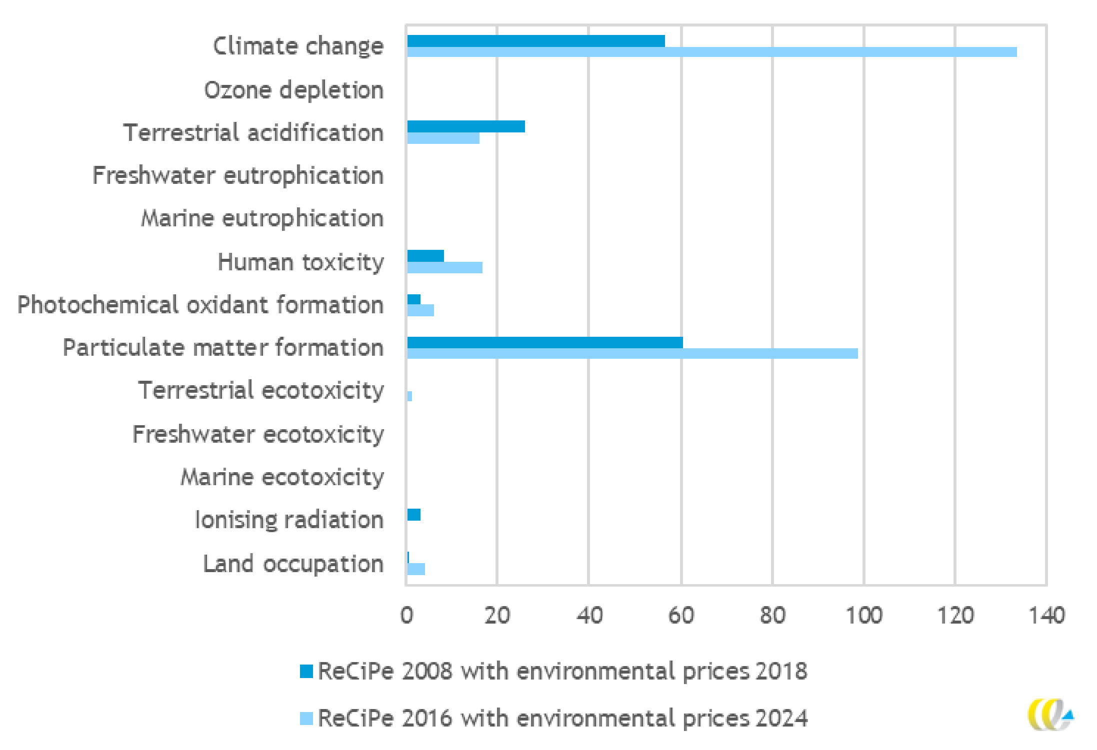
Extract from SimaPro for Polyvinyl chloride, emulsion polymerised {GLO}| market for | Cut-off, S (of project Ecoinvent 3 - allocation, cut-off by classification - system) according to ReCiPe 2008 ReCiPe Midpoint (H) V1.13 / Europe Recipe H and ReCiPe 2016 Midpoint (H) V1.07 / World (2010) H.

The table shows that the total damage costs of emissions of PVC production are more than twice as high as in the previous environmental prices. This is largely because the damage costs of PVC are primarily determined by particulate matter formation, climate change and human toxicity. These are valued considerably higher with the new environmental prices than in the previous Handbook, which is primarily due to an accurate Impact Pathway approach.[83]

[83] The extent to which a score becomes more significant varies by product and depends in part on the characterisation factors and the extent to which those characterisation factors are weighted in the midpoint determination.

Figure 13 provides a similar comparison of a tonne of engineering steel using both methods. This indicates that steel also has higher damage costs in the new Handbook, mainly due to climate change and particulate matter formation, and to a lesser extent human toxicity and photochemical oxidant formation. Acidification leads to slightly lower damage costs than in the previous Handbook, as holds for radiation as well. In contrast, land use and land-based ecotoxicity score higher again. The other midpoints are not relevant to this product.

Figure 13 - Comparison results at LCA midpoint level for the previous and new Handbook



7.3 Specialist application: midpoint prices according to PEF methodology

Environmental prices for use in an LCA are always developed specifically for a characterisation methodology. There is no straightforward way to transfer environmental prices from one characterisation methodology to another: this would require a recalculation of the environmental prices.

The European Union is currently working on the Product Environmental Footprint (PEF) for measuring and assessing the environmental impact of products. The objective of the PEF is to help companies develop responsible products and inform consumers about the environmental impact of their purchases.

The Product Environmental Footprint (PEF) is a European method, co-developed by the Joint Research Centre (JRC) that is recommended by the European Commission for assessing the environmental impact of products and organisations. The PEF partly uses other environmental models and therefore has a different characterisation than ReCiPe (see also

Annex B). The PEF is standard in LCA assessment methods, such as European EN15804-A2, and will gradually be more widely used.

In this Environmental Prices Handbook, we have converted environmental prices at pollutant level to midpoint prices according to the PEF. The PEF assigns three uncertainty categories to the various midpoints. In determining environmental prices, we limited ourselves to the categories rated as ‘recommended and satisfactory’ (CAT I) and ‘recommended but in need of some improvement’ (CAT II). Category III impacts could not be determined within the timeframe of this Handbook and could be added in future.

Although the use of PEF is not mandatory at present for all products, it could play a more central role in the future in substantiating environmental claims. For example, EN15804, a European standard for assessing the environmental impact of buildings and building products, already prescribes the use of the PEF.

For the purpose of this Handbook, we have therefore developed a preliminary valuation for the midpoint characterisation numbers of the European Product Environmental Footprint (PEF) methodology (EF v3.0) for robustness categories I and II. These are effects that can be represented with some degree of certainty in the PEF.

7.3.1 Methodology

The main methodology for determining midpoint prices according to the PEF is exactly the same as for the ReCiPe application: individual environmental prices are derived for about 20 main pollutants, which are then distributed among the various midpoints. In doing so, the following additional calculations were prepared by us:
- For acidification and eutrophication, it was necessary to break down the effects of NO_x and NH_3 emissions for acidification and eutrophication. We are not aware of much literature where such a breakdown has been applied. For example, it is known from (Langner & Bergström, 2005) that NO_x emissions contribute 20% to acidification and 80% to eutrophication in Sweden. Presumably that proportion is even lower now, but this cannot be verified by a good reliable source. We have assumed this percentage and assume that the contribution of NH_3 to acidification is still half as much as NO_x.
- The effects of acidifying emissions on buildings have been added to the acidification theme. The effects of particulate matter formation on buildings were all translated into PM_{10} equivalents and added up when determining the price for the midpoint factor disease-incidence (which therefore encompasses a wider range of damage than human health alone).
- For ozone depletion, climate change and eutrophication to water, the characterisation models in the PEF are mostly the same as in ReCiPe and the valuation from ReCiPe was taken to establish the valuation in the PEF.

7.3.2 Results

Table 63 shows the results for the EU27 for the PEF methodology.

Table 63 - Environmental prices midpoints according to PEF methodology for the EU27, in €$_{2021}$ per unit

Name of Environmental theme PEF	Unit		Lower		Central		Upper
Climate change	kg CO_2-eq.	€	0.05	€	0.130	€	0.16
Ozone depletion	kg CFC-11-eq.	€	15.23	€	29.1	€	69.57
Ionising radiation	kBq U235-eq.	€	0.00046	€	0.00071	€	0.00100
Oxidant formation, human health	kg NCSRC-eq.	€	1.04	€	1.48	€	2.04
Particulate matter formation	Disease incidence	€	538,733	€	890,182	€	1,267,004

Name of Environmental theme PEF	Unit	Lower	Central	Upper
Acidification	Mol H+-eq.	€ 0.58	€ 2.04	€ 4.64
Freshwater eutrophication	kg P-eq.	€ 2.56	€ 3.74	€ 10.13
Marine eutrophication	kg N-eq.	€ 7.64	€ 14.25	€ 27.60
Terrestrial eutrophication	Mol N-eq.	€ 0.23	€ 0.331	€ 0.43

Annex B explains more about the PEF impact categories and how they differ from ReCiPe.

7.4 Environmental prices adjustments

7.4.1 Adjustments over time

The environmental prices from this Handbook can be used during a certain period until the next revision. Because environmental prices are a composite price of effects on human health and effects on ecosystem services, it is not easy to make a conversion based on future adjustments of valuation of human health and ecosystems. Indeed, human health is discounted with an income elasticity (see Paragraph 5.3.5), while ecosystem services have an annual price increase of 1% due to increasing scarcity. Greenhouse gases, on the other hand, have a different rate of value increase, as explained in Paragraph 6.3.

This means that the environmental price of a pollutant such as methane in the Environmental Prices Handbook consists of several components:
- a price for the greenhouse effect of methane;
- a price for ecotoxicity;
- a price for oxidant formation with effects on human health.

Among these, the first effect is dominant, which is also true for other pollutants that have both a greenhouse effect and other effects.

In general, we recommend the following approach:
- Always adjust all prices to the current price level based on the harmonised consumer price index.[84]
- Further increase the price of GHG emissions with autonomous price increases as outlined in Paragraph 6.3
- For prices of emissions other than GHGs, we recommend adjusting the entire pollutant price for income, using income elasticities in the lower, central and upper values of 0.3; 0.65 and 1, respectively.[85] By adjusting the entire pollutant price based on income adjustments and the income elasticities, the most significant category (human health) is corrected. Moreover, this correction, with an economic growth of 1.5% in the central value, also aligns with the increase in the value of biodiversity.

If prices are adjusted in this way, they will not start deviating too much from the methodology used in the Handbook over the next 5-7 years. These adjustments are non-fundamental, as they do not change the underlying methodology for determining environmental prices.[86]

[84] For inflation, we use Eurostat data on ‘HICP - annual data (average index and rate of change) [prc_hicp_aind]’.
[85] We use incomes in Purchains Power Parities from Eurostat: ‘Purchasing power parities (PPPs), price level indices and real expenditures for ESA 2010 aggregates [prc_ppp_ind]’.
[86] For fundamental adjustments, see Section 7.6.2.

We do not recommend corrections for the midpoint characterisation factors, as this is an even more complex process that could alter the relative relationships between midpoints. However, the outcome of an LCA calculation valuing midpoints with environmental prices could be adjusted to the price level with the consumer price index in accordance with the rule above.

7.4.2 Adjustments for location

Environmental prices apply to average emissions from an average location in the EU27. Especially for air pollutants and fertilising emissions, the actual damage costs are highly dependent on the type of emissions (high chimneys or low to the ground) and location of emissions in relation to inhabitants, ecosystems and buildings.

For effects on human health from particulate matter (primary) and NO_x, we calculated and reported specific values for non-average emissions in Paragraph 6.4.10. Such approaches are also possible for other pollutants but have not been conducted in context of this Handbook.

7.4.3 Adjustments for regions or countries

The methodology of the Environmental Prices Handbook can also be applied to other countries if sufficient data are available. Two options are available:
- The models used in the Environmental Prices Handbook are basically European models. This could, in principle, make them applicable to EU27 countries to derive country-specific prices. However, this does involve a significant conversion that has been beyond the scope of the current Handbook.
- One can transform the results for the EU27 using a *benefit transfer model*. A benefit transfer model translates the results from the EU27 to specific countries (inside or outside the EU) by correcting for differences in population density, dispersion of environmentally hazardous substances through the various compartments and physio-chemical processes, such as atmospheric chemistry, among others. (CE Delft, 2011) includes a description of such a benefit transfer model to enable conversion to other countries.

7.4.4 Environmental prices applied to other characterisation models

Environmental prices depend on the method of characterisation. In the Environmental Prices Handbook, we develop weighing ratios associated with the characterisation used in ReCiPe 2016 and the PEF. These prices cannot be used for other characterisation models, such as CML2 or ReCiPe 2008.

However, the prices can in theory be converted to other characterisation models. This follows a two-step procedure in which:
1. All themes for which environmental prices were determined via emission-weighted individual pollutants must be recalculated according to the different characterisation model.
2. All themes for which environmental prices have been determined via endpoint valuation should be converted according to a simulation that links environmental prices to the new characterisation models for a large number of pollutants. The simulation can use Monte-Carlo-like techniques and be based on emissions, or be based on the relative importance of the pollutants in international policy lists.

Both approaches go far beyond the scope of this Handbook.

7.4.5 Adjustments for other pollutants

Finally, it is also possible to convert environmental prices to pollutants that are not in the databases. Conversions have been made in various ways in recent years:

1. Prices for pollutants can sometimes be converted because they have similar effects to substances for which environmental prices have been developed. Through REACH safety sheets (ECHA, 2023) one can check whether chemical and environmental properties of substances match.
2. Toxicological and/or epidemiological literature can be consulted with regard to other substances, especially those with health effects. If such literature is available, an environmental price can be calculated using the methodology explained in Paragraph 5.3 and Annex C, and applied to ultrafine particulate matter, among others, in this Handbook. The calculation becomes even more accurate if a dispersion model is also available that can track the dispersion of emissions through the environment.

7.5 Uncertainty and recommendations

7.5.1 Uncertainty

Environmental prices are subject to uncertainty: in many cases, it is not known exactly how emissions spread through the environment, what effects they cause, and how those effects should be valued. The Handbook does not include a mathematical algorithm of how to quantify this uncertainty. However, Annex G does provide a more intuitive estimate of the extent to which this uncertainty can affect the results.

Finally, below we present a few recommendations for future research that we intend to include in this Handbook to bridge the knowledge gap.

7.5.2 Studies on harmfulness and accumulative substances

For human toxicity, the results are primarily based on the analysis of (EEA, 2021) in which mainly the carcinogenic effects are monetised. In the analysis in Paragraph 6.8, we showed that there is a broader palette of adverse effects for toxic substances that are currently not monetised. Therefore, human toxicity valuations underestimate actual health risks.
In future research, additional analyses specifically for human toxicity could be carried out for a more precise estimate of the damage costs arising from emissions and dispersion of these substances.

Moreover, for pollutants with little or no biodegradation, such as PFAS, the knowledge base is currently insufficient to arrive at an estimate of damage costs. Precisely because these substances play such an important role in the public debate, more research on PFAS would be recommended. This could include risk models to determine the damage costs based on the probability that bioaccumulative substances are also classified as harmful to health at a later stage.

For ecotoxicity, the knowledge gap is even larger. Although the valuation of ecosystem services is in line with the international literature, knowledge on dose-effect relationships is still very much underdeveloped here. To what extent does environmental pollution contribute to a reduction in ecosystem services? More primary research in these areas would help to obtain a more accurate estimate of the harmful effects of environmental pollution.

7.5.3 Recommendations around the sustainability of environmental prices

Environmental prices should be periodically adapted to changing scientific understanding of the harmfulness of environmental pollution, changing background concentrations of pollution, changing valuation as people become richer and changing population size and composition.

Currently, an update of the Environmental Prices Handbook (or its predecessors called shadow prices) has been published every six to eight years. Previous versions of the Environmental Prices Handbook and the Shadow Prices Handbook (for the Netherlands) were published in 1997, 2002, 2010 and 2017 (the latter with an EU-version in 2018). Six years appears to be a reasonable timeframe within which sufficient things have changed to warrant an update.

However, fundamental adjustments are needed reflecting more systematic adjustments of underlying determining factors of damage costs. This could happen, for example, if there are changes to the valuation of a human life year or of ecosystem services. An adjustment may also be needed if the WHO decides to release new insights on the harmfulness of environmental pollution. A new HRAPIE will be released in mid-2024. We have anticipated this by updating the data on relative risk of death with newer data that is likely to be adopted by the new WHO publication.

In addition, the characterisation factors can be further adjusted, either within ReCiPe or within the PEF. This may also lead to a desire to update environmental prices to match what is standard in the LCA.

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A Impact Pathway modelling

A.1 Introduction

The damage calculated on the environmental themes acidification, photochemical oxidant formation and particulate matter formation were determined directly through an adjustment of the results that came from the EEA 2021 study. In this annex, we explain the assumptions behind the original EEA study and which adjustments have been made. Finally, we also compare the results with those obtained in the Environmental Prices Handbook 2018.

A.2 Basic models

The Impact Pathway approach implicitly underpins many environmental prices in this Handbook. We have employed various models striving to maintain consistency in assumptions and valuation methods across the models.

Human health effects of air pollution

We use results from the EEA 2021 project to determine environmental prices related to human health. At our request, the authors of this study also provided some additional data, allowing us to perform our own calculations.

Human health from toxic substances

Here, too, we rely on the results from (EEA, 2021), which provides European values for the impact of toxic emissions. We further differentiate these impacts at the individual country level by using earlier results from the NEEDS 2008 project.

Impact on crops

For oxidant formation, we use data from (EEA, 2021). Acidification impacts were assessed using earlier NEEDS model results, consistent with the Environmental Prices Handbook 2018.

Impact on biodiversity due to acidification and eutrophication

For the impact of air pollution on biodiversity, we rely on NEEDS model 2008 results, in line with the Environmental Prices Handbook 2018.

Effects of radionuclides

For the impacts of radionuclides, we rely on NEEDS modelling results, in line with the Environmental Prices Handbook 2018.

Effects of eutrophication pollutants on water

For the impacts of eutrophication pollutants on water, we rely on the modelling from (IEEP et al. , 2021).

Other effects (ecotoxicity and ozone depletion)

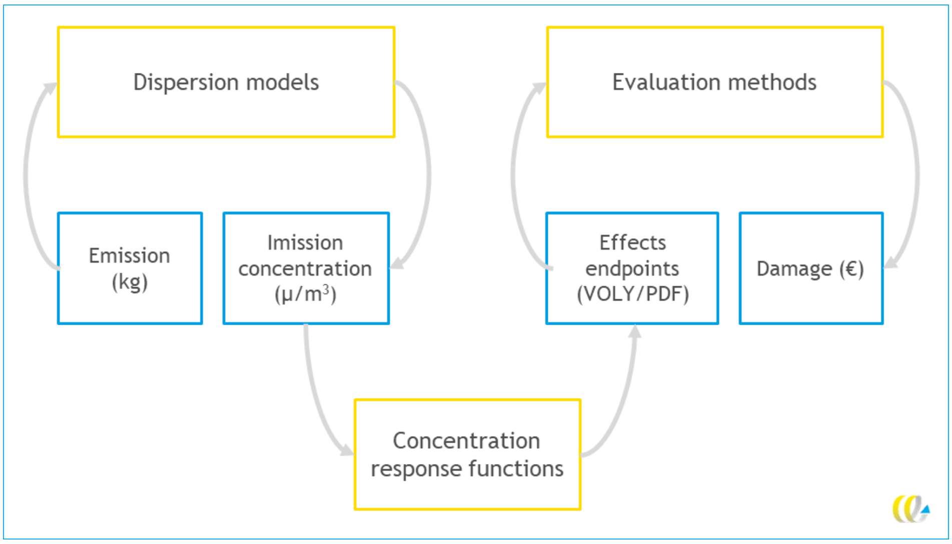
For the remaining impacts, we rely on the Impact Pathway modelling underlying ReCiPe 2016 (Huijbregts, et al., 2016).

A.2.1 Basic methodology: Impact Pathway approach

The group of 'classical air pollutants' includes sulphur dioxide (SO_2), nitrogen oxides (NO_x), particulate matter (PM), ammonia (NH_3) and non-methane volatile organic compounds (NMVOC). These pollutants have been modelled extensively in the EEA project, with additional, separate estimates provided for emissions of toxic substances.

The EEA project employed the Impact Pathway approach. This approach establishes a link between emissions and the resulting damage, expressed in monetary terms. These relationships are developed through a series of models (see the following figure).

Figure 14 - Impact Pathway approach



The different steps are described below.

Step 1: Source emissions

This step identifies, within a geographical grid, the sources of emissions. In the EEA (2021) model, sources are distributed across various grid cells with a horizontal resolution of 0.2° x 0.3°. For NO_2, a more refined resolution of 7.5 x 7.5 km was used. Older models, such as NEEDS (used in this Handbook to determine the impacts of acidification/ eutrophication on biodiversity) have a spatial resolution of approximately 50 x 50 km^2.

After identifying source emissions, scenarios are formulated. For example, EEA (2021) used a 15% reduction, while the NEEDS model included scenario's reflecting current and planned policies. The NEEDS model formulated emission scenarios with firm and intended policies.

Step 2: Dispersion (Dispersion-Receptor sites)

This step translates emissions into concentrations at specific geographically diversified receptor points (sometimes called immissions). For classic air pollutants, dispersion and chemical transformation in Europe have been modelled using the EMEP/MSC-West Eulerian model, which also includes meteorological data.[87] 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 cell across Europe. Model runs have been performed for a 15% reduction of each airborne pollutant (see Step 1).

It should be noted that the chemical reactions and interactions are very complex. For example, a reduction in 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 sites. Because there is a relatively high NH_3 in certain areas of the EU in particular due to intensive agriculture, concentrations of NO_x and SO_2 do not decrease uniformly as emissions are reduced. In the final framework, this means that the damage per kg of emissions of NO_x and SO_2 is higher if NH_3 emissions are not significantly reduced.

Step 3: Concentration response functions and effects

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 (CRF) and with reference to the size of the exposed population, physical impacts have been calculated for each grid cell based on dose-response functions from epidemiological research. These studies link pollutant intake (or air presence) to health or biodiversity impacts. CFRs are applied to mortality and morbidity outcomes (see Annex A.3).

Step 4: Monetary valuation

The final step is monetary valuation. Chapter 5 outlines the valuation framework for health impacts (Paragraph 5.3), biodiversity impact (as a proxy for ecosystem services, see Paragraph 5.4) and the impact of air pollution on buildings and materials (Paragraph 5.5). These valuations are detailed in Chapter 5. Independently, impacts on agriculture crops are quantified based on EEA (2021), where they were valued using market prices.

[87] This model was used in both the EEA and NEEDS projects.

A.3 Update: human health from classic air pollution

Classic air pollution refers to pollutants that have been recognised and regulated since the 1990s under various European clean air treaties, such as the NEC directive or the Gotheburg Protocol. These pollutants include PM_{10} (including $PM_{2.5}$), SO_2, NO_x, NH_3 and NMVOC. These pollutants are known to produce health impacts. Health impacts are endpoints that can be modelled using the IPA. Two crucial elements in this approach are the definition of concentration response functions (CRF) and the monetary valuation of health impacts.

A concentration response function (CRF) establishes the relationship between health impacts and the concentration of air pollutants. For example, it may indicate the years of life lost (YOLL) per 10 µg of $PM_{2.5}$. The CRF depends on two underlying variables:
1. 'Relative Risk' (RR). This indicates the portion of the disease burden attributable to air pollution. Relative risk is a measure that compares the risk of a specific health condition between a group exposed to air pollution and a group not exposed.
2. 'Incidence'. This reflects how often the disease burden occurs within the population. The incidence, or also 'incidence rate', provides an endpoint measure, such as the actual number of hospital admissions within a given year.

For minor variations, the CRF equals the product of the increase in relative risk due to air pollution and the incidence.

The Environmental Prices Handbook 2018 uses CRFs from the NEEDS project, where some CRFs were adjusted for the higher relative risks in (WHO, 2013a) compared to earlier WHO guidelines.[88] However, no correction was made for the change in incidence over time.

