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Preliminary Stocktake Report for Module 2 of the project on ENvironmental Impact Valuation and its Application in private and public DEcision making (ENIVADE)

This document contains a draft of the preliminary stocktake report for Module 2 of the ENIVADE project, intended as a background document for a Module 2 technical workshop on 12-13 November 2025.

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List of Abbreviations

AoP	Area of Protection (e.g., human health, ecosystems, resources)
CF	Characterization Factor
DALY	Disability-Adjusted Life Year(s)
DCE	Discrete Choice Experiment
EP&L	Environmental Profit and Loss
GHG	Greenhouse Gas(es)
HPM	Hedonic Price Method
IF	Intake Fraction (iF)
IPA	Impact Pathway Approach
IPCC	Intergovernmental Panel on Climate Change
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
PDF	Potentially Disappeared Fraction
NPP	Net primary production
RP	Revealed Preference
SETAC	Society of Environmental Toxicology and Chemistry
SIA	Secondary Inorganic Aerosol(s)
SNA	System of National Accounts
SOA	Secondary Organic Aerosol(s)
SP	Stated Preference
UNEP	United Nations Environmental Program

VSL Value of a Statistical Life

VOLY Value of a Life Year

WHO World Health Organization

WMO World Metereological Organization

WTA Willingness to Accept

WTP Willingness to Pay

Table 0.1. Substance abbreviations

Substance	Substance name
CO_2	Carbon dioxide
CFC-11	Chlorofluorocarbons
PM	Particulate matter
$PM_{2.5}$	Particulate matter with aerodynamic diameter ≤ 2.5 µm
PM_{10}	Particulate matter with aerodynamic diameter ≤ 10 µm
PM_1	Particulate matter with aerodynamic diameter ≤ 1 µm
NO_x	Nitrogen oxides
SO_2	Sulphur dioxide
NH_3	Ammonia
NMVOC	Volatile organic compounds (non-methane)
N	Nitrogen
P	Phosphorus
PO_4	Phosphate
H^+	Hydrogen ion

1. Introduction

1.1. Background and objectives

1. There is a growing consensus that environmental damages, including health impacts, resulting from economic activities need to be integrated in public and private decision making to successfully mobilise economic actors for the green transition to reach the UN Sustainable Development Goals. For many decision-making contexts, this is facilitated by quantifying and valuing these impacts, to the extent possible, in monetary terms. Companies are increasingly reporting on their environmental footprint on a voluntary basis, and there are also indications that such reporting is becoming part of public policy requirements.[1]

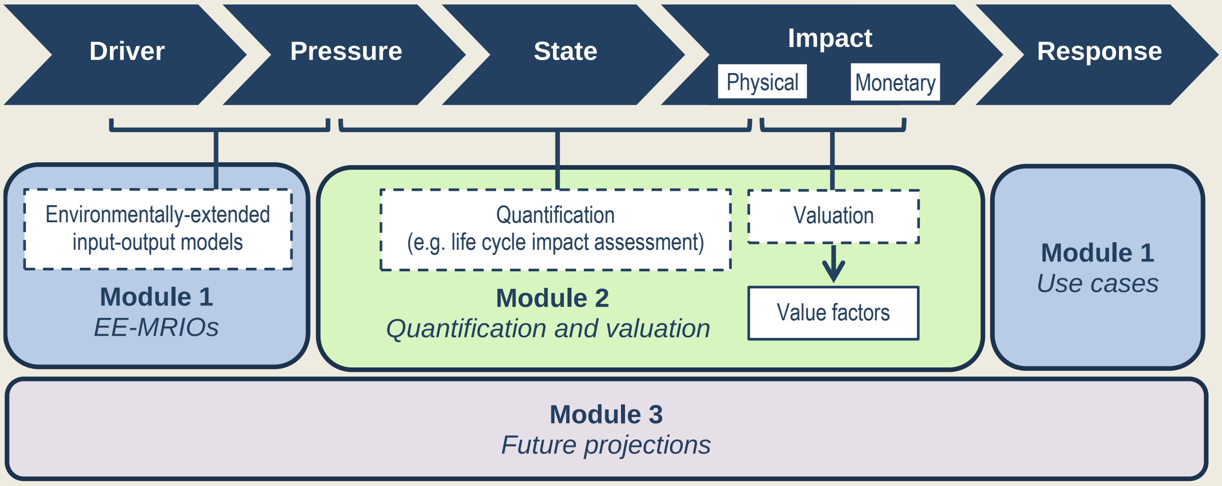
2. Value factors, or metrics that relate environmental pressures from economic activity to their monetary costs to society, are critical in incorporating environmental impacts into public and private decision making. As shown in Section 3.1, a range of value factor estimates exist even within environmental pressure categories. Understanding the sources of heterogeneity in these estimates is important in pursuing standardisation in value factors and their underlying methodologies. Toward this end and for use as a background document for the Module 2 Technical Workshop in November 2025, this report aims to take stock of the methodologies for translating environmental pressures into environmental and health impacts and techniques for valuing these impacts, identify criteria for identifying ready-to-use value factors on the basis of the robustness of the methodologies employed to develop them and identify knowledge gaps that need to be filled in order to advance standardisation in quantification and valuation of environmental pressures. This work is carried out in the context of the OECD Project on Environmental impact valuation and its application in decision making (ENIVADE) (see Box 1.1).

[1] New regulations in California, SB253 and SB261, require companies to disclose their scope 1, 2, and 3 GHG emissions from 2026 and the EU Corporate Sustainability Reporting Directive (CSRD) requires companies to report from 2025.

Box 1.1. The OECD ENIVADE Project

In January 2025, the OECD embarked on a cross-directorate project on Environmental impact valuation and its application in decision making (ENIVADE). The overall objective of the ENIVADE project is to strengthen the quantification and valuation of environmental impacts in private and public decision making. More specifically, the project aims to: (i) provide an assessment of existing methodologies and data with a view to developing recommendations for possible next steps and (ii) provide access to robust methodologies, data, and guidance on their use for environmental impact valuation decision making. The project is undertaken through three modules that aim to provide an assessment of existing approaches and develop indicative guidelines on: (i) the use of input-output models and their environmental extensions (Module 1), (ii) the quantification and valuation of environmental and health impacts (Module 2) and (iii) the projection of future impacts (Module 3).

Figure 1.1. ENIVADE Project modules coverage of the DPSIR impact assessment pathway framework



Note: See Table 1.2 for a definition of terms. Life cycle impact assessment is also used to quantify drivers, pressures, states and impacts (see Section 2.2). Impact pathways are illustrated along the DPSIR (Driver, Pressure, State, Impact, Response) framework, originally used by OECD (1993[1]).
Source: Authors' elaboration on the basis of the DPSIR framework (Verones et al., 2017[2]; Kristensen, 2004[3]; European Environment Agency, 1999[4])

The preliminary stocktake supports Module 2 on the quantification and valuation of environmental and health impacts. It serves as a background document for the ENIVADE Module 2 Technical Workshop on 12-13 November 2025, seeking to establish a basis for expert dialogue on best practice in quantification and valuation of environmental and health impacts. Expert input on best practices will inform the refinement of criteria for identifying fit-for-purpose value factors based on an assessment of the quantification and valuation methodologies on which they are based (Section 3.3). Figure 1.2 depicts in more detail the steps of the DPSIR Framework addressed by Module 2, namely, the quantification and valuation of physical impacts resulting from environmental pressures. This interim version of the preliminary stocktake focuses in particular on a stocktake of existing quantification and valuation methodologies and a survey of existing value factors.

3. Table 1.1 describes known contributions to standardising environmental and health impact accounting. This work brings added value to existing efforts in three main ways. First, it provides a focused comparison of existing value factors across providers, highlighting the magnitude of heterogeneity in estimates of the monetary damages for the same environmental pressures and illustrating a strong need to understand the sources of this variation, and possibly pursuing standardisation in the underlying methodologies used to calculate value factors. Second, based on the existing body of knowledge in this area, this work develops a set of criteria for evaluating the quality of value factors that is distinguished by its applicability to value factors intended for different use cases and its ease of application. Finally, the project provides an updated review of the remaining knowledge gaps in the area of standardizing environmental impact accounting and a roadmap for future work priorities.

Table 1.1. Known contributions to standardising environmental and health impact accounting

Contributors	Project/Publication	Objective	Use case focus	Identifies methodological evaluation criteria [a]	Identifies ready-to-use value factors
Value Balancing Alliance, Capitals Coalition, World Business Council for Sustainable Development	Transparent (2023)	A public-private partnership to develop standardized natural capital accounting and valuation principles as a means of mobilizing the private sector in support of the green transition	Public	No	No
International Foundation for Valuing Impacts, Value Balancing Alliance	The Methodology Guidelines (2024)	A globally applicable and comprehensive methodology for the public good for valuing organizational social and environmental impact that is designed for incorporation into financial analysis and organizational planning and decision-making[2]	Private	No	No
WiFOR	WifOR Impact Valuation (2025)	Describes the methodology underlying WifOR's approach to impact valuation	Private	No	Yes
Value Balancing Alliance	Sprint Report (2024)	Highlights the current state of opportunities and challenges in practice with the goal of raising awareness of Impact Accounting among various stakeholders, accelerating uptake, and informing the different processes in the field of standardisation and integration	Private	No	No
European Commission (coordinator)	ES3P (forthcoming)	Develop solutions to enable data supplies supporting corporate use cases and establish an EE-MRIO reference framework to provide actionable environmental data sets	Private	No	No
Capitals Coalition	Natural Capital Protocol	A decision-making framework that enables organisations to identify, measure and value their direct and indirect impacts and dependencies on natural capital	Private	Yes	Yes

[2] The Transparent project responded to a European Commission Grant through the EU Life program to develop "a standardized natural capital management accounting methodology that would result in the successful development of Environmental Profit and Loss Accounts. The expectation was that the methodology should cover both impacts and dependencies and should be suitable for integration in corporate strategic decision-making processes rather than focused on external reporting covered by other EU and global initiatives."

OECD	ENIVADE (2025-2027)	Strengthen the quantification and valuation of environmental impacts in private and public decision making	Public and private	Yes	TBD

Note: This table refers to efforts that have been undertaken with the explicit purpose of supporting the standardisation of natural capital accounting and impact evaluation. It does not include a large body of existing work by these and other entities to develop value factors. [a] Refers to whether the work sets forth criteria for assessing the robustness of the underlying methodologies used to calculate value factors.

4. In pursuit of these contributions, the detailed objectives of this report are as follows:

a) Provide a preliminary review of impact analysis methodologies, including models estimating midpoint indicators (e.g. changes in pollutant concentrations resulting from emissions of air pollution) and models estimating environmental and health endpoint indicators (e.g. changes in disease incidence resulting from increased concentration of air pollutants). Consider sources of heterogeneity in estimating impacts and identify knowledge gaps with respect to the quantification methodologies reviewed.
b) Provide a preliminary review of valuation concepts and methodologies appropriate for valuing environmental and health impacts, considering sources of heterogeneity in the calculation of monetary values, and identify knowledge gaps with respect to the valuation methodologies reviewed.
c) Provide a preliminary review of existing value factors, identifying broad sources of potential heterogeneity therein, and identify potential criteria for evaluating the suitability of value factors for use in measuring environmental and health impacts for different use cases. Identify areas where more knowledge and/or consensus is needed in order to pursue standardisation in value factor calculation methods.
d) Determine priorities for next steps in better integrating environmental and health impacts into public and private decision making. This could include work to better describe impact pathways that remain poorly understood or fill gaps for valuations that are missing or insufficiently robust

1.2. Analytical scope and framework

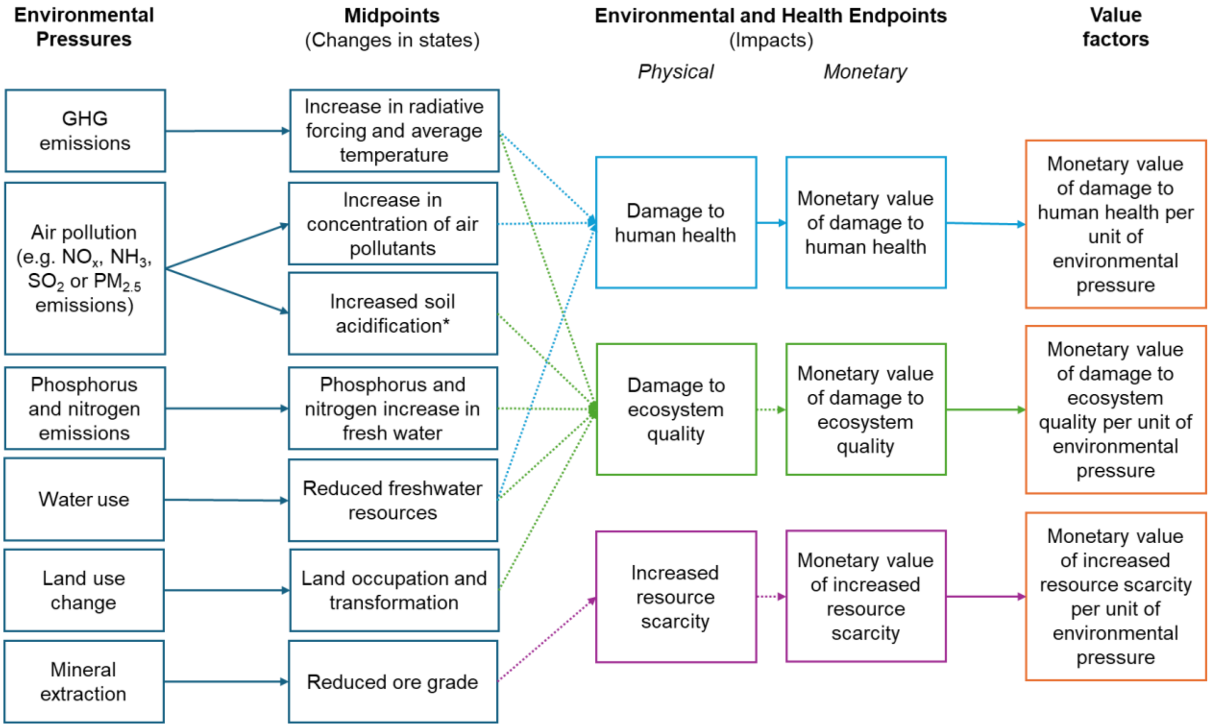
5. Using the DPSIR (Driver, Pressure, State, Impact, Response) as a conceptual framework, Figure 1.1 illustrates how economic activities exert pressures on the environment (e.g. pollution), which cause changes in environmental states and processes. These environmental changes in turn lead to impacts on human health, ecosystems and natural resources that have value to society. Such impacts can induce policy responses to seek to reduce these impacts by addressing their drivers, or by addressing impacts directly through adaptation or restorative action (European Environment Agency, 1999[1]). Noting that terminologies differ across fields and frameworks, Table 1.2 presents a glossary of terms, including equivalent terms that may be found in other impact accounting frameworks, and Figure 1.2 depicts the conceptual steps involved in the quantification and valuation of impacts resulting from environmental pressures.

Table 1.2. Glossary of terms

Term used in this report	Definition	Other terms in use
Driver	Economic activity that generates an environmental pressure (see below)	
Environmental pressure	Source of environmental change (e.g. emissions emitted, natural resources used)	Elementary flows; precursors
State (midpoint)	Biophysical condition of the environment	

Impact (endpoint)	A change in the environment or human health resulting from changes in the biophysical conditions of the environment; can be expressed in physical or monetary terms	Damage
Response	Action taken to address the impact	

Figure 1.2. The ENIVADE Project through the lens of impact assessment pathways



*$PM_{2.5}$ do not contribute to changes in this environmental midpoint.

Note: Impact pathways consist of linked environmental processes, reflecting the causal chain of effects generated by environmental pressures that ultimately result in socioeconomic damages (UNEP, 2016[5]). Midpoints indicate the contribution of an emission or resource use to a change in an environmental state. This biophysical impact is referred to as a midpoint (CE Delft, 2024[6]). Endpoints are defined as the ultimate damages to the environment and human health that are caused by biophysical environmental changes. These include a wide range of impacts to human health, ecosystem services and resource extraction (CE Delft, 2024[6]). As an example, GHG impacts at the midpoint level describe the increase in temperatures relative to pre-industrial levels. At the endpoint level, increased temperatures can result in negative impacts to human health (e.g. heat-related illnesses) and the environment (e.g. reduced ecosystem functioning). The midpoints affected by these environmental pressures depicted are not comprehensive, and the environmental and health endpoints are described in general terms, given their large number. Solid lines indicate relationships that are characterised by relatively greater certainty, dashed lines instead indicate relationships for which there is less certainty, e.g. those that involve additional steps, complex models and/or assumptions.

Source: Author(s) elaboration based on Huijbregts et al. (2016[7]) and CE Delft (2024[6]).

6. As illustrated in Figure 1.2, this work focuses on six environmental pressures or impact categories: greenhouse gas (GHG) emissions, air pollution emissions, water pollution caused by phosphorus and nitrogen emissions, water consumption, land use change and mineral resource extraction. Environmental pressures lead to impacts on environmental and health endpoints through changes in environmental states, or midpoints (see Section 2.2). The monetary value of physical impacts to environmental and health endpoints can be estimated through various valuation techniques (see Section 2.3) Expressing these monetary values as value factors involves translating the endpoint valuations to a corresponding damage value per unit of the relevant environmental pressure. For example, the value of health impacts from air pollution, which at the endpoint level are often reflected as the value per case or per disability-adjusted life year (DALY), could also be expressed in USD per ton of $PM_{2.5}$ emissions.

2. Stocktake of impact quantification and valuation methodologies

7. This chapter provides a brief overview of the methodologies used to quantify the physical environmental and health impacts arising from environmental pressures. The chapter takes stock of methods used to quantify the physical impacts to the environment and human health arising from environmental pressures, as well as the methods used to estimate the monetary value of such impacts. It also seeks to elucidate how different methodological approaches taken to impact quantification and valuation can lead to variation in resulting value factors and assess the degree of consensus surrounding such approaches. Finally, the chapter makes a first effort to identify knowledge gaps. Box 2.1 provides details on the quantification steps considered under Module 2, highlighting relevant definitions that are used throughout this Section.

8. The chapter is organised as follows. Section 0 provides a brief description of input–output models used to estimate environmental pressures from economic activity, providing the data needed to quantify environmental and health impacts. Section 2.2 reviews the frameworks in use to quantify these impacts, including Life Cycle Impact Assessment (LCIA), Impact Pathway Analysis (IPA), and Environmental Priority Strategies (EPS), which link data on emissions and resource extraction to their impacts on the environment and human health, highlighting commonalities and divergencies across impact assessment approaches. Section 2.3 reviews existing methodologies for valuing environmental and health impacts in monetary terms, including a discussion of different value concepts, potential sources of heterogeneity in value factors arising from valuation methodologies, as well as several knowledge gaps in impact valuation.

Box 2.1 Impact quantification: approaches and definitions

Impact pathways consist of linked environmental processes, reflecting the causal chain of effects generated by environmental pressures that ultimately result in socioeconomic damages (UNEP, 2016[5]). Midpoints and endpoints refer to specific stages of the impact pathway processes, and constitute the following:

- **Midpoints** indicate the contribution of an emission or resource use to a change in an environmental state. This biophysical impact is referred to as a midpoint (CE Delft, 2024[6]). Examples of midpoints include temperature increases caused by GHG emissions or increased concentrations of air pollutants.
- **Endpoints** are defined as the ultimate damages to the environment and human health that are caused by biophysical environmental changes. These include a wide range of impacts to human health, ecosystem services and resource extraction (CE Delft, 2024[6]). Examples of endpoints include illness, disease and pre-mature death due to increased heat or air pollution.

There are different approaches that can be employed to quantify each step of an impact pathway. Three well-known and recognized approaches are **Life Cycle Impact Assessment (LCIA)**, **Impact Pathway Analysis (IPA)** and **Environmental Priority Strategy (EPS)[3].**

Life Cycle Impact Assessment (LCIA) is a methodology to quantify the environmental impacts associated with the entire life cycle of a product or economic activity and provides an indication of the impacts associated with production and/or consumption activities. In LCIA, environmental pressures are classified into **impact categories** (environmental domains), as well as common impact units used to render the biophysical impacts of different environmental pressures comparable, also known as **characterisation factors (CFs)**. Common life cycle impact categories include climate change, acidification, $PM_{2.5}$ formation, land use, water use, etc. For example, CO_2 and CH_4 emissions both contribute to climate change, and they can be both expressed as CO_2-equivalent emissions by using their global warming potential (GWP) CFs. Some methodologies adopt CFs at the endpoint level (e.g., expressing the impacts from GHG emissions or air pollutants on human health in terms of disability adjusted life years (DALY)), allowing for a comparison of impacts across environmental pressures.

Impact Pathway Analysis (IPA) is a damage-function approach to environmental quantification and valuation that models the impact chain from emissions emitted at a specific location to the monetized effects of these emissions as realised at various endpoints. Unlike Life Cycle Impact Assessment (LCIA), the IPA approach doesn't estimate characterization factors (CF) to express impacts in substance-equivalent units. Rather, it tracks single substances across causal chain models to estimate their final impacts and monetary value.

Environmental Priority Strategy (EPS) is a method developed by the Swedish Energy Agency FORMAS. First released in 1990 and updated in 2020, the EPS is an impact assessment method assessing the economic damages caused by emissions and the use of energy, material resources and land through a cost-based valuation framework.

Note: Further details on midpoint vs. endpoint indicators and on impact pathways are provided in the Annex Sections 6.1and 6.3.
Source: Authors' elaboration based on (Benini, 2014[8]; Huijbregts et al., 2016[7]; CE Delft, 2024[6])

2.1. Quantifying environmental pressures

9. Multi-regional input-output models (MRIOs) extend conventional input–output analysis to cover multiple countries or regions that are economically interconnected through trade. Like single-region input-output models, MRIOs describe industries that produce outputs while requiring inputs from other sectors, but here the flows are tracked both within and across regions. The core data are organized in an interregional transactions table, where rows show the distribution of each region's sectoral outputs to industries and final demand in all regions, and columns show the mix of domestic and imported inputs needed for production. Final demand includes household consumption, government spending, investment, and exports to other regions. MRIO models thus reveal how much output from each industry and region is required - directly and indirectly - to satisfy final demand anywhere in the system, capturing the entire global supply chain.