In the current Handbook, we adapt the methodology and base it on the HRAPIE >WHO, 2013 #4492< project, which provides recommended relative risks for each health impact. The advantage of this method is that it allows for the use of up-to-date health data, such as the actual number of hospital admissions and deaths in a given year. Many of the incidences from the NEEDS project were based on data from the 1990s, which is now considered outdated for the 2024 update of the Environmental Prices Handbook. Over time, the population has generally become healthier, and survival rates for conditions like cancer have improved.

A.3.1 Emissions and dispersion: EEA

For emissions and dispersion, we use generalised results from (EEA, 2021), which report damage costs per environmental theme ($PM_{2.5}/PM_{10}$; O_3; NO_2). By recalculating based on the valuations and incidences used in the EEA study, we can calculate the per capita reduction in harmful concentrations resulting from emission reductions in the EU.

A.3.2 Relative risks

For the concentration response functions, we base ourselves largely on (WHO, 2013a), as does (EEA, 2021). The HRAPIE project shows a relative risk for each endpoint per 10 µg of the pollutant. The following table lists these relative risks. In the last column, we indicate which adjustment to this overview we are using for this Handbook.

[88] This correction had been applied partially in the Dutch-language Environmental Prices Handbook (CE Delft, 2017) and fully in the EU28 version (CE Delft, 2018) and the Handbook of External Costs of Transport (CE Delft, 2019).

Table 64 – Overview of relative risks from (WHO, 2013a) and adjustments in this Handbook

Endpoint	Age group	RR per 10 µg (WHO)	New RR
$PM_{2.5}$			
Mortality, all natural causes	30+	1.062	1.08
Hospital admissions, cardiovascular disease	All	1.0091	
Hospital admissions, respiratory organ diseases	All	1.019	
Restricted Activity Days (RAD)	All	1.047	
Work Loss Days (WLD)	20-65	1.046	
Days of asthma symptoms among children with asthma	5-19	1.028	
PM_{10}			
Post neonatal infant mortality	0-12 months	1.04	
Incidences of bronchitis in children	6-12	1.08	
Incidences of chronic bronchitis in adults	18+	1.117	
O_3			
Mortality, all natural causes	All	1.0029	
Hospital admissions, cardiovascular disease	65+	1.0089	
Hospital admissions, respiratory organ diseases	65+	1.0044	
Minor Restricted Activity Days (MRAD)	All	1.0154	
NO_2			
New cases of bronchitis symptoms in children with asthma	5-14	1.021	
Mortality, all natural causes (short-term)	All	1.0027	
Hospital admissions, respiratory organ diseases	All	1.018	
Mortality, all natural causes (long-term)	All	1.055*	1.01

* Includes a confounding effect with $PM_{2.5}$.

We made two adjustments to these (WHO, 2013a) recommendations: mortality from $PM_{2.5}$ is based on (Chen & Hoek, 2020) and mortality from NO_2 is based on (COMEAP, 2018). These adjustments are described in Paragraphs 6.4 and 6.5, respectively.

A.3.3 Incidence of disease burden

To estimate an environmental price based on relative risk, we need the incidence or disease burden, for each endpoint. We use 2019 as our reference year, as it is the most recent year unaffected by the COVID-19 pandemic. In the following table, we summarise the data available for incidence data. We consulted a number of sources: data at Eurostat, health data from the Global Burden of Disease database (IHME, 2019), and the WHO sources as followed in the EEA 2021 study, as also reported in (Holland, 2014b). In bold are the data we use in estimating the environmental price and those we consider most reliable. The selection is based on the best available data at, which is EEA in this case. For Eurostat, data for a number of countries are missing, and as such, we do not use Eurostat. For mortality rates, we use the Global Burden of Disease data due to the requirement for detailed data for each year of life. For data on lost work or activity days, we follow the WHO recommendations, which were also followed in the EEA study. These provide guidelines on how to estimate incidence. For more accurate country-specific data (such as for cases of chronic bronchitis), we follow the Global Burden of Disease.

Table 65 - Summary of incidence dates for the year 2019

Endpoint	Unit	GBD	Other (EEA)*
$PM_{2.5}$			
Mortality, all natural causes (30+)	Deceased	**4,477,923**	–
Hospital admissions, cardiovascular disease	Hospital admissions	–	**11,548,480***
Hospital admissions, respiratory organ diseases	Hospital admissions	–	**5,327,107***
Restricted Activity Days (RAD)	Days	–	**6,485,399,759****
Work Loss Days (WLD) (20-65 yr)	Days	–	**1,802,809,294***
Days of asthma symptoms among children with asthma (5-19 yrs)	Days	–	**194,273,383***
PM_{10}			
Post-neonatal infant mortality (< 1 yr)	Deceased	**12,091**	–
Incidences of bronchitis in children (6-12 yrs)	Incidences	540,065	**6,050,773***
Incidences of chronic bronchitis in adults (18+)	Incidences	1,735,167	**1,423,287***
O_3			
Mortality, all natural causes	Deceased	4,506,407	–
Hospital admissions, cardiovascular disease (65+)	Hospital admissions	–	5,128,949*
Hospital admissions, respiratory organ diseases (65+)	Hospital admissions	–	2,057,284*
Minor Restricted Activity Days (MRAD)	Days	–	3,482,282,263*
NO			
Incidences of bronchitis symptoms in children with asthma (5-14 yrs)	Incidences	–	1,837,341*
Mortality, all natural causes (short-term)	Deceased	4,506,407	–
Hospital admissions, respiratory organ diseases	Hospital admissions	–	5,327,107*
Mortality, all natural causes (long-term)	Deceased	4,506,407	–

* Own calculations regarding incidences of use in (EEA, 2021) based on guidelines from (WHO, 2013a) and explained in (Holland, 2014b), adjusted to the 2019 population size.

** This refers to the unadjusted days with limited activity. For application with the relative risk, days with limited activity still need to be corrected for lost working days and days with asthma symptoms among children. It is therefore a residual sum of RAD not covered by work absenteeism or asthma complaints.

A number of endpoints are further explained below with the underlying assumptions. The remaining incidence data comes directly from the aforementioned data source.

Restricted activity days (due to $PM_{2.5}$)

We take the incidence of this from the EEA study. It is based on the recommendations from the WHO publication (WHO, 2013a), which gives an average number of 19 RAD per person. We multiply these by the total population. We then reduce these by the total working days lost and days with asthma symptoms among children, due to the overlap.

Lost working days (due to $PM_{2.5}$)

We base the lost working days on EEA data. An average of 11.11 lost working days per person is given. We multiply this by the total population in the EU between the ages 20 and 65, corrected for the average working population rate (60%). This gives a total of 1.8 billion lost working days in 2019.

Days of asthma symptoms among children with asthma (due to $PM_{2.5}$)

The EEA assumes an average of 4.5% of children with a daily incidence of 17%, in line with WHO reporting recommendations. This amounts to 17% * 365 days * 4.5% = 2.8 days per child per year. We multiply this estimate by the number of children in the age group 5-19.
In total, this yields an estimated 194.3 million days with asthma.

Incidences of bronchitis in children (due to PM_{10})

Again, we base the incidence on the data from the (WHO, 2013a) as also followed in the EEA 2021 study. According to WHO reporting, an average incident rate of 18.6% of the children is assumed. Multiplied by the number of children, this gives the incidence for the year 2019.

Minor Restricted Activity Days (due to O_3)

The EEA follows the WHO report, which has an average of 7.8 MRAD per person. We multiply this number by the size of the age group in 2019 to determine the total incidence.

A.3.4 Years of Life Lost from lifetable analysis

To determine the number of life years lost due to premature mortality, a fixed average of 10.7 years, based on the NEEDS project, was used in the Environmental Prices Handbook 2018. In this update, we refine the method by using 'lifetables'. Such a lifetable uses two types of input: the population composition in a base year by age (the original cohort), and the number of deaths in the base year by age cohort. From this information, the probability of death by age in the base year can be calculated. The hazard rate is calculated as follows (2006):

$$Hazard\ rate\ (h) = \frac{number\ of\ deaths\ in\ age\ group\ (d)}{size\ of\ age\ group\ on\ January\ 1\ (p)}$$

The survival rate is then, logically, 1 minus the hazard rate.[89]

$$Survival\ rate\ (s) = 1 - hazard\ rate$$

[89] The IOM's original method uses a different formula: s = (2 - h) / (2 + h). This is because midyear population data are used, which means that a correction is needed on the survival rate: after all, part of the population has already died by midyear and would otherwise be double counted. However, we calculate with population data on 1 January, making this correction unnecessary. The survival rate then reduces to 1 minus the hazard rate.

The hazard and survival rate can be used to determine how long the original cohort will live *on average*. This method uses a cut-off limit of 105 years: in other words, we disregard the probability of a person becoming older than 105. By continuing the calculation until everyone in the original cohort has passed away, we can sum all the years of life of this cohort, from the time of measurement. This means the oldest individuals in the cohort have a smaller share, as most of their lives have already passed by the measurement year. Newborns in the base year contribute the longest to the calculation: after a maximum of 105 years, they too are assumed to have passed away.

Then, we use the relative risk to increase the hazard rate by year of age. For $PM_{2.5}$, the relative risk is 1.08 per 10 µg/m^3 for the population aged 30 years and over.
To determine the impact on life years for the original cohort due to an additional 1 µg/m^3 $PM_{2.5}$, we multiply the hazard rate from age 30 by 1.007726 in the base year.[90] We then recalculate the total years of life of the original cohort, from the measurement year until everyone in that cohort has passed. In the final step, we subtract these life years from the number of life years in the calculation without increased hazard rate. The result is the number of life years lost due to 1 µg additional concentration of $PM_{2.5}$ in the base year.

In the lifetable, the hazard rate is assumed to remain unchanged in the future. This in itself may not be a realistic assumption, given the advancing medical capabilities to cure diseases. However, when we want to calculate the impact of an increased hazard rate, we are only interested in the difference between developments with and without these increased risks. Scenario analyses have shown that an overall improved survival rate has a negligible impact on future outcomes. Therefore, we maintain the assumption of a constant hazard rate over time, as reliable alternative predictions are not feasible. We summarise the results of the calculation in the following table.

Table 66 - Result lifetable calculations, years of life lost 2019, EU27 for 1 µg/m^3 concentration

Pollutant	Endpoint	Relative Risk	Years of life lost per µg based on 2019 data
$PM_{2.5}$	Mortality, all natural causes (30+)	1.08	346,411
NO_2	Mortality, all natural causes (all)	1.01	43,577

Based on the NEEDS calculation, the number of years of life lost for 1 µg/m^3 $PM_{2.5}$ would be 290,637 (based on a CRF of 0.000651), approximately 55,000 fewer than with the lifetable method. The difference is explained by the updated relative risk, which is higher than the previous recommendation in (WHO, 2013a). For NO_2, the old method, based on a CRF of 0.000083, would yield 43,577 years of life lost, around 18,000 fewer than using the lifetable method. This difference is also explained by the increased relative risk we use in this Handbook and by the fact that the new insights also include the population younger than 30 years.

[90] The increase in the hazard rate was calculated as 1.08 ^ (1 / 10). This method is based on the methodology of IOM lifetables (2006).

A.3.5 Monetary valuation

Valuation endpoints of the WHO

The following table summarises the valuation by endpoint in environmental prices for particulate matter, nitrogen and ozone. The selection is explained under table by endpoint.

Table 67 - Overview of monetary valuation of health effects, in €$_{2021}$ per day or case

Endpoint	Lower value	Central value	Upper value	Source
VOLY/DALY/QALY	57,500	85,000	128,000	CE Delft (2017a)
Post neonatal infant morality	408,2431	6,208,720	8,364,679	OECD (2016)
Prevalence of bronchitis in children	285	407	407	OECD (2016)
Asthma symptoms in asthmatic children	40	57	81	EEA (2021); CE Delft (2017a)
COPD in adults	50,717	72,452	350,498	EEA (2021); OECD (2016)
Hospital admissions, CVDs (excl. stroke)	4,731	6,759	6,759	EEA (2021)
Hospital admissions, respiratory diseases	3,785	5,407	5,407	EEA (2021)
Lost working days	176	211	266	CE Delft (2017a); own calculation based on Eurostat data; National Health Care Institute (2016)
RADs (days of restricted activity)	104	148	190	EEA (2021); CE Delft (2017a)
MRADs (days of small, restricted activity)	57	81	81	EEA (2021); OECD (2016)

VOLY/DALY/QALY

The valuation of VOLY, DALY and QALY is further explained in Paragraph 5.3.6.

Post-neonatal childhood disease

In EEA (2021), a value of €5,860,000 per death is used based on OECD (2012b). This is a value 1.5 times higher than the VSL for adults (€3.9 million). Adjusted to 2021 price levels, this equates to a value of €6,208,720. This aligns closely to the converted upper value from the previous version of the Handbook.

We rely on the VSL value from the OECD (2012b) for the lower value, multiply it by a factor of 1.5 for the middle value and a factor of 2 for the upper value to give an estimate of the impact of premature death in children. This amounts to a value of €4,082,431 (lower value), €6,280,720 (central value) and €8,364,679 (upper value).

Prevalence of bronchitis in children

The EEA (2021) uses a value of €384 per 'event'. This value is based on (Hunt & Arnold, 2009). Adjusted to 2021 price and income levels, this equates to a value of €407. OECD (2016) reports a value of €55 for disutility plus cost of illness for mild bronchitis in the EU. Converted, this is €58.

We adopt the value of (Hunt & Arnold, 2009) as the central and upper value. This amounts to a value of €407 per bronchitis case. This is in contrast to a valuation per new patient, as in the previous Handbook. As the lower value, we assume 70% of the central value (€285).

Asthma symptoms in asthmatic children

The EEA (2021) uses a value of €54 per symptom day. Adjusted to the 2021 price level, this equates to a value of €57. This valuation is based on U.S. EPA (2011). In the CAFE study of Holland (2014b), a value of €295 per 'event' is used, with an additional valuation of €31 per symptom day.

We concur with the findings in EEA for the central value and lower value (2021). That is a value of €57 per day with symptoms in asthmatic children for the 2021 price point. We base the upper value on the central value in the previous Handbook (€81 per day). The lower value is set at 70% of the central value (€40).

COPD in adults

A variety of valuations are available for COPD cases among adults. The EEA (2021) study uses a value of €72,452 per case (in 2021 prices).

In Holland (2014b), a value of €200,000 (2003 price level) is presented as the central value. This value was determined based on a valuation scale factor (WTP risk trade off) of 0.32 relative to the VSL for a fatal traffic accident (€1 million). A second scaling factor of 0.42 was applied to scale down the WTP from 'severe COPD' to 'average COPD'.

The OECD (2016) bases the value for a COPD case on scaling relative to the VSL, representing a central value of $334,750. Adjusted to 2021 price and income levels, this is €350,498 in €2021.

The value used in the EEA study is significantly lower than the OECD recommendation and the value used in the Handbook 2018. The reason for this is unclear. For the time being, we have adopted the value from OECD as the upper value. We have adopted the value from EEA as the central value. The lower value is set at 70% of the central value.

Hospital admissions

The EEA 2021 study uses a value of €6,379 for hospitalisations due to heart disease, and €5,103 for hospitalisations due to lung disease. Adjusted to 2021 price levels, these values are €6,759 and €5,407 per hospital visit, respectively.

OECD (2016) provides various valuations. A specific value is available for the EU. This comprises treatment costs (€1,542) for three days in hospital. Finally, DEFRA provides an estimate of €3,069 for to cardiovascular and €2,873 for respiratory admissions (converted values). These values include the disutility and cost of illness of eight/nine days in hospital.

Most estimates are based on three-day hospital visits, based on either disutility (WTP) or nursing costs. For nursing costs, the average hospital stay duration is relevant. Both OECD (2016) and Defra (2020) are based on an average duration of three days.

For the central value, we use the values from EEA. The lower value is set at 70% of this. Due to lack of more specific information at EU level, we set the upper value equal to the central value.

Lost working days

A value of €166 (€176 in the 2021 price level) is used in EEA (2021). This is based on a publication by Amann (ed.), et al. (2017).
National Health Care Institute (Zorginstituut Nederland) assumes a productivity cost of €34.75 per hour. For an average working day of 6.2 hours (31 hours per week), this amounts to €215 per day ($€_{2014}$). This is €266 in 2021 prices, including income growth.

To best reflect the current situation, we base the value of a WLD on recent Eurostat wage and labour data. Using data from Eurostat allows for a harmonised methodology when determining the value for the EU27, for example. According to Eurostat, the following data apply to the EU27 in 2021:
- Labour costs were €29.1 per hour, averaged across all sectors.
- The average working week of Dutch workers was 36.2 hours.
- Assuming this is five working days, this amounts to 7.2 hours per day. The labour cost per day is then €211. We take this value as the central valuation for a work loss day. The lower value is the converted value from the previous version of the Handbook and the upper value is based on the calculation by the National Health Care Institute (Zorginstituut Nederland).

RADs and MRADs

Considering all other sources, the OECD recommends a value of €178 for RAD and €65 for MRAD for all OECD countries (in 2021 price levels). EEA uses converted values from Hunt, et al. (OECD, 2016) of €148 for RAD and €54 for MRAD.

We adopt the EEA values as the central value. The value corrected for inflation of the previous Handbook is adopted as the upper value for RADs. For MRAD we use the OECD valuation as upper value. We set the lower values at 70% of the values as used in EEA (2021).

A.4 Ultra-fine particulate matter

As explained in Paragraph 6.4.9, ultra-fine particulate matter has damaging effects on top of the effects of fine particulate matter. Based on RIVM (2022), we include the effects of four health risks in our calculation. Only those health effects with a clear association to increased concentrations of ultrafine particles — and that do not overlap with the health effects already included in the pricing for $PM_{2.5}$ — were considered. This led to the quantification of four health effects in the environmental prices: three from the research within the health monitor (diabetes, medication use for diabetes, and medication use for high blood pressure) and one from the medication use study (for dementia). We examined these further for quantification.

Table 68 – Relative Risks for health effects of ultrafine particulate matter

Health effect	Age group	Relative Risk (per 3,500 particles/cm^3)
High blood pressure (medication use)	19+	1.05 (1.00-1.11)
Diabetes (medication use)	19+	1.08 (1.00-1.17)
Diabetes (self-reported)	19+	1.16 (1.02-1.33)
Medication for dementia	40+	1.141 (1.013-1.286)

Source: (RIVM, 2022).

Relative risks were calculated based on an elevated concentration of 3,500 particles per cm^3. There is little data on the average concentration of ultrafine particulate matter in the EU or the Netherlands. To translate concentrations measured in particles per cm^3, to emissions measured in kg, we use data from the UK. In the UK, the National Atmospheric Emissions Inventory has recorded the emissions of ultrafine particulate matter. Over the years, $PM_{0.1}$ emissions have consistently been around 10% of the total PM_{10} emissions, measured in kilotonnes. We assume that the same ratio between PM_{10} and $PM_{0.1}$ applies to the situation in the EU. We therefore apply a 10% factor to the total PM_{10} emissions in the EU to estimate the total $PM_{0.1}$ emissions.

A more detailed explanation of the data used can be found below. Based on the four identified health effects, the associated health costs, the average concentration of ultrafine particulate matter in the EU and the estimated $PM_{0.1}$ emissions, the environmental price for ultrafine particulate matter is €438/kg $PM_{0.1}$ for the Netherlands. This represents the central value and is considered additional to the environmental price for the $PM_{2.5}$ emissions. Due to the limited data available, we have not established an uncertainty range. Instead, for simplicity, we applied the same spread as in the VOLY to estimate the lower and upper values of the environmental price. This results in an environmental price with a lower and upper estimate of €296 and €660/kg $PM_{0.1}$, respectively.

This environmental price is more than 3.5 times higher than the Dutch environmental price for $PM_{2.5}$. It is likely that this price is still an underestimate. To determine the environmental price more precisely, we would need translate emissions into ultrafine particulate matter concentrations, which would require a dispersion model that is currently unavailable for ultrafine particles. For $PM_{2.5}$, it is known that to result in a net export of emissions, from the Netherlands to neighbouring countries due to the Netherlands' coastal location and prevailing westerly winds. This means there is no direct relationship between emissions and concentrations, and the level of immissions is lower than the level of emissions. Some of the emissions therefore cause damage beyond national borders and this has not been quantified in this damage estimate. This means that the actual health damage is higher.

Monetary valuation

The health effects included in the environmental prices of ultrafine particulate matter consist of medication use for dementia, diabetes and high blood pressure. In addition, a value is given to the disease burden of diabetes. The values below are specific to the Netherlands.

The GIP database was used for the costs of medication. It provides information for each type of medicine on an annual basis regarding the number of users and daily doses (DDD) per user, as well as the reimbursement of the costs of a daily dose.

Table 69 – Medication costs per user per year

	DDD per user per year	Reimbursement per DDD	Cost per user per year
Diabetes medication	595	€ 0.46	€ 276
Dementia medication	254	€ 0.60	€ 152
High blood pressure medication	793*	€ 0.13*	€ 102

* Weighted average of different types of medication for hypertension with C02, C03, C07, C08, C09 and C10 classifications in the BIP database.

Costs for diabetes cases are based on the valuation for a DALY. Based on research by IHME, diabetes (without associated complications) has a disability weight of 0.049. This means that by having diabetes, on average, one lives almost 5% of the year in poorer health. By multiplying this weight by the value for a DALY, a life year with diabetes can be valued. Therefore, the valuation for living with diabetes comes to €4,165 per year, based on a DALY of €85,000.

Disease burden of ultrafine particulate matter

For the Netherlands, we also determined the price for ultrafine particulate matter. For the valuation of ultra particulate matter, not shown in Table 69, we use data on the number of people with diabetes. These figures are from CBS, from the healthcare monitor data that tracks self-reported diabetes cases. According to these data, there will be 848,000 people with diabetes in the years 2020/2021 in the Netherlands.

The other health effects concern medication use. For incidence data of medication use, we make use of the BIP database. This records the number of users per year, the number of doses issued and the reimbursements per medicine for each medicine group. According to this database, more than 836,000 people aged 19 years and overtake medication for diabetes, 19,000 over-40s take medication for dementia, and more than 3.5 million people aged 19 years and overtake one or more medicines for high blood pressure (in the Netherlands).

B Characterisation factors

B.1 Introduction and use goals characterisation

Characterisation factors are key indicators used to express how much a standard amount of a pollutant contributes to a particular environmental impact. Characterisation factors are developed using LifeCycle Impact Assessment models (LCIA). Many types of LCIA models are distinguished in the literature. For this Handbook, the LCIA models of ReCiPe and the PEF are of specific interest. These models provide characterisation factors for a wide range of pollutants.

Characterisation factors, as outcomes of LCIA models, have been used in three different ways in this Handbook:
1. As weighting factors for the damage costs of primary pollutants when determining the midpoint environmental price. This is Step 3, outlined in the framework from Chapter 4. In this step environmental prices are determined for each midpoint characterisation factor by dividing the total damage costs of individual pollutants by the total scores of the characterisation factors for a given set of emissions. This is explained in more detail in Paragraph 4.5.
2. As a characterisation factor to translate the environmental price per midpoint into a damage cost for each individual pollutant on that environmental theme. This is in line with Step 4 in the framework and is explained in Paragraph 4.6.
3. For the themes of ozone depletion, water consumption and ecotoxicity, endpoint-level characterisation factors are used to arrive at an environmental price valuation for that theme. This is explained in Paragraph 4.5.3.