10. In their environmental extensions (EE), MRIOs assign coefficients for resource use or emissions (e.g., energy consumption, greenhouse gas emissions, water withdrawals) to each sector and region. These models are therefore able to link consumer and industrial purchases to the primary resources needed to produce them, including associated emissions, providing an estimate of the environmental pressures (e.g., GHG emissions, litres of water, hectares of land) caused by production or consumption. In the context of the DPSIR framework, these pressures are then linked to their environmental and health impacts via impact pathways, which are described in the subsequent Section.[4]

2.2. Quantifying environmental and health impacts

11. This section reviews three well-known approaches for determining the environmental and health effects of emissions and resource use from environmental pressures in physical terms, namely life cycle impact assessments (LCIA), impact pathway analysis (IPA) and environmental priority strategies (EPS). It provides a general description of each approach, considers potential sources of heterogeneity in estimating impacts among reported methodologies and identifies knowledge gaps in the reviewed approaches.

2.2.1. *Life Cycle Impact Assessment*

12. A number of LCIA methods exist that vary across multiple dimensions. One main distinction in LCIA methods pertains to whether the method estimates impacts at the midpoint level or at the endpoint (damage) level along the impact pathway. Another distinction can be made according to the range of environmental pressures covered, e.g. methods focusing solely on GHG emissions versus those addressing multiple impact categories. LCIA methods can differ in their geographical scope. Finally, another important source of variation in impact estimates can arise from differences in the underlying life cycle inventory (LCI) data used in LCIAs.[5] Even when the same LCIA methods are applied, differences in

[3] It should be noted that both LCIA and IPA impact assessments align with damage-cost approaches for valuing impacts, whereas they do not relate directly to other valuation approaches such as mitigation or abatement costs.

[4] In the context of the ENIVADE Project, EE-MRIOs are addressed in detail in Module 1, and so an in-depth review of their data sources and methodologies falls outside the scope of the current stocktake.

[5] One example is the ecoinvent life cycle inventory LCI database, which implements several internationally recognized LCIA methods to provide indicators and results.

input data can lead to variation in estimated environmental and health impacts.[6] Box 2.2 provides an overview of relevant efforts toward an international harmonization of LCIA approaches and methodologies.

Box 2.2. The UNEP/SETAC Global LCIA Guidance and GLAM Process

The UNEP (2016[5]) **Global Guidance for Life Cycle Impact Assessment Indicators** represents the achievement of a series of complementary initiatives for LCIA consensus-building that have taken place since the early 1990s. Since that time, the UNEP (United Nations Environment Programme)/SETAC (Society of Environmental Toxicology and Chemistry) Life Cycle Initiative has played a central role in advancing the convergence of LCIA methods. The Initiative is a multi-stakeholder partnership launched in 2002 that includes governments, businesses, scientific and civil society organizations. The main goal of the partnership is to provide global guidance and build consensus on environmental LCIA indicators. Through these coordinated efforts, the Initiative has facilitated the transition from diverse practices toward internationally aligned methodologies. Recognizing the variety of available LCIA approaches, the Initiative aims to identify good practice, address challenges and resolve key questions.

The scope of the guidance includes: describing the impact pathway and reviewing potential indicators, selecting the best-suited indicator or set of indicators based on well-defined criteria, developing methods to quantify indicators using a sound scientific basis and providing characterization factors with corresponding uncertainty and variability ranges. The impact pathways described include climate change, fine particulate matter health impacts, water use and land use. Consensus is still needed for additional impact indicators, including acidification, eutrophication, human toxicity, and mineral resource depletion.

Building on these foundations, the Initiative the Life Cycle Initiative started the **Global Guidance for Life Cycle Impact Assessment Indicators and Methods (GLAM)** in 2013 to develop a full LCIA methodology, co-lead by the University of Michigan, Norwegian University of Science and Technology (NTNU) and Denmark's Technical University (DTU). Phase 3 of GLAM Project (2020–2024) has generated a scoping paper on this activity.

Source: Life Cycle Initiative (2023[9]).

13. This section provides an initial stocktake of LCIA approaches, identifying their key differences and similarities in order to highlight potential sources of heterogeneity that may contribute to variations in derived value factors. Table 2.1 reports several of the most widely-recognised and utilised LCIA methodologies. Each method is described according to whether it estimates impacts at the midpoint and/or endpoint level, the scope of midpoint and endpoint categories included, the geographical resolution of impacts estimated, whether it is regionalized, as well as whether it is are documented and applied in the calculation of the value factors reported in Section 3.1. More detailed descriptions of LCIA methodologies are provided in Table 6.5 in Section 6.2.

Table 2.1 Existing LCIA methodologies and potential sources of heterogeneity

Method	Developer	Last version	Type of method	Geographic Scope	Used in value factors reports

[6] For example, most LCI databases do not handle spatially differentiated data well, being mostly restricted to country scales, thus undermining the broader applicability of spatially differentiated LCIA methods (Verones et al., 2020[12]).

"Carbon Footprint" – IPCC	Intergovernmental Panel on Climate Change (UN)	2021, Regular reports	Midpoint method	Global	Yes
CML Method	University of Leiden	2016	Midpoint method	Global	No
IMPACT World+	CIRAIG, University of Michigan, Quantis International, Technical University of Denmark (DTU), and école Polytechnique de Lausanne (EPFL).	2019	Midpoint and Endpoint method	Global	No
LC-Impact	Norwegian University of Science and Technology (NTNU), PRé Sustainability B.V., Radboud University Nijmegen ETH Zurich, (DTU, Denmark	2020	Endpoint method	Global	Yes
Product Environmental Footprint (PEF)	European Commission, JRC	2022	Midpoint method	EU	No
ReCiPe	National Institute for Public Health and the Environment, Radboud University Nijmegen, Leiden University, and	2016	Midpoint and Endpoint method	Global	Yes
TRACI	U.S. Environmental Protection Agency (US EPA)	2012	Midpoint method (Chemicalfocus)	Global	No
USEtox	UNEP	2017	Midpoint and Endpoint method	Global	Yes

Source: Authors' compliation based on (IPCC, 2021[10]; Bulle et al., 2019[11]; Verones et al., 2020[12]; JRC, 2019[13]; Huijbregts et al., 2016[7]; Fantke, 2023[14]; Bare, 2012[15]).

14. Across all methods, the main midpoint impact categories typically include climate, ozone depletion, acidification, eutrophication, ecotoxicity, human toxicity, photochemical oxidant formation, particulate matter formation, ionizing radiation, land use, water use and resource depletion (abiotic or fossil). At the endpoint level, impacts are assessed in three key areas: human health, quantified using Disability-Adjusted Life Years (DALY); ecosystem quality, expressed as Potentially Disappeared Fraction of species (PDF × unit × year); and resource scarcity, monetized through USD or $ per kg dissipated.

15. Each LCIA method has distinctive features. The IPCC provides a single-issue midpoint method that focuses exclusively on climate change using global warming potential (GWP) and global temperature change (GTP). The CML method from Leiden University covers a broad range of midpoint categories but does not extend to endpoint indicators. IMPACT World+ and ReCiPe 2016 are midpoint-endpoint frameworks that estimate and link detailed midpoint results to outcomes relating to human health, ecosystem quality and resource scarcity. LC-Impact is similarly comprehensive, providing categories only at the endpoint level and integrating spatial differentiation. It covers all three endpoint areas using DALY, PDF and $ as indicators. In contrast, the PEF, developed by the European Commission, standardizes midpoint indicators for European applications, but does not include endpoint modelling. TRACI, used by the US EPA, focuses on chemical-related midpoints, while USEtox provides specialized midpoint and endpoint characterization for toxicity impacts only, relating to ecosystems and human health.

16. For midpoint impacts, there is substantial overlap among methods. In terms of indicators, there appears to be convergence on some of the characterization factors and relative reference substance to

estimate impacts for several impact categories, including for climate change, fine particulate matter (PM), acidification, eutrophication, ecotoxicity and human toxicity. All methods considered include Global Warming Potential (GWP100)[7] by the IPCC (2014[16]; 2021[10]) as the indicator for shorter-term impacts on climate change. Notably, UNEP/SETAC life cycle initiative guidelines confirmed the indicator as an "high quality and robust reference" (UNEP, 2016[5]). Ozone Depletion Potential (ODP), and most methods also quantify ecotoxicity and human toxicity (both related to cancer and not-cancer effects) potentials. In this regard, the majority appears to be using Comparative Toxic Unit (CTU) indicators. The main reference for ecotoxicity and human toxicity metrics is the USEtox method developed by the UNEP, which is applied in PEF, IMPACT World+, and LC-IMPACT. Water consumption impacts on both freshwater ecosystems and human health are estimated using the scarcity indicator AWARE[8]. Acidification is typically expressed in SO_2 or H+ equivalents and eutrophication in P or N equivalents.

17. Endpoint coverage is limited to just a few comprehensive methods: ReCiPe, IMPACT World+ and LC-IMPACT. All of these methods consider the three areas of damage identified by the UNEP/SETAC Life Cycle Initiative guidelines: human health, ecosystem quality and natural resources. DALY is used as the damage metric for impacts on human health, $ for impacts on natural resources, and PDF for impacts on ecosystem quality. However, ReCiPe uses "species × year", i.e. time-integrated local species loss. The use of DALY as an endpoint metric for human health and PDF as an endpoint metric for ecosystem quality is recommended by the UNEP/SETAC guidelines (UNEP, 2016[5]), while acknowledging divergences in modelling approaches for damage to natural resources. Despite some convergence in endpoint damage metrics, comparability is challenged where the models and assumptions differ.

18. In terms of coverage of endpoint categories, LC-IMPACT and ReCiPe 2016 use the same categories for impacts on human health and ecosystem quality. LC-IMPACT and ImpactWorld+ also align in impact categories, with the exception of four categories covered in LC-IMPACT that are either not covered or covered with an "interim" method in IMPACT World+.[9] On the other hand, IMPACT World+ addresses four endpoint categories that are not covered in LC-IMPACT.[10] In LC-IMPACT, all impact categories provide characterization factors with spatial detail, with the exception of climate change, ionizing radiation, and stratospheric ozone depletion.

19. With respect to value factor calculation, ReCiPe 2016 and PEF are explicitly cited in CE Delft (2024[6]). USEtox is employed in value factor calculations by the Value Balancing Alliance, GIST Impact, the International Foundation for Valuing Impacts and WifOR. IPCC (2021[10]) midpoint indicators are widely adopted in numerous valuation exercises, such as those by WifOR and the International Foundation for Valuing Impacts.

[7] The Global Temperature Potential (GTP100) is an instantaneous, not time-integrated indicator as currectly used in LCIA, that has been recommended by the IPCC () as an appropriate proxy to replace the GWP100 for longer time-horizons (i.e. infinite horizon) because of high uncertainty.

[8] The AWARE indicator is a water use midpoint indicator representing the relative Available WAter Remaining. It is developed by WULCA, a group not anymore active, founded in August 2007 under the auspices of the UNEP/Society for Environmental Toxicology and Chemistry (SETAC) Life Cycle Initiative, WULCA Water LCA website, Water Use in Life Cycle Assessment (LCA) - WULCA, (accessed 18 October 2025).

[9] These categories include photochemical ozone formation on terrestrial ecosystems, marine and terrestrial ecotoxicity and mineral resources extraction.

[10] These categories include marine and freshwater acidification, thermally polluted water and the impacts of ionizing radiation on ecosystem quality.

2.2.2. Environmental Priority Strategies

20. The EPS method (Environmental Priority Strategies) is a variation of LCIA approach from the EPS system at Chalmers University and Swedish Life Cycle Center. It starts by modeling endpoint level indicators (e.g., human health in DALY/YOLL, ecosystem service capacities, biodiversity, abiotic resources, access to water) and then converts those physical outcomes into monetary damage costs using market-based or prevention/restoration value estimates. The total damage cost for a flow is computed as the sum over pathways of (characterization factor × monetary value of the state indicator), with a long-term perspective and 0% discounting with uncertainties and value choices made explicit. The result is a single monetized score intended to support design and decision-making by internalizing environmental externalities. Differently from other valuation methodologies which are based on welfare values (e.g. stated preference method), the EPS impact monetization is based on costs, where values are derived from principles and assumptions rather than from preferences and behaviour. While the advantage of such an approach is its ease of use, the cost-based valuations it generates should be considered a proxy for estimating the actual value of environmental and health damages. With regards to valuation of environmental impacts, the EPS method is typically used by WiFOR (2023[17]) to construct value factors for land-use (see also Section 3.1.5).

2.2.3. Impact Pathway Analysis

21. Impact Pathway Analysis (IPA) is a damage-function approach to environmental quantification and valuation that models the process through which location-specific emissions result in monetized effects to environmental and health endpoints. In practice, IPA traces a substance from release at its source through its environmental fate and transport (air, water, soil), to changes in ambient concentrations and exposures (midpoints), to physical impacts on human health, ecosystems and materials (endpoints) and finally to a monetary valuation of those damages (CE Delft, 2024[6]; DEFRA, 2023[18]).

22. Unlike LCIA, the IPA approach does not estimate characterization factors (CF) to express impacts in substance-equivalent units. Instead, it follows single substances across causal chain models to estimate the final impact and its monetary value. Both the objective and final output of IPA are thus different from those of LCIA insofar as the main objective of LCIA is the quantification and monetization of environmental damages, the main objective of IPA is the translation of environmental pressures into a select set of impact scores, i.e. characterization factors.

23. Among the models developed in the context of the European Framework Projects, IPA was developed by NEEDS models, CAFE-CBA, Gains and EEA. The United Kingdom also publishes IPA-based damage costs for air pollutants by sector and pollutant to inform policy appraisal. European IPA applications differ by outcomes and assumptions, but typically follow the same approach, notably:

1. Emission data/forecasts – inventories and policy scenarios.
2. Dispersion and chemistry-transport modelling – translating emissions into spatially and temporally resolved concentrations using meteorology and atmospheric chemistry.
3. Concentration–response (dose–response) functions (CRFs) – converting concentration changes into physical impacts at endpoints (e.g. mortality and morbidity, crop yields, ecosystem indicators, materials corrosion).
4. Monetary valuation – converting physical impacts to damages using values such as VSL/VSLY, cost-of-illness, willingness-to-pay for ecosystem services and repair/replacement costs for materials.

24. IPA is often applied to emissions of air pollutants. In its impact valuation guidance, DEFRA (2023[18]) uses IPA for assessing the impacts of air pollution to human health from PM, NO_x, SO_2, NH_3 and volatile organic compounds (VOCs) by means of dispersion modelling and concentration response

functions. EEA (2021[19]) also applies IPA to estimate air pollution impacts considering effects from PM, NO_x, SO_2, NH_3, non-methane volatile organic compounds (NMVOCs), heavy metals, organic pollutants and GHGs. The EEA models impact pathways of pollutants with dispersion models, intake fractions and exposure response functions. In recent years, socio-economic assessments for chemicals have also used the IPA (EEA, 2021[19]). It should also be noted that IPA can be combined with LCIA. For example, IPA-derived damages can be expressed as midpoint-equivalent prices (or monetary characterization factors) to aggregate and compare heterogeneous impacts across categories, an approach used in CE Delft (2024[6]).

2.2.4. Economic Valuation of Air Pollution (EVA)

25. Economic Valuation of Air pollution (EVA) (Brandt et al., 2013[20]) is an integrated model system to assess the health-related economic externalities of air pollution. EVA is based on the impact pathway chain consisting of emissions, chemical dispersion and transformation of air pollutants, population exposure, health impacts and associated external costs. Originally developed to value site-specific health costs related to air pollution, such as from specific power plants, the system was extended to assess health cost externalities for Europe and Denmark related to entire emission sectors (Brandt et al., 2013[20]).

26. The system is coupled with two chemistry transport models: DEHM (Danish Eulerian Hemispheric Model) and UBM (Urban Background Model) for regional-scale and local-scale health impact assessments, respectively. To estimate the effect of a specific emission source or emission sector, the system uses a newly developed “tagging” method which is capable of calculating the contribution from a specific emission source or sector to the overall air pollution levels, taking into account the non-linear atmospheric chemistry[11].

27. EVA system includes gridded population data[12] which are combined with results from the chemistry transport model to estimate human exposure and response. Impacts from $PM_{2.5}$ are estimated both on chronic mortality and morbidity. EVA valuation of air pollution mortality impacts is based on OECD guidelines for environmental cost-benefit analysis and apply VSL approach to valuation for acute mortality and a VOLY approach for chronic mortality. For morbidity effects EVA adopts a cost-of-illness approach, with the exception of cough and lower respiratory symptoms which are based on WTP approaches.

2.2.5. Impact pathway models

28. Variation in quantification estimates can arise from differences in the impact mechanisms that are assumed when calculating midpoint and endpoint-level impacts. This subsection reviews the methodologies used to model the cause-effect chains that describe how emissions or resource use for each of the impact categories considered in Module 2 (see Figure 1.2) lead to final impacts in three endpoint areas: human health, ecosystem quality and resource depletion. Additional information on the impact categories considered and their impact pathways is provided in Annex Section 6.2. In this impact pathway review, ReCiPe 2016 (Huijbregts et al., 2016[7]) is taken as methodological anchoring.

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[11] The reliability of results from the chemistry transport models can be challenged if the differences in concentrations due to the specific emissions of interest are relatively small compared to the model numerical noise, tagging method increases reliability of results by accounting for non-linear processes.

[12] For Denmark, population gridded data are created using data on addresses, gender and age from the Denmark central registry, while for Europe population uses EMEP (https://www.emep.int/), a co-operative programme for monitoring and evaluation of the long-range transmission of air pollutants in Europe.

Climate change

29. The standard midpoint characterization factor for climate change, used in most of the LCIA methods reviewed in this document, is the Global Warming Potential (GWP). GWP expresses the amount of additional radiative forcing integrated over time (20, 100, or 500 years in IPCC (2021[10])) caused by an emission of 1 kg of GHG relative to the additional radiative forcing integrated over that same time horizon caused by the release of 1 kg of CO_2. At the endpoint level, human health impacts can be quantified using epidemiological models that estimate disability-adjusted life years (DALYs) associated with climate-related diseases (De Schryver et al., 2009[21]). Effects on terrestrial ecosystems are assessed through global projections of species loss (Urban, 2015[22]), while damages to freshwater ecosystems rely on global species–discharge models (Hanafiah et al., 2011[23]).

Fine particulate matter formation

30. Changes in ambient $PM_{2.5}$ concentrations following the release of emissions are usually estimated using source–receptor relationships derived from the global chemical transport model TM5, applied through the TM5-FASST (Fast Scenario Screening Tool for Global Air Quality and Instantaneous Radiative Forcing).[13] The spatial characteristics of emissions releases are important in determining where pollutants accumulate, and demographic factors (e.g. population density and age distribution) influence the extent to which pollutants impact health outcomes. Human intake of $PM_{2.5}$ is quantified using intake fractions (iF, kg intake per kg emission), which link emissions to population exposure. The midpoint characterisation factor (CF) calculated in ReCiPe 2016 and other methods is the particulate matter formation potential (PMFP), expressed in primary $PM_{2.5}$-equivalents. The PMFP directly depends on $PM_{2.5}$ intake fractions, calculated at the regional and global level. As for endpoint indicators, the relationship between exposure health effects is established through concentration–response functions (WHO, 2013[24]), which estimate the increase in mortality risk associated with incremental increases in $PM_{2.5}$ concentration.

Terrestrial acidification

31. Midpoint indicators of soil acidification are based on fate factors measuring the persistence of an acidifying substance, calculated using atmospheric deposition models and geochemical soil acidification models (Roy, Deschênes and Margni, 2012[12]; Roy et al., 2012[12]). At the endpoint level, the damage to ecosystems caused by acidifying substances can be calculated using dose-response functions of the potential occurrence of plant species derived from logistic regression functions (Azevedo et al., 2013[25]).

Freshwater eutrophication

32. Midpoint indicators for impacts of freshwater eutrophication are based on fate factors (FFs) for phosphorus and nitrogen emissions to fresh water. The FF represents the net residence time in the freshwater body (in years). Impacts at the endpoint level are derived from effect factors which model the probability of PDF of species for total concentration of phosphorous (Azevedo et al., 2013[26]).

Water use

33. For determining endpoint impacts from water consumption, the CF at the midpoint level is m^3 of water consumed per m^3 of water extracted, i.e. the water consumption potential (WCP). This measure reflects how much of withdrawn water is actually lost from a watershed through consumption rather than

[13] TM5 simulates atmospheric dispersion, chemistry, and transport, while TM5-FASST uses sensitivity matrices to estimate concentration changes from precursor emissions. TM5 is a global chemical transport model hosted by the European Commission Joint Research Centre (JRC).

returned to the environment.[14] At the endpoint level, ReCiPe 2016 estimation impacts on human health uses the Water Stress Index (WSI) developed by Pfister, Koehler and Hellweg (2009[27])and regionalized fate factors to link water consumption to freshwater availability. Impacts are quantified by coupling water availability with food production, and linking malnutrition vulnerability, measured through HDI, to disability-adjusted life years (DALYs). Terrestrial ecosystem impacts are modeled by estimating the fraction of NPP limited by water availability, which is then translated into species loss (Pfister, Koehler and Hellweg, 2009[27]). For freshwater ecosystems, global species-discharge models are applied, relating reductions in river discharge to the PDF of freshwater species (Hanafiah et al., 2011[23]).

Land use

34. Impacts of land use on ecosystem quality (e.g. biodiversity) are conventionally assessed using a (semi-) natural reference habitat by comparing species richness and community composition under a certain type of land use with those in a natural counterfactual or reference state. Several reference states have been proposed for land use impact assessments. ReCiPe 2016, for example, utilises the concept of potential natural vegetation (PNV), which describes the expected state of mature vegetation that would develop with the cessation of human activities. The species richness of the PNV is approximated using monitoring data from current, (semi-) natural habitats and is considered a valid ecological reference if it is located in the same ecoregion (De Baan, Alkemade and Koellner, 2013[28]) or biome[15] (Elshout et al., 2014[29]). Moreover, rather than selecting a reference habitat based on a specific ecoregion or biome, simplified schemes distinguish only between forest and open vegetation as global types of natural reference vegetation (Huijbregts et al., 2016[7]). Biodiversity impacts also vary by taxonomic groups which react differently to land use due to the variety of requirements for food, shelter and breeding or nesting (Huijbregts et al., 2016[7]). This poses a particular challenge considering that data availability is biased on a limited set of well-studied species groups and that empirical studies from different world regions tend to have little predictive power in other settings (De Baan, Alkemade and Koellner, 2013[28]).