This makes characterisation factors, in addition to the valuation framework and dose-effect determination of primary pollutants, a third important pillar of the Handbook. In this Annex, we explain which characterisation factors we have chosen.

B.2 Comparison ReCiPe 2016 and 2008

The following table presents the modelling for each environmental theme within ReCiPe 2016, compared to the 2008 version. This table shows that the main adjustments in the ***update*** to ReCiPe 2016 are:
1. Characterisation factors now represent impacts on a global scale rather than a European scale. However, it is still possible to apply characterisation factors at national and regional levels for the environmental themes of photochemical ozone formation, particulate matter formation, terrestrial acidification, freshwater eutrophication and water consumption.
2. Consistency between models to determine midpoint and endpoint impacts has been improved by using the same time horizon per cultural perspective (see Paragraph 3.2) across different environmental impacts.
3. The number of environmental interventions has been expanded, and the following new *damage pathways* have been added;
   - impact of water consumption on human health;
   - impact of water consumption and climate change on freshwater ecosystems;
   - impact of water consumption and tropospheric ozone formation on terrestrial ecosystems.

Table 70 - Overview of midpoints used in ReCiPe 2016 and differences from the earlier ReCiPe 2008 (in the 2013 version)

ReCiPe 2016			
ReCiPe impact category	**Unit**	**Reference (complete list in ReCiPe 2016 report** (Huijbregts, et al., 2016)	**Change from ReCiPe 2008** (Goedkoop, et al., 2013)
Global warming	kg CO_2-eq.	(IPCC, 2013), (Joos, et al., 2013), (Hanafiah, et al., 2011), (De Schryver, et al., 2009), (Urban, 2015)	– A much larger set of greenhouse gas emissions (207 GHGs in total) is included on the basis of the latest IPCC report. – Climate-carbon feedbacks are now included for the hierarchist perspective. – Midpoint to endpoint factors for human health and terrestrial ecosystems are corrected on the basis of De Schryver, et al. (2009) and Urban (2015), respectively. – Damage to freshwater (river) ecosystems is included, as derived from Hanafiah, et al. (2011).
Stratospheric ozone depletion	kg CFC-11-eq.	(WMO, 2011), (Hayashi, et al., 2006), (De Schryver, et al., 2011)	– New semi-empirical ODPs were included with more specification between various chlorofluorocarbons (CFCs). – A preliminary ODP for N_2O was included. – Three time horizons have now been consistently implemented: 20 years (Individualist), 100 years (Hierarchist). – Midpoint to endpoint factors were recalculated, based on (Hayashi, et al., 2006)
Human carcinogenic toxicity	kg 1.4-DCB	See Terrestrial ecotoxicity	– Separate midpoint factors for human cancer and non-cancer effects. – Fate and exposure for dissociating organics were included. – USEtox organic and inorganic database was implemented (3,094 substances in total). – Time horizon of 20 years was included for the Individualist perspective. – Linear approach only for damage factor calculations. – Effects on agricultural soil were excluded to prevent double counting with the land use impact category.
Human non-carcinogenic toxicity	kg 1.4-DCB	See Terrestrial ecotoxicity	– Separate midpoint factors for human cancer and non-cancer effects. – Fate and exposure for dissociating organics were included. – USEtox organic and inorganic database was implemented (3,094 substances in total). – Time horizon of 20 years was included for the Individualist perspective. – Linear approach only for damage factor calculations. – Effects on agricultural soil were excluded to prevent double counting with the land use impact category.
Ozone formation, Human health	kg NO_x-eq.	(Van Zelm, et al., 2016)	– The European factor was replaced by a world average factor, based on region-specific factors. – Respiratory mortality has been included. – NO_x equivalents instead of NMVOC equivalents, because NMVOC is a mixture of substances.

ReCiPe 2016			
ReCiPe impact category	**Unit**	**Reference (complete list in ReCiPe 2016 report** (Huijbregts, et al., 2016)	**Change from ReCiPe 2008** (Goedkoop, et al., 2013)
			– To derive intake fractions for individual VOCs, the latest POCPs from Derwent, et al. (2007) were used. – Damage to terrestrial ecosystems was included as well. – World-region-specific characterisation factors were added. – Midpoint and endpoint characterisation factors available at a country level.
Ozone formation, Terrestrial ecosystems	kg NO_x-eq.	See Ozone formation, Human health	– See Ozone formation, Human health.
Fine particulate matter formation	kg $PM_{2.5}$-eq.	(Van Zelm, et al., 2016)	– The European factor has been replaced by a world average factor, based on region-specific factors. – Lung cancer and cardiovascular mortality have been included, no morbidity. – Value choices have been added. – World-region specific characterisation factors have been added. – Midpoint and endpoint characterisation factors available at a country level.
Ionising radiation	kBq Co-60-eq.	(Frischknecht, et al., 2000), (De Schryver, et al., 2011)	– Three time horizons have now been consistently implemented: 20 years (Individualist), 100 years (Hierarchist). – Dose and dose-rate effectiveness factors (DDREFs) were specified per cultural perspective. – Updated DALYs per fatal cancer incidence were applied.
Terrestrial acidification	kg SO_2-eq.	(Roy , et al., 2014)	– The European factor was replaced by a world average factor, based on grid-specific factors. – Soil sensitivity was based on pH indicator H+ concentration instead of base saturation. – Effects on all vascular plant species included, not only forest species. – No value choices included. – Midpoint and endpoint characterisation factors available at a country level.
Freshwater eutrophication	kg P-eq.	(Helmes, et al., 2012) (Azevedo, et al., 2013a), (Azevedo, et al., 2013b), (Azevedo, 2014)	– The European characterisation factor was replaced by a world average factor, based on grid-specific factors. – Fate factors were derived using a state-of-the-art global fate model for phosphorus instead of a European fate model. – The effect factor was updated based on Azevedo, et al. (2013b, 2014), including heterotrophic and autotrophic species. – No marine eutrophication was included, because there is no endpoint model. – Midpoint and endpoint characterisation factors available at a country level.

ReCiPe 2016			
ReCiPe impact category	**Unit**	**Reference (complete list in ReCiPe 2016 report** (Huijbregts, et al., 2016)	**Change from ReCiPe 2008** (Goedkoop, et al., 2013)
Marine eutrophication	kg N-eq.	See Freshwater eutrophication	– The European characterisation factor was replaced by a world average factor, based on grid-specific factors. – Fate factors were derived using a state-of-the-art global fate model for phosphorus instead of a European fate model. – The effect factor was updated based on Azevedo, et al. (2013b, 2014), including heterotrophic and autotrophic species. – No marine eutrophication was included, because there is no endpoint model.
Terrestrial ecotoxicity	kg 1.4-DCB	Van Zelm, et al. (2009; 2013)	– Fate and exposure for dissociating organics were included. – USEtox organic and inorganic database was implemented (3,094 substances in total). – Time horizon of 20 years was included for the Individualist perspective. – Linear approach only for damage factor calculations. – Effects on agricultural soil were excluded to prevent double counting with the land use impact category.
Freshwater ecotoxicity	kg 1.4-DCB	See Terrestrial ecotoxicity	– See Terrestrial ecotoxicity.
Marine ecotoxicity	kg 1.4-DCB	See Terrestrial ecotoxicity	– See Terrestrial ecotoxicity.
Land use	m^2a crop-eq.	(De Baan, et al., 2013a), (De Baan , et al., 2013b), (Elshout, et al., 2014), (Köllner & Scholz, 2007), (Curran, et al. (2014)	– The CFs are now based on global scale data, whereas the previous versions focused on Europe. – The local impact of land use is only covered, as we found the methods for regional impact too arbitrary to take into account. – CFs specific to several species groups are now provided. – In this document, we use the general term 'land use' when referring to the complete cycle of land transformation, occupation and relaxation.
Water consumption	m^3	(Pfister , et al., 2009) (De Schryver, et al., (2011), (Hanafiah, et al., 2011)	– Provide consumption/extraction ratios. – The inclusion of characterisation factors at an endpoint level for human health, terrestrial and aquatic ecosystems. – Midpoint and endpoint characterisation factors available at a country level.
Mineral resource scarcity	kg Cu-eq.	(Vieira, et al., (2012) (Vieira, et al., (2016a) (Vieira, et al., (2016b)	– Developing log-logistic regressions to determine cumulative grade-tonnage relationships and cumulative cost-tonnage relationships. – Use of mine-specific cost and production data. – Average modelling approach, considering all future production and without discounting.

ReCiPe 2016			
ReCiPe impact category	**Unit**	**Reference (complete list in ReCiPe 2016 report** (Huijbregts, et al., 2016)	**Change from ReCiPe 2008** (Goedkoop, et al., 2013)
Fossil resource scarcity	kg oil-eq.	(Ponsioen, et al., (2014) (Vieira and Huijbregts, (In preparation.=>2018)	– Use of more recent cost and future production data. – Use of log-linear cumulative cost-tonnage relationships. – Average modelling approach for endpoint indicator considering all future production and without discounting.

B.3 Comparison of ReCiPe and PEF on characterisation

This Handbook uses impact assessment methods as described in ReCiPe 2016 and the EF impact assessment from the PEF (product environmental footprint) method.[91] The EF impact assessment method should be considered complementary to other methods such as ReCiPe 2016. In this Annex, we discuss the similarities and differences between the methods.

Table 71 - Similarities and differences between ReCiPe 2016 and EF impact assessment

ReCiPe 2016			EF impact assessment (PEF)		
ReCiPe impact category	**Unit**	**Reference (complete list in ReCiPe 2016 report)**	**EF impact category**	**Unit**	**Reference (complete list in PEF Annex 1 & 2)**
Global warming	kg CO_2-eq.	(IPCC, 2013), (Joos, et al., 2013), (Hanafiah, et al., 2011), (De Schryver, et al., 2009), (Urban, 2015)	Climate change* CAT I	kg CO_2-eq.	Bern model - Global warming potentials (GWP) over a 100-year time horizon IPCC (2013)
Stratospheric ozone depletion	kg CFC-11-eq.	(WMO, 2011), (Hayashi, et al., 2006) (De Schryver, et al., 2011)	Ozone depletion* CAT I	kg CFC-11-eq.	EDIP model based on the ODPs of the World Meteorological Organisation (WMO) over an infinite time horizon WMO (2014 + integrations)
Human carcinogenic toxicity	kg 1.4-DCB	See Terrestrial ecotoxicity	Human toxicity, cancer CAT III (not included in the Environmental Prices Handbook)	CTUh	Based on USEtox2.1 model (Fantke, et al., 2017), adapted as in (Saouter, et al., 2018)
Human non-carcinogenic toxicity	kg 1.4-DCB	See Terrestrial ecotoxicity	Human toxicity, non-cancer CAT III (not included in the Environmental Prices Handbook)	CTUh	Based on USEtox2.1 model (Fantke, et al., 2017) adapted as in (Saouter, et al., 2018)

[91] Commission Recommendation (EU) 2021/2279 of 15 December 2021 on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organisations C/2021/9332

ReCiPe 2016			EF impact assessment (PEF)		
ReCiPe impact category	**Unit**	**Reference (complete list in ReCiPe 2016 report)**	**EF impact category**	**Unit**	**Reference (complete list in PEF Annex 1 & 2)**
Ozone formation, Human health	kg NO_x-eq.	(Van Zelm, et al., 2016)	Photo-chemical ozone formation, human health* CAT II	kg NCSRC-eq.	LOTOS-EUROS model (Van Zelm, et al., 2008) as applied in ReCiPe 2008
Ozone formation, Terrestrial ecosystems	kg NO_x-eq.	See Ozone formation, Human health		(Fantke, et al., 2016)	
Fine particulate matter formation	kg $PM_{2.5}$-eq.	(Van Zelm, et al., 2016)	Particulate matter* CAT I	Disease incidence	PM model (Fantke, et al., 2016)
Ionising radiation	kBq Co-60-eq.	(Frischknecht, et al., 2000), (De Schryver, et al., 2011)	Ionising radiation, human health* CAT II	kBq U235-eq.	Human health effect model as developed by (CEPN, 1995) (Frischknecht, et al., 2000)
Terrestrial acidification	kg SO_2-eq.	(Roy , et al., 2014)	Acidification* CAT II	mol H+-eq.	Accumulated exceedance (Seppälä, et al., 2006)
Freshwater eutrophication	kg P-eq.	(Helmes, et al., 2012), (Azevedo, et al., 2013a), (Azevedo, et al., 2013b), (Azevedo, 2014)	Eutrophication, freshwater* CAT II	kg P-eq.	EUTREND model (Struijs, et al., 2009) as applied in ReCiPe
Marine eutrophication	kg N-eq.	See Freshwater eutrophication	Eutrophication, marine* CAT II	kg N-eq.	EUTREND model (Struijs, et al., 2009) as applied in ReCiPe
			Eutrophication, terrestrial* CAT II	mol N-eq.	Accumulated exceedance (Seppälä, et al., 2006), Posch et al. (2008)
Terrestrial ecotoxicity	kg 1.4-DCB	(Van Zelm, et al., 2009; 2013)			
Freshwater ecotoxicity	kg 1.4-DCB	See Terrestrial ecotoxicity	Ecotoxicity, freshwater CAT III (not included in the Environmental Prices Handbook)	CTUe	based on USEtox2.1 model (Fantke, et al. 2017), adapted as in (Saouter, et al., 2018)
Marine ecotoxicity	kg 1.4-DCB	See Terrestrial ecotoxicity			
Land use	m^2a crop-eq.	(De Baan, et al., 2013a), (De Baan , et al., 2013b), (Elshout, et al., 2014) (Köllner & Scholz, 2007), (Curran, et al., 2014)	Land use CAT III (not included in the Environmental Prices Handbook)	Dimensionless (pt)	Soil quality index based on LANCA model (De Laurentiis , et al., 2019) and on the LANCA CF version 2.5 (Horn & Maier, 2018)
Water consumption	m^3	(Pfister , et al., 2009) and (De Schryver, et al., 2011) (Hanafiah, et al., 2011)	Water use CAT III (not included in the Environmental Prices Handbook)	m^3 water eq. of deprived water	(EU, 2021)

ReCiPe 2016			EF impact assessment (PEF)		
ReCiPe impact category	**Unit**	**Reference (complete list in ReCiPe 2016 report)**	**EF impact category**	**Unit**	**Reference (complete list in PEF Annex 1 & 2)**
Mineral resource scarcity	kg Cu-eq.	(Vieira, et al., 2012) (Vieira, et al., 2016a) and (Vieira, et al., 2016b)	Resource use, minerals and metals CAT III (not included in the Environmental Prices Handbook)	kg Sb eq.	(Van Oers, et al., 2002) as in CML 2002 method, v.4.8
Fossil resource scarcity	kg oil-eq.	Ponsioen et al. (2014) and Vieira and Huijbregts (In preparation.=>2018)	Resource use, fossils CAT III (not included in the Environmental Prices Handbook)	MJ	(Van Oers, et al., 2002)as in CML 2002 method, v.4.8

The table above shows that the main differences between ReCiPe 2016 and the PEF can be explained by the following aspects:
1. Different units are used for a number of environmental themes, such as ozone formation, fine particulate matter formation, acidification, human toxicity and eco-toxicity.
2. Different environmental themes are included. For example, ReCiPe 2016 (as in 2008), does not include the theme 'eutrophication from air', while air pollutant emissions such as NH_3 and NO_x do have a mutual impact on soil conditions, namely acidification and eutrophication. This means that these impacts have been aggregated based on ReCiPe, as in the previous Handbook. The PEF methodology does distinguish soil acidification and soil eutrophication due to emissions to air.
   In addition to midpoint environmental impacts, ReCiPe 2016 also provides estimates of final damage to human health and ecosystems (endpoints). PEF does not provide endpoint characterisation.

For the update of the Environmental Prices Handbook 2024, midpoint-level environmental prices were determined based on ReCiPe 2016 as well as PEF CAT I and II, and endpoint-level environmental prices on ReCiPe 2016. CAT III impact categories are shown in the table for completeness, however, they are not used for the calculation of environmental prices.

Below, we indicate for each environmental theme how the characterisation differs between ReCiPe 2016 and PEF and whether these differences can be bridged by, for example, converting characterisation factors:
- **Ozone depletion**
  There is no difference in unit, as both methods use the same underlying model of the WMO on ozone depletion. For this, PEF refers to a more recent report (WHO, 2014) where ReCiPe 2016 refers to (WMO, 2011).
- **Climate change**
  There is no difference in unit, as the same underlying model is used (IPCC 2013 GWP100).
- **Oxidant formation**
  ReCiPe 2016 relies on the work of (Van Zelm, et al., 2016) and expresses the characterisation factor in kg NO_x-eq. PEF relies on van Zelm (2008) which was also used for ReCiPe 2008, the unit used is kg NCSRC-eq. The relationship between these two units is described in (Van Zelm, et al., 2016).

- **Particulate matter formation**
  ReCiPe 2016 relies on the work of (Van Zelm, et al., 2016) and expresses the characterisation factor in kg $PM_{2.5}$-eq./kg. In contrast, PEF takes the UNEP/Life Cycle Initiative (UNEP, 2016) characterisation as its starting point and expresses the characterisation factor as *disease incidence*. The relationship between $PM_{2.5}$ and disease incidence is explained in Chapter 4 of the UNEP report (UNEP, 2016).
- **Acidification**
  ReCiPe relies on the work of (Roy , et al., 2014) and expresses the characterisation factor in kgSO_2-eq. PEF expresses the characterisation factor in *accumulated exceedance* (AE), which is explained in (Seppälä, et al., 2006).[92] The relationship between kgSO_2-eq. and AE can be derived from these studies.
- **Eutrophication**
  No difference in units. PEF relies on the underlying model used in ReCiPe 2008. For ReCiPe 2016, an update was carried out based on the latest scientific findings. (Helmes, et al., 2012; Azevedo, et al., 2013a; 2013b; 2014).
- **Radiation**
  Both ReCiPe 2016 and PEF base the characterisation of radiation on the work of (CEPN, 1995). ReCiPe 2016 expresses this in kBq Co-60-eq.

B.4 Perspectives chosen from ReCiPe

In the Environmental Prices Handbook, we have been guided by the ReCiPe 2016 characterisation method. Three perspectives can be chosen in that characterisation method. In this Annex, we indicate the perspectives we chose for the characterisation method in the Handbook.

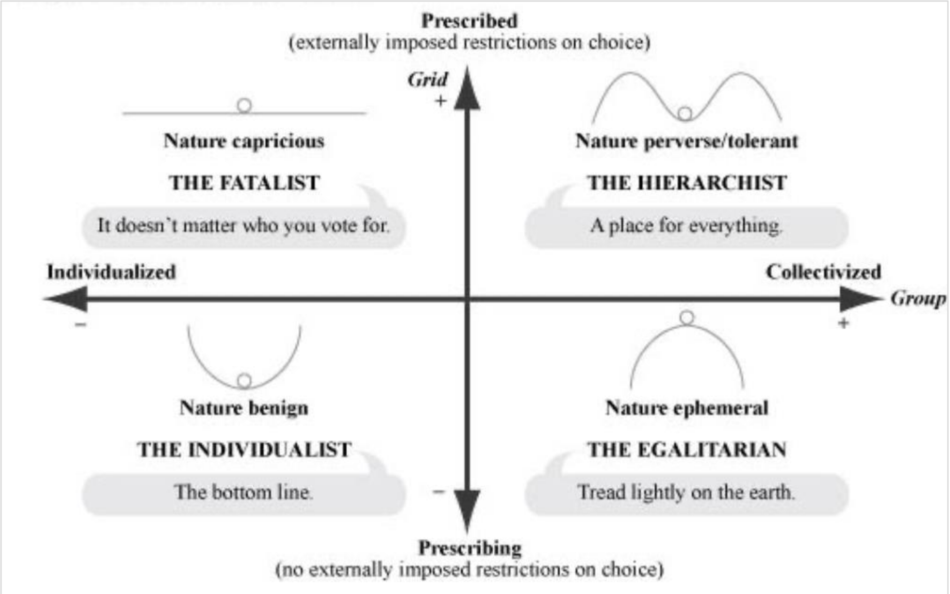
B.4.1 Background of cultural theory

The characterisation models used in the ReCiPe project are subject to uncertainty. The main source of uncertainty is that the modelled relationships reflect incomplete and uncertain knowledge of environmental mechanisms. This uncertainty is elaborated in different perspectives according to 'Cultural Theory' as elaborated by (Thompson, et al., 1990).

Thompson, et al. identify a number of basic value systems by looking at the strength of the relationships people have with their group and the extent to which an individual's life is constrained by externally imposed demands (called a 'grid'). The following figure shows perspectives applied to thinking about nature and the environment.

[92] No reference details given in PEF background information.

Figure 15 – Cultural perspectives applied to nature and the environment in the theory of Thompson, et al. (1990)



Source: (Schwarz & Thompson, 1990).

Based on these five groups, ReCiPe detailed three 'human images' that influence the extent to which interests of the group and others are included in their considerations:

1. **Individualist**: in this scenario, only proven cause-effect relationships are included and are used for the short term only. There is technological optimism regarding human adaptation. In practice, this means that all effects between now and 20 years is counted as relevant.
2. **Hierarchist**: included in this scenario are facts supported by scientific and political bodies. The hierarchical attitude is common in the scientific community and among policymakers. In practice, this means counting all effects for up to 100 years.
3. **Egalitarian**: this scenario uses the precautionary principle and the very long term. All effects between now and 500 years are counted as relevant here.

B.4.2 Use perspectives in this Handbook

The following decisions were made for the three user goals of characterisation factors in this Handbook:

Step 3 (user goal 1, weighting): hierarchical perspective
The hierarchical worldview was used for weighing the damage costs of the primary pollutants to arrive at a midpoint price. We did this because it is the most commonly used worldview in LCA. The underlying damage at the midpoint are one-to-one related to the results from life cycle analyses.

Step 4 (user goals 2 and 3): Pollutant prices and midpoint prices via endpoint valuation: combination of individualistic and hierarchical perspective
The valuation for the non-primary pollutants is obtained via the summation of their characterisation factor times the midpoint price. For this summation, we have been guided by the individualist perspective for most themes in the lower value and the hierarchical perspective in the upper value. The reason for this is that the individualist perspective has

a non-discountable time horizon of only 20 years, while in the hierarchical worldview there is a non-discountable time horizon of 100 years. The central value is in between.

For the central value, an intermediate value was chosen in which the difference between individualistic and hierarchical is discounted at a rate of 2.25%. This is explained in more detail in Annex D.4.3.

For themes for which there were many primary pollutants, such as fine particulate matter and oxidant formation, this methodology did not work well because it allowed a higher valuation in the lower value than in the upper value for pollutants that were not primary pollutants. For this reason, we have allocated these themes entirely via the hierarchical worldview. This also applies with regard to climate change because the non-discounted impact of 100 years (i.e. the hierarchical worldview) has become the norm internationally. Therefore, it would be unwise to deviate from that in the lower and upper values.