Mineral resource scarcity

35. In assessing the impacts of mineral resource scarcity at the midpoint level, the modelling of Surplus Ore Potential (SOP) follows two steps (Vieira et al., 2017[30]). First, cumulative grade–tonnage relationships are established, describing how ore grades decline as cumulative extraction increases. Second, these relationships are used to estimate the increase in ore production required for future extraction, which translates into additional costs. This forms the basis of the Surplus Cost Potential (SCP), the endpoint indicator that reflects the extra economic burden imposed on future resource users.

2.2.6. LCIA integration with EE-MRIOs

36. LCIA primary function is to assess the final impacts of products' life cycle. Input data usually come from the so-called Life Cycle Inventories (LCI), that is, the inventory of all resources used and pollutants released for a product or service. Recent developments, however, have expanded LCIA to include spend-based or economic data, rather than being strictly product-based.

[14] The greater the water use efficiency, the greater the proportion of water withdrawn that is consumed. In contrast, if efficiency is lower, more water needs to be withdrawn to achieve the same result, and much of this water will not be consumed but returned to the environment. Therefore, industrialised nations tend to have high efficiency levels thanks to good irrigation infrastructure.

[15] A biome is a broad global ecosystem type (like desert or rainforest), while an ecoregion is a smaller area within a biome with its own local climate, soils, and species.

37. Insofar as EE-MRIOs provide comprehensive accounts of production and consumption patterns across global supply chains, combining these two approaches makes it possible to link economic activities with spatially explicit environmental consequences (Verones et al., 2017[31]). While MRIOs provide economy-wide estimates of the environmental pressures associated with economic activities in sectors across the economy, they lack the spatial specificity in estimating environmental pressures that characterizes the inputs needed, and outputs produced, by LCIA. Several efforts have sought to connect MRIO pressure accounts with LCIA characterization factors (CFs). The goal of such efforts has been to hybridize the two techniques in order to mitigate their respective shortcomings and to connect drivers to impacts.

38. To study the impact on biodiversity of consumption and production, for example, Verones et al., (2017[31]) combine Eora MRIO global supply chain database with the LC-IMPACT LCIA model. The authors match the spatial granularity of the CFs used in the LCIA model with the national averages provided by the MRIO models in order to estimate the total impacts of consumption via LCIA. Since the MRIO pressure accounts are generally not well spatially differentiated (often reflecting national averages), the authors calculated an average CF for each pressure at the country level by calculating emissions-weighted (or resource-weighted, for water and land use) average CFs for each of the environmental pressures in each country.

39. Beylot, Corrado and Sala (2020[32]) quantify the impact of final consumption in EU Member States across LCIA PEF method impact categories, including climate change, acidification, terrestrial eutrophication, marine eutrophication, freshwater eutrophication, particulate matter formation, photochemical ozone formation, human toxicity, ecotoxicity, land use, water use and materials use. The authors combine EE-MRIOs (using EXIOBASE 3) with the European Environmental Footprint (EF2017) LCIA method and assign a CF to each of the flows addressed in the EXIOBASE environmental extensions. When matching EXIOBASE extensions to a flow in EF2017 is not possible, a proxy CF is used. Each CF is estimated either as a weighted average of the emissions or resource use at the EU-27 or the global level, when feasible, or as an arithmetic average of the CFs available in EF2017.

40. Davin et al. (2024[33]) combine MRIO EXIOBASE with LC-IMPACT data to assess the impact of agricultural land use on biodiversity and measure the sensitivity of CFs to different aggregation methods. Disaggregating the number of LC-IMPACT CFs – which are aggregated by ecoregion land areas - Davin et al. (2024[33]) created national CFs following a different aggregation approach based on elementary flows. Results show that impact estimates on biodiversity differ greatly between the two aggregation methods, suggesting that aggregation of CFs leads to poor proxies of country-level ecosystem damages. Authors push for the need to regionalize MRIO models beyond the national scale to capture the heterogeneity of biodiversity impacts.

41. Collectively, these studies illustrate both the methodological advances as well as the challenges inherent in linking LCIA with EE-MRIO models. Different aggregation strategies have been proposed to address the mismatch between the spatially explicit resolution of LCIA CFs and the nationally aggregated nature of MRIO pressure accounts. Approaches include the use of emission- or resource-weighted averages at the country level (Verones et al., 2017[31]), aggregation based on elementary flows (Davin and others, 2024[33]) and weighted or arithmetic averages of characterization factors for grouped flows (Beylot, Corrado and Sala, 2020[32]). While these methods enable the quantification of environmental impacts, they also introduce uncertainties due to the loss of spatial detail and reliance on proxy variables. Nevertheless, the extension of MRIO–LCIA integration from biodiversity-focused assessments to a broad range of 14 endpoint categories, as in Beylot, Corrado and Sala (2020[32]), highlights the potential of hybrid models to capture a more comprehensive set of environmental pressures and impacts across global supply chains.

2.2.7. Sources of heterogeneity in quantification methodologies

42. Differences in both input data and impact pathway models represent a significant source of heterogeneity in impact quantification across all the methods reviewed.

Uncertainty from data sources

43. Many impact categories, such as climate change, are based on data from existing scientific literature. Depending on how results are reported in the original studies, the statistical uncertainty of the literature-derived data can only be quantified and reported to a limited extent (Verones et al., 2020[12]). That can result in greater heterogeneity in reported impacts, even when applying the same LCIA method. Moreover, because input data for water and land impacts vary strongly over space and time (e.g., across freshwater types and species groups), resorting to proxies or extrapolation increases uncertainty and reduces the comparability of results.

Impact pathway modeling

44. For some environmental mechanisms, scientific consensus is more established, enabling comparability across impact results and value factors. For others, however, models' assumptions can vary more widely due to limited knowledge and the difficulty of standardizing methodologies.

45. This preliminary stocktake indicates that broad consensus exists on impact pathways for climate change and fine particulate matter formation (PM) from emissions to exposure, where the IPCC and WHO provide methodological benchmarks (IPCC, 2021[10]; WHO, 2013[24]). For relationships between emissions and exposure (i.e. midpoint impacts), these two categories show limited challenges in developing standardised impact pathways that be utilised across contexts and datasets. As highlighted in the review of quantification methods, the IPCC GWP indicator, measuring the impact on climate change from GHG emissions, is widely adopted, as well as dispersion models and intake fractions used to quantify the dispersion and intake of air pollutants, with similar models applied in IPA frameworks.

46. When moving from exposure to impacts at the endpoint level, in contrast, uncertainty increases, often due to limited knowledge of the exact impact mechanisms leading to endpoint impacts (i.e. a limited understanding of how environmental changes (e.g. increasing temperatures) translate into damages (e.g. changes in species richness) (Verones et al., 2020[12]). Moreover, aspects related to exposure (e.g. population composition and susceptibility to different substances) also contribute to increase uncertainty in this step of impact quantification (Verones et al., 2020[12]).

47. As noted in Section 2.2.1, methods for developing impact indicators at the endpoint level have increased in recent years, but remain fewer than the range of midpoint methods available. In addition, among the relatively few LCIA methods calculating impacts at the endpoint level, some differences are observed in the scope of their coverage, especially among those related to impacts on ecosystem quality. Nevertheless, a high degree of consistency exists in both the midpoint and endpoint indicators developed through different methodologies. These indicators are widely applied across LCIA methods and beyond (e.g. IPA frameworks and EPS), supporting potential methodological alignment.

48. Impact pathway models are nevertheless able to provide transparent and interpretable outcomes when key limitations, assumptions and uncertainties are explicitly identified. in Sections 6.1 - 6.3 of the Annex describes in more detail several sources of heterogeneity in LCIA methods, including midpoint vs. endpoint levels, impact categories considered and geographical scope.

Water, land and biodiversity challenges

49. Impact pathways involving water and land processes remain far less mature relative to those involving other environmental pressures. This is primarily because regional applicability and transferability are substantially more challenging for these impact pathways. Metrics for freshwater eutrophication, for instance, vary remarkably across regions, freshwater types (e.g. lakes vs. streams), and species groups, introducing significant uncertainty in regional applicability (Azevedo et al., 2013[14]). Similar issues arise in water use assessments, where estimating water return flows, and sectoral efficiencies can be complex because of uncertainty introduced by data availability and variability in the conditions of local hydrological and ecological systems (Huijbregts et al., 2016[7]).

50. Biodiversity poses an even greater challenge. It is multidimensional by nature, spanning numerous species and ecosystem levels, and its responses to land-use change are complex and context-dependent. Because ecosystems and species react differently to disturbances, land-use impacts are highly heterogeneous across regions and land-use types, significantly complicating their quantification in LCIA. In particular, the diversity of responses among species groups and ecosystems underscores the need for region- and context-specific models to capture land use impacts accurately. For example, most databases do not currently provide spatially-differentiated data beyond the country level, undermining the broader applicability of spatially-differentiated models (Verones et al., 2020[21]).

2.2.8. Knowledge gaps

51. When knowledge on impact pathways is limited and methodologies aren't standardized (or at least well documented), cross-study comparability decreases. In such cases, a greater understanding of the exact impact mechanisms (i.e. the cause-effect chain itself is well known) is needed to reduce divergencies across methods. Likewise, a greater uniformity in approaches and models will facilitate more harmonised quantification of physical impacts contributing to a greater and more transparent comparability of results.

52. Uncertainty aspects are also important to include in quantification models. Better and more standardized uncertainty reporting in both source data and impact pathways would narrow unexplained differences and make results more comparable and interpretable across studies, both for physical impacts and value factors.

2.3. Valuing environmental and health impacts

2.3.1. Measures of value

53. The value of a given environmental or health impact is comprised of several components reflecting different value concepts. The plurality of existing value concepts and valuation methods for measuring them highlights the need for standardization in the valuation methods used for incorporating environmental and health impacts into public and private decision making. Various typologies have been developed across different fields (e.g. natural capital accounting and environmental economics) to distinguish between different concepts of value.

54. Perhaps the most fundamental distinction relevant for incorporating environmental and health impacts into public and private decision making is the distinction between **market value** and **non-market value**. The market value of an impact is its value as measured through marketplace transactions that put a direct price on the value of a good or impact. For example, one market value of the impact of forest loss is the loss of park entry fees to the forest or the market value of the timber resources. Non-market value refers to the monetary value of any impact that is not directly priced by marketplace transactions. The monetary value of non-market impacts can either be deduced from related market transactions through

what is known as 'revealed preference' methods, or by asking individuals to what extent they value these impacts through 'stated preference' methods.

55. In natural capital accounting contexts, another distinction in value concepts is made between **exchange value** and **welfare value**. Exchange values reflect "the values at which goods and other assets, services, and labor are exchanged or else could be exchanged for cash" (UN et al., 2024[34]).[16] Exchange values reflect the prices at which goods and services are actually sold, whereas welfare values reflect the total benefit that a person receives from a good or service, which may be greater than its exchange value.[17] As a result, exchange values are lower than welfare values. Natural capital accounting, which seeks to quantify extended measures of national wealth, takes into account the non-market benefits that environmental amenities provide to the economy insofar as these benefits are reflected in observed exchange values obtained through various revealed preference valuation methods (UN et al., 2024[34]).[18]

56. In contrast, cost-benefit analyses of environmental policies in the public sector, as well as environmental profit and loss statements in the private sector, rely on welfare values insofar as these uses seek to assess the net welfare cost or benefit that a policy change or business activity implies for society. Different measures of monetary value can play complementary roles in supporting decision making.

57. While the concepts of market value and exchange value are closely aligned, the concepts of non-market value and welfare value diverge. The welfare value of a good or impact, which represents its total utility to an individual, can be measured by both its market and non-market value. For example, the welfare value of a forest may be measured by both the monetary value of an entry fee as well as the additional amount that an individual would hypothetically be willing to pay to access the non-market benefits that accessing the forest may offer them, such as mental or health benefits. Thus, while non-market values often contribute to welfare and welfare value, not all welfare values are non-market values. When the price of a good is equal to a consumer's willingness-to-pay,[19] for example, its welfare value is entirely reflected by its market value. Additionally, not all non-market values are equal to welfare values. This is notably the case for goods or impacts have value that are difficult to capture by non-market valuation methods, for example because individuals refuse or are unable to monetize their value.[20]

58. Another distinction that is made in environmental economics is that of total economic value, which is typically understood as a measure of welfare value. It is comprised of use values and non-use values. Figure 2.1 depicts different concepts of value as typically considered in environmental economics applications, which also are typically assumed to refer to welfare values (rather than exchange) insofar as

[16] Accounting practices rely on observed market prices (i.e. current exchange values) as the basis for valuation.

[17] In economic terms, exchange values encompass producer surplus only, while welfare values encompass both producer surplus and consumer surplus. For goods for which there is a market, changes in welfare are reasonably well represented by changes in national domestic product, i.e. changes in gross domestic product less any changes in depreciation. This is due to the fact that, in perfectly competitive markets with no externalities, the marginal value of a unit of a good reflects the welfare that that unit provides (UN et al., 2024[34]) (Weitzman, 1976[69]). For further elaboration on the concept of exchange vs. welfare value, see Appendix A12.1 in UN et al. (2024[34]).

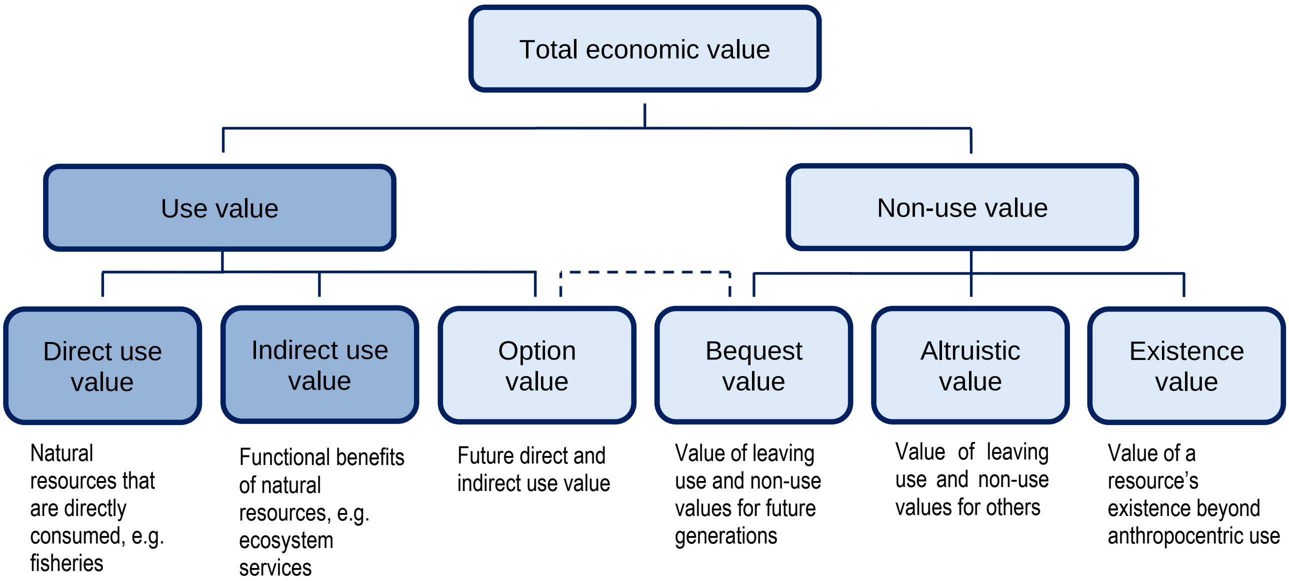
[18] Since no explicit markets exist for non-market goods, these exchange values of non-market goods cannot reflect the general equilibrium effects that would be expected if a market did exist. The extent to which the various valuation methods provide a good approximation varies, and it is to be noted that all methods reflect prices of a partial equilibrium.

[19] I.e. in cases where consumer surplus is equal to zero.

[20] Non-market values may notably be lower than welfare values for goods that individuals may oppose putting a monetary value on (e.g. cultural heritage sites, endangered species, or organ donation), or are ill-equipped to determine how much they value (e.g. biodiversity or existence values).

they are assumed to reflect consumer surplus in addition to producer surplus. Direct and indirect use values can be explicitly or implicitly reflected in market values, while non-use values are non-market values, as markets do not typically exist in which these values are materialised.

Figure 2.1. Measures of economic value



Note: Dark boxes indicate values that can be explicitly or implicitly reflected in market values. Light boxes indicate values that are typically non-market in nature.
Source: Adapted from Turner, Pearce and Bateman (1994[35])

59. Equivalent variation and compensating variation (measures of Hicksian surplus, which reflect only substitution effects) are the theoretically appropriate measures to capture non-market impacts because they capture the monetary compensation required to hold individuals' utility constant in the face of a change in the provision of non-market good. Travel cost and hedonic methods typically estimate Marshallian surplus, which reflects the income effect, as well. For this reason, Marshallian surplus is typically smaller than Hicksian surplus. Because income effects are expected to be small with respect to the consumption of non-market goods, travel cost and hedonic price methods are generally considered to be a measure of the use value of consumer surplus.

60. Finally, it should be noted that monetary values of environmental and health impacts can also be measured in different units to monetize changes at different points of the impact process – namely valuing impacts at the emissions, midpoint and damage levels.[21] Importantly, estimates of monetary value at the midpoint level are determined by characterization of emissions, i.e. the quantification of how much a given environmental pressure contributes to a particular environmental mechanism (midpoint), for example the contribution of CO_2 to climate change (CE Delft, 2024[6]). As reported in Section 3.1, most value factors are expressed at the emissions level, that is expressing the damage caused by a specific emission. An advantage of measuring the monetary value of impacts at the emissions level is that changes in emissions may be more easily linked to policies. In cases where the impacts of a given economic activity are more

[21] Valuations at the 'emissions' and 'damage' levels referred to here correspond to the 'substance' and 'prosperity' levels, respectively, in CE Delft (2024[6]).

readily measured in terms of damages (e.g. lives saved), then value factors at the damage level are likely to be the most useful metric for evaluating impacts.

61. To facilitate standardization of valuation practices, Table 2.2 presents a synthesis of valuation concepts as used across different fields. The table illustrates each value concept using the example of forest loss as an environmental pressure. It should be noted that not all valuation methods depicted in Table 2.2 can be applied for all non-market impacts, and in some cases, valuation methods are only applied for specific types of non-market impacts.

Table 2.2. Measures of economic value and their measurement methods

	Valuation Method Category [a]	**Valuation methods**	**Value concepts**	**Example (forest loss)**
			Market value	
	1. Prices are directly observable [b]	Market prices	Direct use value: Explicit exchange value of lost uses reflected by economic units	Lost revenue from park entry fees to the forest
			Non-market values	
Revealed preferences	2. Prices from similar markets	Market prices in similar markets	Direct use value: Implicit exchange value of lost uses at a similar forest reflected by economic units	Lost revenue from park entry fees to a similar forest
	3. Prices embodied in market transactions	Residual value; net factor income	Direct use value: Implicit exchange value of lost ecosystem services	Lost rent from timber, groundwater supply, foraged edible foods
		Hedonic pricing	Direct and indirect use value: Implicit exchange value of the lost benefits of proximity to the forest	Loss of price premiums for nearby properties
		Production function method	Direct and indirect use value: Implicit exchange value reflecting lost productivity of ecosystem services	Cost of productivity loss of nearby agricultural operations due to lost forest ecosystem services
	4. Prices from revealed expenditures on related goods and services	Averting behaviour	Direct use value: Implicit exchange value reflecting loss of subjective forest benefits	N/A (value applies to environmental disamenties)
		Travel cost	Direct use value: Implicit exchange value of lost recreation services	Lost of recreational value as measured by the cost of travel to the forest (e.g. gasoline, tolls)
		(Abatement cost [c])	Direct use value: Implicit exchange value reflecting cost of maintaining or improving forest benefits to the public	Cost of preventing forest loss, e.g. purchasing forestland for conservation
	5. Prices from expected or simulated expenditures or expected markets	(Replacement cost [c])	Direct use value: Implicit exchange value of the cost of replacing forest ecosystem services	Cost of replacing pollination services for nearby agriculture
		(Avoided damage cost [c])	Direct use value: Implicit exchange value of expected damage costs of forest loss	Loss of economic revenue from tourism
Stated preferences		Simulated exchange value [d]	Use and non-use value: Estimated exchange value of non-market benefits (i.e. excluding consumer surplus)	Estimated price of non-market benefits if a market existed
	Other valuation measures	Choice experiments Contingent valuation	Total economic value: Welfare value of use and non-use benefits	Stated willingness to pay for the forest
		(Qualitative methods)	N/A (non-monetary measure of value)	Subjective evaluation of the value of the forest to well-being

Note: Further details on valuation methods are provided in the Annex. Methods in parentheses are not the focus of the current report. Replacement cost and avoided damage cost approaches are relevant for measuring the total value of an asset, rather than changes in an asset, and the mitigation cost approach is less relevant for measuring the value of damages. UN et al. (2024) also identifies the following as types of other valuation measures: shadow project cost method, opportunity costs of alternative uses, as well as prices from economic modelling. These are not included in the above due to the fact that their applicability is limited to very specific cases and/or require significant data inputs.[22]
[a] Valuation method categories as identified by the SEEA EA (System of Environmental-Economic Accounting – Ecosystem Accounting) (UN et al., 2024[34]). [b] Exchange values from directly observable prices are the only values reflected in Systems of National Accounts (SNA). Environmental extensions can incorporate exchange values measured by exchange values not in the SNA. Welfare values are theoretically not aligned with a SNA framework. [c] Cost-based methods seek to measure the cost of compensating for damages, the cost of replacing an asset, or the cost paid to mitigate a risk to an asset. [d] Simulated exchange values are estimated by combining information on the relevant demand function with a supply function and an appropriate market structure and applying microeconomic methods to arrive at a simulated price that could be expected if a market for the non-market good existed (UN et al., 2024[34]).
Source: The illustrative example of forest loss is adapted from Barton et al. (2019[36]). Valuation concepts synthesized from OECD (2018[37]), UN et al. (2024[34]), CE Delft (2024[6]), HM Treasury (2022[38]), UBA (2024[39]), IFVI and VBA (2024[40]), WiFOR Institute (2025[41]) and VBA, Capitals Coalition and WBCSD (2023[42]).