B.4.3 Calculation of intermediate value with discounting

The characterisation factors from ReCiPe do not take time preferences into account. However, considering time preferences is common in economics: costs in the future are *discounted* so that they carry less weight than costs in the present. Especially in ReCiPe's 100-year perspective (the hierarchical perspective), this lack of discounting leads to a significant overestimation of damage costs. In the 20-year perspective (the individualistic perspective), this overestimation is more limited due to the shorter time horizon. To arrive at more accurate estimates of environmental prices within the 100-year perspective, we use the following method:

1. We assume that the characterisation factor increases linearly when the time horizon used increases from 20 to 100 years. In reality, most damage cost curves are concave when plotted against the time horizon. This assumption therefore leads to a limited overcorrection of total damage costs.
2. We discount the annual increase in the characterisation factor between 21 and 100 years using the social discount rate (2.25%). We do not discount the damage costs between 0 and 20 years because we have no information on the damage cost curve between 1 and 20 years (the starting point at 1 year is unknown, as is the further course of the function). This omission leads to a limited under-correction of total damage costs.
3. We add the calculated discounted characterisation factor from Step 2 to the characterisation factor associated with the 20-year perspective to capture the damage costs over the entire period (0-100 years).

The linear assumption in the first step leads to a limited overcorrection of damage costs, while the partial discounting in the second step leads to a limited under-correction of damage costs. Implicitly, then, we assume that these two opposite effects cancel each other out.

The following example illustrates the approach:

Suppose the 20-year characterisation factor of an environmental pollutant is 100, and the 100-year characterisation factor equals 180. The annual assumed linear increase is then equal to (180-100)/80 = 1. The cumulative discounted characterisation factor between the 20 and 100 years is equal to 1 * 0.9775 ^ 21 + 1 * 0.9775 ^ 22 + ... + 1 * 0.9775 ^ 100 = 24.[93]

[93] The factor 0.9775 occurs from applying the social discount rate of 2.25%: 1 - 0.0225 = 0.9775.

We then add this value to the 20-year characterisation factor: 100 + 24 = 124. This 124 forms the new discounted characterisation factor for the 100-year perspective.

The approach presented in formulaic form:

$$H^{*} = I + \sum_{i=21}^{100} \frac{H - I}{80} * (1 - d)^{i}$$

Where H* represents the new 100-year discounted characterisation factor, where I represents the 20-year characterisation factor, H represents the unadjusted 100-year characterisation factor and d represents the discount rate.

C Valuation of nature

C.1 Introduction

This annex provides additional information and calculation steps we made when determining the valuation for natural values. This valuation plays a role in the themes of eutrophication, acidification, photochemical oxidant formation, ecotoxicity and ozone depletion.

C.2 Metrics of biodiversity

The traditional approach to measuring biodiversity focuses on four dimensions of species-level diversity - also called alpha (α), beta (β), gamma (γ) and omega (o) diversity. α diversity is most commonly used in impact assessment models. It determines the taxonomic diversity of species in a given system and thus indicates species richness. It can be measured by different indices. The most commonly used index in LCA is relative species richness. For land use, this can be measured as the relative richness of under land use type *i* in region *j* of taxa *g*.

$$S_{rel,LU_{i,j,g}} = \frac{S_{LU_{i,j,g}}}{S_{ref_{j,g}}}$$

Other commonly used indices for α diversity are the Shannon and Simpson indices.

Shannon index

$$H = -\sum_{i=1}^{S} Pi * \ln pi$$

Where p_i is the ratio of the number of individuals of the *i* species to the total number of individuals, S. The higher the index, the greater the species richness at a site and the more even their relative abundance.

$$D = -\sum_{i=1}^{S} p^2 i$$

The Simpson's index defines p_i as the fraction of all organisms represented by the *i* species and has a value between 0 (zero diversity) and 1 (infinite diversity) and 1 (infinite diversity).

Another example is Fischer's α which relates the number of species to the total number of individuals within the species.

$$\frac{N}{S} = \frac{e^{\frac{S}{a}} - 1}{\frac{S}{a}}$$

β-diversity measures the difference in species diversity between systems. The metric compares systems based on the number of taxa unique to each system. Sørensen's similarity index of β-diversity:

$$S_s = \frac{2c}{S_{LUi} + S_{ref}}$$

For two systems, the number of species common to both systems (c) is related to the total number of species found in those systems. It also has a value of 0 when there is no species overlap between the communities and a value of 1 when exactly the same species are found in both communities.

γ-diversity is a measure of taxonomic diversity in all systems evaluated. For two systems, it is a count of the number of distinct species in both systems (Whittaker 1972).

$$\gamma = S_1 + S_2 - c$$

o-diversity measures the phylogenetic diversity of a system (Schweiger, et al., 2008). The most common measurements use a minimum spanning path or pairwise spacing. The minimum spanning path approach sums the branch lengths of the phylogenetic tree containing all species in the area or the number of nodes separating the species (Hanley, 2019).

C.3 Indicators of biodiversity

Many types of indicators have been proposed in the context of biodiversity. We developed three different indicators in the main text (Paragraph 5.4.4):
1. Potentially Disappearing Fraction (PDF).
2. Potentially Affected Fraction (PAF).
3. Biodiversity Damage Potential (BDP)

Many other relevant indicators have been identified in the literature. The two main indicators that still have a function in this Handbook are:

Ecosystem Damage Potential (EDP) is a life cycle impact assessment method for the characterisation of land use and land transformation, developed by the Swiss Federal Institute of Technology (ETH) and often used in the context of LCAs (Köllner & Scholz, 2007). The EDP is based on an ecological model that describes the transformation of species richness over time and between areas through land transformation. A high EDP of a land use type indicates lower ecological quality of that land. The model uses as a reference the EDP at t0, which indicates the quality of the land before the transformation. EDP_{occ} is the characterisation function for a specific land use type to be calculated for the duration of occupation. The indicator only considers the species richness of vascular plants, which is the main limitation of the metric. If the country of the 'natural state' is transferred to another state, the EDP can be considered similar to the PDF. (Kuik, et al., 2008) suggest that EDP and PDF should be considered comparable in practical applications.

Mean Species Abundance (MSA) is another indicator of biodiversity and is defined as the average abundance of native species relative to their abundance in an undisturbed ecosystem. The MSA metric was developed in the GLOBIO3 framework for the analysis of land use scenarios to capture changes in community composition due to human pressures (Alkemade, et al., 2009). MSA is a useful indicator because it can also be linked to

conservation objectives because it is more sensitive than the S_{rel} which is more difficult to link to conservation objectives (De Baan, et al., 2013a). The new version of the GLOBIO model, GLOBIO4, also evaluated the intactness of terrestrial biodiversity using MSA as an indicator for three socio-economic trajectories.
The global average MSA was estimated at 0.56 for 2015. Land-use change was considered the biggest cause of MSA decline and was found to be responsible for 70% of MSA loss. (Schipper, et al., 2020). MSA metrics do not include spatial variability in species richness and thus cannot capture disproportionate species loss in species-rich regions (e.g. tropical forests). (Barlow, et al., 2018), while also neglecting other aspects such as β diversity in the model (Schipper, et al., 2020). The MSA can be calculated as follows:

$$MSA = \frac{1}{S_{ref}} * \sum\nolimits_k \frac{n_{g,LUi}}{n_{g,ref}}$$

Where n_g represents the number of individuals in species g.

Other indicators of biodiversity

Other indicators relevant to biodiversity have been proposed in the literature. These are based on biotic production: HANNP, Life Support Functions using soil organic matter, Living Planet Index, Biodiversity Intactness Index, e. The following table shows the key indicators.

Despite criticism of an indicator such as species richness, many authors argued that species richness was still the best possible indicator for LCIA, due to data availability and requirements (De Baan, et al., 2013a). This is the main reason why the framework in the Environmental Prices Handbook includes a species richness analysis.

Table 72 - Overview of indicators relevant to biodiversity

Name of CF	Base indicator	Units	Equations	Reference study
EDP	Species richness	m^2*yr	$CF(a,t) = c_{trans} + m * t$ all $a \in [0, a1]$ and $t \in [t0,\ t1]$	(Köllner & Scholz, 2007)
PDF	Species richness	Species*year	$CF_{occA,reg} = z_j * ED$ $CF_{occB,reg} = (z_j - z_i) * SD$	(Goedkoop, et al., 2009)
BDP	Species richness	PDF/Species/year	$CF_{Occ,LU,ij} = 1 - S_{rel,LU,i,j}$	(De Baan, et al., 2013a)
Matrix-SAR_{PDF}	Species richness	Species*m^2*yr	$CF_{Occ,reg,i,j,g} = \frac{S_{lost,nonend,j,g} * a_{i,j}}{A_{i,j}}$	(De Baan , et al., 2013b)
BDP_{crop}	Species richness	PDF/Species/year	$CF_{cr,j,g} = \frac{S_{crop,cr,j,g}}{S_{ref,j,g}}$	(Elshout, et al., 2014)
PDF	Species richness	PDF/annual crop-eq.	$CF_{m_{occ,i}} = \frac{S_{rel,i}}{S_{rel,annualcr}}$	(Huijbregts, et al., 2016)
C-SAR_{PDF}	Species richness	PDF/Species/year	$CF_{occ,i,j,g} = \frac{\delta S_{lost,j,g} * a_{i,j}}{\delta A_{lost,j,g} * P_{i,j}} * VS_{g,j}$	(Chaudhary, et al., 2015)
C-SAR_{PDF}	Species richness	PDF/Species/year	$CF_{global,g,i,j} = \frac{S_{loss,g,i,j}}{A_{i,j}} * VS_{g,j}$	(Chaudhary & Brooks, 2018)
FD	Functional diversity	PDF/Species/year	$CF_{FD} = ln\left(\frac{FD_i}{FD_{ref}}\right)$ $CF_{SR} = ln\left(\frac{SR_i}{SR_{ref}}\right)$	(De Souza, et al., 2013)
LSF	Soil organic material (SOM) content	kg C m−2 yr−1	$CF_{use,a} = A_a \sum_{i=t_{ini}}^{t_{fin}} (SOM_{ref,i} - SOM_{a,i}) t_i$	(Mila i Canals, et al., 2007)
BPP	Soil organic carbon content	kg C m−2 yr−1	$CF_{occ} = \frac{(SOC_{pot} - SOC_{LU2}) * (t_{fin} - t_{ini})}{(t_{fin} - t_{ini})}$	(Brandão. & i Canals, 2013)

Name of CF	Base indicator	Units	Equations	Reference study
LANCA	Soil quality (5 indicators)		$CF_{occ} = -(Q_{LU,current} - Q_{ref})$	(Bos, et al., 2016)
SQI (LANCA)	Soil quality	Pt/m-2 yr-1	$\overline{CF_{occ,i}} = \sum_{y=1}^{4}\left(\frac{CF_{y,e}}{CF_{y}^{95}}\right)$	(De Laurentiis , et al., 2019)
NPPD	Net primary production (soil)	MJex m−2 yr−1	$CF = \frac{NPP_{0,1}}{NPP_{0,ref}}$	Nunez, et al., 2013
$HANPP_{NPP}$	Net primary production	MJex m−2 yr−1	$CF_{OCC,i,c} = \Delta NPP_{LC,i} + NPP_{h,c}$	(Taelman, et al., 2016)
$Naturalness_{NPP}$	Net primary production	MJex m−2 yr−1	$CF_{OCC,i,c} = HANPP_c = NPP_{0,c} * NDP_i$	(Taelman, et al., 2016)
$Naturalness_{NI}$	Qualitative management parameters		$CF_{PNI_x} = (\frac{1}{n} * \sum_{l=1} n * Con_{PNI_l}) * (1 - \sum_{o=1} k * NDP_o)$	Cote, et al., 2019
Hemeroby	Qualitative – management parameters	Naturalness score	$LUI_i = \frac{F_i}{F_{max}} + \frac{Me_i}{Me_{max}} + \frac{Pe_i}{Pe_{max}} + \frac{I_i}{I_{max}} + \frac{P_i}{P_{max}}$	Meier, et al., 2019
BVI	Qualitative score – mostly management parameters	BVI/m²/year	$CF = EF * \sum_{f=1}^{n} Z_f * Y_{mp,cp}(X_{mp}, X_{cp})_f$	(Lindner, et al., 2019) (Lindner, et al., 2021)

c_{trans}: y intercept; m: damage function slope; ED: ecosystem damage; SD: species density, z: species accumulation factor; i: land use type; j: region; g: taxonomic group; cr: crop type; VS: vulnerability score; P_i: relative area share of each land use type; a: any moment; t_{ini}: occupation period starts; t_{fin}: occupation period ends; SOC_{pot}: potential soc level; SOC_{LU2}: SOC level at occupation; y: indicator; CF_y^{95}= the 95th percentile of the distribution of country-specific CFs; c: country; h: amount harvested; LC: land conversion; NDP: naturalness damage potential; x: characteristic; l: condition indicator; k: number of NDPo; F: fertilisation level; Me: mechanisation level; Pe: pesticide application; I: irrigation; P: further parameters; Y: biodiversity contribution; Z_f: weighing ratios; X_{mp}: management parameters; X_{cp}: context parameters

C.4 Overview of studies valuing ecosystem services per hectare

Constanza, et al. developed an early but highly influential global estimate of ecosystem services (ES) in 1997. The study estimated the economic value of 17 ecosystem services for 16 different biomes, as a meta-analysis based on 'benefit transfers' with values from about 100 studies. The authors calculated an average value of at least $33 trillion per year as the global flow value of ecosystem services, roughly equivalent to the then global GDP, or about $650 per hectare (including oceans), or $2,250 per hectare, including oceans.

Several more complete studies have been published since 1997. In these studies, the main changes were attributed to the improved understanding of the functions of ecosystem services and their contribution to human, social and built capitals. Kuik, et al (2008) conducted a meta-analysis of economic valuation studies, and studied the impact of land-use change on terrestrial biodiversity loss. A total of 160 valuation studies were collected, but data could only be extracted and standardised from 24 studies for the meta-analysis, yielding 42 data points. The average value of ecosystem services was found to be €4,706 per hectare, with a much lower central value of €604.

Another noteworthy study was an examination of the cost of policy inaction in preserving ecological values. The authors estimate a total global monetary value for ES loss related to biodiversity loss. The study estimated the loss of ES-value due to biodiversity loss between 2000 and 2050 at €14 trillion (IEEP, 2009).

De Groot, et al. (2012) provided a newer estimate of the global monetary value of ES. The monetary values of ES were calculated for ten major biomes (out of twelve globally identified). A total of 22 ES were identified for each biome. The study authors developed the Ecosystem Service Database (ESVD) with about 320 local case studies and with more than 1,350 data points. About half of these data points were used by them for the meta-analysis. The studies presented in the meta-analysis by De Groot, et al. (2012) included mostly studies that estimated monetary values based on direct market value, with almost half of the values obtained through direct market prices. In addition, contingent valuation was used in 15% of cases, while production functions, avoided costs and replacement costs each accounted for about 10% of data points. In terms of ES, supply services were over-represented with 43% of all data points, with food and raw materials being the most valued. Regulatory services accounted for about 22% of all data points, with climate control and moderate disturbance being the most common. Cultural services accounted for 21% of all data points, with recreational services rated at 84%. Finally, habitat services accounted for only 12% of data points.

Based on this study, Constanza, et al. (2014) added a new valuation to their 1997 study. Complementing the ten biomes in De Groot, et al. (2012), the new study included additional estimates for urban and agricultural systems. The authors used a comprehensive ground cover database, GlobCover, developed by the European Space Agency, in collaboration with UN-FAO. The definitions from the dataset matched well with those in the report by Constanza, et al. (1997) allowing further comparison. Changes in land cover between 1997 and 2011 were taken into account, which eventually showed lower values due to the loss of more precious ecosystems and the increase of ecosystems with lower values (e.g. 'cropland'). Aggregate values were calculated by multiplying land area by unit values. Estimates showed that global land-use changes between 1997 and 2011 resulted in a loss of ES value between €3.1 and 14.8 trillion per year (Constanza, et al., 2014).

The studies by De Groot, et al. (2012) and Costanza, et al. (2014) are up-to-date still the most comprehensive studies on ecosystem service valuation. As such, they are used in our Handbook to calculate the value of biodiversity, assuming that biodiversity as an indicator includes all relevant ecosystem services. However, the study by Costanza, et al. (2014) also has its limitations. Data were only found for 12 ES per biome (out of a potential maximum of 22 recognised ES). Moreover, some services, such as carbon sequestration, important in the tundra ecosystem, were not valued at all. Moreover, in practice, values depend on the local context and the same services can have a wide range of values in different locations. These originate from differences in ecological and socio-economic backgrounds and different valuation methods. The supply of ES is certainly not homogeneous, so using constant values per hectare for all biomes may distort the overall average results. Due to the large differences between studies, selection bias was also significant. The valuation method also had a significant effect on estimated values. Thus, conditional valuation produced higher values on average than other methods.

As shown in the following table, the total value of ecosystem services varies widely among the different land types, ranging from $490 dollars (€358)/ha/year ('open ocean') to over $350,000 (€256,000)/ha/year ('coral reefs').

Table 73 - Valuation of different land types in Constanza, et al. (2014)

	Area	Avg value ($/ha)	Median value
Marine	**36,302**	**1,368**	–
Of which Open ocean	33,200	660	135
Of which Coastal	3,102	8,944	–
– *Estuaries*	180	28,916	26,760
– *Seagrass/Algae beds*	234	28,916	26,760
– *Coral reefs*	28	352,249	197,900
– *Shelf*	2,660	2,222	–
Terrestrial/freshwater	15,323	4,901	–
Of which Forest	4,261	3,800	–
– *Tropical*	1,258	5,382	2,355
– *Temperate/Boreal*	3,003	3,137	1,127
Of which Grass/Rangelands	4,418	4,166	2,698
Of which Wetlands	188	140,174	–
– *Tidal Marsh/Mangroves*	128	193,843	12,163
– *Swamps/Floodplains*	60	25,681	16,534
Of which Lakes/Rivers	200	12,512	3,938
Cropland	622	5,567	–
Urban	352	6,661	–
Non-allocated areas*	4,232	-	–
Total	**51,625**	**2,417**	–

* These include tundra, desert and ice plains. No valuation was established for this in Costanza, et al. (2014).

Based on this study, we made a conversion to saltwater, terrestrial and freshwater, with the 'tidal marshes/mangroves' category allocated 50% to saltwater and terrestrial, and the 'marsh' category allocated 50% to freshwater and terrestrial. For Table 18 in Paragraph 5.4.5, the corresponding values are still converted into euros (2007 exchange rate) and the values are inflated by a 1% autonomous growth per year in the value of biodiversity and inflation between 2007 and 2021.

There are two regional studies that are also included in the overview of project results. These are not based on meta-analysis but include a comprehensive accounting framework to determine the value of ecosystem services. One is the Phase II report of the EU's INCA project (Vysna, et al., 2021) and the other is the US Federal Emergency Management Agency (FEMA) report (2022).

The EU's INCA project was set up as a pilot project for an integrated natural capital accounting system. The project involves five key partners: Eurostat, the Directorate-General for the Environment, the Directorate-General for Research and Innovation, the Joint Research Centre and the European Environment Agency.

The economic valuation is based on the Corine Land Cover data and runs from 2012-2021 to the present. In their new Phase II report, the authors published values for ten ecosystem services with a total estimated value of €234 billion per year in 2019. The study sought to identify the real contribution of ES to the economy and society (Vysna, et al., 2021). Interestingly, the INCA report found significantly lower values than other estimates. However, when one examines the actual number of Vysna, et al. (2021) in more detail with the meta-analysis of De Groot, et al. (2024) one has to conclude that they are relatively equal among the different ecosystem services, but that the INCA study examined only ten ES (and excludes, for example, habitat and cultural services) and De Groot, et al. (2022) examine a much larger number of ecosystem services. Thus, the lower results in the EU INCA study are purely due to the inclusion of a limited number of ES for which the results are more certain.

In the US, the Federal Emergency Management Agency (FEMA) is a government organisation that provides billions of dollars each year to communities to reduce or eliminate the long-term risk of natural disasters. FEMA requires hazard mitigation projects to be cost-effective for the federal government; therefore, the project must show a cost-benefit analysis comparing the net present value of a project's future benefits and costs. To include nature in cost-benefit analysis, FEMA has developed a valuation database in which different types of land use have been valued for their ecosystem services. FEMA recognises 23 ES, making it the most comprehensive ES valuation study known. Each of these values is based on an extensive literature review and often an average of literature values has been proposed.

This study is important for our Handbook to show that the valuation of ecosystem services can be even higher if more recent studies are published. The study arrived too late for our project to be part of the valuation framework, but we recommend that future updates to the Environmental Prices Handbook look specifically at this study, as the valuation methodology may be important in determining the environmental prices of nature.

C.5 Implicit valuation for species

The indicator PDF/m^2/yr is used in (Kuik, et al., 2008) and includes a valuation of species richness per m^2 per year. We use the value of PDF to arrive at a valuation per species per year. We do this on a global scale, since a specific adaptation to European biodiversity is beyond the scope of this Handbook. On the contrary, intrinsic valuations are also assigned to biodiversity on a global scale. Although most residents in the EU will never encounter giraffes or lions in the wild, value is placed on preventing the extinction of these species.

The following table shows the calculations done to arrive at an ecosystem damage valuation of 1 kg 1.4-dichlorobenzene. This pollutant is used in the theme 'terrestrial ecotoxicity'. Here, we first assume the valuation of PDF/m^2/yr, as used in this Handbook. We then multiply this value by the m^2 area of land at this value. So this represents a value for the total biodiversity on this planet. In this case, it is €11.8 trillion.[94] This value can then be divided by the number of species on land. Thus, we obtain a valuation by species or species.yr. This valuation can then be multiplied by the characterisation factor.

Table 74 - Explanation of the calculation valuation characterisation factor 'ecotoxicity' for the central value

Step	Indicator	Value	Source
1	€/PDF/m^2/yr	0.32	Our adjustment of the results of (Kuik, et al., 2008) scaled to (Costanza, et al., 2014).
2	m^2 Earth's surface	1.48E+14	ReCiPe 2013 (Goedkoop, et al., 2013).
3	€/PDF/yr	4.82E+13	= 1 x 2 and thus represents the total euros for species richness on land on this Earth.
4	Number of species on land	1.600.000	ReCiPe 2013 (Goedkoop, et al., 2013).
5	€/PDF/type	30.114.219	= 3/4 and thus reflects the implicit valuation for one species.
6	Factor from midpoint to endpoint species*yr/kg 1.4-DBC emitted to industrial soil-eq.	1.14E-11	ReCiPe 2013 shows the characterisation factor for 1 kg of dichlorobenzene on species.yr. (Goedkoop, et al., 2013).
7	€/kg 1.4 DBC to soil	0.000343	= 5 * 6

For freshwater and saltwater, we correct the valuation for PDF with the difference in species density between land and water. We then determine the number of species on land, freshwater and marine water based on ReCiPe and calculate the valuation per species.

€/PDF/species/year = €/PDF/yr/number of species

This leads to the following valuation of species per year:

Table 75 - Valuation PDF/species for use in midpoint ecotoxicity, in €$_{2021}$

Category	Lower value	Central value	Upper value
Land (m^2)	€ 21,236,112	€ 30,114,219	€ 38,992,327
Freshwater (m^3)	€ 15,522,553	€ 22,012,012	€ 28,501,471
Saltwater (m^3)	€ 15,522,553	€ 22,012,012	€ 28,501,471

By multiplying the valuation per species by a characterisation factor per pollutant (in species.yr/kg), the damage costs per kilogram of emissions of a given pollutant are determined.

94 Here we note that this value is well below the total ecosystem value of US $33 trillion calculated in (Constanza, et al., 1997). The value presented here is 0.04% of the value reported in (Constanza, et al., 1997).

D Noise valuations: the Netherlands

This annex describes the valuation of noise as valid for the Netherlands. The resulting environmental prices are specific to the Netherlands. To value noise for other countries, an adjustment needs to be made. This is currently not available in this handbook.

D.1 Indicators of noise exposure

The A-weighted decibel value dB(A) is the most common unit for noise exposure. The decibel is a logarithmic measure of sound level: a 3 dB increase reflects a doubling of the sound level. A-weighting is applied to this to correct for the human ear's sensitivity to the pitch of the sound.

Besides sound level and pitch, the time of day and duration of sound also play an important role. These factors are included in the noise measure. There are a large number of noise metrics, each differing in how they account for these factors. In this Handbook, we assume the noise measure Lday-evening-night (Lden), the current legal standard for measuring traffic noise. The Lden is determined by setting equivalent noise levels during the day (07:00-19:00 hours), evening (19:00-23:00 hours) and night (23:00-07:00 hours), with levels for the evening and night being increased by 5 and 10 dB(A), respectively, before calculating a 24-hour average. This metric assumes that noise in the evening and especially at night is more disturbing than during the day. The Lden is based on annual average noise levels measured at the most exposed exterior façade of a dwelling.

D.2 Monetary valuation

In this paragraph, we look at how noise valuations for nuisance and health effects are determined and discuss the role of threshold values.

Valuation methods for noise nuisance

Broadly speaking, three methods for the valuation of ambient noise nuisance can be distinguished in the literature:

1. **Stated preference (SP)**. Under the SP method, respondents are asked through surveys or experiments to give their Willingness-to-Pay (WTP) for noise reduction. This method leads directly to a WTP per dB per person or per household. SP methods have the advantage of allowing the researcher to control for all external factors and thus isolate the value of noise nuisance. In addition, SP methods allow researchers to calculate the WTP at different noise levels and thus determine whether there is non-linear growth of the WTP (recent literature indicates this is the case). 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 (see Paragraph 5.2).

2. **Revealed preference (RP)**. Under the RP method, the cost of noise nuisance is derived from actual observed market effects. By far the most frequently used RP method for valuing the impact of noise is hedonic pricing, deriving the Willingness-to-Pay (WTP) for noise reduction from variations in house prices. For this purpose, a Noise Sensitivity Depreciation Index (NSDI) is usually used. The great advantage of RP methods is that the valuation is based on actual behaviour of people (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., see Paragraph 5.2). Estimates based on revealed preference methods therefore exhibit a very wide range. In addition, most RP studies assume a linear relationship between NSDI and noise level - an assumption that seems to contradict results from the SP literature.

3. **Environmental Burden of Disease (EBD)**. Under the EBD method, nuisance is valued using DALYs (Bruitparif ; ORS Ile-de-France; WHO, 2011); (Defra, 2014) (WHO, 2011). Within the broad definition of health used by the WHO ('a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity') (WHO, 2011), nuisance can be understood as a health effect and thus its impact can be expressed in DALYs (however, DALYs for noise nuisance cannot simply be added to DALYs due to disease as calculated in the Global Burden of Disease studies). The advantage of the EBD 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 appropriate factor. In addition, there is as yet little literature on this issue. The range (WHO, 2011) represents is therefore quite large: 0.01 to 0.12. (Defra, 2014) and (WHO, 2018) use a disability weight of 0.02 as central value from a conservative point of view. A second drawback of the EBD method is that disability weights are only assigned to respondents who report being 'highly annoyed'. This is likely to lead to an underestimation of nuisance costs: after all, people who are only 'moderately annoyed' also experience nuisance.

There is no clear agreement in the literature as to which of the three methods is preferable (Andersson, et al., 2013). In line with the previous edition of the Environmental Prices Handbook, we base the valuation of nuisances mainly on the SP method, but compare the results with estimates based on the RP and EBD methods. We adopt this approach because (i) the EBD method is likely to underestimate nuisance costs (due to its focus on 'highly annoyed'); (ii) the RP method does not sufficiently account for the possibility that the marginal cost of noise nuisance increases with noise levels; and (iii) both methods have large uncertainty ranges.

Valuation methods for health effects

In this paragraph, we discuss the valuation methods used to monetise the health effects of noise exposure. Before turning to the valuation methods, we briefly consider the additionality of health effects and nuisance.

In determining the valuation of the health effects of noise, the extent to which they are not yet included in the valuation of noise nuisance should be considered. Indeed, if people are well aware of the various health effects of noise, it can be assumed that these costs are reflected in the WTP values produced by the SP survey. In this context, (HEATCO, 2006) assumes that the cost of sleep disturbance is already part of the nuisance cost and therefore no longer needs to be considered separately when valuing health costs. This approach seems plausible to us, and we therefore also followed it in the Environmental Prices Handbook 2018 and it is currently adhered to. In Paragraph 6.11.2, we show that nuisance costs calculated using the EBD method are significantly lower than nuisance costs determined using the SP method. In the EBD method, when we tentatively add the health effects of sleep disturbance to the health effects of nuisance, the cost estimates come out much better.[95] We see this as additional support for the hypothesis that sleep disturbance is taken into account in SP research. In the case of the other health effects, such as an increased probability of ischaemic heart disease, we assume that the costs *have not yet* been included in the WTP values; it is ever obvious that most people are unaware of such health risks from noise.

For the health effects of noise, a distinction can be made between the effects for the person himself (pain, discomfort, etc.) and the effects for the rest of society (e.g. medical costs). The first type of costs can be valued using VOLYs or DALYs, while the Cost of Illness (COI) method can be used to value the second type of effects. Under the latter method, the valuation is based on market prices. For example, it is possible to estimate the average number of days someone is hospitalised due to a morbidity caused by noise, which is then multiplied by the cost of a day of hospitalisation to determine the economic cost.
As specific cost figures are lacking, we assume an average COI equal to 8% of the calculated DALYs, in line with (HEATCO, 2006).

The number of DALYs can be determined by calculating a Population Attributable Fraction (PAF) using an exposure distribution and the relative risk per 10 dB. Based on this PAF, it can then be estimated how many people *as a result of noise exposure* develop ischemic heart disease and how many DALYs this costs. (Defra, 2014) uses this method in its public model, on which the health costs of noise nuisance in the Handbook on Environmental Prices 2018 are based. Note that this method only calculates with VOLYs, QALYs and DALYs, and not with the VSL; as the probability of developing the health effects studied (such as heart disease, hypertension, stroke and dementia) increases with age. A VSL approach would therefore lead to overestimation of health costs (see Paragraph 5.3).

Threshold and rail bonus

When valuing noise exposure, threshold values are often used. Below this threshold, no noise costs are assigned. Threshold values are used in practice because effects below a certain exposure limit appear negligible, or because reliable indices cannot be established below a certain value.

In addition, the noise nuisance literature sometimes employs a ‘rail bonus’. This rail bonus should reflect the fact that people generally perceive noise exposure from rail traffic as less disturbing than noise exposure from road and air traffic. When a rail bonus of 5 dB(A) is applied, the threshold value for rail traffic is increased by 5 dB(A) (e.g. from 50 to 55 dB(A)) and all valuation numbers move up by 5 dB(A). In Paragraph D.5, we describe why we no longer use this rail bonus in this update of the Handbook.

[95] In principle, nuisance and sleep disturbance cannot simply be added together because nuisance is usually calculated on the basis of Lden, and sleep disturbance on the basis of Lnight.

D.3 Approach in the previous edition of the Handbook

Based on the analysis of available literature in 2016, the previous edition of the Environmental Prices Handbook recommended using the results of (Bristow, et al., 2015) for noise nuisance costs. These values are based on a very extensive meta-analysis of stated preference studies on the valuation of noise nuisance. Moreover, in 2016, these valuations were fairly in line with the average valuation of noise nuisance found in revealed preference surveys.

The Dutch Environmental Prices Handbook 2017 recommended the use of increasing marginal valuations for noise nuisance for the first time. This was in line with scientific findings in this area, which showed that a 1 dB(A) increase is perceived as more disturbing at high than at low noise levels. Increasing marginal costs were also prescribed in other EU countries (Denmark, UK, Sweden).

The results of (Defra, 2014) were used for the health effects of noise, which were applied to the Dutch situation. These results were based directly on the epidemiological insight presented by (WHO, 2011). The range in the valuation of health effects reflects the range used in the Environmental Prices Handbook 2017 for the valuation of DALYs.

When valuing health effects, sleep disturbance costs were excluded to avoid overlap with nuisance costs. In the same way as (HEATCO, 2006), we assumed that people are aware of the effects of noise on sleep disturbance and that costs of this are therefore reflected in WTP values for nuisance.

The previous edition of the Handbook used a threshold of 50 dB(A) for both health effects and nuisance costs. As early as 2016, it was known that nuisance occurs even at lower noise levels (WHO, 2011); (EEA, 2010). However, valuation studies for lower noise levels were almost entirely lacking and ways of extrapolating existing valuation numbers below 50 dB(A) were considered insufficiently reliable.

D.4 New insights

Since 2016, a number of new studies and reports have come out in the field of noise valuation. Of particular importance are the WHO Environmental Noise Guidelines published in 2018 (WHO, 2018), as mentioned above. Many of the new insights are consistent with WHO findings and underlying systematic reviews. RIVM recently produced GGD Guidelines on ambient noise (RIVM, 2019) that incorporates most of the WHO findings and recommendations:

1. In the Environmental Prices Handbook 2018, health costs were based on the (Defra, 2014) model, where a positive causal relationship with heart attacks, hypertension, stroke and dementia was assumed. However, the WHO concludes in its Guidelines that there is no strong evidence as yet for these effects (WHO, 2018). The WHO only indicates that there is 'high or moderate quality' evidence for the relationship between noise exposure and sleep disturbance, and the relationship between road traffic noise exposure and ischaemic heart disease. For rail traffic, no evidence has been found for a relationship with ischaemic heart disease and for air traffic, the evidence is labelled 'very low quality'. As most of the WHO systematic reviews were conducted several years ago, new literature on the health effects of noise has now been published. (Clark, et al., 2020) conclude that most of the WHO's findings still apply, but that in the meantime 'low-quality' evidence exists for a relationship with depression (based on interview criteria) and breast cancer.

2. There is strong evidence that noise nuisance also occurs at noise levels below 50 dB (WHO, 2018). The systematic review into noise nuisance conducted by the WHO (Guski et al., 2017) includes dose-response relationships between noise levels and the percentage of respondents reporting being 'highly annoyed' (%HA). These dose-response functions show a clear increase in the %HA from 40 dB(A) Lden for rail and air traffic, and from around 45 dB(A) for road traffic. Based on these findings, the WHO strongly recommends limiting noise exposure from air traffic to a maximum of 45 dB(A).[96] Because the marginal cost of noise nuisance below 50 dB(A) was not included in the previous Handbook, past nuisance costs appear to have been underestimated.

3. The rail bonus no longer seems supported by the scientific literature (WHO, 2018). Some recent studies conclude that the relationship between noise exposure and reported annoyance is stronger for rail noise than for road noise (see, for example (Guski et al., 2017). Other studies, using physical indicators of nuisance such as heart rate, systolic blood pressure and stress biomarkers, show no different response to track noise than to road noise (Gallash et al., 2016). In addition, because there was already conflicting evidence from RP studies, CE Delft concluded back in 2019 that the rail bonus was no longer supported by academic literature (CE Delft, 2019). The valuation numbers for rail traffic therefore appear to have been slightly underestimated in the previous Handbook.

4. New SP and RP studies on noise nuisance valuation have appeared since 2016. Some of these studies find substantially lower valuation numbers than Bristow, et al. (see e.g. (Kim., et al., 2019), (Bravo-Moncayo, et al., 2017) and (Huh, 2018) while other studies come up with substantially higher valuations (e.g. (Mouter, et al., 2019) and (Winke, 2017)). All in all, we thus still label the results of the (Bristow, et al., 2015) meta-analysis as the most reliable estimates for the nuisance costs of noise.

5. Average estimates in the RP literature based on hedonic pricing are still largely in line with results from the SP literature. A very recent Italian hedonic pricing study (Morano, et al., 2021) finds a decrease in house price of 1-7% per 5 dB(A) increase, which is consistent with the range found earlier in the literature. (Sparrow et al., 2019) based on an analysis of RP literature find a range of 0 to 2.3% with mean value of 0.55% per dB(A). Another recent study by (Ahlfeldt, et al., 2019) observes an average house price drop of 0.4% per dB(A), again in line with previously observed values. Assuming an average house price of €230,000, an average household size of 2.2 people, an interest rate of 5% and a ten-year term, the NSDI of 0.55 corresponds to a marginal WTP of €75 per person per dB(A) per year (CE Delft, 2019). This value is a good match with the values found by (Bristow, et al., 2015) for higher noise levels.

[96] In its recommendations, the WHO assumes a somewhat arbitrary limit of 10% HA, above which noise nuisance should be avoided. Nevertheless, nuisance can also occur below 45 dB(A). The recently published GGD Guidelines (RIVM, 2019) for noise do not adopt the WHO's advisory value for air traffic because a high percentage of severe annoyance (10%) and severe sleep disturbance (11%) still occurs at a noise level of 45 dB(A).

D.5 Choices in the Environmental Prices Handbook 2023

In this paragraph, we describe the new noise valuation recommendations. Note that these recommendations are based on current scientific literature available in 2023. If convincing new evidence is found in the future on the relationships between noise exposure and health effects, this could lead to the application of current recommendations underestimating noise costs.

Nuisance costs

Since 2016, some new SP and RP studies have come out on noise valuation. Because the associated results are not uniform, we have based the nuisance costs of ambient noise on (Bristow, et al., 2015), as in the previous version of the Handbook.

As explained in the previous paragraph, the nuisance costs found in the RP literature are quite similar to the valuation numbers of (Bristow, et al., 2015). We can also compare SP valuations with valuations that follow from the EBD method. For this purpose, the new dose response functions of the WHO, which represent the relationship between noise levels and %HA, can be converted into new estimates of nuisance costs using the Defra model.
The marginal valuations determined in this way are between €0 and €75 (price level 2021), which are considerably lower than those of Bristow, et al. that rise to €262 at 80 dB(A) traffic noise due to air traffic. However, if we add the costs of sleep disturbance (€52, €148 and €108, respectively, for road, rail and air traffic at 80 dB(A)) to the costs of nuisance in the Defra model, the values found are quite similar.[97] We see this as additional substantiation for the assumption that sleep disturbance is taken into account in SP studies and for use of the valuation numbers of (Bristow, et al., 2015).

Health costs

In the previous Handbook, the health costs of noise exposure were determined by adapting the Defra model (Defra, 2014). Among other things, this model attributes costs to an increased probability of hypertension, and thus an increased probability of stroke and dementia. However, in its Environmental Noise Guidelines, the WHO concludes that the evidence for a relationship with hypertension, stroke and dementia is insufficiently supported by science. WHO does find evidence for a relationship between traffic noise exposure and ischaemic heart disease. In the monetisation, we have chosen to include all effects for which the WHO qualifies the evidence as ‘moderate’ or ‘high quality’.
This means that only road traffic includes health costs for ischaemic heart disease.
The relationships with depression and breast cancer found by (Clark, et al., 2020) are labelled ‘low quality’ and therefore we do not include them in the health costs.
We calculate residual health costs by adapting the Defra model, which assumes a relative risk method.

[97] This addition is only possible for road traffic; for rail and air traffic, there are no scientifically substantiated methods for aggregating nuisance and sleep disturbance costs (Defra, 2014).

This adjustment concerns:
- a conversion to euros;
- an adjustment of the value of a QALY;
- an adjustment of the PAF for acute myocardial infarction to be applicable for ischaemic heart disease;
- an adjustment of the burden of disease figures to match the 2019 Dutch ischaemic heart disease figures (based on data from the Global Burden of Disease studies).[98]

Threshold and rail bonus

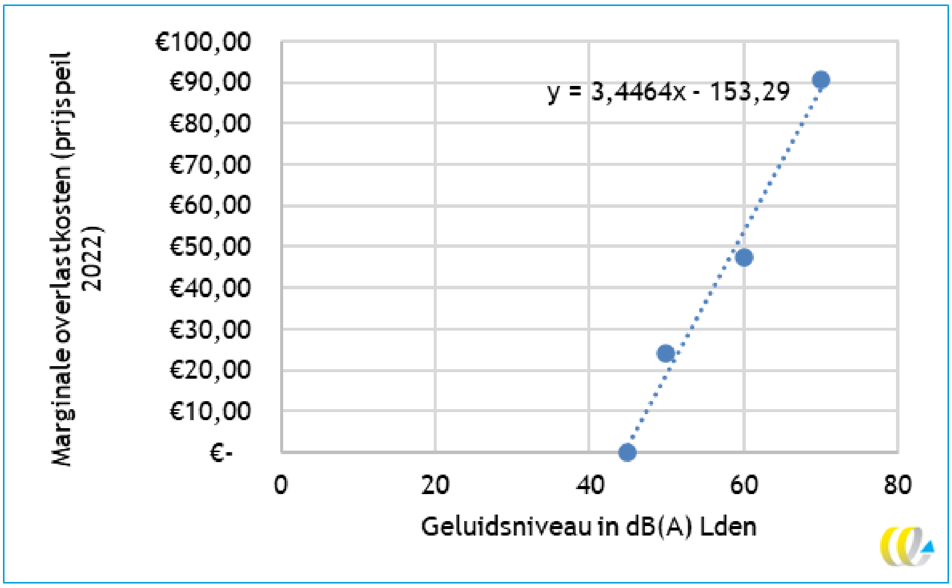
Based on the new dose response functions determined by WHO, we conclude that the previously used threshold of 50 dB(A) Lden is no longer supported by scientific evidence. Indeed, from 40-45 dB(A) onwards, we see a clear increase in the percentage of respondents stating they are 'highly annoyed'. We therefore choose to assume a threshold of 45 dB(A) for all three modalities in the new central value of environmental prices. As in the previous Handbook, we apply this threshold to both nuisance and health costs.
The chosen threshold of 45 dB(A) reflects, on the one hand, the finding that %HA increases as early as 40-45 dB(A) and, on the other hand, is a pragmatic choice: noise models used to calculate noise contours around Schiphol Airport, for example, do not generally correct for background noise. Background noise with an intensity greater than 40 dB(A) exists in much of the Randstad region (Atlas Leefomgeving, 2022). Direct application of environmental prices at 40 dB(A) would therefore potentially lead to an overestimation of noise costs.

In order to do justice to the WHO findings, we recommend an upper value that assumes a threshold of 40 dB(A) for rail and air traffic. In the lower values presented, we assume a conservative threshold of 50 dB(A). We have omitted the rail bonus used in the previous Handbook for all valuations, due to insufficient evidence.

To arrive at a valuation between 40 dB(A) and 50 dB(A), we extrapolate the valuation numbers of (Bristow, et al., 2015). For this purpose, we add an additional data point to the estimates of Bristow, et al. where the %HA equals 0%. We assume that for a %HA of 0%, the nuisance cost is equal to €0. The following figure shows what this extrapolation looks like for road traffic. Note that the linear extrapolation may underestimate environmental costs: after all, at a %HA of 0%, costs of people who are only 'moderately annoyed' may still occur. We choose this conservative assumption because of the complications due to background noise mentioned above and the lack of understanding of how the valuation of nuisance below 50 dB proceeds.

[98] Defra is currently updating its model to include the latest health cost insights. We approached Defra to ask if they were able to share any results. However, researchers said they were not at this stage as yet.

Figure 16 - Extrapolation of marginal nuisance costs to 45 dB(A) for road traffic for the central value



D.6 New environmental prices in the Environmental Prices Handbook 2023 for the Netherlands

In the previous edition of the Environmental Prices Handbook, key valuation indicators were shown as marginal costs. As this created a lot of confusion, we present the new prices as total cost per person per year. If someone is exposed to road traffic of, say, 55 dB(A) Lden for a year, associated costs can now be read directly at 55 dB(A), without further calculations. The new environmental prices are shown in Table 76 to Table 78 in classes of 5 dB(A).

Table 76 - New environmental prices for road traffic noise exposure, in €$_{2021}$ per person per year for the Netherlands

Noise class (dB(A) Lden)	Lower value	Central value	Upper value
40-45	–	–	–
45-50	–	€ 51	€ 68
50-55	€ 93	€ 201	€ 248
55-60	€ 313	€ 439	€ 517
60-65	€ 620	€ 765	€ 875
65-70	€ 1,015	€ 1,180	€ 1,323
70-75	€ 1,498	€ 1,683	€ 1,861
75-80	€ 2,069	€ 2,276	€ 2,489
80+	€ 2,450	€ 2,670	€ 2,906

Table 77 - New environmental prices for noise exposure from rail traffic, in €2021 per person per year for the Netherlands

Noise class (dB(A) Lden)	Lower value	Central value	Upper value
40-45	–	–	€ 11
45-50	–	€ 52	€ 78
50-55	€ 93	€ 188	€ 215
55-60	€ 298	€ 393	€ 420
60-65	€ 572	€ 667	€ 694
65-70	€ 914	€ 1,009	€ 1,036
70-75	€ 1,325	€ 1,421	€ 1,447
75-80	€ 1,805	€ 1,900	€ 1,927
80+	€ 2,123	€ 2,218	€ 2,245

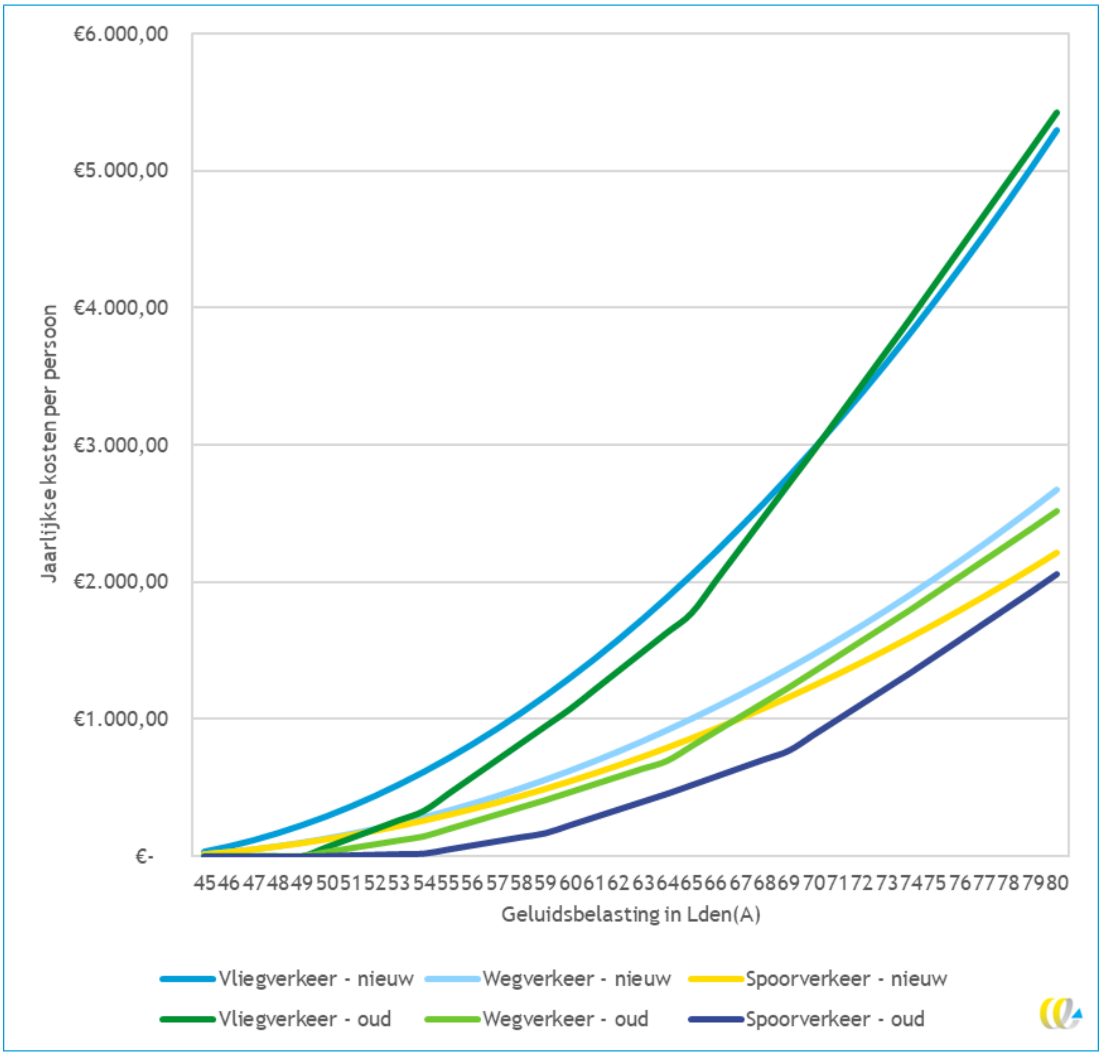
Table 78 - New environmental prices for noise exposure from air traffic, in €2021 per person per year for the Netherlands

Noise class (dB(A) Lden)	Lower value	Central value	Upper value
40-45	–	–	24
45-50	–	€ 121	€ 183
50-55	€ 220	€ 445	€ 506
55-60	€ 708	€ 933	€ 994
60-65	€ 1,361	€ 1,585	€ 1,647
65-70	€ 2,179	€ 2,403	€ 2,465
70-75	€ 3,161	€ 3,386	€ 3,447
75-80	€ 4,309	€ 4,533	€ 4,595
80+	€ 5,069	€ 5,294	€ 5,355

D.7 Comparison between the new and the previous environmental prices

The following figure shows a comparison between the previous and the new central environmental prices for noise in the Netherlands. To this end, the previous environmental prices have been adjusted for inflation, but not for the increased valuation of a VOLY. For lower noise levels, the new environmental prices are significantly higher, due to the introduction of a lower threshold (45 dB(A) vs 50 dB(A)) and the expiry of the rail bonus. For air traffic, at higher noise levels, we see that the previous prices are higher than the new prices. This is because a number of health effects are no longer included in the new Handbook because they are not supported by the literature.