Health impacts

62. Human health effects are classified into mortality, i.e. premature death, and morbidity, i.e. disease. The health effects due to environmental pollution can thus be divided into two main types of effects, namely mortality and morbidity. Mortality is expressed as an increase in mortality risk. Certain types of environmental pollution, such as smog, are also associated with reduced lifespan. Morbidity is expressed as an increased in the disease burden. Environmental pollution leads to a higher incidence of numerous health problems, including asthma and pulmonary disorders, allergies, eczema and reduced IQ development.

63. The value of prevented fatalities can be expressed in terms of a Value of a Statistical Life, which reflects the value a given population places on avoiding the death of an unidentified individual (OECD, 2025[43]). It is constructed from measures of willingness to pay at the individual level that reflect the maximum amount that can be subtracted from an individual's income or wealth in exchange for a given improvement in mortality risk without leaving the individual worse off.[23] VSL can thus be defined as the rate at which individuals are willing to trade off wealth or income for a reduction in their risk of dying. It is important to note that VSL is based on the value of marginal variations in mortality risk, rather than the value of risk attached to a specific death.

64. The nature and time horizon of risk to life and health can differ. For example, some risks pose an immediate danger to life (e.g. road accidents) while others have lifetime impacts (e.g. air pollution). Furthermore, the implications of some risks may be uncertain or occur in the future (e.g. climate change). Consequently, it may sometimes be more appropriate to value some risks insofar as they impact the *length* of a life. To this end, a standard adaptation of the VSL concept is the value of a statistical life-year (VOLY or VSLY) which was first introduced by Moore and Viscusi (1988[44]).

[22] The 'shadow project cost method' is a variant of the replacement cost method that estimates the hypothetical costs of providing the same ecosystem service elsewhere and is valid only valid only if the shadow project is actually realized or planned to be realized. The 'opportunity costs of alternative uses' method estimates values of ecosystem services by measuring the forgone benefits of not using the same ecosystem asset for alternative uses, which is significantly complicated by the difficulty of determining a realistic alternative use. The 'prices from economic modelling' derives prices for non-market benefits from computable general equilibrium models that are extended to include environmental factors, however, the data requirements for applying these models are very restrictive.

[23] Such a valuation may also be treated as the willingness to accept (WTA) a higher mortality risk in exchange for compensation, such as accepting a more hazardous job in exchange for a higher salary.

65. In addition to the VSL and VOLY, other metrics seek to capture the combined effects of mortality and morbidity. One example where such approaches could be useful is in estimating the benefits of a policy aimed at reducing the occurrence of a disease that leads to both pain and suffering and premature death. The most common metrics in this context are Quality-Adjusted Life-Years[24] (QALYs) and Disability-Adjusted Life-Years[25] (DALYs). While QALYs and DALYs can be converted into monetary terms, they are more often used without monetisation for comparing the relative impacts of policies (e.g. UNEP (2013[45]), IHME (2024[46])).

2.3.2. Approaches for measuring value

66. Natural capital accounting contexts differ from cost-benefit analysis and EP&L contexts. Whereas natural capital accounting attempts to capture the total value of environmental assets, the cost-benefit analysis and EP&L seek to capture the value of changes in, or impacts to, these assets. This report focuses on valuing impacts, or changes in the provision of non-market goods due to the environmental pressures generated by economic activities. Measuring the value of the impacts to the environment and human health that are quantified by impact pathway analyses described in Section × calls for measures of damage cost. Measures value as reflected by abatement costs seek to measure the cost of mitigating risk to an asset.[26]

67. Damage cost approaches fall into two categories: revealed preference methods and stated preference methods. Revealed Preference (RP) methods use observed market behaviour in related markets to infer the economic value of non-market environmental goods or impacts. Common applications include travel cost methods for recreational sites, hedonic pricing of property attributes, and defensive expenditures to avoid environmental harms. Stated Preference (SP) methods rely on surveys and interviews to directly elicit individuals' WTP for environmental changes, capturing both use and non-use values. Common methods include contingent valuation and discrete choice experiments. Further details on the variety of RP and SP methods are provided in the Annex.

2.3.3. Sources of heterogeneity in valuation methodologies

Inconsistencies in how additivity, uncertainty and other factors are addressed can lead to substantial variation in valuation results and ultimately value factors. This section provides a preliminary survey of potential sources of heterogeneity arising from differences in the approaches taken to several methodological choices involved in valuation.

[24] For a QALY, a life-year in perfect health is represented by a value of 1. The life-year can then be adjusted downwards to account for disability and loss of quality of life at different ages in a population, including full life-years lost due to premature death. To use QALYs for policy analysis, the sum of QALYs for different policy alternatives can be compared, where a larger number of QALYs would normally be preferred for a given cost.

[25] DALYs also vary from 0 to 1, but for an individual over his/her lifetime. In contrast to QALYs, DALYs measure the life-years lost due to disability and premature death and can be defined as the sum of life-years lost to premature death and life-years lost due to disability and/or pain and suffering. A weighting factor is usually applied to a year lived with disabilities to signify its severity relative to a life-year lost. DALYs can be summed over affected populations by policy alternatives and are often used in cost-effectiveness analysis, where for a given cost, the policy with the lowest DALY score is generally preferred.

[26] HM Treasury (2022[38]) and CE Delft (2024[6]) use mitigation cost approach to value GHG emissions for reasons related to national policy context and uncertainty, respectively.

Adjustments

68. A number of standard adjustments are recommended in order to ensure that monetary values of environmental and health impacts are comparable across applications (VBA, Capitals Coalition and WBCSD, 2023[42]). Such adjustments include:

- Adjusting for foreign exchange rates: For impacts valued using different currencies, the exchange rate needs to match the time period defined in the scope of the study. Use data published by the World Bank, IMF, or similar recognized institutions. Depending on the business application, it may be useful to use five-year rolling averages to avoid currency conversion artifacts.
- Adjusting for inflation: Costs and benefits in appraisal of social value should be estimated in 'real' base year prices. When using data sets for valuation developed in the past, these should be adjusted to the time period considered in the scope of the study. You should use official sources of inflation such as the IMF or the World Bank.[27]
- Adjusting for Purchasing Power Parity (PPP) (optional): Adjustments for purchasing power parity may be performed to ensure that the values expressed across country contexts reflect the same effective purchasing power.
- Calculating net present value: Impacts that occur in the future should be discounted to take into account the fact that individuals tend to place greater value on outcomes that occur today than outcomes that occur in the future. A social discount rate can be used to convert impacts into their present value to allow for a comparison between costs and benefits that occur at different times. The social discount rate reflects the value for society of future costs and benefits compared to present ones. See the subsection on 'Discounting' below for further details.

Discounting

69. Discounting is a technique used to compare costs and benefits occurring over different periods of time on a consistent basis, and the choice of discount rate to use in evaluating the net present value of future environmental and health impacts can have a particularly significant impact on valuation results. The discount rate reflects the extent to which future impacts are given less weight relative to current impacts. The discount rate can be decomposed into several components, including a pure rate of time preference (reflecting the extent to which individuals weight current outcomes more than future outcomes), the elasticity of marginal consumption (reflecting the extent to which consumption affects utility) and the expected growth rate of future real per capita consumption.

70. Because environmental and health impacts resulting from environmental pressures have inter-generational relevance, guidance generally recommends using a pure rate of time preference of zero, an elasticity of marginal consumption of 1, and an expected growth rate of future real per capita consumption of 2% (UN et al., 2024[34]), resulting in a recommended social discount rate of 2%.[28] Best practice approach is to first convert costs or benefits to a real price basis, and then perform the discounting adjustment (HM Treasury, 2022[38]). Uncertainty about future economic growth or future interest rates, combined with caution in the face of future risks implies the use of a declining discount rate, whereby the discount rate

[27] It has been recommended to use the most recent estimates of whole economy inflation for short time horizons and forecasts of this inflation for longer time horizons. Longer appraisal periods may be suitable where intervention is likely to have significant social costs or benefits beyond 60 years.

[28] In a survey of more than 200 experts, Drupp et al. (2018[70]) find that social discount rates used range from 1 to 3.5%, and that more than 75% consider a median risk-free social discount rate of 2% acceptable.

applied at points in time farther in the future is lower than the rate applied at points nearer in time (OECD, 2018[37]).

Additivity

71. Ensuring accurate measurements of the monetary value of environmental and health impacts, and comparability across estimates involves clear delineation and alignment, respectively, of the scope of the impacts considered. Double counting, for example, can lead to inflated measurements of value, and differences in scope between measurements of value can reduce comparability of estimates. With respect to double counting, for example, HMT notes that insofar as “biodiversity may be reflected by, or associated with other benefits e.g. recreation, pollination, water quality and amenity… to avoid double counting, biodiversity should only be valued where it directly impacts human wellbeing and where it is additional to other benefits. For example, non-use value for biodiversity represents a legitimate additional category of value that can be added to direct and indirect use values for final goods and services” (HM Treasury, 2022[38]).

72. Considered design, modelling, and interpretation of SP and RP valuation methods are essential to avoid double counting. When conducting SP studies, such as contingent valuation or choice experiments, a primary strategy is to clearly define the good or service being valued in a way that avoids overlapping benefits. For example, if a survey asks for willingness to pay to improve river quality, and that improvement includes benefits like aesthetics, recreation, and wildlife, the values for those components should not be separately summed unless they were independently evaluated. Similarly, summing willingness to pay across overlapping scenarios, such as estimating separate values for water quality and biodiversity when respondents have already valued the full package, can lead to overestimation.[29]

73. Similar risks apply to RP methods. For example, the travel cost method (see Annex Section 6.4.1) captures the total value of a recreational visit, which may already include benefits from scenery or wildlife, implying that adding separate WTP estimates for these could constitute double counting of value. Likewise, to the extent that the hedonic price method (see Annex Section 6.4.1) reflects environmental qualities (e.g. clean air, noise reduction) in property values, extracting and adding these values separately without careful modelling could inflate the value of environmental amenities. Averting behaviour studies (see Annex Section 6.4.1) also risk overlap if combined with SP measures that capture similar motivations.

74. Generally speaking, to mitigate the risk of double counting, the scope of a given valuation exercise should be clearly defined and practitioners should avoid overlapping methods unless properly adjusted and use integrated or functionally separated models when possible. Total values should not be the simple sum of multiple components unless it's clear they represent distinct, non-overlapping aspects of value.[30]

Uncertainty

75. Uncertainty in estimates of the monetary value of environmental and health impacts can stem from a multitude of sources. In addition to the way in which spatial and temporal variation in the processes

[29] These issues can be mitigated by, for example, employing scope tests and embedding checks to determine whether respondents are valuing a specific change or just expressing general support for environmental protection. Pretesting instruments with focus groups and cognitive interviews is also key to ensuring respondents clearly understand what they’re valuing and don’t assume overlapping benefits. In choice experiments, attributes should be independent (orthogonal) to reduce the risk of correlation-based double counting, and marginal WTPs derived from choice experiments should only be aggregated when they reflect realistic, well-defined policy packages.

[30] For example, using TCM to estimate the value of visiting a national park and then separately estimating WTP for birdwatching at the same park could result in double counting, especially if birdwatching is a main reason for the visit.

through which environmental pressures translate to physical impacts generates uncertainty in LCIA, variation in market conditions and preferences can generate uncertainty in the monetary value of physical impacts to the environment and human health. In general, parametric uncertainty refers to uncertainty within models, such as concerning the value of specific parameters like economic growth rate, and structural uncertainty refers to uncertainty about the models themselves, such a not taking into account relevant ecological tipping points.[31]

76. In non-market valuation, uncertainty in estimates of value can arise from issues related to data quality, model specification and behavioral assumptions. One major source of uncertainty pertaining to revealed preference methods is measurement error in data. This includes inaccuracies in recording key variables such as prices, travel distances, or property values. For example, travel time may be estimated imprecisely, or housing prices might not reflect the actual transaction values, leading to biased or inconsistent estimates of monetary value. Another critical issue is omitted variable bias, which occurs when models exclude important factors that influence behavior. For example, failing to account for substitute sites in a travel cost model or neglecting neighborhood crime rates in a hedonic pricing study could significantly distort estimates of how much people value environmental or public goods. Functional form misspecification also contributes to uncertainty. This occurs when the mathematical form of the utility or demand function is incorrectly assumed, which results in poor model fit and erroneous WTP estimates.

77. Multicollinearity, or high correlation between explanatory variables (like income and housing quality), can further complicate estimation by making it difficult to isolate individual effects and increasing uncertainty in the final estimates of value. The problem of limited variation in key variables also arises in RP methods, limiting the ability to identify marginal WTP. Endogeneity, where explanatory variables are correlated with the model's error term, can occur due to reverse causality (e.g. house prices influencing proximity to amenities) or omitted variables (e.g. that affect both location choice and WTP), leading to biased estimates.

78. RP models also rely on assumptions about the relationships between goods, such as substitutes and complements. If these assumptions are incorrect, the resulting demand and WTP estimates may be inaccurate. Additionally, non-monetary influences on behavior can distort valuation. People might make choices for social, cultural, or emotional reasons unrelated to economic value. For example, a person may visit a site out of tradition rather than environmental appreciation, which can mislead inference about the site's value. Market imperfections, such as discrimination or credit constraints in the housing market, can also lead to inaccurate estimations of value.

79. Estimates from stated preference methods can also vary due to the econometric issues above. Uncertainty in estimates generated using SP method can also arise due to cognitive biases (e.g. hypothetical bias of strategic bias). As in RP methods, a lack of knowledge or understanding of the good or the scenario being valued can also lead to inaccurate estimates of value. Estimates from SP methods have also been shown to be sensitive to various survey design factors such as the complexity of the elicitation method, wording of survey questions, the type of payment vehicle used (e.g., taxes versus donations), and the order or framing of choices, which can influence how respondents interpret and answer the questions. Because SP methods rely on surveying a sample of the population, sampling error, or statistical variation in how representative the sample is of the general population, can also affect value estimates. This can be compounded by non-response and self-selection bias, whereby estimates may not

[31] In the calculation of the social cost of carbon, for example, the two largest sources of uncertainty are the damage function employed and the pure rate of time preference assumed (OECD, 2018[37]). Structural uncertainty can be addressed by using multiple models, where relevant and justified (e.g. integrated assessment models such as DICE, FUND and PAGE in the calculation of the social cost of carbon). Monte Carlo simulations are used to handle parametric uncertainty by assigning probability distributions to inputs.

accurately reflect the broader population's preferences if non-respondents differ systematically from respondents.

80. To the extent that context-specific factors are important determinants of valuation results, valuation should therefore be undertaken at scales appropriate for reflecting such differences, rather than attempting to estimate average values at scales that do not adequately reflect important differences across contexts. Following current practice, the results of valuation can also be presented by categorizing impacts according to the certainty with which they occur and using a range of lower, central and upper values, as in CE Delft (2024[6]). Supplementary information can also be provided to inform users of the range of the alternative estimates and may be complemented by sensitivity analysis for different methods.

Value transfer

81. Value transfer involves applying economic value estimates from existing studies, e.g. those contained in the Environmental Valuation Reference Inventory (Environment and Climate Change Canada, 2025[47]), to new contexts, adjusting for spatial, temporal, and other contextual differences. This approach helps estimate values where original valuation studies are unavailable.

82. Two main types of monetary value transfer methods exist. The first is referred to as unit value transfer, which applies value estimates directly from one site to another. Unadjusted value transfer directly applies values obtained from one site to another, and adjusted value transfer involves adjusting the values for factors such as income differences or purchasing power parity between the original and target sites as relevant. Although unadjusted value transfer is simple to carry out, it fails to capture important differences between the characteristics of an original study site (or sites) and a new policy site. If these differences are significant determinants of values, then this transfer approach will fail to reflect likely divergences in values between the study and policy sites.[32]

83. A more sophisticated method is value function transfer, which uses valuation functions based on physical and socioeconomic variables to estimate values at new sites. If it is known that values at the study site is a function of certain physical features of the site and the socioeconomic characteristics of the population at the site, then the value at the target site can be estimated by using the target characteristics in this function. The method can employ value functions derived either from a single study or from multiple studies in meta-analysis function transfer to develop generalized value functions applicable across different contexts. In general, more sophisticated value transfer methods are considered more accurate (OECD, 2018[37]).

2.3.4. Knowledge gaps

Aggregating different value concepts into single measures of monetary value

84. In light of the diversity of measures of value concepts and methodologies to estimate them, a better understanding is needed regarding how such measures and methodologies can be meaningfully aggregated, notably while avoiding double-counting. This involves, for example, the aggregation of exchange and welfare values, morbidity and mortality effects and stated and revealed preference methods.

[32] Determinants of WTP that might differ between study and policy sites include the socio-economic and demographic characteristics of the relevant populations (e.g. income, educational attainment and age), the physical characteristics of the study and policy sites (e.g. the environmental services that a good provides such as, in the case of a river, opportunities for recreation in general and angling), the proposed change in provision between the sites of the good to be valued, differences in the market/institutional conditions applying to the sites (e.g. in the availability of substitutes in the case of recreational resources), and temporal changes (e.g. trend in income or environmental quality) (OECD, 2018[37]).

While some work has been done in this area, for example on estimating simulated exchange values of consumer surplus as a way of incorporating welfare values into national accounting (Caparrós et al., 2017[48]; Oviedo et al., 2016[49]) and on combining revealed and stated preference methods to value welfare impacts (Adamowicz, Louviere and Williams, 1994[50]), more standardised guidance is needed if such methods are to be more widely practiced. Other questions, such as include whether marginal prices can be used to value non-marginal changes in environmental and health conditions, should also be addressed therein.

Treating uncertainty and risk

85. Uncertainty in valuation can arise from uncertainties embedded in impact pathway models, but also from uncertainties in valuation procedures. These include for example, how to reflect the risk of irreversible environmental damages in discounting practices and how the precautionary principle should be reflected. Standard discounting frameworks are not well-equipped to handle irreversible losses and deep uncertainty, leading to underestimation of the true economic value of environmental impacts that are irreversible (e.g. species extinction or abrupt catastrophic ecosystem changes), which may be unknown and are often difficult to model. A better understanding and standardisation of methods to measure and report uncertainty and risk will contribute to more comparable value factors.

Identifying consensus on ethically relevant methodological choices

86. Shared approaches to methodological choices involving ethical considerations will also facilitate more comparable value factors. This pertains for example to choices regarding discount rates, VSL values and equity weighting approaches. Variation in approaches to equity weighting can significantly affect aggregated estimates of non-market values.[33] Further, once non-market impacts are measured and reported in a standardised way, consideration should also be given to boundary conditions, i.e. whether the extent or type of certain impacts should be considered unacceptable for policy purposes (e.g. surpassing certain ecosystem threshold effects or generating impacts characterised by high levels of inequality).

[33] When aggregating non-market values to the population level, some guidance has recommended the use of equal weighting of human health values, and using the highest candidate value of statistical life (VBA, Capitals Coalition and WBCSD, 2023[42]).

3. Stocktake of value factors

3.1. Overview of existing value factors

87. The valuation methodologies described in Section 2.3produce values at the level of damages (e.g. the Value of a Statistical Life or of deforestation of a given amount of land). Incorporating the value of these impacts in public and private decision making generally requires monetary values expressed at either the emissions level (e.g. USD per unit of air pollution emitted) or at the midpoint level (e.g. USD per unit increase in the concentration of air pollution per cubic meter). As value factors at the emissions level are most useful for the majority of applications of impact accounting or cost benefit analysis, the current review focuses on value factors measuring monetary impacts per unit of environmental pressure generated.

88. Value factors are prepared by various entities for different purposes, such as by governments to support appraisal and evaluation of public projects, as well as by the private sector and NGOs to help assist companies undertake environmental impact valuation, such as in the form of environmental profit and loss (EP&L). Under this preliminary stocktake, five main sources are reviewed (see Table 3.1). The rationale for selecting these sources is that they cover multiple environmental domains (including climate change, air pollution, water and land use) which may better serve to comprehensively inform public and private decision making. These five sources have been also referred to in recent international discussions, including at the OECD kick-off workshop on environmental impact valuation (OECD, forthcoming[51]).

89. To note, additional value factors may be available from other sources that focus on specific environmental domains such as GHG emissions. These individual studies are excluded from this initial review, and subject to further analysis.[34]

[34] For example, value factors from GIST impact were not included as these are provided for a fee. The authors are also aware of value factors provided by IVL Swedish Environmental Research Institute, Environmental Priority Strategies (EPS), that could additionally be considered in the next update of this preliminary stocktake.

Table 3.1. Sources of value factors under the stocktake exercise

Source	Name of publication / database	Year of valuation	Currency	Geographical coverage	Provider status
IFVI (2024)	IFVI (2024) Global Value Factor Database, 15 Oct 2024	2023	USD	218 countries and 50 US states	Nonprofit organisation
WiFOR (2023)	WiFOR (2023) WifOR Impact Valuation, METHODOLOGICAL REPORT	2020	EUR	N/A	Private sector
UBA (2024)	UBA (2024) Methodological Convention 3.2 Value Factors, Version 10/2024	2024	EUR	Germany	Government agency
CE-DELFT (2024)	CE-DELFT (2024) Environmental Prices Handbook 2024: EU27 version	2021	EUR	EU27	Government agency
UK (2022)	UK (2022) THE GREEN BOOK CENTRAL GOVERNMENT GUIDANCE ON APPRAISAL AND EVALUATION	2021	GBP	United Kingdom	Government agency

Source: Author(s) based on listed references: (IFVI, 2024[52]), (WiFOR, 2023[17]), (UBA, 2024[39]), (CE Delft, 2024[6]), and (UK, 2022[53]).

91. The five sources cover different environmental domains with differences in scope and detail. Table 3.2 maps out these five sources against the 6 environmental domains covered in this stocktake, namely, GHG, air pollution, water (consumption, pollution), land use, biodiversity and material resources.

Table 3.2. Sources of value factors and environmental domains covered

Env domains	Climate change (GHG)	Air Pollution ($PM_{2.5}$, NO_x)	Water Consumption	Water Pollution (N, P)	Land Use	Biodiversity	Material Resources
Units / Source	per ton CO_2e	per ton of emissions	per m3 of water	per kg of pollutants	per ha	per unit (per kg of pollutants / per ha of land)	per unit (kg / Nm^2)
IFVI (2024)	✓	✓	✓	✓	✓	(✓)	
WiFOR (2023)	✓	✓	(✓)	(✓)		✓	
UBA (2024)	✓	✓		✓		(✓)	
CE-DELFT (2024)	✓	✓	✓	✓	✓	(✓)	(✓)
UK (2022)	✓	✓	(✓)	(✓)	(✓)		

Note: Check mark in brackets indicates partial coverage or where the methodology is available, however, value factors are not publicly available. For air pollution, additional pollutants beyond $PM_{2.5}$ and NO_x are covered by IFVI, WiFOR, UBA, and CE DELFT. For water pollution, additional pollutants beyond Nitrogen (N) and Phosphorus (P) are considered by CE DELFT and IFVI. Biodiversity is considered as a separate environmental domain by WiFOR, whereas IFVI and CE Delft integrates them in land-use and UBA integrates them under air pollution. Material resources are only covered by CE DELFT, in terms of energy resources (oil, gas coal). Other environmental domains that are covered and not on this list include noise and waste (which both are beyond the scope of this review)
Source: Author(s) based on listed references: (IFVI, 2024[52]), (WiFOR, 2023[17]), (UBA, 2024[39]), (CE Delft, 2024[6]), and (UK, 2022[53]).

92. Environmental domains such as climate change (GHG) and air pollution ($PM_{2.5}$, NO_x) are well covered by these five sources. Other domains such as water consumption and water pollution, land use are partially covered by these five sources. Biodiversity is covered as a separate environmental domain in WiFOR (2023[17]), whereas this is considered as a part of air pollution impacts under UBA (2024), and as part of air pollution, water pollution land use under CE DELFT (2024[6]) and IFVI (2024[52]). Value factors for material resources are not covered, except for energy resources (oil, gas, coal) under CE DELFT (2024[6]). Additional environmental domains such as waste (e.g. IFVI, WiFOR, UBA, and UK Green Book) and noise (e.g. UBA and UK Green Book) are covered by these sources (however, beyond the scope of review of this stocktake exercise).

93. The following subsections illustrate the value factors for each environmental domain and provides an illustrative comparison of the values.

3.1.1. Climate change (GHG)

94. Climate change can have severe economic consequences, including damage to agriculture, fisheries and capital due to increased severe weather events and rising sea levels, as well as impacts via health diseases and labour productivity losses from heat stress (OECD, 2015[54]).

95. Climate change is one of the environmental domains that are well supplied with value factors. These values factors provide the net present value of emissions in units of the monetary value of per tonne of CO_2 equivalent. All five sources under this review provide value factors for GHG emissions. While the primary methodology appears to be a damage cost approach, such as estimating the impact value on human health and agricultural productivity (applied by IFVI (2024[52]), WiFOR (2023[17]),[35] and UBA (2024[39])), CE Delft (2024[6]) applies an abatement cost approach and the UK (2022[53]) Green Book applies a mitigation cost approach.[36] See Table 3.3 for further details.

[35] Damage cost approaches closely align with LCIA approaches that specify impact pathways and characterisation factors, whereas other approaches (e.g. abatement cost approaches, mitigation cost approaches) may not require the consideration of impact pathways (e.g. based on LCIAs) in order to produce estimates.

[36] Abatement cost approaches and mitigation cost approaches are similar concepts, both focusing on cost estimation related to reducing GHG emissions. An abatement cost approach focuses on cost of reducing emissions directly, whereas a mitigation cost approach focuses on the broader economic cost of mitigation.

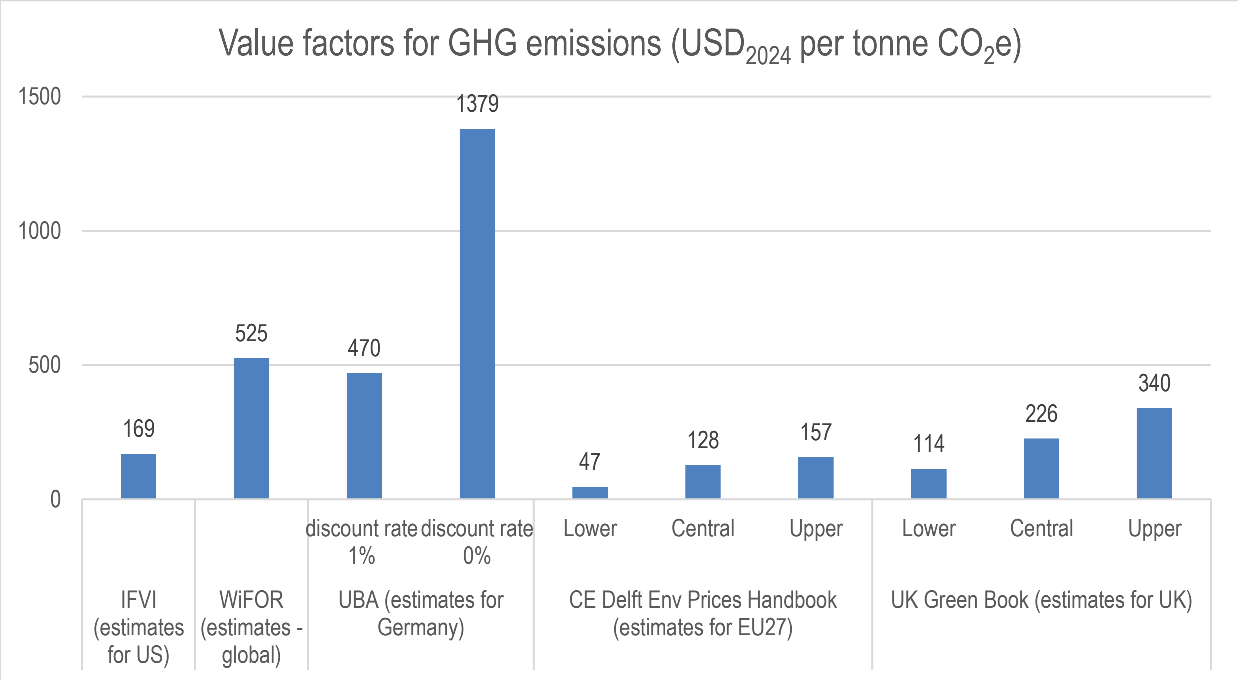
Table 3.3. Overview of value factors for GHG emissions

Source		Value factor	Units	Discount Rate	GHGs considered	Impacts considered	Remarks
IFVI (2024)		236.00	USD 2023 per ton CO_2e	Uses dynamic discount rate (Ramsey formula) calibrated to meet a near term discount rate of 2%; modelling of future impacts due to GHG emissions until 2300.	Carbon dioxide (CO_2) Methane (CH4) Nitrous oxide (N2O) Hydrofluorocarbons (HFCs) Perfluorinated compounds (PFCs) Sulfur hexafluoride (SF6) Nitrogen trifluoride (NF3) Other GHGs	Damage cost approach 1. Reduced human health via heat/cold-related mortality (VSL uses EPA's 1990 guidance value of $4.8 million, inflation adjusted) 2. Increased energy demand 3. Reduced agriculture production 4. Damages to the built environment and human health via sea-level rise 5. Losses in labour availability	Valuation of impacts follows a damage cost approach. A global value factor is used since climate change is a global phenomenon, which does not materialize at the source of the emissions. Two models are used to determine the GHG value factor: The Greenhouse Gas Impact Value Estimator (GIVE) and the Data-driven Spatial Climate Impact Model (DSCIM). The value factor developed from each model is averaged to produce a single value factor.
WiFOR (2023)		224.71	USD 2020 per ton CO_2e	Social discount rate1%	Carbon dioxide (CO_2) Methane (CH_4), Nitrous oxide (NO_2)	Damage cost approach: 1. Built environment 2. Agriculture and timber 3. Desertification 4. Other ecosystem services 4. Economic disruption 5. Human health	Costs of climate change are assessed in terms of their damage Valuation coefficient is equal across all countries and industries
UBA (2024)		300.00	EUR 2024 per ton CO_2e	Social discount rate1% - higher weight on current over future welfare	Carbon dioxide (CO_2) Methane (CH_4), Nitrous oxide (NO_2)	Damage cost approach: 1. agriculture 2. health (heat related mortality) 3. building energy consumption 4. sea level rise	Social cost of carbon (SCC) or damage cost approach are based on the Greenhouse Gas Impact Value Estimator (GIVE) model.
		880.00	EUR 2024 per ton CO2e	Social discount rate 0% - equal weight on current and future welfare			
CE-DELFT (2024)	Upper	160.00	EUR 2021 per ton CO_2e	2.25% discount rate	Carbon dioxide (CO_2) Methane (CH_4), Nitrous oxide (NO_2) + others?	Abatement cost approach: Estimations based on abatement cost method in-line with Environmental Price Handbook of 2018 and 2021.	The SCC is highly dependent on the chosen discount rate, global climate policy and the damage function included in the Integrated Assessment Model (IAM).
	Central	130.00					
	Lower	50.00					
UK (2022)	Upper	362.00	GBP 2021 per ton CO_2e	3.5% discount rate	Carbon dioxide (CO_2) Methane (CH4) Nitrous oxide (N2O) Hydrofluorocarbons (HFCs) Sulfur hexafluoride (SF6)	Mitigation cost approach: GHG values are based on the economic cost of mitigating a unit of carbon.	Carbon value assumptions for the traded and non-traded sectors are available for 3 different scenarios (low, central, and high) to enable sensitivity analysis. Further information and details are available in the BEIS guidance and Valuation of energy use and greenhouse gas (GHG) emissions.
	Central	241.00					
	Lower	121.00					

Source: Author(s) based on listed references: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]; UK, 2022[53]).

96. As in Table 3.3 above, value factors can range between EUR 50-880 per tonne of CO_2 equivalents depending on the source. Figure 3.1 further provides a comparison of value factors for GHG emissions in USD. The values for GHG emissions range between USD 47-1379 per tonne of CO_2 equivalents in 2024 prices. The main source of this heterogeneity appears to be from: (i) the discount rate applied, (ii) the differences in valuation approaches used (i.e. damage cost approach or mitigation/abatement cost approach), and (iii) the integrated assessment models that are used to combine and estimate damage costs and welfare costs. Applied discount rates range from 0% (treating current and future generations equally by UBA (2024[39])), to 3.5% by the UK Green Book (UK, 2022[53]).

Figure 3.1. Comparison of Value factors for GHG emissions



Note: Value factors for IFVI show estimates for the US and are indicative as the database has a global geographical coverage as shown in Table 3.1. OECD (2024) purchasing power parities referred for currency conversion from EUR and GBP into USD. OECD (2025) Consumer Price Index used to adjust for inflation to USD 2024 prices.
Source: Author(s) based on listed references: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]; UK, 2022[53])

3.1.2. Air pollution

97. Air pollution is another environmental domain that is well covered by value factors among the sources of this review. They can have impacts on human health, as well as crop harvest, material infrastructure and biodiversity loss. Value factors for air pollution are usually expressed in monetary values per tonne of emissions.

98. At minimum, pollutants considered include $PM_{2.5}$ and NO_x emissions, such as by UK (2022[53]). Other pollutants often covered include PM coarse particles (also known as $PM_{10-2.5}$), PM_{10}, sulfur dioxide (SO_2) organic chemical compounds (VOC/NMVOC), ammonia (NH_3), and methane (CH_4). These pollutants are largely in-line with the System of Economic-Environmental Accounting (SEEA) air emission accounts, with the exception of carbon monoxide (CO) that are considered as air emission accounts under the SEEA Central Framework (UN, 2014[55]). See Table 3.4 for an overview of covered pollutants among the five sources under this review.

Table 3.4. Coverage of pollutants

Source	$PM_{2.5}$	PM coarse	PM_{10}	NO_x	SO_2	VOC / NMVOC	NH_3	CH_4	Impacts considered	Remarks
IFVI (2024)	✓		✓	✓	✓	✓	✓		Impact pathways are determined based on several studies including from the US, EU and OECD. 1. Primary Health (mortality & morbidity of respiratory diseases) 2. Secondary Health (health impacts from ozone (O_3). 3. Agriculture (loss of crop output) 4. Visibility (aviation disruptions)	For mortality, OECD (2009) value of statistical life (VSL) used. For morbidity, Willingness to Pay (WTP) estimates from peer reviewed literature used. Value factors separately prepared for urban, peri urban, rural & transport, for 218 countries/50 US states.
WiFOR (2023)	✓		✓	✓	✓	✓	✓		Valuation methodology follows recommendations by UBA. 1. Human health (respiratory or cardiac illness, premature death) 2. Agriculture (change in crop yields due to air quality/ acid rain) 3. Man-made materials (corrosion) 4. Ecosystems (from air pollution)	Value factors are estimated for urban, peri-urban, rural, and transport, and calculated for countries based on geographical density, dependence on agriculture sector, and number of red list species.
UBA (2024)	✓	✓	✓	✓	✓	✓	✓		Air quality and exposure modelling based on impact pathway approach (EcoSenseWeb model). 1. Health damage ($PM_{2.5}$, PM_{coarse}, PM_{10}, NO_x, SO_2, NMVOC, NH_3) 2. Biodiversity loss (NO_x, SO_2, NH_3) 3. Crop damage (NO_x, SO_2, NMVOC, NH_3) 4. Material damage (NO_x, SO_2)	Value factors are specific for Germany. Publishes value factors for air pollution from emissions from (i) unknown source (ii) power stations, combustion processes in industry and small scale combustion plants and (iii) transport.
CE-DELFT (2024)	✓		✓	✓	✓	✓	✓	✓	Impact pathway approach based on EEA 2021 study. 1. Human health (mortality-stated preferences, morbidity-stated and revealed preferences. 2. Ecosystem services (stated preferences). 3. Damage to buildings and materials (restoration costs).	Damage costs of most pollutants can vary due to local conditions (e.g. population density) and nature of emissions (e.g. hight of exhaust pipe). As averages for EU27, environmental prices cannot readily be applied to specific local pollution, other countries, and non-average emission sources.
UK (2022)	✓			✓					Based on Defra's impact pathway approach. 1. Public health (mortality and morbidity effects of long-term and short-term exposure). 2. Natural environment (damage to materials and ecosystems) 3. Economy (work-days lost ($PM_{2.5}$) & minor restricted activity days for employees ($PM_{2.5}$, O_3).	Impact values based on damage cost approach, derived from more detailed Impact Pathway Approach of Defra. Value factors available for $PM_{2.5}$ and NO_x with estimates for low, central and high bound.

Notes: Coarse particles (also known as $PM_{10\text{-}2.5}$): particles with diameters generally larger than 2.5 µm and smaller than, or equal to, 10 µm in diameter. VOCs include such compounds as methane, benzene, xylene, propane and butane. Methane is primarily emitted from agriculture (from ruminants and cultivation), whereas non-methane VOCs (or NMVOCs) are mainly emitted from transportation, industrial processes and use of organic solvents.

Source: Author(s) based on listed references: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]; UK, 2022[53]).

99. Table 3.5 and Table 3.6 provide detailed value factors for $PM_{2.5}$ and NO_x respectively. Value factors for air pollutants are expressed in terms of their monetary value in units of per tonne of emissions. The values are further specified according to the specific location of the emissions taking place. For example, IFVI (2024[52]) and WiFOR (2023[17]) provide value factors for four different situations, which includes, (i) urban, (ii) peri-urban, (iii) rural, and (iv) transport. In particular, exposure to air pollutants can have severe health impacts on humans. Population density and the height of the exhaust pipe can make a difference, and transport is often handled as a separate specific case. UBA (2024[39]) provides value factors for five categories of: (i) emissions from unknown source, (ii) transport emissions with surroundings unknown, (iii) transport emissions in an urban setting, (iv) transport emissions in an suburban setting, and (v) transport emissions in a rural setting. Furthermore, in addition, UBA (2024[39]) provides detailed value factors depending on the hight of the exhaust pipe. CE Delft (2024[6]) and UK (2022[53]) provide value factors in a range for upper, central and lower bound estimations.

Table 3.5. Value factors for air pollution from $PM_{2.5}$ emissions

Source		Value factor	Units	Remarks
IFVI (2024)	Urban	46,950	USD 2023 per ton of $PM_{2.5}$ emissions	Value factors per air pollutant are country specific; a region can be urban, rural, peri-urban, or transportation. Indicative value chosen here from the entire database for primary health impacts for Peri-Urban areas in the US
	Peri urban	36,846		
	Rural	19,310		
	Transport	34,278		
WiFOR (2023)	Urban	90,520	EUR 2016 per ton of $PM_{2.5}$ emissions	Air pollutants that are being valued are particulate matter with a diameter 2.5 µm or less ($PM_{2.5}$), particulate matter with a diameter 10 µm or less (PM_{10}), nitrogen oxides (NO_x), sulphur oxides (SO_x), non-methane volatile organic compounds (NMVOC) and ammonia (NH_3).
	Peri urban	58,400		
	Rural	34,456		
	Transport	59,568		
UBA (2024)	Emissions from unknown source	227,273	EUR 2024 per ton of $PM_{2.5}$ emissions	For the air quality and exposure modelling we use EcoSenseWeb model developed for the EU project NEEDS (New Energy Externalities for Sustainability), Version v1.3 (Preiss et al. 2008), that has already been used in previous versions of this handbook (Methodological Conventions (2.0, 3.0 and 3.1). Other UBA estimates go more into details of the source and high of air emissions release
	Transport - surroundings unknown	222,571		
	Transport - Urban	905,956		
	Transport - Suburban	260,188		
	Transport - Rural	153,605		
CE-DELFT (2024)	Upper	210,031	EUR 2021 per ton of $PM_{2.5}$ emissions	These prices at the pollutant level are different from the mid point level.
	Central	126,959		
	Lower	82,810		
UK (2022)	Upper	378,885	GBP 2021 per ton of $PM_{2.5}$ emissions	Atmospheric pollution can have significant effects on health, quality of life, economic activity and the functioning of ecosystems. Three approaches can be used for valuation, if impacts are likely to be less than £50 million and do not affect compliance with legal limits then a “damage cost” approach is appropriate. This involves multiplying emissions changes by pre-calculated unit costs, described further below. This is often used to value the consequences of changes in pollution e.g. on health, crops and buildings. If impacts are greater than £50 million then the “impact pathway” approach should be considered. This involves bespoke modelling specific to the intervention. An “abatement cost” approach should be used in the limited instances where a proposal could affect compliance with legal limits. This involves estimating the least costly way of mitigating the impact of the proposal to ensure continued compliance with legal obligations.
	Central	122,342		
	Lower	26,481		

Source: Author(s) based on listed references: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]; UK, 2022[53]).

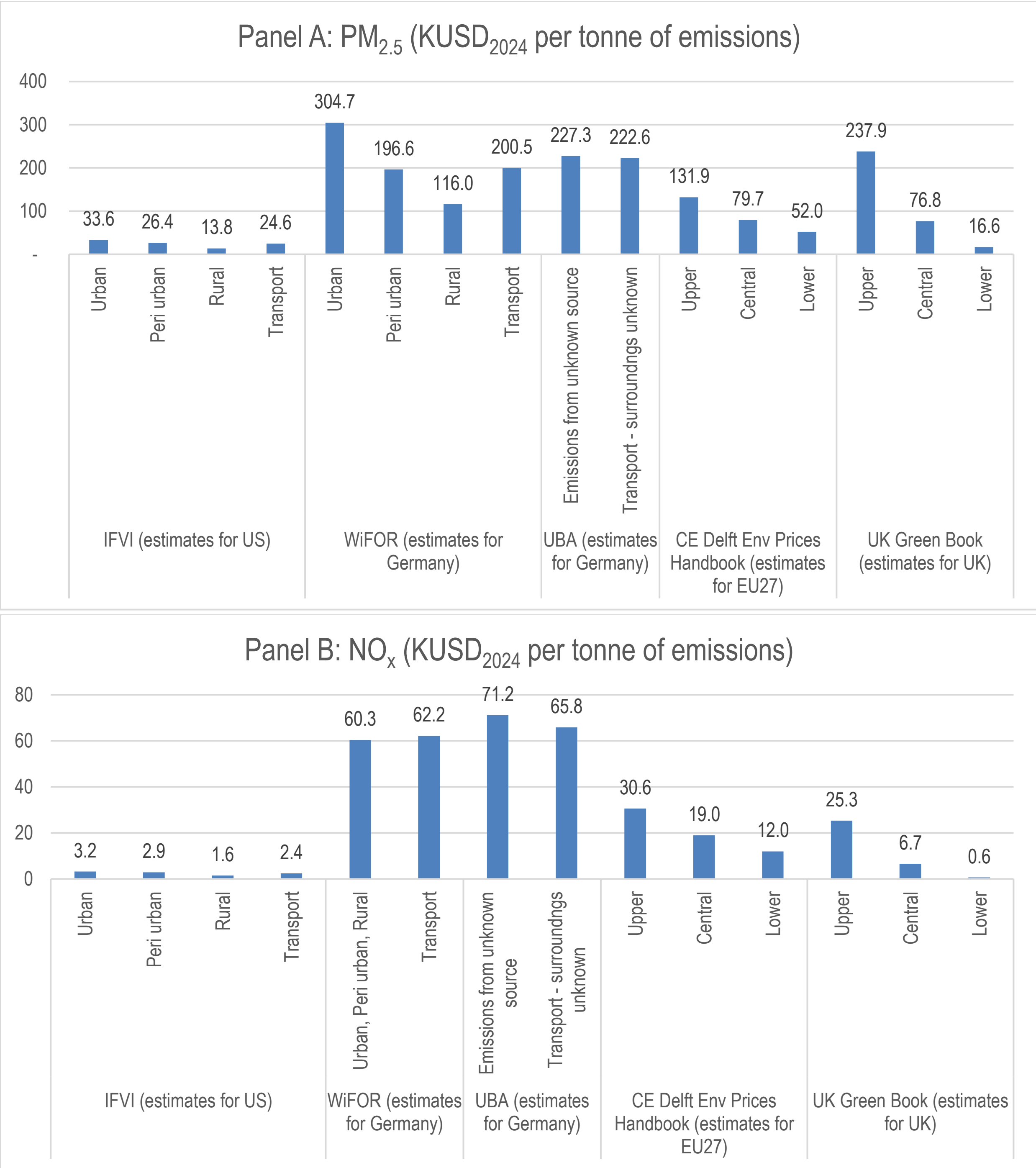
Table 3.6. Value factors for air pollution from NO_x emissions

Source		Value factor	Units	Remarks
IFVI (2024)	Urban	4486	USD 2023 per ton of NO_x emissions	Value factors per air pollutant are country specific; a region can be urban, rural, peri-urban, or transportation. Indicative value chosen here from the entire database for primary health impacts for Peri-Urban areas in the US
	Peri urban	4115		
	Rural	2181		
	Transport	3340		
WiFOR (2023)	Urban	17930	USD 2020 per ton of NO_x emissions	Air pollutants that are being valued are particulate matter with a diameter 2.5 µm or less ($PM_{2.5}$), particulate matter with a diameter 10 µm or less (PM_{10}), nitrogen oxides (NO_x), sulphur oxides (SO_x), non-methane volatile organic compounds (NMVOC) and ammonia (NH_3). This is assumed as peri urban. This is then estimated for urban, rural, and transport (based on UBA studies). These values are then calculated for countries - based on geographical density, dependance on the agricultural sector, and number of red list species
	Peri urban			
	Rural			
	Transport	18,468		
UBA (2024)	Emissions from unknown source	71160	EUR 2024 per ton of NO_x emissions	For the air quality and exposure modelling we use EcoSenseWeb model developed for the EU project NEEDS (New Energy Externalities for Sustainability), Version v1.3 (Preiss et al. 2008), that has already been used in previous version. Other UBA estimates go more into details of the source and high of air emissions release.
	Emissions from Transport	65831		
CE-DELFT (2024)	Upper	48746	EUR 2021 per ton of NO_x emissions	These prices at the pollutant level are different from the mid point level. When comparing the table above with the table from Section 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.
	Central	30251		
	Lower	19133		
UK (2022)	Upper	40351	GBP 2021 per ton of NO_x emissions	Atmospheric pollution can have significant effects on health, quality of life, economic activity and the functioning of ecosystems. Three approaches can be used for valuation. if impacts are likely to be less than £50 million and do not affect compliance with legal limits then a "damage cost" approach is appropriate. This involves multiplying emissions changes by pre-calculated unit costs, described further below. This is often used to value the consequences of changes in pollution e.g. on health, crops and buildings. 2. if impacts are greater than £50 million then the "impact pathway" approach should be considered. This involves bespoke modelling specific to the intervention. 3. an "abatement cost" approach should be used in the limited instances where a proposal could affect compliance with legal limits. This involves estimating the least costly way of mitigating the impact of the proposal to ensure continued compliance with legal obligations.
	Central	10643		
	Lower	1018		

Source: Author(s) based on listed references: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]; UK, 2022[53])

100. A comparison of value factors for air pollution are provided in Figure 3.2 ($PM_{2.5}$ in panel A and NO_x in panel B). It appears that there is heterogeneity between the value factors that are available. This can be due to the impact pathways and the characterisation factors that are applied under each methodology, the specificity of impacts from air pollution depending on the geographical area (e.g. EU27, Germany, United States, United Kingdom).