Figure 17 - Comparison between previous and new environmental prices for noise (total cost in €$_{2021}$ per person per year)



E Treatment of uncertainty

The valuations per unit of pollutants, as reported in this Handbook, have been estimated with different assumptions and models. In doing so, each analysis step has a certain degree of uncertainty. These uncertainties accumulate and effectively increase with each analysis step. There are three major components of uncertainty:
1. Uncertainty about the distribution of pollutants throughout the environment and the change in concentrations due to emissions.
2. Uncertainty about dose-effect relationships: what effects occur due to a change in concentration.
3. Uncertainty about valuation.

In the Environmental Prices Handbook, only the latter uncertainty is explicitly included in a lower and upper range of estimates for all themes. In addition, for ecotoxicity and human toxicity, it has been estimated that (a) and (b) involve such a high degree of uncertainty (much more than for other environmental themes) that in the lower and upper values we have also included this uncertainty in the determination of environmental prices where for the lower value the more *certain* effects have been included and in the upper value the more *uncertain* effects have been included.

It is useful to distinguish between risk and uncertainty. Risk refers to a situation in which we have some idea of the possibility of a particular effect occurring. Often, however, we do not know the probability at all. That is uncertainty. For example, we are not yet able to assign probabilities to the harmfulness of bioaccumulative substances: substances that do not degrade in the environment. For these reasons, these substances are largely excluded in underlying models such as Usetox (see Annex B) and no environmental price is determined for them either.

E.1 Uncertainty in determining health damage

Health damage is the biggest harm in most environmental pollutants. We therefore focus mainly on uncertainties related to valuation of mortality and morbidity effects.

The EEA (2021) and NEEDS (2008a) projects assessed the effects of classic pollutants on human health by aggregating the effects of specific pollutants on various forms of health damage that are well documented in medical literature. In doing so, we largely followed the 2013 WHO guidelines. However, it is increasingly evident that air pollution is also a co-morbidity at endpoints that we have not quantified, such as diabetes or dementia. In addition, all dose-response relationships from WHO (2013) *did* include a margin of uncertainty.

The treatment of uncertainty was discussed in (NEEDS, 2008b). The methodology used there involves assessing geometric standard deviations (σg) of the damage cost estimates, assuming a lognormal distribution. For classical pollutants, NEEDS (2008b) suggested that the geometric standard deviation of these damage costs is about 3. According to the characteristics of a lognormal distribution, this means that for classical pollutants the true value lies in the interval between the middle value divided by three and the middle value multiplied by three with a probability of 68%. For toxicity, (Rabl, et al., 2014) report an even larger uncertainty with a standard deviation of a factor of 4. In the previous

Environmental Prices Handbook (CE Delft, 2017a), it was argued that an even higher uncertainty for toxic emissions seemed likely to us.

We did not perform a new formal treatment of uncertainty in this Handbook but analysis of the calculation routines underlying this study showed that the standard deviation is now comparable. The analyses showed the following effects:
- If the concentration response functions were modelled at the lower value of the 95% confidence interval, harmfulness would decrease by about 30-40%: it would increase by the same amount for the upper value.[99]
- In general, the uncertainty of the valuation reveals that the standard deviation for the central value is also around 40%.
- The effects of a different dispersion of emissions are unknown.

In summary, a standard deviation of a factor of 3 does not seem unrealistic to us. About 1/3 of this is made explicit in the variation of the valuation, but there may be as much variation in the other effects.

However, a large uncertainty range does not always provide a meaningful route for inclusion in SCBAs, for example, because the ranges between lower and upper values can become very large and can also be larger than is common in adjacent domains such as healthcare (see SEO, 2016a). Although we consider that communicating uncertainty as openly as possible is appropriate, including a wider range in the upper and lower limits of estimates would affect the SCBA of environmental measures in a predictable way: at the upper limit, environmental impacts would dominate other costs and benefits, and at the lower limit, environmental impacts could be neglected. In an SCBA, all other items, including financial items, are also subject to uncertainties. If we represented all items with the fully statistically correct margins of uncertainty, an SCBA could no longer be conducted.

Therefore, we recommend using the lower and upper values as presented in this Handbook in SCBAs. If the results from the SCBA are very sensitive to the exact values of the environmental prices, it may make sense to perform additional sensitivity analyses taking the 1/3 to 3 factor for the central values and additional compensation should be made for missing health impact studies for the upper values. However, this should not be applied to toxic effects because there is already a more extensive uncertainty margin for these effects, where uncertainty about dose-effect relationships and dispersion is explicitly included in the estimates.

E.2 Uncertainty regarding other effects

There is even less literature available that attempts to estimate the uncertainty in the remaining endpoints, but in all likelihood, it is even higher. However, we have generally made a conservative assumption here, so that there is a *higher* probability that the actual damage is higher than lower.

[99] This is a tentative analysis and should be determined more precisely in a future study.

F Environmental prices of individual pollutants

F.1 Introduction

This annex contains the environmental prices for common pollutants for which a valuation has been calculated. This table should be read alongside the tables set out in Chapter 2 and complement Chapter 2. In compiling this list, we were guided by the following considerations:
- For emissions to soil and air, the classification as *substances of very high concern* was followed. Only for a limited number of these susbtances environmental prices could be established.
- For emissions to water, the pollutants listed in the Water Framework Directive (WFD) and the priority pollutants listed in the Water Quality Requirements and Monitoring Decree 2009 (Besluit kwaliteitseisen en monitoring water 2009) have been followed. We have divided the environmental prices into emissions to inland waters and emissions to oceans.

F.2 Environmental prices for emissions to air

Table 79 - Environmental prices (damage costs) for emissions to air in the EU, in €$_{2021}$/kg

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
(epoxyethyl) benzene	000096-09-3	4.08	6.03	9.07
1,2,3,6,7,8-hexachlorodibenzo-p-dioxine	057653-85-7	0	1,730	2,240
1,2,3,7,8-pentachlorodibenzodioxin	040321-76-4	0	61,991	80,267
1,2,3-trichlorobenzene	000087-61-6	0.0111	0.0157	0.0203
1,2,3-trichloropropane	000096-18-4	720	1,064	1,603
1,2,4-trichlorobenzene	000120-82-1	0.168	0.248	0.373
1,2-benzenedicarboxylic acid, di-C6-10-alkyl esters	068515-51-5	0	0	0
1,2-benzenedicarboxylic acid, di-C7-11 branched and linear alkyl esters	068515-42-4	0	0	0
1,2-dibromo-3-chloropropane	000096-12-8	2,148	3,175	4,781
1,2-dibromoethane	000106-93-4	249	367	553
1,2-dichloropropane	000078-87-5	20.4	33.8	51
1,2-epoxy-3-phenoxypropane	000122-60-1	3.04	4.5	6.78
1,3,5,7,9,11-hexabromocyclododecane	025637-99-4	0	1.26	1.63
1,3,5-trichlorobenzene	000108-70-3	0	0.00383	0.00496
1,3-butadiene	000106-99-0	1.4	2.01	2.88
1,3-dichloro-2-propanol	000096-23-1	6.34	8.99	11.6
1,3-propanesultone	001120-71-4	163	241	362
1,3-propiolactone	000057-57-8	963	1424	2146
1,4,5,8-tetraaminoanthraquinone	002475-45-8	0.357	0.528	0.795
1,4-dichlorobut-2-ene	000764-41-0	0	567	854
1,5,9-cyclododecatrine	004904-61-4	0	0.00000249	0.00000323

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
1-bromopropane	000106-94-5	0	0.0000631	0.0000817
1-methyl-3-nitro-1-nitrosoguanidine	000070-25-7	2078	3072	4627
1-methylnaphthalene	000090-12-0	0.0000703	0.0000996	0.000129
2-(2-aminoethylamino)ethanol	000111-41-1	0	0	0
2-(2-methoxyethoxy)ethanol	000111-77-3	0	0.00336	0.00435
2,2'-(nitrosoimino)bisethanol	001116-54-7	274	404	609
2,2-bis(bromomethyl)propane-1,3-diol	003296-90-0	13.9	20.5	30.9
2,3,7,8-tetrachloordibenzodioxine	001746-01-6	34,071,638	50,367,195	75,846,980
2,3,7,8-tetrachlorodibenzofuran	051207-31-9	0	46,685	60,448
2,3-dibromopropane-1-ol	000096-13-9	0	0.126	0.163
2,3-dinitrotoluene	000602-01-7	0.942	1.34	1.73
2,3-epoxypropyl-trimethylammoniumchloride	003033-77-0	0	0.00629	0.00814
2,4,5-trimethylaniline	000137-17-7	0	13.1	19.7
2,4,5-trimethylanilinehydrochloride	021436-97-5	0	1.76	2.65
2,4,6-tri-tert-butylphenol	000732-26-3	0	0.146	0.189
2,4-diaminoanisole sulfate	039156-41-7	0	15.6	23.5
2,4-dinitrotoluene	000121-14-2	10.6	15.6	23.2
2,5-dinitrotoluene	000619-15-8	0	0.791	1.02
2,6-dinitrotoluene	000606-20-2	1,477	2,184	3,288
2-butenal	004170-30-3	0.0042	0.00595	0.00771
2-ethoxyethanol	000110-80-5	0.325	0.455	0.625
2-ethoxyethyl acetate	000111-15-9	0.012	0.017	0.0219
2-ethylhexanoic acid	000149-57-5	0	0.0056	0.00725
2-methoxyethanol	000109-86-4	0.588	0.851	1.23
2-ethoxyethyl acetate	000110-49-6	0	0.117	0.151
2-methylimidazole	000693-98-1	0	2.72	4.1
2-methylnaphthalene	000091-57-6	0.0746	0.11	0.166
2-naphthylamine	000091-59-8	1.88	2.78	4.19
2-nitroanisole	000091-23-6	23	34	51.2
2-nitropropane	000079-46-9	0.615	0.909	1.37
2-nitrotoluene	000088-72-2	0.00997	92.6	140
3,3'-dichlorobenzidine	000091-94-1	31.6	47	70.7
3,3'-dichlorobenzidine dihydrochloride	000612-83-9	0	472	710
3,3'-dimethoxybiphenyl-4,4'-ylenediammonium dichloride	020325-40-0	0	2425	3651
3,3'-dimethylbenzidinedihydrochloride	000612-82-8	0	661	995
3,4-dinitrotoluene	000610-39-9	0	0.358	0.464
3,5-dinitrotoluene	000618-85-9	0	0.173	0.224
4,4'-(4-iminocyclohexa-2,5-dienylidene methylene)dianiline hydrochloride	000569-61-9	0	127	191
4,4'-methyleen-bis(2-chlooraniline) hydrochloride	064049-29-2	0	7.9	11.9
4,4'-bi-o-toluidine	000119-93-7	0	0.834	1.08
4,4'-bis(dimethylamino)benzophenone	000090-94-8	0	271	408
4,4'-methylene bis(2-chloroaniline)	000101-14-4	34.5	51	76.8
4,4'-methylenedianiline	000101-77-9	0	0	0
4,4'-methylenedi-o-toluidine	000838-88-0	48.9	72.4	109
4,4'-oxydianiline	000101-80-4	108	159	240
4,4'-thiodianiline	000139-65-1	74.6	110	166
4-aminoazobenzene	000060-09-3	0	1.99	2.58

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
4-aminobiphenyl	000092-67-1	58.1	85.9	129
4-chloroaniline	000106-47-8	0.0429	0.0624	0.0891
4-chloro-o-toluidine hydrochloride	003165-93-3	7.26	10.7	16.2
4-methyl-m-phenylenediamine	000095-80-7	146	215	324
4-nonylphenol, branched	084852-15-3	0.0272	0.0386	0.05
4-tert-butylbenzoic acid	000098-73-7	0	0.436	0.564
4-tert-butylphenol	000098-54-4	0.00247	0.00351	0.00455
5-allyl-1.3-benzodioxole	000094-59-7	2.31	3.42	5.15
5-nitroacenaphthene	000602-87-9	25.1	37.1	55.8
6.6‘-di-tert-butyl-2.2’-methylenedi-p-cresol	000119-47-1	0	0	0
6-methoxy-m-toluidine	000120-71-8	0.995	1.47	2.22
7-oxa-3-oxiranylbicyclo[4.1.0]heptane	000106-87-6	0	0	0
8-hydroxyquinoline	000148-24-3	0	0.00653	0.00845
Acenaphthene	000083-32-9	0.0111	0.0165	0.0247
Acridine	000260-94-6	0.184	0.261	0.337
Acrylamide	000079-06-1	307	455	685
Acrylonitrile	000107-13-1	30.5	45.1	67.9
Aldrin	000309-00-2	24.1	225	339
Alpha-endosulfan	000959-98-8	10.8	15.4	19.9
Alpha-hexachlorocyclohexane	000319-84-6	153	227	341
Anthracene	000120-12-7	0.0706	0.1	0.13
Anthraquinon	000084-65-1	0.138	0.195	0.253
Arsenic	007440-38-2	6,271	9,275	13,980
Arsenic pentoxide	001303-28-2	1,713	5,010	18,520
Aziridine	000151-56-4	829	1,226	1,846
Azobenzene	000103-33-3	0.196	18.7	28.1
Azocyclotin	041083-11-8	142	203	271
Benomyl	017804-35-2	0.44	0.632	0.861
Benz[a]acridine	000225-11-6	0	12.9	16.7
Benz[c]acridine	000225-51-4	0	155	201
Benzene	000071-43-2	0.278	0.405	0.593
Benzidine	000092-87-5	89,692	132,591	199,666
Benzidine dihydrochloride	000531-85-1	0	107	160
Benzo[a]anthracene	000056-55-3	0	0.996	1.29
Benzo[a]pyrene	000050-32-8	3,859	5,704	8,590
Benzophenone	000119-61-9	0	0.123	0.159
Benzotrichloride	000098-07-7	1,523	2,252	3,391
Benzyl butyl phthalate	000085-68-7	0.0434	0.588	0.881
Benzyl chloride	000100-44-7	7.36	10.9	16.4
Beryllium	007440-41-7	31,345	46,380	69,972
Beta-endosulfan	033213-65-9	5.13	7.27	9.43
Beta-hexachlorocyclohexane	000319-85-7	34.2	51.3	77.1
Binapacryl	000485-31-4	0.288	0.409	0.53
Bis(2-ethylhexyl)phthalate	000117-81-7	1.82	5.89	8.87
Bis(chloromethyl)ether	000542-88-1	97,380	143,955	216,787
Bis(pentabromophenyl)ether	001163-19-5	28.3	44.2	66.7
Bisphenol A	000080-05-7	0.56	0.825	1.23
Brodifacoum	056073-10-0	0.00834	0.0119	0.0158
Butane	000106-97-8	0.23	0.319	0.43

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
Butanon-oxime	000096-29-7	0	5.67	8.54
Butylparaben	000094-26-8	0	0	0
C.I. Basic Violet 3 [containing 0.1 per cent or more Michler's ketone (EC No 202-027-5)]	000548-62-9	0	494	655
Cadmium	007440-43-9	105,034	155,294	233,924
Captafol	002425-06-1	11.5	16.7	23.8
Carbendazim	010605-21-7	6.25	8.95	12.1
Carbetamide	016118-49-3	0.841	1.19	1.54
Catechol	000120-80-9	5.05	7.52	11.3
Quinoline	000091-22-5	0.00787	251	378
Chlordecone	000143-50-0	8,829	13,052	19,654
Chlorodimethyl ether	000107-30-2	45.6	67.4	101
Chlorfenvinphos	000470-90-6	85	125	183
Chloromethyl mercury	000115-09-3	0	658	991
Chlorotriethyl lead	001067-14-7	5.43	7.7	9.96
Chloroprene	000126-99-8	30.7	45.4	68.3
Chromium (VI)	018540-29-9	1,815	2,703	4,121
Chrysotile	012001-29-5	0	0.00426	0.0252
Cumatetralyl	005836-29-3	0.0142	0.0201	0.026
Cumene	000098-82-8	0.246	0.342	0.463
Cyclododecane	000294-62-2	0	0.00000247	0.0000032
Cycloheximide	000066-81-9	0	58.5	75.7
Cyhexatin	013121-70-5	7.92	11.7	17.4
DDT, 2,4'-isomer	000789-02-6	0	2.64	3.42
DDT, 4,4'-isomer	000050-29-3	684	1,013	1,530
Delta-hexachlorocyclohexane	000319-86-8	1.2	1.7	2.21
Dibenzo[a,h]anthracene	000053-70-3	331	489	736
Dibromo-nitrilopropiamide	010222-01-2	0	33.4	43.2
Dibutyl phthalate	000084-74-2	0.0814	0.117	0.16
Dibutyltin di(acetate)	001067-33-0	0	0.102	0.133
Dibutyltin dichloride	000683-18-1	0.0153	0.0217	0.0281
Dibutyl tindilaurate	000077-58-7	0	0.00205	0.00266
Dibutyltinoxide	000818-08-6	0	0.00000442	0.00000572
Dicofol	000115-32-2	142	578	870
Dieldrin	000060-57-1	368	4,473	6,735
Di-ethylenetriamine pentaacetic acid	000067-43-6	0	0.236	0.306
Diphenacoum	056073-07-5	0	0.00111	0.00144
Diphenylchlororesin	000712-48-1	0	1.31	1.69
Dihexyl phthalate	000084-75-3	0	0.00186	0.0024
Diisobutyl phthalate	000084-69-5	0	0.00454	0.00588
Dimethomorph	110488-70-5	0	1.01	1.3
Dimethylcarbamoyl chloride	000079-44-7	137	203	305
Dimethyl sulphate	000077-78-1	0	0.00642	0.00832
Dimethyltin dichloride	000753-73-1	9.13	13	16.8
Disodium-{5-[(4'-((2,6-dihydroxy-3-((2-hydroxy-5-sulfophenyl)azo)phenyl)azo)(1,1'-biphenyl)-4-yl)azo]sa	016071-86-6	0	3,961	5,964
Disodium 3,3'-[[1,1'-biphenyl]-4,4'-diyl bis(azo)]bis(4-aminonaphthalene-1-sulphonate)	000573-58-0	0	0	0

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
Dinatrium-4-amino-3-[[4'-[(2,4-diaminofenyl)azo][1,1'-bifenyl]-4-yl]azo]-6-(fenylazo)-5-hydroxynaftal	001937-37-7	0	5107	7,690
Dinitrotoluene	025321-14-6	0	67.3	101
Dinocap	039300-45-3	3.76	5.54	8.24
Dinoseb	000088-85-7	14.1	20.6	29.5
Dinoterb	001420-07-1	6.04	8.56	11.1
Dioxane	000123-91-1	1.76	2.61	3.93
Diuron	000330-54-1	17.4	25	33.7
E-2-butenal	000123-73-9	0	38.6	58.1
Endosulfan	000115-29-7	3	4.39	6.41
Endrin	000072-20-8	46.5	68.3	100
Epichlorohydrin	000106-89-8	13.6	20	30.2
Ethanal	000075-07-0	1.79	2.6	3.83
Ethenyl ester of neodecanoic acid	051000-52-3	0	0	0
Ethyl-1-(2,4-dichlorophenyl)-5-(trichloromethyl)-1H-1,2,4-triazole-3-carboxylate	103112-35-2	19.3	27.4	35.6
Ethylenediamine	000107-15-3	0.302	0.428	0.554
Ethylene oxide	000075-21-8	11.4	16.9	25.5
Ethylenethiourea	000096-45-7	4.97	154	232
Ethyl-p-nitrophenylthiobenzene-phosphenate	002104-64-5	174	258	388
Phenanthrene	000085-01-8	0.0351	0.0498	0.0645
Fenantridine	000229-87-8	0	3.3	4.27
Fenbutatin	013356-08-6	316	465	688
Phenolphthalein	000077-09-8	2.58	3.81	5.74
Fentin hydroxide	000076-87-9	59	84.6	114
Phenylarsenic acid	000098-05-5	0	0.0543	0.0703
Phenylhydrazine	000100-63-0	0	0.1	0.13
Phenylhydrazine chloride	000059-88-1	0	31.7	47.7
Phenylmercury acetate	000062-38-4	2,122	3,117	4,594
Fluazifop-butyl	069806-50-4	0.3	0.425	0.551
Flucythrinate	070124-77-5	398	564	732
Fluoranthene	000206-44-0	0.217	0.309	0.411
Fluorene	000086-73-7	0.0344	0.0507	0.076
Flurochloridone	061213-25-0	2.92	4.14	5.37
Formaldehyde	000050-00-0	0.491	0.694	0.967
Formamide	000075-12-7	0	0	0
Furan	000110-00-9	258	382	575
Gamma hexachlorocyclohexane	000058-89-9	106	157	235
Glufosinate ammonium	077182-82-2	2.62	3.85	5.72
Glutaaraldehyde	000111-30-8	0	0	0
Glycidol	000556-52-5	118	175	264
Heptachloride	000076-44-8	117	172	260
Heptachlor epoxide	001024-57-3	4,613	6,828	10,311
Heptachloronorbornene	028680-45-7	0	0.00387	0.00501
Hexachlorobenzene	000118-74-1	258	382	576
Hexachlorobutadiene	000087-68-3	0.0247	34	51.2
Hexachlorocyclohexane	000608-73-1	79.7	118	176
Hexachlorocyclopentadiene	000077-47-4	143	212	319

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
Hexadecafluorheptane	000335-57-9	434	1,129	1,389
Hexahydrophthalic acid anhydride	000085-42-7	0	0	0
Hexamethylphosphoramide	000680-31-9	3,302	4,881	7,350
Hydrazine	000302-01-2	696	1,027	1,534
Hydrazobenzene	000122-66-7	0	17.7	26.6
Imidazole	000288-32-4	0	0	0
Isobutane	000075-28-5	0.208	0.288	0.388
Isobutyl nitrite	000542-56-3	0	6.94	10.5
Isodrin	000465-73-6	0	0.652	0.844
Isoprene	000078-79-5	1.19	1.69	2.35
Isoquinoline	000119-65-3	0.0158	0.0224	0.029
Cacodylic acid	000075-60-5	73.6	109	164
Cobalt	007440-48-4	11.8	23.2	50.1
Mercury	007439-97-6	9,983	14,951	23,019
Linuron	000330-55-2	35	51.7	77.6
Lead	007439-92-1	18,455	27,287	41,106
Lead styphnate	015245-44-0	0	0.00115	0.00149
Mancozeb	008018-01-7	22.7	32.3	42.5
M-bis(2,3-epoxypropoxy)benzene	000101-90-6	11.5	17	25.6
Methoxy chloride	000072-43-5	0.845	1.21	1.61
Methyl hydrazine	000060-34-4	2.01	38	56.9
Mirex	002385-85-5	1,032	1,546	2,404
Musk xylene	000081-15-2	0	48	72.2
N-(1,3-dimethylbutyl)-N'-phenyl-1,4-benzenediamine	000793-24-8	0	0	0
N,N,N',N'-tetramethyl-4,4’-methylenedianiline	000101-61-1	0	5.72	8.61
N,N-dimethylacetamide	000127-19-5	0	0	0
N,N-dimethylformamide	000068-12-2	1.17	1.74	2.62
N,N-dimethylhydrazine	000057-14-7	99.6	147	221
Naphthalene	000091-20-3	0.527	10.4	15.6
Sodium cacodylate	000124-65-2	0.566	0.803	1.04
Sodium pentachlorophenolate	000131-52-2	1.72	2.45	3.18
N-butyltin trichloride	001118-46-3	0.365	0.518	0.67
Nickel	007440-02-0	67.6	126	257
Nitrobenzene	000098-95-3	15.1	22.3	33.6
Nitrophene	001836-75-5	13	19.1	28.8
Nitrosodipropylamine	000621-64-7	731	1,081	1,628
N-methyl-2-pyrrolidon	000872-50-4	0	0.532	0.801
N-methylolacrylamide	000924-42-5	0	83.5	126
N-nitrosodimethylamine	000062-75-9	3,178	4,699	7,076
Nonylphenol	025154-52-3	0.0017	0.00241	0.00312
O-aminoazotoluene	000097-56-3	127	188	283
O-anisidine	000090-04-0	0	0.0515	0.0667
Octachlorinated naphthalene	002234-13-1	0	0	0
Octamethyltetrasiloxane	000556-67-2	0	0.0195	0.0252
O-toluidine	000095-53-4	0.00105	0.00149	0.00193
P-(1,1-dimethylpropyl)phenol	000080-46-6	0	0.0552	0.0715
Para-tert-octylphenol	000140-66-9	0.0109	0.0154	0.02
PCB 101	037680-73-2	0	1.58	2.05

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
PCB 77	032598-13-3	0	1.16	1.5
Pentachloranisole	001825-21-4	0	27.6	41.5
Pentachlorobenzene	000608-93-5	5.98	8.87	13.5
Pentachloroethane	000076-01-7	0.00277	44.2	66.6
Pentachlorophenol	000087-86-5	15	22.1	32.3
Pentasodium diethylene-triaminepentaacetic acid	000140-01-2	0	0.0628	0.0813
P-nonylphenol	000104-40-5	0.00183	0.0026	0.00336
Polychlorinated biphenyls	001336-36-3	0	467	704
Propiconazole	060207-90-1	3.1	4.54	6.63
Propylene oxide	000075-56-9	6.91	10.2	15.4
Pyrene	000129-00-0	0.338	0.48	0.625
Pyrithione zinc	013463-41-7	0	0	0
Roxarsone	000121-19-7	0	0	0
Sulphallate	000095-06-7	2.23	3.3	4.97
Tellurium	013494-80-9	0	0.261	1.55
Terphenyl	026140-60-3	0	0	0
Tetrabromobisphenol A	000079-94-7	0.178	0.254	0.335
Tetrabutyltin	001461-25-2	0.00000194	0.00000275	0.00000356
Tetraethyl lead	000078-00-2	3,320	4,908	7,391
Tetrafluoroethylene	000116-14-3	13.1	19.3	29.1
Tetrahydro-2-furylmethanol	000097-99-4	0	0.00604	0.00782
Tetramethyl lead	000075-74-1	0.0000324	0.0000459	0.0000594
Tetrasodium-3,3'-[[1,1'-biphenyl]-4,4'-diylbis(azo)]bis[5-amino-4-hydroxynaphthalene-2,7-disulfonate]	002602-46-2	0	9,514	14,327
Tetrasul	002227-13-6	0	0.00176	0.00228
Theophylline	000058-55-9	0	0	0
Thioacetamide	000062-55-5	59.1	87.8	132
Thiomersal	000054-64-8	1.99	2.83	3.66
Toxaphene	008001-35-2	2,269	3,379	5,176
Triadimenol	055219-65-3	4.63	6.77	9.8
Tributyltin	000688-73-3	0.00019	0.00027	0.000349
Tributyltin oxide	000056-35-9	0.73	1.08	1.62
Trichlorobenzene	012002-48-1	0	0.00338	0.00437
Trichloroethylene	000079-01-6	0.62	0.896	1.3
Tridemorph	024602-86-6	0	3.92	5.08
Triphenyltin acetate	000900-95-8	55.1	81.3	122
Triphenyltin chloride	000639-58-7	22.7	33.6	50.3
Triflumizole	068694-11-1	20.7	29.3	38
Trifluraline	001582-09-8	0.489	1.93	2.91
Tris(2,3-dibromopropyl)phosphate	000126-72-7	33.9	50.1	75.4
Tris(2-chloroethyl)phosphate	000115-96-8	0.0016	2.49	3.74
Triton X-100	009002-93-1	0.000000175	0.000000249	0.000000322
Trixylyl phosphate	025155-23-1	0	0.0256	0.0332
Urethane	000051-79-6	37	54.7	82.3
Vinchlozolin	050471-44-8	3.33	4.92	7.35
Vinyl bromide	000593-60-2	12.4	18.4	27.7
Vinyl chloride	000075-01-4	36.4	53.8	81
Warfarin	000081-81-2	114	169	254

F.3 Environmental prices for emissions to water

Environmental prices for emissions to water are broken down into emissions to inland waters and emissions to marine waters.

F.3.1 Emissions to inland waters

Table 80 - Environmental prices (damage costs) for emissions to inland waters in the EU, in €$_{2021}$/kg

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
1,2-Dichloropropane	000078-87-5	7.67	12.7	19.2
4-Chloroaniline	000106-47-8	6	0.189	0.253
Aclonifen	074070-46-5	2.13	3.02	3.91
Alachlor	015972-60-8	1.44	2.04	2.64
Aldrin	000060-57-1	5,646	68,740	103,505
Anthracene	000120-12-7	5.35	7.59	9.84
Antimony	007440-36-0	6.37	23.5	86
Arsenic (and inorganic compounds thereof)	007440-38-2	171	2,411	11,361
Atrazine	001912-24-9	2.97	10.4	14.8
Azinphos-ethyl	002642-71-9	151	214	277
Azinphos-methyl	000086-50-0	55.9	79.5	104
Barium	007440-39-3	2.07	6.02	19.7
Bentazon	025057-89-0	0.141	0.209	0.311
Benz(a)anthracene	000056-55-3	0	33.9	43.9
Benzene	000071-43-2	0.971	1.44	2.16
Benzo(a)pyrene	000050-32-8	100	148	223
Benzyl chloride (alpha-chlorotoluene)	000100-44-7	1.26	1.86	2.8
Beryllium	007440-41-7	0.977	2.7	9.34
Bifenox	042576-02-3	1.08	1.54	1.99
Borium	007440-42-8	0	0.00161	0.00956
Cadmium and cadmium compounds	007440-43-9	3.06	31.5	144
Captan	000133-06-2	5.47	8.71	11.6
Carbendazim	010605-21-7	0.771	1.1	1.45
Chlorpropham	000101-21-3	1.09	1.59	2.29
Chlorotoluron	015545-48-9	0.277	0.394	0.51
Chlorpyrifos	002921-88-2	358	515	709
Chlorfenvinphos	000470-90-6	341	504	756
Chromium	007440-47-3	0.031	0.0477	0.0734
Cybutryne	028159-98-0	0	2,141	2,772
Cyclodiene and pesticides:	000309-00-2	1,544	14,168	21,334
Cypermethrin	052315-07-8	957	1,360	1,770
Deltamethrin	052918-63-5	97.8	139	180
Di(2-ethylhexyl)phthalate (DEHP)	000117-81-7	1.13	3.61	5.42
Diazinon	000333-41-5	35.1	51.5	75.6
Dichloromethane	000075-09-2	0.849	1.25	1.89
Dichlorvos	000062-73-7	38.4	56.5	84.2
Dicofol	000115-32-2	1,539	6,305	9,493
Dieldrin(7)	000072-20-8	2,065	3,045	4,549
Dimethoate	000060-51-5	0.941	1.36	1.87
Diuron	000330-54-1	3.89	5.55	7.38

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
Endosulfan	000115-29-7	25.7	36.8	49.7
Endrin(7)	000465-73-6	0	59.1	76.5
Esfenvalerate	066230-04-4	1,241	1,760	2,280
Ethylbenzene	000100-41-4	0.27	0.398	0.597
Fenamiphos	022224-92-6	29.5	42	55.2
Phenanthrene	000085-01-8	0.875	1.24	1.61
Fenitrothion	000122-14-5	14.6	21.2	29.5
Phenoxycarb	072490-01-8	1.43	2.03	2.63
Fenthion	000055-38-9	49.1	71.7	103
Fluoranthene	000206-44-0	10.3	14.6	19.2
Heptachloride	000076-44-8	8,105	11,980	18,040
Heptachlor epoxide	001024-57-3	286,217	423,113	637,187
Heptenophos	023560-59-0	3.25	4.6	5.97
Hexachlorobenzene	000118-74-1	3,164	4,678	7,046
Hexachlorobutadiene	000087-68-3	0.227	103	155
Hexachlorocyclohexane	000608-73-1	225	332	498
Something with xylene	000108-38-3	0.0073	0.0103	0.0134
Something with xylene	000106-42-3	0.00773	0.011	0.0142
Imidacloprid	138261-41-3	0.267	0.386	0.541
Isoproturon	034123-59-6	1.18	1.67	2.17
Cobalt	007440-48-4	0.0867	0.225	0.747
Copper	007440-50-8	2.21	3.46	5.56
Mercury and mercury compounds	007439-97-6	9.1	1,346	6,802
Lambda-cyhalothrine	091465-08-6	734	1,045	1,364
Linuron	000330-55-2	7.13	10.2	13.8
Lead and lead compounds	007439-92-1	0.626	9.02	42.6
Malathion	000121-75-5	2.35	3.34	4.33
MCPA	000094-74-6	1.03	1.53	2.29
Mecoprop-P	016484-77-8	0	0.0275	0.0356
Metazachlor	067129-08-2	0.32	0.454	0.588
Methabenzthiazuron	018691-97-9	0.24	0.341	0.442
Metolachlor	051218-45-2	2.37	3.37	4.41
Metsulfuron-methyl	074223-64-6	75.8	107	139
Mevinphos	007786-34-7	19.4	27.9	38.3
Molybdenum	007439-98-7	1.27	3.51	11.5
Monolinuron	001746-81-2	0.254	0.361	0.467
Naphthalene	000091-20-3	0.0844	3.73	5.6
Nickel and nickel compounds	007440-02-0	9.86	37.7	140
Nonylphenols	084852-15-3	3.29	4.67	6.05
Octamethylcyclotetrasiloxane	000556-67-2	0	22.7	29.4
Omethoate	001113-02-6	4.32	6.38	9.59
para-para-DDT	000050-29-3	3,237	4,799	7,278
Parathion	000056-38-2	31.5	45.3	61.5
Parathion-methyl	000298-00-0	16	23.6	35
Pentachlorobenzene	000608-93-5	24.1	35.6	53.7
Pentachlorophenol	000087-86-5	120	177	265
Pirimicarb	023103-98-2	0.255	0.373	0.541
Pirimiphos-methyl	029232-93-7	21	30.4	42.2
Propoxur	000114-26-1	2.9	4.16	5.63

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
Pyrazon (Chloridazon)	001698-60-8	0.066	0.0936	0.121
Pyridaben	096489-71-3	166	235	304
Pyriproxyfen	095737-68-1	42.3	60	78.1
Selenium	007782-49-2	0.215	0.513	1.59
Simazine	000122-34-9	2.46	3.59	5.14
Teflubenzuron	083121-18-0	45.8	118	167
Tellurium	013494-80-9	0	0.261	1.55
Terbutryn	000886-50-0	77.6	114	170
Terbuthylazine	005915-41-3	0.987	1.4	1.81
Tetrachloroethylene(7)	000127-18-4	3.65	5.4	8.13
Carbon tetrachloride	000056-23-5	698	1,062	1,706
Thallium	007440-28-0	7.32	149	733
Tin	007440-31-5	0.065	0.17	0.567
Titanium	007440-32-6	0	0.0123	0.0726
Tolclofos-methyl	057018-04-9	1.02	1.51	2.24
Triazophos	024017-47-8	139	206	308
Tributyl phosphate	000126-73-8	0.0321	0.65	0.97
Trichlorobenzene	012002-48-1	0	0.0496	0.0642
Trichloroethylene	000079-01-6	0.103	0.152	0.228
Trichlorfon	000052-68-6	3.57	5.11	6.9
Trichloromethane (chloroform)	000067-66-3	2.94	4.35	6.55
Trifluraline (19)	001582-09-8	6.65	25	37.4
Uranium	007440-61-1	0.000242	0.000404	0.000242
Vanadium	007440-62-2	4.07	18.2	70.5
Xylenes	000095-47-6	0.00779	0.0111	0.0143
Silver	007440-22-4	9.75	29.9	101
Zinc	007440-66-6	4.44	178	882

F.3.2 Emissions in marine waters

Table 81 - Environmental prices (damage costs) for emissions to marine waters in the EU, in €$_{2021}$/kg

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
1,2-Dichloropropane	000078-87-5	3.42	5.67	8.55
4-Chloroaniline	000106-47-8	0.00313	0.00444	0.00577
Aclonifen	074070-46-5	0.0521	0.0738	0.0956
Alachlor	015972-60-8	0.0339	0.0481	0.0622
Aldrin	000060-57-1	194	2,345	3,532
Anthracene	000120-12-7	0.701	0.993	1.28
Antimony	007440-36-0	0.033	1.14	5.61
Arsenic (and inorganic compounds thereof)	007440-38-2	0.061	188	958
Atrazine	001912-24-9	0.0637	0.101	0.133
Azinphos-ethyl	002642-71-9	3.13	4.43	5.72
Azinphos-methyl	000086-50-0	1.07	1.52	1.97
Barium	007440-39-3	0.00433	0.161	0.792
Bentazon	025057-89-0	0.00052	0.000756	0.00108
Benz(a)anthracene	000056-55-3	0	5.44	7.04

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
Benzene	000071-43-2	0.433	0.64	0.964
Benzo(a)pyrene	000050-32-8	7.7	11.4	17
Benzyl chloride (alpha-chlorotoluene)	000100-44-7	0.228	0.336	0.506
Beryllium	007440-41-7	0.333	1.9	8.55
Bifenox	042576-02-3	0.0335	0.0475	0.0615
Borium	007440-42-8	0	0.00161	0.00956
Cadmium and cadmium compounds	007440-43-9	0.119	8.96	44.8
Captan	000133-06-2	0.121	0.176	0.228
Carbendazim	010605-21-7	0.0138	0.0195	0.0252
Chlorpropham	000101-21-3	0.0114	0.0164	0.0222
Chlorotoluron	015545-48-9	0.00559	0.00792	0.0102
Chlorpyrifos	002921-88-2	15.8	22.6	30.1
Chlorfenvinphos	000470-90-6	1.69	2.49	3.68
Chromium	007440-47-3	0.0174	0.0455	0.123
Cybutryne	028159-98-0	0	58.1	75.2
Cyclodiene and pesticides:	000309-00-2	64.6	589	888
Cypermethrin	052315-07-8	122	174	230
Deltamethrin	052918-63-5	8.96	12.7	16.5
Di(2-ethylhexyl)phthalate (DEHP)	000117-81-7	0.0129	0.04	0.06
Diazinon	000333-41-5	0.24	0.346	0.48
Dichloromethane	000075-09-2	0.398	0.588	0.886
Dichlorvos	000062-73-7	0.291	0.426	0.618
Dicofol	000115-32-2	14.5	59.3	89.6
Dieldrin(7)	000072-20-8	60.6	89	131
Dimethoate	000060-51-5	0.00933	0.0132	0.0172
Diuron	000330-54-1	0.0681	0.0965	0.125
Endosulfan	000115-29-7	4.11	5.86	7.73
Endrin(7)	000465-73-6	0	11.2	14.5
Esfenvalerate	066230-04-4	33.8	48	62.1
Ethylbenzene	000100-41-4	0.0774	0.114	0.172
Fenamiphos	022224-92-6	0.802	1.14	1.47
Phenanthrene	000085-01-8	0.113	0.16	0.207
Fenitrothion	000122-14-5	0.216	0.307	0.406
Phenoxycarb	072490-01-8	0.0301	0.0427	0.0551
Fenthion	000055-38-9	0.552	0.793	1.08
Fluoranthene	000206-44-0	0.893	1.27	1.65
Heptachloride	000076-44-8	356	527	794
Heptachlor epoxide	001024-57-3	9,205	13,640	20,656
Heptenophos	023560-59-0	0.0687	0.0973	0.126
Hexachlorobenzene	000118-74-1	268	396	598
Hexachlorobutadiene	000087-68-3	0.0634	24.1	36.3
Hexachlorocyclohexane	000608-73-1	10.1	14.8	22.2
Something with xylene	000108-38-3	0.000664	0.000943	0.00122
Something with xylene	000106-42-3	0.000705	0.001	0.0013
Imidacloprid	138261-41-3	0.00325	0.00461	0.00598
Isoproturon	034123-59-6	0.0239	0.0338	0.0437
Cobalt	007440-48-4	0.0294	0.176	0.781
Copper	007440-50-8	0.785	2.28	6.6
Mercury and mercury compounds	007439-97-6	0.4	554	2,819

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
Lambda-cyhalothrine	091465-08-6	140	202	270
Linuron	000330-55-2	0.247	0.35	0.456
Lead and lead compounds	007439-92-1	0.00519	3.8	19.3
Malathion	000121-75-5	0.0377	0.0535	0.0693
MCPA	000094-74-6	0.00142	0.00208	0.00301
Mecoprop-P	016484-77-8	0	0.000547	0.000708
Metazachlor	067129-08-2	0.0075	0.0106	0.0138
Methabenzthiazuron	018691-97-9	0.00486	0.00688	0.00889
Metolachlor	051218-45-2	0.053	0.0752	0.0975
Metsulfuron-methyl	074223-64-6	1.77	2.51	3.25
Mevinphos	007786-34-7	0.203	0.289	0.374
Molybdenum	007439-98-7	0.00143	0.614	3.45
Monolinuron	001746-81-2	0.00541	0.00767	0.00991
Naphthalene	000091-20-3	0.0135	0.296	0.444
Nickel and nickel compounds	007440-02-0	0.219	4.7	22.6
Nonylphenols	084852-15-3	0.379	0.537	0.695
Octamethylcyclotetrasiloxane	000556-67-2	0	3.04	3.94
Omethoate	001113-02-6	0.00203	0.00292	0.00401
para-para-DDT	000050-29-3	238	356	550
Parathion	000056-38-2	0.519	0.737	0.963
Parathion-methyl	000298-00-0	0.0888	0.129	0.182
Pentachlorobenzene	000608-93-5	3.99	5.93	9.03
Pentachlorophenol	000087-86-5	1.04	1.53	2.26
Pirimicarb	023103-98-2	0.00195	0.00278	0.0037
Pirimiphos-methyl	029232-93-7	0.318	0.453	0.597
Propoxur	000114-26-1	0.0435	0.0617	0.0799
Pyrazon (Chloridazon)	001698-60-8	0.00132	0.00188	0.00242
Pyridaben	096489-71-3	18.3	26	33.6
Pyriproxyfen	095737-68-1	1.97	2.79	3.62
Selenium	007782-49-2	0.0728	0.358	1.53
Simazine	000122-34-9	0.022	0.0313	0.0411
Teflubenzuron	083121-18-0	0.335	2.45	3.28
Tellurium	013494-80-9	0	0.261	1.55
Terbutryn	000886-50-0	0.479	0.7	1.01
Terbuthylazine	005915-41-3	0.0243	0.0345	0.0446
Tetrachloroethylene(7)	000127-18-4	1.71	2.53	3.8
Carbon tetrachloride	000056-23-5	397	605	977
Thallium	007440-28-0	0.106	25	140
Tin	007440-31-5	0.0334	0.155	0.641
Titanium	007440-32-6	0	0.0123	0.0726
Tolclofos-methyl	057018-04-9	0.0409	0.0595	0.0851
Triazophos	024017-47-8	0.612	0.897	1.31
Tributyl phosphate	000126-73-8	0.000539	0.00276	0.004
Trichlorobenzene	012002-48-1	0	0.00947	0.0123
Trichloroethylene	000079-01-6	0.0305	0.045	0.0677
Trichlorfon	000052-68-6	0.0629	0.0892	0.116
Trichloromethane (chloroform)	000067-66-3	1.28	1.9	2.85
Trifluraline (19)	001582-09-8	0.411	1.34	1.99
Uranium	007440-61-1	0	0	0

Pollutant name	CAS Registry Number	Lower (€/kg)	Median (€/kg)	Upper (€/kg)
Vanadium	007440-62-2	0.35	1.62	6.18
Xylenes	000095-47-6	0.000712	0.00101	0.00131
Silver	007440-22-4	3.54	15.4	58.2
Zinc	007440-66-6	0.182	7.57	37.3

F.4 Environmental prices for emissions to soil

Table 82 - Environmental prices (damage costs) for emissions to soil in the EU, in €$_{2021}$/kg