Figure 3.2. Comparison of value factors for air pollution



Note: Value factors for IFVI show estimates for the US and are indicative as the database has a global geographical coverage as shown in Table 3.1. OECD (2024) purchasing power parities referred for currency conversion from EUR and GBP into USD. Consumer Price Index used to adjust for inflation to USD 2024 prices. Additional air pollutants are considered by estimates developed by IFVI, WiFOR, UBA, and CE DELFT. More detailed value factors are developed by UBA concerning the source of emissions and the hight of the exhaust pipe.

Source: Author(s) based on listed references: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]; UK, 2022[53])

3.1.3. Water consumption

101. Value factors for water consumption are provided by IFVI (2024), WiFOR (2023), CE-DELFT (2024), and the UK Green Book (2022). This value factor estimates the monetary value of cubic square meters of water (with the exception of the UK Green Book (2022) that provides the industry average present value lifetime social cost of providing water supply per mega litre of water per day). Some characterisation factors that are considered include: (i) health damages, (ii) economic damages (e.g. damages from inadequate freshwater supplies for agriculture or the price to access freshwater supplies) (iii) damages to ecosystems. For health damages, these are often estimated from the DALYs lost using a value of statistical life (VSL), and LCIA characterisation factors, which express the domestic impacts on human health through water consumption. The impact of water consumption also depends on the water scarcity of each region. For this reason, some sources recommend to specify the source of water to the extent possible and consider water scarcity, for example the World Resources Institute's Aqueduct Water Risk Atlas (IFVI, 2024[52]). See Table 3.7 for further details.

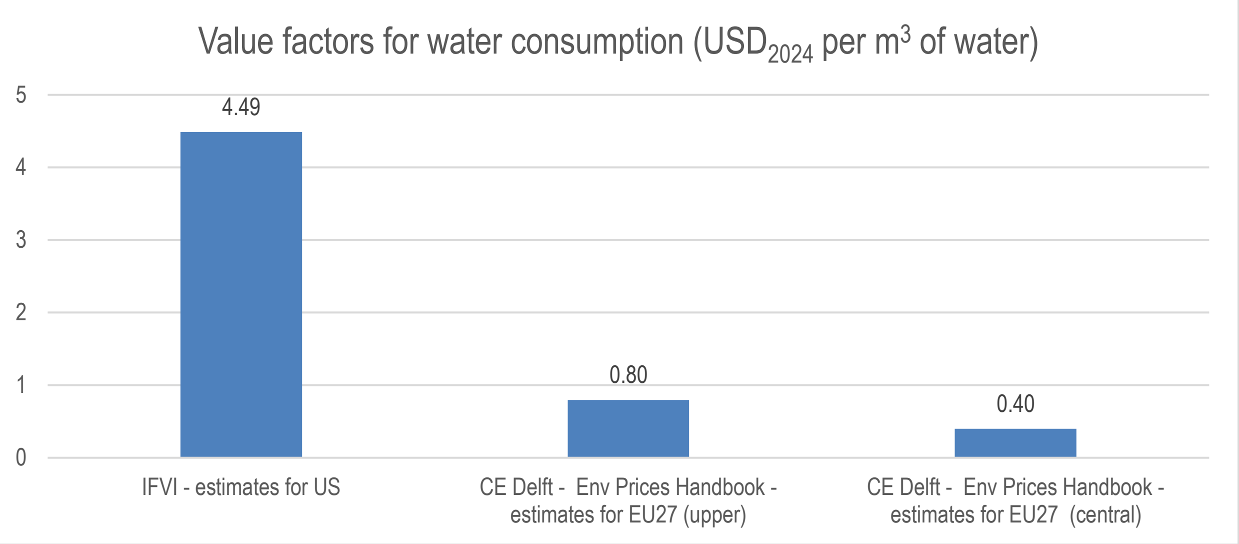
Table 3.7. Overview of value factors for water consumption

<table>
<tr><th colspan="2">Source</th><th>Value factor</th><th>Units</th><th>Impacts considered</th><th>Remarks</th></tr>
<tr><td colspan="2">IFVI (2024)</td><td>6.2613</td><td>USD 2023 per m3 of water</td><td>Impacts include (i) affected health from malnutrition, (ii) water-borne disease, (iii) altered ecosystem services, and (iv) financial costs to access future water.</td><td>For (i) malnutrition and (ii) water-borne disease, DALYs lost are valued using the value of statistical life (VSL) of the OECD (2005) and inflation adjusted, and this value is then multiplied by the DALYs from ReCiPe and corrected for inflation. For (iii) altered ecosystem services measures impacts to health, social connections, economic capital, human capital, and social capital that come through the lost ecosystem services, by using the Ecosystem Services Valuation Database (ESVD), a meta-analysis of 9,500 ecosystem service estimates across 6 continents. For (iv) financial costs to access future water, this determines when water demand is greater than available renewable surface and groundwater supplies using the World Resources Institute's Aqueduct Water Risk Atlas. This is a ratio of water demand over water supply with values above 1 representing unsustainable rates of extraction. Indicative aggregate values for the US are shown here for illustration.</td></tr>
<tr><td colspan="2">WiFOR (2023)</td><td>N/A</td><td>N/A</td><td>The valuation methodology includes: (i) economic damages, and (ii) health damages.</td><td>For (i) economic damages, the global water shadow price by Lightart and Harmelen (2019) is used as a baseline. The value represents the loss of economic gains in agriculture due to inadequate freshwater supplies that result from 1 m3 water usage. This is adjusted by country specific water scarcity factors according to the AWARE (Available WAter Remaining) model. For (ii) health damages, Disability Adjusted Life Years (DALYs) are valued from the statistical value of life (VSL), and LCIA characterisation factors (Debarre, et al, 2022) are used to express the domestic impacts on human health through water consumption.</td></tr>
<tr><td colspan="2">UBA (2024)</td><td>N/A</td><td>N/A</td><td colspan="2">N/A</td></tr>
<tr><td rowspan="3">CE-DELFT (2024)</td><td>Upper</td><td>0.811</td><td rowspan="3">EUR 2021 per m3 of water</td><td rowspan="3">Water consumption includes: (i) human health Daly/m3 consumed, (ii) terrestrial ecosystems species yr/m3 consumed, and (iii) aquatic ecosystems species yr/m3 consumed.</td><td rowspan="3">LCIA characterization factors are based on ReCiPe.</td></tr>
<tr><td>Central</td><td>0.407</td></tr>
<tr><td>Lower</td><td>0.00</td></tr>
<tr><td colspan="2">UK (2022)</td><td>5.7</td><td>Million GBP 2021 per mega litre of water/ day</td><td>Industry average present value lifetime social cost of providing water supply.</td><td>Value factor of water availability (rather than water consumption) for appraisal and evaluation of public projects</td></tr>
</table>

Source: Author(s) based on: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]; UK, 2022[53]).

102. Figure 3.3 further illustrates the comparison of value factors for water consumption. These estimates vary considerably depending on the source from which they are derived.

Figure 3.3. Comparison of value factors for water consumption



Note: Value factors for IFVI show estimates for the US and are indicative as the database has a global geographical coverage as shown in Table 3.1. OECD (2024) purchasing power parities referred for currency conversion from EUR and GBP into USD. Consumer Price Index used to adjust for inflation to USD 2024 prices.
Source: Author(s) based on: (IFVI, 2024[52]; CE Delft, 2024[6]).

3.1.4. Water pollution

103. Value factors from water pollution are mainly expressed in terms of monetary value per unit of kg of pollutants. The main pollutants of concern include Nitrogen (N) and Phosphorus (P), which are a source of eutrophication (excessive richness of nutrients in bodies of water), as covered by UBA (2024). There are a range of other toxic pollutants that can lead to health damages, which are additionally covered by IFVI (2024), WiFOR (2023), and CE Delft (2024).

104. Table 3.8 provides an overview of value factors for Nitrogen (N) and Phosphorus (P) emissions into bodies of water. The impact of eutrophication can be different from N and P, depending on their release to different bodies of water. Therefore, UBA (2024) and CE Delft (2024) provide separate value factors for emissions to different types of bodies of water (e.g. fresh water/inland waters, salt water/costal and marine waters). IFVI (2023) provides aggregate values, and WiFOR (2023) provide estimates for fresh waters only, as emissions directly into the sea are considered rare.

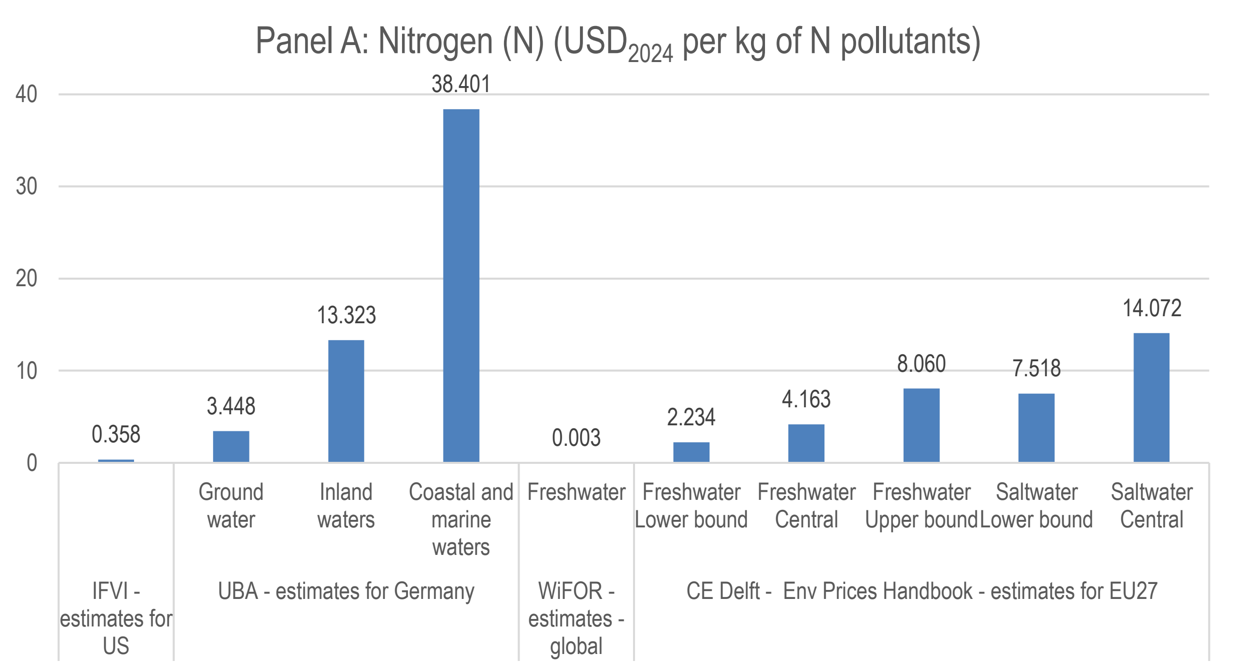
Table 3.8. Overview of value factors for water pollution from Nitrogen (N) and Phosphorus (P)

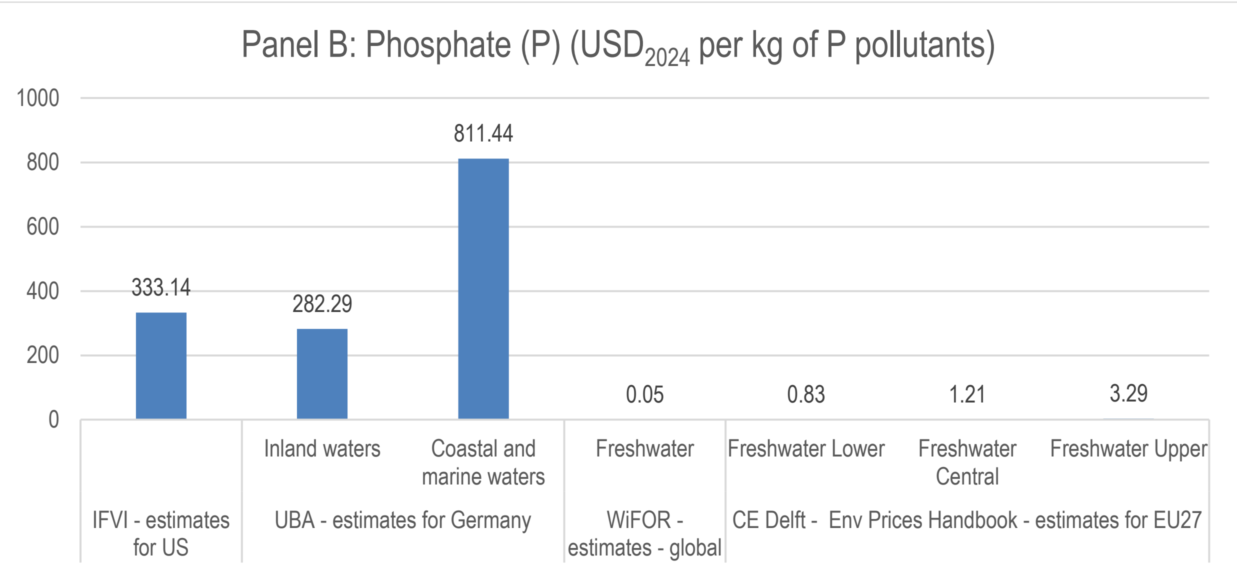
Source		Value factor		Units	Impacts considered	Remarks
IFVI (2024)		N	0.05	USD 2023 per kg of pollutant	Impacts from emissions to water pollution include (i) human health impacts from toxic pollutants, and (ii) eutrophication. For N and P, the impacts are solely for eutrophication. The method includes P for emissions to freshwater, and N and P for marine water.	For (i) human health, DALYs lost are valued using the value of statistical life (VSL) of the OECD (2005) and inflation adjusted, and this value is then multiplied by characterisation factor (i.e. number of disease incidences per kilogram of substance released to the waterbed of a substance in a given country) based on LCA of USEtox. For (ii) eutrophication, impacts caused by excessive nutrients is valued by a welfare-based approach to calculate generic damage values, based on Ahlroth (2009) who uses willingness to pay to estimate damage values per kg of N or P, combined with benefits transfer to other countries from the OECD. Characterization factors are calculated by the LC Impact. Indicative aggregate values of eutrophication impacts from N and P are shown here for the US as an illustration.
		P	464.93			
WiFOR (2023)		N	0.0024	USD 2018 per kg of pollutant	Human health impacts and eutrophication are considered as impacts from water pollution. For N and P, impacts from eutrophication are considered.	Estimates shown here are for N and P impacts to fresh water and adjusted to USD 2018. Estimates of pollutants are based on and expand upon Steen (2020) and include those for Nitrogen (N), Phosphorus (P), Arsenic (As), Cadmium (Cd), Mercury (Hg), Chromium (Cr), Lead (Pb), Nickel (Ni), Copper (Cu), Zinc (Zn), and Antimony (Sb). Human health characterization factors (DALY/kg emitted) are based on USEtox_2.0.
		P	0.0455			
UBA (2024)	Ground water	N	2.2	EUR 2024 per kg of pollutants	Monetized environmental impacts of N and P are specified assuming that the respective substance is the limiting factor for the eutrophication of the water body.	In most inland waters, plant growth is limited by phosphorus, whereas in most marine and coastal waters nitrogen is the limiting substance. Methodology is based on Schäppi et al. (2019).
		P	N/A			
	Inland waters	N	8.5			
		P	180.1			
	Coastal/ marine waters	N	24.5			
		P	517.7			
CE-DELFT (2024)	Freshwater Upper	N	8.190	EUR 2021 per m3 of water	Methodology includes freshwater eutrophication and marine eutrophication.	For other pollutants beyond N and P, freshwater ecotoxicity and marine ecotoxicity are considered. Characterisation factors are based on Recipe.
		P	0.845			
	Freshwater Central	N	4.230			
		P	1.23			
	Freshwater Lower	N	2.270			
		P	3.34			
	Saltwater Upper	N	27.600			
		P	0			
	Saltwater Central	N	14.300			
		P	0			
	Saltwater Lower	N	7.640			
		P	0			

Source: Author(s) based on: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]).

105. Figure 3.4 further provides a comparison of value factors for water pollution based on emissions of Nitrogen (N) and Phosphorus (P). As shown in this figure, there are significant variations of value factors provided depending on the source. While value factors for N and P emissions to water mainly quantify their effects on eutrophication, the values may differ depending on the geographic location of estimates, as well as the impact pathways and characterisation factors applied.

Figure 3.4. Comparison of value factors for water pollution





Note: Value factors for IFVI show estimates for the US and are indicative as the database has a global geographical coverage as shown in Table 3.1. OECD (2024) purchasing power parities referred for currency conversion from EUR and GBP into USD. Consumer Price Index used to adjust for inflation to USD 2024 prices.
Source: Author(s) based on: (IFVI, 2024[52]; WiFOR, 2023[17]; UBA, 2024[39]; CE Delft, 2024[6]).

3.1.5. Land use

106. Value factors for land use illustrate the monetary value in units of hectors (ha) of land. There are important nuances between land use and land conversion and details are provided in Box 3.1 below.

Box 3.1. Valuing land use and land conversion

The concept of **land use** refers to the ongoing utilisation of land for specific human activities such as agriculture, forestry, urban development, or industrial production. It reflects a state of use, meaning the land has already been altered from its original natural condition and is now providing benefits aligned with human activities. **Land conversion,** on the other hand, is the specific act of changing land from its natural state into a state used for industrial or productive purposes.

Such concepts are also reflected in the impact pathway framework for land use. ReCiPe 2016 distinguishes between two phases of land use transformation and occupation. During the transformation phase, the land is made suitable for its new function, with a rapid reduction in land quality; during the occupation phase, the land is utilised for a certain period in which land quality stays constant. Land transformation, or change of land cover, directly affects the original habitat, while land occupation, or intensification, further disqualifies the land as a suitable habitat.

The difference between land use and land conversion can be relevant when valuing impacts of land utilisation. In fact, value factors of land conversion assess the discounted total impact of land conversion process in loss of ecosystem quality, until that area of land could theoretically provide the pristine state ecosystem services again. Whereas land use value factors value the ecosystem services lost due to land use, or occupation, in a given year. In other words, land use values represent opportunity costs, while land conversion values are calculated over multiple years. The rationale underlying such value factors calculation is the attributability of the impacts of the conversion of the land from its natural state. Land use value factors can be used in cases of land occupation where land conversion has already occurred and cannot be attributable to the occupying entity. Value factors for land conversion on the other hand, specifically attribute the future consequence of the land conversion act to the entity responsible for it (i.e. for the loss of all ecosystem services that would have been provided by the land in its natural state). The conversion coefficient is therefore the total value of lost ecosystem services until the land could theoretically regenerate and discounted to reflect the higher value society places on lost services today.

Source: (IFVI, n.d.[56]; Huijbregts et al., 2016[7]).

107. Table 3.9 provides further details of value factors of land use. These estimates mainly include ecosystem services lost due to land use change in a specific year. However, they may also include other damages such as working capacity, and effects on drinking water, as the case for WiFOR (2023). As values of land use depend on their specific location, value factors are provided separately, such as for (i) animal rearing, (ii) crops, (iii) forestry, and (iv) pavement.

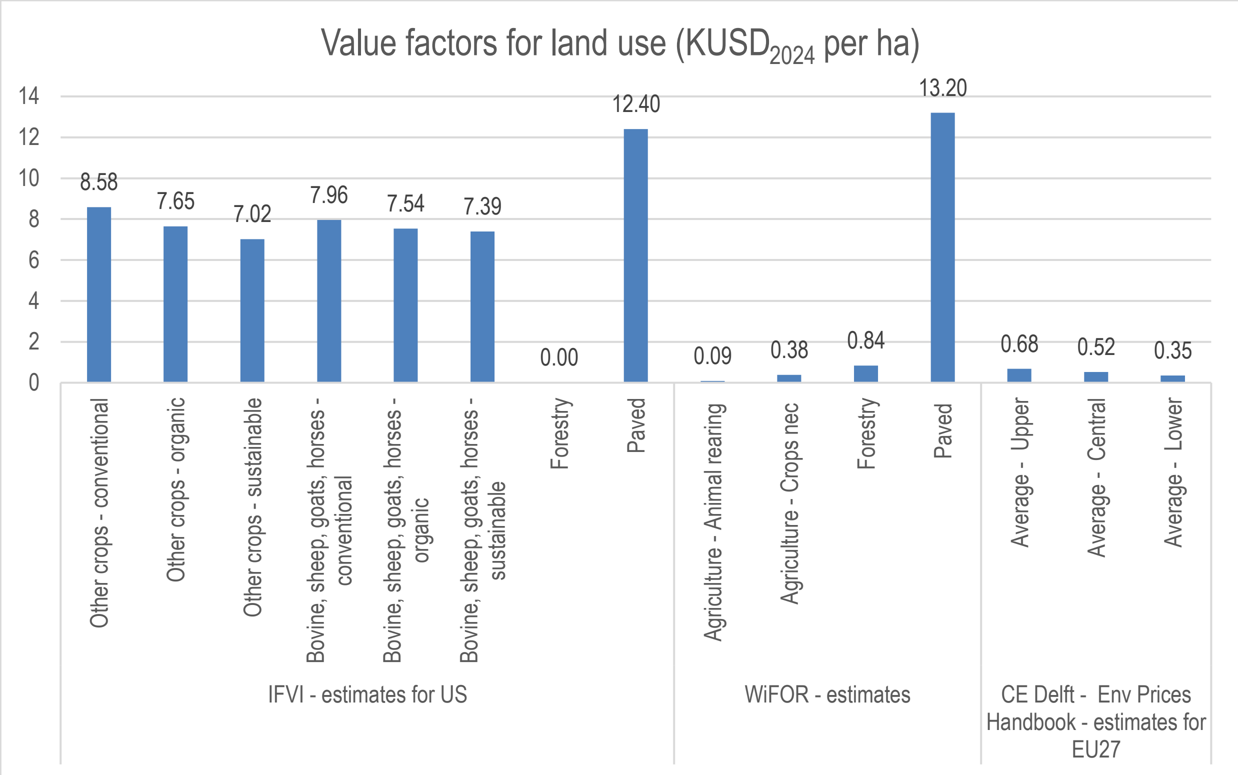
Table 3.9. Overview of value factors for land use

Source	Category	Value factor	Units	Impacts considered	Remarks
IFVI (2024)	Other crops - conventional	11,975	USD 2023 per ha of land	The value factors for land use specifically value the ecosystem services lost due to land use change in a given year (i.e. the difference in value between the ecosystem service provided by a pristine biome and a changed land use).	The value of land is determined through the ecosystem services framework, based on the Ecosystem Services Valuation Database (ESVD), which is a meta-analysis of 9,500 ecosystem service estimates across 6 continents. Studies in the ESVD analysis use numerous objective and subjective wellbeing indicators and methods to determine impacts across 23 individual ecosystem services that fall into four broad categories – provisioning services, regulating services, habitat services, and cultural services. Selected categories are shown for value factors of the US for illustration.
	Other crops - organic	10,681			
	Other crops - sustainable	9,800			
	Bovine, sheep, goats, horses - conventional	11,111			
	Bovine, sheep, goats, horses - organic	10,523			
	Bovine, sheep, goats, horses - sustainable	10,316			
	Forestry	0			
	Paved	17,302			
WiFOR (2023)	Agriculture - Animal rearing	36	USD 2020 per ha of land	The estimates cover: (i) the impact of urban land use on working capacity, (ii) the impact on the effect on drinking water, and (iii) impact from land use on biodiversity costs.	Estimates are build on the Swedish Life Cycle Center (2015) Environmental Priority Strategies (EPS).
	Agriculture - Crops nec	161			
	Forestry	353			
	Paved	5,519			
CE-DELFT (2024)	Upper	690	EUR 2021 per ha of land	The impact from land use is estimated as external costs resulting from the loss of biodiversity due to average land use in the EU27.	Impact pathway and characterisation factors are based on ReCiPe. Values (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 and valuing it over a 50-year period. 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.
	Central	530			
	Lower	370			

Source: Author(s) based on: (IFVI, 2024[52]; WiFOR, 2023[17]; CE Delft, 2024[6]).

108. Figure 3.5 provides a comparative overview between the different value factors in USD. Across the different providers, paved areas and roads seem to have the biggest impact, due to its heat capturing effects and impacts on labour productivity etc. The values available diverge between sources. This may be due to the geographical location of these estimates, as well as the impact pathways and characterisation factors applied.

Figure 3.5. Comparison of value factors for land use



Note: Value factors for IFVI show estimates for the US and are indicative as the database has a global geographical coverage as shown in Table 3.1. OECD (2024) purchasing power parities referred for currency conversion from EUR and GBP into USD. Consumer Price Index used to adjust for inflation to USD 2024 prices.
Source: Author(s) based on: (IFVI, 2024[52]; WiFOR, 2023[17]; CE Delft, 2024[6]).

3.1.6. Biodiversity

109. Biodiversity encompasses both species and ecosystems and underpins economic activities and human well-being. They are critical for sustaining our economic activities and livelihoods through the provision of numerous ecosystem services, such as food and habitat provisioning, clean water provisioning, pollution absorption, nutrient cycling and pollination, and flood protection and carbon storage. More than half of global GDP is highly or moderately dependent on nature and its ecosystem services (WEF, 2020[57]).

110. While biodiversity loss can result in economic damages, they may or may not be considered as a separate impact category, especially as a part of Life Cycle Impact Assessment (LCIA) or Impact Pathway Analysis (IPA) (see Box 3.2). Among the sources for this review, WiFOR (2023[17]) is the only provider that specifies biodiversity as a separate impact category. Other sources such as UBA (2024[39]) cover them as a part of impacts from air pollution, while IFVI (2024[52]) and CE DELFT (2024[6]) cover them as a part of air pollution, water pollution and land use.

Box 3.2. Valuing biodiversity as a part of environmental impacts?

According to the Convention on Biological Diversity, biodiversity refers to the variability of living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part. This variability occurs at three levels: within species, between species, and across ecosystems. The same Convention defines an ecosystem as a dynamic complex of plant, animal, and micro-organism communities and their non-living environment, interacting as a functional unit.

Within the SEEA Ecosystem Accounting (EA) framework, two of the five core accounts are "stock accounts," or physical accounts: ecosystem extent and ecosystem condition. These measure respectively the total area of each ecosystem type (i.e. the quantity) and the physical and chemical quality of those ecosystems. While extent reflects the availability and distribution of habitats, condition provides information on their ecological state. SEEA EA considers a reference condition against which past, present and future ecosystem condition is compared to measure relative change over time, reflecting the optimal endpoint which reflects the highest ecological integrity.

In Life Cycle Impact Assessment (LCIA), biodiversity follows the concept of ecosystem quality configuring as one of the three endpoint Areas of Protection (AoP), as implemented in well-established methods such as ReCiPe 2016 and the EU PEF. The endpoint indicator for quality of ecosystem is expressed in terms of local species loss integrated over time (species·yr). This can be referred to damages to terrestrial or freshwater ecosystems. Such impacts are conventionally assessed against a (semi)-natural reference habitat, such as the expected ecosystem state that would develop if all human activities were to be stopped at once. The relative impact pathways models consider the main sources of impact on biodiversity to be land use, GHG emissions, air pollution via soil acidification and freshwater eutrophication.

In conclusion, biodiversity is generally not treated as a separate LCIA midpoint category; instead, LCIA represents biodiversity loss as an endpoint (ecosystem quality, e.g. species·yr) that is driven by midpoint categories such as land occupation/transformation, eutrophication, water use and climate change - SEEA EA's ecosystem extent aligns with these midpoint drivers, while ecosystem condition aligns with endpoint measures of ecosystem quality.

Source: UNEP, (1992[58]); SEEA (2024[59]); (Huijbregts et al., 2016[7]); JRC (2019[13]).

111. WiFOR (2023[17]) values biodiversity impacts by combining the impacts from air pollution, water pollution and land use, and their respective share of threats to red-listed species, as a proxy indicator for biodiversity loss. The methodology is based on (Steen, 2020[60]), which develops a model to estimate impacts on biodiversity from single human activities. The valuation is based on an abatement cost approach, i.e. cost of preventing biodiversity from declining. Based on Steen (2020[60]), the following physical interventions are considered: (i) air pollution (CO_2, CO, CH_4, N_2O, NH_3, NO_x, SO_x, PM_{10}, NMVOC) measured in terms of the value of pollutants per kg; (ii) water pollution (nitrogen, phosphorus) measured in terms of the value of pollutants per kg; and (iii) land use (agriculture, animal husbandry, forestry and paved areas) measured in terms of the value per square metre of land. Specific value factors for each of these physical interventions and their share that contribute to biodiversity loss (impacts on red-listed species) are provided by WiFOR (2023[17]).

3.1.7. Material resources

112. In the context of this review, existing value factors for material resources are extremely limited. The exception is energy resources (oil, gas and coal), which are provided by CE DELFT (2024[6]). See Table 3.10 for details.

Table 3.10. Value factors for material resources

Source	Category	Value factor	Units	Impacts considered	Remarks
CE DELFT (2024)	Crude oil	0.457	USD 2013 per kg	Changes in extraction cost. Impact pathway and characterisation factors are based on ReCiPe: which identifies change in extraction costs due to consumption of raw materials.	This assumes that the most economically viable raw materials are extracted first, and future generations will be left with only raw materials that are more expensive to extract.
	Coal	0.034	USD 2013 per kg		
	Natural gas	0.301	USD 2013 per Nm3		

Source: Author(s) based on (CE Delft, 2024[6]).

3.2. Sources of heterogeneity

113. A number of factors can be responsible for observed heterogeneity in value factors from different sources, including: (i) discount factors applied (in the case of applying net present values as in GHG emissions), (ii) the valuation methods used (e.g. damage cost approach versus mitigation cost approach as in GHG emissions) (iii) conceptual differences in impact pathways and characterisation factors considered (e.g. ReCiPe, LC-Impact), (iv) conceptual differences in integrated assessment models applied (e.g. whether air pollution only applies to health impacts or cover other damages to buildings and ecosystem services), (v) asymmetries in the coverage of pollutants (especially in the case of air pollution and water pollution, which can be different between the value factor providers), (vi) geographical specifics of the values (e.g. global, national or subnational), (vii) the differences in application of exchange values (market values) and welfare values (non-market values), as well as (viii) divergence in data sources (e.g. sources of Value of Statistical Life).[37]

114. In many cases, the data sources and the methodology used were not available nor transparent in the documents and data sets reviewed. In certain instances, this required delving further into the referenced secondary sources. Enhancing transparency regarding data sources and methodologies would enable further refinement of the underlying approaches and help close existing methodological gaps. For this reason, it is paramount to ensure transparency of sources when producing value factors.[38]

115. Potential areas for further convergence of methodologies may include (i) climate change (GHG), (ii) air pollution, and (iii) water pollution, as similar logic is used across the different value factor providers. In contrast, (iv) water consumption and (v) land use may be of second order priority, due to gaps the underlying methodologies. Finally, (v) biodiversity and (vi) material resources, may take time to develop common methodologies as these are often unconsidered (at the time of writing this report).

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[37] For example, this preliminary stocktake exercise revealed that different VSL sources were sometimes used for health impacts from GHG emissions, air pollution and water pollution, despite coming from the same value factor provider (for details, see Table 3.3, Table 3.4, and Table 3.8). In other cases, the data sources were unavailable. Transparency of data sources can help improve the underlying methodologies and data used for consistency in future applications.

[38] As an example, governance frameworks on valuation are made available by the Capitals Coalition (2025[77]).

3.3. Potential evaluation criteria for identifying ready-to-use value factors

116. Identifying value factors appropriate for use entails a consideration of a number of process- and substance-related criteria. Table 3.11 describes potential criteria that can be used for this purpose.

Table 3.11. Potential evaluation criteria for identifying ready-to-use value factors

		Criteria assessment category		
		Green	Yellow	Red
Data	Impact pathway analyses and revealed preference data	Data is widely used and comprehensive documentation is available	Data is less widely used, but documentation available	No known precedent for data use and no documentation available
	Stated methods data	Primary studies provide evidence that cognitive biases are minimised and statistics indicate the study sample is representative of the population	Primary studies describe efforts taken to minimise cognitive biases and report statistics regarding representativeness are reported	Primary studies provide no evidence of efforts taken to minimise cognitive biases, no statistics regarding representativeness are reported
Value factor calculation	Transparency	All assumptions and justifications provided	Some assumptions provided, but no justifications	No information on assumptions provided
	Methodological robustness and consistency	Impact pathways are well described and reference LCIA frameworks that are well-cited in peer-reviewed literature; consistent approaches and data across value factors	Some impact pathway details missing and/or LCIA frameworks are not referenced or inconsistent; some consistency in approaches and data across value factors	Some impact pathway details missing and/or LCIA frameworks are not referenced; inconsistent approaches and data across value factors
	Scientific uncertainty	Ranges for value factors are provided and explained; sensitivity analyses are reported	Ranges for value factors are provided but not well explained; sensitivity analyses are not reported	Ranges for value factors are not provided; sensitivity analyses are not reported
	Additivity within or across impact categories	Double-counting is minimized methodologically and/or implications acknowledged	Double-counting is not methodologically addressed and/or possible implications not acknowledged	Double-counting is not addressed and possible implications not acknowledged
	Flexibility to accommodate different data inputs	Detailed value factors (or formula to customise them) are available to match different data sources with finer (spatial) resolution	Detailed value factors (or formula to customise them) are partially available to match different data sources with finer (spatial) resolution	Detailed value factors (or a formula to customise them) are not available to match different data sources with finer (spatial) resolution
Value factor use	Precedent	Multiple known use cases, including in public sector and academia	Some known use cases, e.g. in private sector; limited precedent of use in public sector	Few known use cases, no precedent for use in public sector or academia
	Generalisability	Value factors require few adjustments to apply to other contexts	Average values require some adjustments to apply to other contexts	Average values are highly context specific and cannot be applied in other contexts

Note: Potential synergies with the Governance of Valuation of 53 criteria regarding transparency requirements, confidence criteria, and value notes in development by the Value Commission of the Capitals Coalition (2025[61]) have not been explored.

3.4. Ready to use value factors

[For further development following the ENIVADE Module 2 Technical Workshop]

117. This section will identify value factors that the OECD considers suitable for well-considered use in relevant contexts based on the final set of evaluation criteria developed in Section 3.3.

3.5. Transferring value factors across use cases

[For further development following the ENIVADE Module 2 Technical Workshop]

118. As detailed in Sections 2.2 and 2.3, the valuation of environmental and health impacts can be highly context-specific due to biophysical processes, population characteristics and economic conditions. It should be noted that although these factors can be taken into account to some extent, transferability across regions, sectors or contexts may nevertheless remain limited or inadvisable in certain cases. Midpoint value factors, for example, are generally considered not transferrable from one characterisation methodology to another (CE Delft, 2024[6]).

119. Transferring emissions-level value factors to other contexts depend highly on the characteristics of the value factor being transferred (e.g. impact category, characterisation factors used in impact pathway analysis, valuation assumptions made regarding discounting, equity weighting, etc.) and the context in which it is being transferred to – in particular the extent to which it is characterised by the same features as that of the original value factor. For certain impact categories, such as climate change, values can be considered relatively transferable insofar as the impact pathways are shared across geographies. For environmental pressures other than GHG emissions, it has been recommended for example to adjust value factors for income differences across contexts to reflect expected variations in willingness to pay.

120. Depending on the environmental pressure in question, other factors may be important in determining variation in damage costs (e.g. the type and location of the air pollution emissions in relation to populations and ecosystems) (CE Delft, 2024[6]). For this reason, meaningful transfer of value factors to other contexts depends on the specificity of the original value factor and the data availability and methodological capability to adjust for the specificities of the context in question.[39]

3.6. Knowledge gaps

[For further development following the ENIVADE Module 2 Technical Workshop]

121. Several knowledge gaps remain with respect to the calculation and use of value factors, including:

- Identifying minimum evaluation criteria requirements that value factors should satisfy in order to be considered appropriate for use
- Expanding value factors to additional environmental pressures (e.g. the bioaccumulation of chemicals such as PFAs (CE Delft, 2024[6]))
- Establishing common practices to represent uncertainty in value factors, to reflect their sensitivity to various data and modelling choices
- Better understanding to what extent value factors can vary across geographies and being able to account for such variation through value transfer
- Managing sectoral heterogeneity with single estimates, e.g. whether national estimates suffice or finer spatial resolutions are necessary for specific environmental pressures or impacts
- Better understanding how value factors may change over time due to changing pressures, prices and preferences

[39] Some general guidelines for adjusting value factors (or environmental prices) across locations and substances are provided in CE Delft (2024[6]).

4. Use of OECD existing valuation methodologies for health and environmental endpoints

122. OECD has performed extensive work on economic valuation of health effects, including both mortality effects (value of statistical life) and morbidity effects associated with chemicals that examine the willingness to pay to avoid chemicals-related health effects. In addition, work on developing new estimates of willingness to pay to avoid chemicals related environmental endpoints is in progress. This suite of internationally comparable and transferrable valuations for health and environmental endpoints could be an important source of endpoint valuations for input into the methodologies and value factors for the environmental pressures that are considered in the ENIVADE project. This section provides a brief summary of each of the valuation work streams in OECD that could support the project.

4.1. Value of Statistical Life (VSL)

123. The "Value of Statistical Life" (VSL) is a concept that has for decades helped policymakers to assess the costs and benefits of any policy that involves changes in mortality risk, such as air pollution regulations. In 2012, the OECD produced a seminal report that included the largest VSL meta-analysis study ever conducted. Significant advances have been made since the publication of that report and the evidence base on valuation of mortality effects has grown substantially. Despite this evolution, no globally consistent valuation updates have been developed. As a result, practitioners have grappled with applying earlier estimates in the face of fragmented research, evolving circumstances and a global pandemic. OECD's 2025 report Mortality Risk Valuation in Policy Assessment: A Global Meta-Analysis of Value of Statistical Life Studies (OECD 2025) provides the first comprehensive global review of the VSL since the 2012 OECD report. This new report includes a meta-analysis of nearly 4 000 individual estimates from 280 studies across 49 countries between 1970-2023, from which about 2 400 VSL estimates between 2009-2023 were used for the base VSL estimates. The current report is once again the largest meta-analysis on VSL ever attempted and is the first to develop VSL estimates based on both revealed and stated preference methodologies. While most of the primary VSL studies in the database are from Europe and North America, it also includes numerous studies from Latin America, Africa, Australia and Asia.

124. The report provides base VSL estimates (mean, calculated median values, and confidence bands) for six country groups: OECD Member Countries, European Union, United States, Low-and middle-income countries, High-income countries and a Global estimate. The estimated mean base VSL estimates range from about USD 1 million in low- and middle-income countries to USD 7.1 million and USD 7.9 million for OECD and high-income countries, respectively.

4.2. Valuation of morbidity for chemicals-related health endpoints

125. Chemicals are part of our daily life and must be soundly managed to limit risks to human health and the environment. While countries around the world are setting up legal frameworks to address these risks, the cost of policy inaction is still poorly understood. Assessment of chemicals management options and environmental policies can be considerably improved by better estimating their costs and benefits. The resourcing of national chemicals management programmes also often requires economic justification of the benefits of such investment. However, current socio-economic analyses of chemical regulations use values for morbidity impacts that are often incomplete. In most cases, these values cover only lost productivity, lost earning or cost of illness and disregard the disutility costs of pain and suffering from the illnesses.

126. The OECD project Surveys on Willingness to Pay to Avoid Negative Chemicals-Related Health Impacts (SWACHE) brings together expertise on chemical safety and economic analysis to fill this gap. The project aims to establish internationally comparable values for the willingness-to-pay (WTP) to avoid negative health effects due to exposure to chemicals. Such values can be used to demonstrate and measure the economic benefits of minimising the impacts of chemicals on human health. Moreover, by using similar methodologies, survey design, approach to analyse survey data across many health impacts and implementing the surveys in parallel in a large number of countries, the SWACHE project offers a unique perspective that make it easier to compare the value of health impacts across health outcomes as well as across countries.

127. The only way to capture the full WTP to avoid illness is to conduct a stated-preference study, i.e., surveys where individuals are asked to report their WTP to reduce their risk of negative health impacts due to chemicals exposure. Contingent valuation methods and discrete choice experiments do just that, and WTP figures based on these methods have been used in assessment efforts. To derive WTP values, surveys of a large number of citizens of countries have therefore been conducted under the SWACHE project. Particularly, these stated preference surveys provide data that can shed light on the disutility in terms of symptoms and lower quality of life of a given disease or health effect, which is not captured by existing metrics such as those based on the cost of illness.

128. The SWACHE project thus far is organised in two rounds, each focusing on 5 health effects each. The first round of health effects included asthma, fertility loss, IQ loss, serious kidney disease and very low birth weight. The first round of surveys was implemented in 2022 in at least five countries each where representative samples of at least 1 200 respondents each were collected. Overall, one to five of the surveys were implemented in 22 countries, totalling 46 surveys conducted. Survey responses were empirically analysed to estimate mean WTP for a given reduction in health risk for each country surveyed. The second round of surveys includes hypertension, miscarriage, skin sensitisation, thyroid dysfunction and non-fatal cancer, and was implemented between 2023 and 2025. Overall, in the second round, one to five surveys were implemented in 19 countries, totally 50 surveys conducted. In total, across the 2 rounds of surveys, 96 surveys were executed covering more than 110 000 respondents.

129. SWACHE results are presented in working papers, one for each health effect. The research described in individual working papers makes a variety of empirical contributions to health valuation in the context of chemicals exposure, although, by design, the approach was not to break new conceptual, theoretical, or econometric ground. Moreover, the comparison of the estimated WTP across health effects and across countries is being carried out in a separate summary paper, which will also provide guidance for the transfer of WTP value over time and to non-surveyed countries. The first five working papers were published in 2023.