Pollutant name	CAS Registry Number	Lower (€/kg)	Central (€/kg)	Upper (€/kg)
(epoxyethyl) benzene	000096-09-3	0.348	0.516	0.776
1,2,3,6,7,8-hexachlorodibenzo-p-dioxine	057653-85-7	0	17.4	22.6
1,2,3,7,8-pentachlorodibenzodioxin	040321-76-4	0	590	764
1,2,3-trichlorobenzene	000087-61-6	0.00826	0.0117	0.0151
1,2,3-trichloropropane	000096-18-4	142	212	324
1,2,4-trichlorobenzene	000120-82-1	0.0482	0.071	0.106
1,2-benzenedicarboxylic acid, di-C6-10-alkyl esters	068515-51-5	0	0	0
1,2-benzenedicarboxylic acid, di-C7-11 branched and linear alkyl esters	068515-42-4	0	0	0
1,2-dibromo-3-chloropropane	000096-12-8	328	485	731
1,2-dibromoethane	000106-93-4	55.5	82.6	127
1,2-dichloropropane	000078-87-5	6.8	11.3	17
1,2-epoxy-3-phenoxypropane	000122-60-1	0.855	1.28	1.93
1,3,5,7,9,11-hexabromocyclododecane	025637-99-4	0	0.626	0.81
1,3,5-trichlorobenzene	000108-70-3	0	0.00397	0.00514
1,3-butadiene	000106-99-0	0.747	1.1	1.95
1,3-dichloro-2-propanol	000096-23-1	3.92	5.55	7.19
1,3-propanesultone	001120-71-4	104	154	233
1,3-propiolactone	000057-57-8	806	1,196	1,814
1,4,5,8-tetraaminoanthraquinone	002475-45-8	1.15	1.7	2.56
1,4-dichlorobut-2-ene	000764-41-0	0	88.3	133
1,5,9-cyclododecatrine	004904-61-4	0	0.000115	0.000149
1-bromopropane	000106-94-5	0	0.000146	0.00019
1-methyl-3-nitro-1-nitrosoguanidine	000070-25-7	56.1	82.9	125
1-methylnaphthalene	000090-12-0	0.000488	0.000692	0.000897
2-(2-aminoethylamino)ethanol	000111-41-1	0	0	0
2-(2-methoxyethoxy)ethanol	000111-77-3	0	0.000436	0.000565
2,2'-(nitrosoimino)bisethanol	001116-54-7	61.1	90.3	136
2,2-bis(bromomethyl)propane-1,3-diol	003296-90-0	34.9	51.8	78.3
2,3,7,8-tetrachloordibenzodioxine	001746-01-6	0	18.5	24
2,3,7,8-tetrachlorodibenzofuran	051207-31-9	0	8,427	10,911
2,3-dibromopropane-1-ol	000096-13-9	0	0.0574	0.0744
2,3-dinitrotoluene	000602-01-7	0.655	0.928	1.2
2,3-epoxypropyl-trimethylammoniumchloride	003033-77-0	0	0.000211	0.000273
2,4,5-trimethylaniline	000137-17-7	0	15.3	23.1
2,4,5-trimethylanilinehydrochloride	021436-97-5	0	2.16	3.26

Pollutant name	CAS Registry Number	Lower (€/kg)	Central (€/kg)	Upper (€/kg)
2,4,6-tri-tert-butylphenol	000732-26-3	0	0.0122	0.0158
2,4-diaminoanisole sulfate	039156-41-7	0	7.6	11.4
2,4-dinitrotoluene	000121-14-2	1.37	2.01	2.95
2,5-dinitrotoluene	000619-15-8	0	0.558	0.722
2,6-dinitrotoluene	000606-20-2	412	609	916
2-butenal	004170-30-3	0.00707	0.01	0.013
2-ethoxyethanol	000110-80-5	0.00446	0.0066	0.00994
2-ethoxyethyl acetate	000111-15-9	0.00868	0.0123	0.0159
2-ethylhexanoic acid	000149-57-5	0	0.00148	0.00191
2-methoxyethanol	000109-86-4	0.933	1.38	2.08
2-ethoxyethyl acetate	000110-49-6	0	0.0483	0.0625
2-methylimidazole	000693-98-1	0	0.0337	0.0505
2-methylnaphthalene	000091-57-6	0.0028	0.00405	0.00568
2-naphthylamine	000091-59-8	3.81	5.64	8.49
2-nitroanisole	000091-23-6	7.23	10.7	16.1
2-nitropropane	000079-46-9	0.0202	0.0299	0.045
2-nitrotoluene	000088-72-2	0.00967	14.3	21.6
3,3'-dichlorobenzidine	000091-94-1	69.6	104	156
3,3'-dichlorobenzidine dihydrochloride	000612-83-9	0	0.974	1.47
3,3'-dimethoxybiphenyl-4,4'-ylenediammonium dichloride	020325-40-0	0	87.3	131
3,3'-dimethylbenzidinedihydrochloride	000612-82-8	0	362	545
3,4-dinitrotoluene	000610-39-9	0	0.32	0.415
3,5-dinitrotoluene	000618-85-9	0	0.0583	0.0755
4,4'-(4-iminocyclohexa-2,5-dienylidene methylene)dianiline hydrochloride	000569-61-9	0	6.85	10.3
4,4'-methyleen-bis(2-chlooraniline) hydrochloride	064049-29-2	0	13.8	20.8
4,4'-bi-o-toluidine	000119-93-7	0	0.375	0.485
4,4'-bis(dimethylamino)benzophenone	000090-94-8	0	23.5	35.4
4,4'-methylene bis(2-chloroaniline)	000101-14-4	60.2	89.1	134
4,4'-methylenedianiline	000101-77-9	0	0	0
4,4'-methylenedi-o-toluidine	000838-88-0	36.3	53.6	80.8
4,4'-oxydianiline	000101-80-4	6.92	10.2	15.4
4,4'-thiodianiline	000139-65-1	23	33.9	51.1
4-aminoazobenzene	000060-09-3	0	0.732	0.947
4-aminobiphenyl	000092-67-1	85	126	189
4-chloroaniline	000106-47-8	0.0682	0.0974	0.13
4-chloro-o-toluidine hydrochloride	003165-93-3	3.28	4.86	7.31
4-methyl-m-phenylenediamine	000095-80-7	109	161	242
4-nonylphenol, branched	084852-15-3	0.00649	0.00921	0.0119
4-tert-butylbenzoic acid	000098-73-7	0	0.0726	0.0941
4-tert-butylphenol	000098-54-4	0.00307	0.00435	0.00563
5-allyl-1.3-benzodioxole	000094-59-7	0.723	1.07	1.61
5-nitroacenaphthene	000602-87-9	2.1	3.1	4.67
6.6'-di-tert-butyl-2.2'-methylenedi-p-cresol	000119-47-1	0	0	0
6-methoxy-m-toluidine	000120-71-8	0.993	1.47	2.21
7-oxa-3-oxiranylbicyclo[4.1.0]heptane	000106-87-6	0	0	0
8-hydroxyquinoline	000148-24-3	0	0.025	0.0323
Acenaphthene	000083-32-9	0.00955	0.0137	0.0183

Pollutant name	CAS Registry Number	Lower (€/kg)	Central (€/kg)	Upper (€/kg)
Acridine	000260-94-6	0.152	0.215	0.279
Acrylamide	000079-06-1	559	827	1,247
Acrylonitrile	000107-13-1	4.08	6.05	9.16
Aldrin	000309-00-2	24.6	226	340
Alpha-endosulfan	000959-98-8	6.76	9.6	12.4
Alpha-hexachlorocyclohexane	000319-84-6	132	195	294
Anthracene	000120-12-7	0.119	0.168	0.218
Anthraquinon	000084-65-1	0.0294	0.0416	0.0537
Arsenic	007440-38-2	19.5	168	884
Arsenic pentoxide	001303-28-2	19.5	168	884
Aziridine	000151-56-4	69.2	102	154
Azobenzene	000103-33-3	0.166	5.03	7.52
Azocyclotin	041083-11-8	40.7	59	83.4
Benomyl	017804-35-2	0.0966	0.137	0.179
Benz[a]acridine	000225-11-6	0	11	14.3
Benz[c]acridine	000225-51-4	0	132	171
Benzene	000071-43-2	1.04	1.54	2.68
Benzidine	000092-87-5	5,630	8,323	12,534
Benzidine dihydrochloride	000531-85-1	0	8.43	12.7
Benzo[a]anthracene	000056-55-3	0	0.219	0.284
Benzo[a]pyrene	000050-32-8	1.57	2.31	3.49
Benzophenone	000119-61-9	0	0.122	0.158
Benzotrichloride	000098-07-7	478	706	1,064
Benzyl butyl phthalate	000085-68-7	0.0166	0.0784	0.114
Benzyl chloride	000100-44-7	0.746	1.1	1.66
Beryllium	007440-41-7	0.0566	2.43	13.6
Beta-endosulfan	033213-65-9	3.2	4.54	5.89
Beta-hexachlorocyclohexane	000319-85-7	30.1	45.1	67.9
Binapacryl	000485-31-4	0.174	0.247	0.32
Bis(2-ethylhexyl)phthalate	000117-81-7	0.0123	0.0397	0.0599
Bis(chloromethyl)ether	000542-88-1	55,279	81,719	143,521
Bis(pentabromophenyl)ether	001163-19-5	10.5	16.3	24.7
Bisphenol A	000080-05-7	0.0107	0.0153	0.0203
Brodifacoum	056073-10-0	0.0000524	0.0000742	0.000096
Butane	000106-97-8	0	0	0
Butanon-oxime	000096-29-7	0	0.228	0.343
Butylparaben	000094-26-8	0	0	0
C.I. Basic Violet 3 [containing 0.1 per cent or more Michler's ketone (EC No 202-027-5)]	000548-62-9	0	20.4	27.6
Cadmium	007440-43-9	9.34	2,224	11,320
Captafol	002425-06-1	9.84	14.4	21.2
Carbendazim	010605-21-7	0.442	0.63	0.831
Carbetamide	016118-49-3	0.0599	0.0849	0.11
Catechol	000120-80-9	0.0883	0.154	0.227
Quinoline	000091-22-5	0.00552	20.3	30.6
Chlordecone	000143-50-0	1,285	1,904	2,882
Chlorodimethyl ether	000107-30-2	5.53	8.22	12.5
Chlorfenvinphos	000470-90-6	201	297	445
Chloromethyl mercury	000115-09-3	0	652	982

Pollutant name	CAS Registry Number	Lower (€/kg)	Central (€/kg)	Upper (€/kg)
Chlorotriethyl lead	001067-14-7	1.45	2.06	2.67
Chloroprene	000126-99-8	3.49	5.16	10.5
Chromium (VI)	018540-29-9	0.363	22,830	34,385
Chrysotile	012001-29-5	0	0.00426	0.0252
Cumatetralyl	005836-29-3	0.00307	0.00435	0.00561
Cumene	000098-82-8	0.000778	0.00115	0.00172
Cyclododecane	000294-62-2	0	0.000014	0.0000181
Cycloheximide	000066-81-9	0	1.53	1.99
Cyhexatin	013121-70-5	0.227	0.335	0.496
DDT, 2,4'-isomer	000789-02-6	0	0.372	0.482
DDT, 4,4'-isomer	000050-29-3	89.1	132	200
Delta-hexachlorocyclohexane	000319-86-8	0.978	1.39	1.8
Dibenzo[a,h]anthracene	000053-70-3	0.421	0.622	0.937
Dibromo-nitrilopropiamide	010222-01-2	0	5.52	7.15
Dibutyl phthalate	000084-74-2	0.063	0.0899	0.119
Dibutyltin di(acetate)	001067-33-0	0	0.513	0.665
Dibutyltin dichloride	000683-18-1	0.0924	0.131	0.17
Dibutyl tindilaurate	000077-58-7	0	0.0000078	0.0000101
Dibutyltinoxide	000818-08-6	0	0.000263	0.00034
Dicofol	000115-32-2	254	1,039	1,565
Dieldrin	000060-57-1	425	5,173	7,789
Di-ethylenetriamine pentaacetic acid	000067-43-6	0	0.00244	0.00316
Diphenacoum	056073-07-5	0	0.00011	0.000142
Diphenylchlororesin	000712-48-1	0	0.853	1.1
Dihexyl phthalate	000084-75-3	0	0.000849	0.0011
Diisobutyl phthalate	000084-69-5	0	0.0105	0.0136
Dimethomorph	110488-70-5	0	0.115	0.15
Dimethylcarbamoyl chloride	000079-44-7	47	69.5	105
Dimethyl sulphate	000077-78-1	0	0.0097	0.0126
Dimethyltin dichloride	000753-73-1	5.64	8	10.4
Disodium-{5-[(4'-((2,6-dihydroxy-3-((2-hydroxy-5-sulfophenyl)azo)phenyl)azo)(1,1'-biphenyl)-4-yl)azo]sa	016071-86-6	0	1,260	1,897
Disodium 3,3'-[[1,1'-biphenyl]-4,4'-diyl bis(azo)]bis(4-aminonaphthalene-1-sulphonate)	000573-58-0	0	0	0
Dinatrium-4-amino-3-[[4'-[(2,4-diaminofenyl)azo][1,1'-bifenyl]-4-yl]azo]-6-(fenylazo)-5-hydroxynaftal	001937-37-7	0	67.2	101
Dinitrotoluene	025321-14-6	0	15.7	23.7
Dinocap	039300-45-3	0.027	0.0394	0.0567
Dinoseb	000088-85-7	4.61	6.74	9.77
Dinoterb	001420-07-1	1.3	1.84	2.37
Dioxane	000123-91-1	1.01	1.52	2.39
Diuron	000330-54-1	1.72	2.46	3.26
E-2-butenal	000123-73-9	0	26.3	39.6
Endosulfan	000115-29-7	0.959	1.38	1.86
Endrin	000072-20-8	118	174	259
Epichlorohydrin	000106-89-8	14.4	21.5	33.3
Ethanal	000075-07-0	0.0427	0.0631	0.0951

Pollutant name	CAS Registry Number	Lower (€/kg)	Central (€/kg)	Upper (€/kg)
Ethenyl ester of neodecanoic acid	051000-52-3	0	0	0
Ethyl-1-(2,4-dichlorophenyl)-5-(trichloromethyl)-1H-1,2,4-triazole-3-carboxylate	103112-35-2	2.48	3.51	4.55
Ethylenediamine	000107-15-3	0.00477	0.00677	0.00878
Ethylene oxide	000075-21-8	10.4	17.8	35.5
Ethylenethiourea	000096-45-7	0.452	14	21
Ethyl-p-nitrophenylthiobenzene-phosphenate	002104-64-5	245	363	546
Phenanthrene	000085-01-8	0.0284	0.0403	0.052
Fenantridine	000229-87-8	0	0.571	0.739
Fenbutatin	013356-08-6	1,064	1,574	2,376
Phenolphthalein	000077-09-8	0.0101	0.0149	0.0225
Fentin hydroxide	000076-87-9	14.5	20.6	27
Phenylarsenic acid	000098-05-5	0	0.00146	0.00189
Phenylhydrazine	000100-63-0	0	0.0196	0.0254
Phenylhydrazine chloride	000059-88-1	0	4.89	7.37
Phenylmercury acetate	000062-38-4	256	378	568
Fluazifop-butyl	069806-50-4	0.234	0.331	0.427
Flucythrinate	070124-77-5	161	228	296
Fluoranthene	000206-44-0	0.129	0.184	0.243
Fluorene	000086-73-7	0.00584	0.00843	0.0117
Flurochloridone	061213-25-0	0.448	0.635	0.823
Formaldehyde	000050-00-0	0.00275	0.0039	0.00505
Formamide	000075-12-7	0	0	0
Furan	000110-00-9	27.1	40.1	86.8
Gamma hexachlorocyclohexane	000058-89-9	77.2	114	170
Glufosinate ammonium	077182-82-2	0.106	0.154	0.224
Glutaaraldehyde	000111-30-8	0	0	0
Glycidol	000556-52-5	170	254	389
Heptachloride	000076-44-8	19.6	29	43.6
Heptachlor epoxide	001024-57-3	16,657	24,631	37,115
Heptachloronorbornene	028680-45-7	0	0.0615	0.0797
Hexachlorobenzene	000118-74-1	180	267	403
Hexachlorobutadiene	000087-68-3	0.021	19.5	29.3
Hexachlorocyclohexane	000608-73-1	79.4	117	175
Hexachlorocyclopentadiene	000077-47-4	55.1	81.4	123
Hexadecafluorheptane	000335-57-9	0	0	0
Hexahydrophthalic acid anhydride	000085-42-7	0	0	0
Hexamethylphosphoramide	000680-31-9	861	1,273	1,917
Hydrazine	000302-01-2	64.3	94.8	141
Hydrazobenzene	000122-66-7	0	12.7	19.1
Imidazole	000288-32-4	0	0	0
Isobutane	000075-28-5	0	0	0
Isobutyl nitrite	000542-56-3	0	0.688	1.04
Isodrin	000465-73-6	0	1.1	1.43
Isoprene	000078-79-5	0.0693	0.102	0.207
Isoquinoline	000119-65-3	0.00963	0.0137	0.0177
Cacodylic acid	000075-60-5	3.08	4.55	6.85
Cobalt	007440-48-4	0.000557	0.0941	0.551
Mercury	007439-97-6	1.69	280	1,425

Pollutant name	CAS Registry Number	Lower (€/kg)	Central (€/kg)	Upper (€/kg)
Linuron	000330-55-2	0.153	0.219	0.295
Lead	007439-92-1	0.969	23	118
Lead styphnate	015245-44-0	0	0.000198	0.000256
Mancozeb	008018-01-7	0.927	1.32	1.74
M-bis(2,3-epoxypropoxy)benzene	000101-90-6	55.9	82.6	124
Methoxy chloride	000072-43-5	0.326	0.464	0.607
Methyl hydrazine	000060-34-4	0.459	8.17	12.2
Mirex	002385-85-5	293	443	704
Musk xylene	000081-15-2	0	14.2	21.3
N-(1,3-dimethylbutyl)-N'-phenyl-1,4-benzenediamine	000793-24-8	0	0	0
N,N,N',N'-tetramethyl-4,4’-methylenedianiline	000101-61-1	0	68.5	103
N,N-dimethylacetamide	000127-19-5	0	0	0
N,N-dimethylformamide	000068-12-2	0.572	0.845	1.27
N,N-dimethylhydrazine	000057-14-7	8.55	12.6	19
Naphthalene	000091-20-3	0.0383	3.11	4.68
Sodium cacodylate	000124-65-2	0.0164	0.0233	0.0302
Sodium pentachlorophenolate	000131-52-2	0.198	0.281	0.364
N-butyltin trichloride	001118-46-3	1.11	1.58	2.04
Nickel	007440-02-0	5.8	45.1	287
Nitrobenzene	000098-95-3	1.87	2.76	4.16
Nitrophene	001836-75-5	6.97	10.3	15.5
Nitrosodipropylamine	000621-64-7	125	185	278
N-methyl-2-pyrrolidon	000872-50-4	0	0.0641	0.0965
N-methylol acrylamide	000924-42-5	0	420	632
N-nitrosodimethylamine	000062-75-9	1,223	1,810	2,731
Nonylphenol	025154-52-3	0.000445	0.000631	0.000817
O-aminoazotoluene	000097-56-3	125	185	278
O-anisidine	000090-04-0	0	0.118	0.153
Octachlorinated naphthalene	002234-13-1	0	0	0
Octamethyltetrasiloxane	000556-67-2	0	0.0334	0.0433
O-toluidine	000095-53-4	0.00494	0.00701	0.00909
P-(1,1-dimethylpropyl)phenol	000080-46-6	0	0.172	0.223
Para-tert-octylphenol	000140-66-9	0.00357	0.00506	0.00654
PCB 101	037680-73-2	0	0.681	0.882
PCB 77	032598-13-3	0	0.386	0.5
Pentachloranisole	001825-21-4	0	46.4	69.8
Pentachlorobenzene	000608-93-5	3.34	4.96	7.56
Pentachloroethane	000076-01-7	0.00233	29.4	44.3
Pentachlorophenol	000087-86-5	14.8	21.9	32.8
Pentasodium diethylene-triaminepentaacetic acid	000140-01-2	0	0.00064	0.000829
P-nonylphenol	000104-40-5	0.000479	0.00068	0.000879
Polychlorinated biphenyls	001336-36-3	0	386	582
Propiconazole	060207-90-1	0.605	0.88	1.25
Propylene oxide	000075-56-9	2.49	3.84	6.37
Pyrene	000129-00-0	0.27	0.383	0.498
Pyrithione zinc	013463-41-7	0	0	0
Roxarsone	000121-19-7	0	0	0

Pollutant name	CAS Registry Number	Lower (€/kg)	Central (€/kg)	Upper (€/kg)
Sulphallate	000095-06-7	1.96	3.07	4.59
Tellurium	013494-80-9	0	0.261	1.55
Terphenyl	026140-60-3	0	0	0
Tetrabromobisphenol A	000079-94-7	0.00366	0.00527	0.00708
Tetrabutyltin	001461-25-2	0.0000177	0.0000251	0.0000326
Tetraethyl lead	000078-00-2	324	478	720
Tetrafluoroethylene	000116-14-3	5.94	8.79	13.3
Tetrahydro-2-furylmethanol	000097-99-4	0	0.0016	0.00208
Tetramethyl lead	000075-74-1	0.00184	0.0026	0.00337
Tetrasodium-3,3'-[[1,1'-biphenyl]-4,4'-diylbis(azo)]bis[5-amino-4-hydroxynaphthalene-2,7-disulfonate]	002602-46-2	0	23,068	34,737
Tetrasul	002227-13-6	0	0.000558	0.000723
Theophylline	000058-55-9	0	0	0
Thioacetamide	000062-55-5	7.67	11.3	17.1
Thiomersal	000054-64-8	0.0571	0.0811	0.105
Toxaphene	008001-35-2	1,365	2,032	3,110
Triadimenol	055219-65-3	0.411	0.598	0.849
Tributyltin	000688-73-3	0.00795	0.0113	0.0146
Tributyltin oxide	000056-35-9	0.00235	0.00347	0.00522
Trichlorobenzene	012002-48-1	0	0.0035	0.00453
Trichloroethylene	000079-01-6	0.084	0.124	0.263
Tridemorph	024602-86-6	0	46.5	60.2
Triphenyltin acetate	000900-95-8	0.253	0.362	0.484
Triphenyltin chloride	000639-58-7	0.146	0.215	0.32
Triflumizole	068694-11-1	1.02	1.44	1.85
Trifluraline	001582-09-8	0.715	2.68	4.02
Tris(2,3-dibromopropyl)phosphate	000126-72-7	2.18	3.24	4.87
Tris(2-chloroethyl)phosphate	000115-96-8	0.00321	0.628	0.945
Triton X-100	009002-93-1	0.0000213	0.0000302	0.0000392
Trixylyl phosphate	025155-23-1	0	0.00169	0.00219
Urethane	000051-79-6	5.43	8.02	12.1
Vinchlozolin	050471-44-8	1.68	2.48	3.71
Vinyl bromide	000593-60-2	2.24	3.31	5.6
Vinyl chloride	000075-01-4	6.49	9.59	14.6
Warfarin	000081-81-2	22.4	33.1	49.8