4.3. Valuation of chemicals-related environmental endpoints

130. The Surveys to elicit willingness to pay to Avoid Chemicals Related Environmental effects (SACRE) project aims to develop new methodologies for economic valuation of chemicals-related environmental impacts that can be directly applied in regulatory assessments and benefit cost analysis. The project will facilitate the estimation of internationally comparable willingness-to-pay values for the reduction in the risk of adverse environmental impacts due to chemicals.

131. An initial step in the project is to properly define environmental endpoints and their attributes that the methodology will be developed around. A survey instrument is currently under development which focuses on the Potentially Disappeared Fraction (PDF) of species as a measure of the health of the ecosystems that are subject to valuation. Two general ecosystems are targeted for testing of the survey instrument – one aquatic freshwater ecosystem and one terrestrial ecosystem. The survey instrument will be tested in pilot studies in two to three countries, following which full-scale surveys are foreseen in OECD countries over the next few years. The endpoint valuations resulting from this effort are also expected to be highly applicable for the ENIVADE project.

4.4. Knowledge gaps and future work

This section will identify current knowledge gaps and outline further necessary research, for example, with respect to natural capital accounting:

- The need for reliable measures of the extent and condition of environmental assets
- Medium-term planning for data and research to support the improvement and timeliness of natural capital accounts (e.g. land cover data)
- Datasets that are partial or have insufficient spatial resolution to measure actual ecosystem services (such as reduction in flood risk)
- Frequency of updating datasets and a consistent time series to identify trends

5. Knowledge gaps and next steps

5.1. Cross-cutting challenges

[For further development following the ENIVADE Module 2 Technical Workshop]

132. As described in Sections 2.2.7, 2.3.4, and 0, significant knowledge gaps remain with respect to assessing how environmental pressures result in impacts to human health and the environment, how such impacts can be measured, and the calculation and use of value factors linking environmental pressures to monetary impacts at the emissions level. In addition to area-specific challenges, a number of cross-cutting challenges characterize efforts to incorporate environmental and health impacts into public and private decision making, including:

- Ensuring transparency in underlying data and methodologies
- Clarifying definitions and terminology across different communities
- Evaluating counterfactual scenarios and factoring in unforeseen costs
- Downscaling macro-level data from MRIOs or CGE to the corporate level to fill gaps in supply-chain data
- Accounting for planetary boundaries
- Accounting for future impacts

5.2. Next steps for the ENIVADE Project

[For further development following the ENIVADE Module 2 Technical Workshop]

133. Based on the outcomes of the workshop and in particular on the elaboration of knowledge gaps, this section will prioritise next steps for Module 2 of the ENIVADE Project.

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6. Annex

6.1. Midpoint vs. endpoint indicators

Compared to midpoint, endpoint indicators are sometimes considered to lead to more understandable results for decision makers and society carrying a higher environmental relevance of the environmental flows (e.g. speaking in terms biodiversity loss or human health outcomes) (Bare, 2000[5]; Huijbregts et al., 2016[5]). Nevertheless, midpoint indicators that can be crucial to support policy decisions. IPCC, 2021[6], for instance, suggests that the Global Warming Potential (GWP)[40] midpoint characterization factor is a useful metric to employ in the context of multi-component policies to prioritize which emissions to abate. Moreover, midpoint measures may be preferred for specific communication purposes and for reducing the risk of unwarranted conclusions in communicating scientific results. Finally, in cases where aggregation is desired, endpoint indicators may be preferred, for example to compare human health impacts associated to climate change with those of air pollution using a common basis such as DALYs (Disabilty Adjusted Life Years) (Bare, 2000[62]). Table 2.1 summarizes the main differences among the two indicators.

Table 6.1 Comparison of midpoint and endpoint indicators

Aspect	Midpoint Indicators (mCFs)	Endpoint Indicators (eCFs)
Position in the pathway	Intermediate stage	Final stage (areas of protection)
What they measure	Potential impact within a category (e.g. kg CO_2-eq, kg SO_2-eq)	Actual damage to areas of protection (e.g. DALYs, species lost, resource scarcity)
Purpose	Make substances comparable within an impact category	Compare damages across categories on a common scale
Certainty	Higher (closer to physical emissions, less modelling)	Lower (more modelling steps, higher uncertainty)
Environmental relevance	Indirect (technical, category-focused)	Direct (health, ecosystems quality, resource scarcity)
Use in policy decisions	Yes (e.g. multi-component policies to prioritize emissions to abate)	Yes (e.g. comparison of human health effects from different sources)
Examples	Global Warming Potential (GWP) (CO_2-eq), Acidification Potential (SO_2-eq), $PM_{2.5}$ formation eq	Climate change damages in DALYs, species extinction risk, additional cost of resource use

Source: Authors' elaboration based on Bare, 2000[5] and ReCiPe 2016 (Huijbregts et al., 2016[6]).

These conceptual differences between midpoint and endpoint modelling have been the subject of international debate. On May 25-26, 2000 in Brighton (England), the third in a series of international workshops was held under the umbrella of UNEP addressing issues in Life Cycle Impact Assessment (LCIA). Experts discussed the respective advantages and limitations of midpoint and endpoint indicators, focusing on issues of uncertainty, transparency and comparability across impact categories. The workshop concluded that both approaches are useful for decision-making and highlighted the need for tools that integrate midpoint and endpoint indicators within a consistent framework (Bare, 2000[62]).

[40] The GWP is a midpoint measure of global warming in the context of impacts to human health and ecosystems.

Table 2.2 lists midpoint categories and their related indicators, as well as the reference units used in LCIA framework to express impacts in substance-equivalence terms within the same impact category.

Table 6.2 Overview of midpoint categories and related indicators

Midpoint category	Indicator	Unit	Reference substance Unit
Climate Change	Infra-red radiative forcing increase	W × yr/m^2	kg CO_2 to air
Fine Particulate Matter formation	$PM_{2.5}$ population intake increase	kg	kg $PM_{2.5}$ to air
Terrestrial acidification	Proton increase in natural soils	Yr × m^2 × mo l/l	kg SO_2 to air
Freshwater Eutrophication	Phosphorus increase in fresh water	Yr × m^3	kg P to fresh water
Water Use	Increase of water consumed	m^3	m^3 water consumed
Land Use	Occupation and time-integrated transformation	Yr × m^2	m^2 × yr annual crop land
Mineral resource scarcity	Ore grade decrease	kg	kg Cu

Note: The table reports the six midpoint categories from ReCiPe 2016 considered in this report, along with their corresponding indicators and measurement units. For each midpoint, the reference substance and its unit are also indicated. The reference units are used in the LCIA framework to convert different impacts into substance-equivalents, ensuring comparability and allowing aggregation within the same impact category.
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

Table 2.3 illustrates the three Areas of Protection (i.e. areas of impacts at the endpoint level) with indication of their indicator and unit of measure.

Table 6.3 Overview of endpoint categories and related indicators

Endpoint category	Indicator	Unit
Human health	DALY (Disability Adjusted Life Years)	Year
Ecosystem quality	Time-integrated species loss	Species × yr
Resource scarcity	Surplus cost	Dollar

Note: Table shows the three endpoint categories human health, ecosystem quality and resource scarcity considered from ReCiPe 2016 considere in this report and their related indicators and measurement unit.
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7]).

Table 2.4 presents the relationships between the selected midpoint categories and their corresponding endpoints following ReCiPe 2016 impact pathways. It illustrates which midpoint mechanisms are linked to each of the three Areas of Protection (AoP) at the endpoint level, human health, ecosystem quality, and resource scarcity.

Table 6.4 Relationships between midpoint and endpoints categories

Midpoint \ Endpoint	Human health	Ecosystem quality	Resource scarcity
Climate Change	✓	✓	
$PM_{2.5}$ formation	✓		
Terrestrial acidification		✓	
Water use	✓	✓	
Water pollution		✓	
Land use		✓	
Mineral resource scarcity			✓

Note: Table shows the six midpoint impact categories considered in this report and their impact on the three endpoint categories human health, ecosystem quality and resource scarcity. Check mark in brackets indicates if the midpoint category eventually results in a specific endpoint category in the ReCiPe 2016 impact pathways modelling.
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

6.2. Life cycle impact assessment methods

Table 6.5. Existing LCIA methodologies and potential sources of heterogeneity including midpoint and endpoint level impact categories

Method	Developer	Last version	Type of method	Midpoint level impact categories	Endpoint level impact categories	Geographic Scope	Used in value factors reports
"Carbon Footprint" – IPCC	Intergovernmental Panel on Climate Change (UN)	2021, Regular reports	Midpoint method	- **Climate change** (Global Warming Potential (GWP) and Global Temperature Change Potential (GTP), kg of CO_2-eq)	n/a	Global	Yes
CML Method	University of Leiden	2016 (CML Handbook on LCA not available)	Midpoint method	**- Climate change** (Global Warming Potential (GWP), kg of CO_2-eq) - **Ozone depleting substances** Photochemical ozone creation potential (POCP), kg of CFC-11-eq - **Human toxicity (**Human toxicity potential (HTP), kg of dichlorobenzene-eq) - **Ecotoxicity** (Freshwater aquatic ecotoxicity potential (FAETP), Marine aquatic ecotoxicity potential (MAETP), Terrestrial ecotoxicity potential (TETP), Freshwater sediment ecotoxicity potential (FSETP), Marine sediment ecotoxicity potential (MSETP), kg of dichlorobenzene-eq) - **Photo-oxidant formation** (Photochemical ozone creation potential (POCP)) - **Acidification** (acidification potential (AP), kg of SO_2-eq) - **Water pollution** (eutrophication potential (EP), kg of PO_4-eq) - **Material resources** (Abiotic depletion potential (ADP), kg antimony-eq) - **Land use** (Land occupation, m^2xyr)	n/a	Global	No
IMPACT World+	CIRAIG, University of Michigan, Quantis International,	2019	Midpoint and Endpoint method	- **Climate change** (kg CO_2-eq) - **Ozone layer depletion** (kg CFC-eq) - **Mineral resource use** (kg) - **Terrestrial acidification** (kg SO_2-eq)	- **Climate change, human health** (DALY/kg emitted) - **Climate change, ecosystem quality** (PDF × m^2 × yr/kg emitted)		

	Technical University of Denmark (DTU), and école Polytechnique de Lausanne (EPFL).			- **Freshwater acidification** (kg SO_2-eq) - **Marine eutrophication** (kg N-eq) - **Freshwater eutrophication** (kg of P_4-eq) - **Freshwater ecotoxicity** (Comparative Toxic Unit, ecosystems (CTUe)) - **Human toxicity (cancer, non-cancer)** (Comparative Toxic Unit, human (CTUh)) - **Particulate matter formation** (kg of $PM_{2.5}$ emitted) - **Photochemical oxidant formation** (kg of NMVOC-eq) - **Ionizing radiation** (Bq C-14-eq) - **Water scarcity** (m^3) - **Land transformation** (m^2) - **Land occupation** ($m^2 \times yr$)	- **Climate change, resource & ecosystem services** ($) - **Ozone layer depletion** (DALY/kg emitted) - **Mineral resource use** ($/kg) - **Terrestrial acidification** (PDF × m^2 × yr/kg emitted) - **Freshwater acidification** (PDF × m^2 × yr/kg emitted) - **Marine eutrophication** (PDF × m^2 × yr/kg emitted) - **Freshwater eutrophication** (PDF × m^2 × yr/kg emitted) - **Freshwater ecotoxicity** (PDF × m^2 × yr/kg emitted) - **Terrestrial ecotoxicity** (PDF × m^2 × yr/kg emitted) - **Marine ecotoxicity** (PDF × m^2 × yr/kg emitted) - **Human toxicity(cancer, non-cancer**) (DALY/kg emitted) - **Particulate matter formation** (DALY/kg emitted) - **Photochemical oxidant formation**(DALY/kg emitted) - **Ionizing radiation human health** (DALY/kg emitted) - **Ionizing radiation ecosystem quality** (PDF × m^2 × yr/kg emitted) - **Water availability, human health** (DALY/m^3 consumed) - **Water availability, freshwater ecosystem** (PDF × m^2 × yr/m^3 dissipated) - **Water availability, terrestrial ecosystem** (PDF × m^2 × yr/m^3 dissipated) - **Land transformation, biodiversity** (PDF × m^2 × yr/m^2 transformed) - **Land occupation, biodiversity** (PDF × m^2 × yr/m^2 occupied × yr) - **Mineral resource use** ($/kg dissipated)	Global	No

LC-Impact	Norwegian University of Science and Technology (NTNU), PRé Sustainability B.V., Radboud University Nijmegen ETH Zurich, (DTU, Denmark	2020	Endpoint method	n/a	- **Climate change, human health** (DALY/kg emitted) - **Climate change, ecosystem quality** (PDF × m^2 × yr/kg emitted) - **Climate change, resource & ecosystem services** ($) - **Ozone layer depletion** (DALY/kg emitted) - **Mineral resource use** ($/kg) - **Terrestrial acidification** (PDF × m^2 × yr/kg emitted) - **Freshwater acidification** (PDF × m^2 × yr/kg emitted) - **Marine eutrophication** (PDF × m^2 × yr/kg emitted) - **Freshwater eutrophication** (PDF × m^2 × yr/kg emitted) - **Freshwater ecotoxicity** (PDF × m^2 × yr/kg emitted) - **Human toxicity(cancer, non-cancer)** (DALY/kg emitted) - **Particulate matter formation** (DALY/kg emitted) - **Photochemical oxidant formation** (DALY/kg emitted) - **Ionizing radiation human health** (DALY/kg emitted) - **Ionizing radiation ecosystem quality** (PDF × m^2 × yr/kg emitted) - **Water availability, human health** (DALY/m^3 consumed) - **Water availability, freshwater ecosystem** (PDF × m^2 × yr/m^3 dissipated) - **Water availability, terrestrial ecosystem** (PDF × m^2 × yr/m^3 dissipated) - **Land transformation, biodiversity** (PDF × m^2 × yr/m^2 transformed) - **Land occupation, biodiversity** (PDF × m^2 × yr/m^2 occupied × yr)	Global	Yes

Product Environmental Footprint (PEF)	European Commission, JRC	2022	Midpoint method	- **Climate change** (Global Warming Potential (GWP), kg CO_2-eq) - **Ozone depletion** (Ozone depleting potential (ODP), kg of CFC-11-eq) - **Freshwater ecotoxicity** (Comparative Toxic Unit, ecosystems (CTUe)) - **EF-Particulate matter** (Impact on human health, kg of $PM_{2.5}$ emitted) - **Marine eutrophication** (Fraction of nutrients reaching marine and compartment (N)) - **Freshwater eutrophication** (Fraction of nutrients reaching freshwater and compartment (P)) - **Terrestrial eutrophication** (Accumulated Exceedance (AE)) **- Acidification** (Accumulated Exceedance (AE) mol H+ eq) - **Human toxicity (cancer, cancer inorganics, cancer metals, cancer organics, non-cancer, non-cancer inorganics, non-cancer metals, non-cancer organics)** (CTUh) - **Ionising radiation, human health** (Human exposure efficiency relative to U235) - **Photochemical ozone formation** (Tropospheric ozone concentration increase) - **Land use** (Soil quality index) - **Resource use, minerals and metals** (Abiotic resource depletion (ARD)) - **Water use** (User deprivation potential)	n/a	EU	No
ReCiPe	National Institute for Public Health and the Environment, Radboud University Nijmegen, Leiden University, and	2016	Midpoint and Endpoint method	- **Climate change** (Global Warming Potential (GWP), kg CO_2-eq) - **Ozone depletion** (Ozone depleting potential (ODP), kg CFC-11-eq) - **Ionizing radiation** (Ionizing Radiation Potential (IRP), kBq Co-60-eq) - **Photochemical ozone formation** (Photochemical ozone creation potential (POFP), kg NO_x-eq) - **Fine particulate matter formation** (Particular	- **Climate change, human health** (DALY/kg emitted) - **Climate change, ecosystem quality** (PDF × m^2 × yr/kg emitted) - **Ozone layer depletion** (DALY/kg emitted) - **Mineral resource use** ($/kg) - **Terrestrial acidification** (PDF × m^2 × yr/kg emitted)	Global	Yes

				matter formation potential (PMFP), kg $PM_{2.5}$-eq) - **Terrestrial acidification** (Terrestrial Acidification Potential (TAP), kg SO_2-eq) - **Freshwater eutrophication** (Eutrophication Potential (FEP), kg P-eq) - **Marine eutrophication** (Eutrophication Potential (MEP), kg N-eq) - **Human toxicity (cancer)** (Human Toxicity Potential (HTPc), kg 1,4-DCB-eq) - **Human toxicity (non-cancer)** (Human Toxicity Potential (HTPnc), kg 1,4-DCB-eq) - **Terrestrial ecotoxicity** (Terrestrial Ecotoxicity Potential (TETP), kg 1,4-DCB-eq) - **Freshwater ecotoxicity** (Freshwater Ecotoxicity Potential (FETP), kg 1,4-DCB-eq) - **Marine ecotoxicity** (Marine Ecotoxicity Potential (METP), kg 1,4-DCB-eq) - **Land use** (Land use occupation potential (LOP), m^2a crop-eq) - **Water consumption** (Water consumption Potential (WCP), m^3) - **Mineral resource scarcity** (Mineral resource scarcity potential (MSP), kg Cu-eq) - **Fossil resource scarcity** (Fossil resource scarcity potential (FFP), kg oil-eq)	- **Freshwater acidification** (PDF × m^2 × yr/kg emitted) - **Marine eutrophication** (PDF × m^2 × yr/kg emitted) - **Freshwater eutrophication** (PDF × m^2 × yr/kg emitted) - **Freshwater ecotoxicity** (PDF × m^2 × yr/kg emitted) - **Human toxicity (cancer, non-cancer)** (DALY/kg emitted) - **Particulate matter formation** (DALY/kg emitted) - **Photochemical oxidant formation** (DALY/kg emitted) - **Ionizing radiation human health (**DALY/kg emitted) - **Ionizing radiation ecosystem quality** (PDF × m^2 × yr/kg emitted) - **Water availability, human health** (DALY/m^3 consumed) - **Water availability, freshwater ecosystem** (PDF × m^2 × yr/m^3 dissipated) - **Water availability, terrestrial ecosystem** (PDF × m^2 × yr/m^3 dissipated) - **Land transformation, biodiversity** (PDF × m^2 × yr/m^2 transformed) - **Land occupation, biodiversity** (PDF × m^2 × yr/m^2 occupied × yr)		
TRACI	U.S. Environmental Protection Agency (US EPA)	2012	Midpoint method (Chemicalfocus)	- **Climate change** (Global Warming Potential (GWP)) - **Ozone depletion** (Ozone depletion potential (ODP)) - **Acidification Air** Acidification potential (AP), kg SO_2-eq - **Freshwater ecotoxicity** Comparative Toxic Unit (CTU) - **Human health particulate** Particular matter formation potential (PMFP), $PM_{2.5}$-eq	n/a	Global	No

USEtox	UNEP	2017	Midpoint and Endpoint method	- **Ecotoxicity** (Comparative Toxic Unit, ecosystems (CTUe)) - **Human toxicity** (Comparative Toxic Unit, human health (CTUh))	- **Ecotoxicity** (PDF × m^3 × day per kg emitted) - **Human toxicity** (DALY × kg emitted)	Global	Yes

6.3. Impact pathways

In LCIA, an impact pathway represents the environmental cause-effect chain that follows a substance from its emission or resource use through the resulting environmental changes, midpoint impacts, toward the final endpoints, such as damages to human health, ecosystem quality, or resource availability. Mapping these pathways allows the assessment of how different human activities contribute to overall environmental damages.

Environmental impact pathways can be framed as going from pressures, to state to impacts: pressure is defined as a source of environmental change exerting stress on the environment (emissions, land transformation, water consumption); state is the resulting changes in the environment (air quality, water scarcity, biodiversity condition); impact is the effects on ecosystems, human health, or resources (disease, crop yield loss, species extinction) (European Environment Agency, 1999[4]).

In this overview of impact pathways, the ReCiPe 2016 method (Huijbregts et al., 2016[7]) is used as the methodological anchor, as it is well-documented, widely cited, and provides explicit midpoint and endpoint pathways across many impact categories. In addition, it has been directly applied in CE Delft *Environmental Prices Handbook* calculations (CE Delft, 2024[6]), which further supports its relevance for the present analysis. The LCIA method ReCiPe 2016 classifies impacts into 17 midpoint categories[41], each representing an environmental mechanism or process to which substances contribute. Substances are grouped according to the mechanism they influence, for example, climate change (i.e. increase in global mean temperature), particulate matter (PM) formation, or water use, reflecting their interactions with ecosystems or human health. Each category is linked to a reference substance, such as CO_2 for climate change, which provides a common basis for expressing the impacts of different substances within the same category, allowing for comparisons and aggregations.

In this report, the analysis focuses exclusively on the following environmental domains: climate change, fine particulate matter formation (i.e. air pollution), water use, freshwater eutrophication (i.e. water pollution), land use, and mineral resource scarcity.

Moreover, ReCiPe 2016 considers three Areas of Protection (AoP) to which the single endpoints can be attributed[42]. Those are: human health, ecosystem quality and resource scarcity. DALYs (disability adjusted life years), relevant for human health, represent the years that are lost or that a person is disabled due to a disease or accident. The unit for ecosystem quality is the local species loss integrated over time (species year). Finally resource scarcity is measured in dollars ($), representing the extra costs involved for future mineral and fossil resource extraction.

The following sections describe in more detail each of the impact categories considered in this report, beginning with Climate Change. For each category, following an introduction to the environmental topic

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[41] Climate Change, Ozone Depletion, Ionizing Radiation (Human Health), Fine Particulate Matter Formation, Photochemical Oxidant Formation (Ecosystem Quality), Photochemical Oxidant Formation (Human Health), Terrestrial Acidification, Freshwater Eutrophication, Human Toxicity (Cancer), Human Toxicity (Non-Cancer), Terrestrial Ecotoxicity, Freshwater Ecotoxicity, Marine ecotoxicity, Land use, Water use, Mineral resource scarcity, Fossil resource scarcity.

[42] The Areas of Protection (AoPs) represent the broad environmental, health, or resource systems that LCIA considers, while endpoints are the specific quantitative indicators (e.g., DALYs, species·year, resource depletion) used to measure potential damage within each AoP. In other words, endpoints operationalize the AoPs.

and its economic and social relevance, the impact pathways will be illustrated.

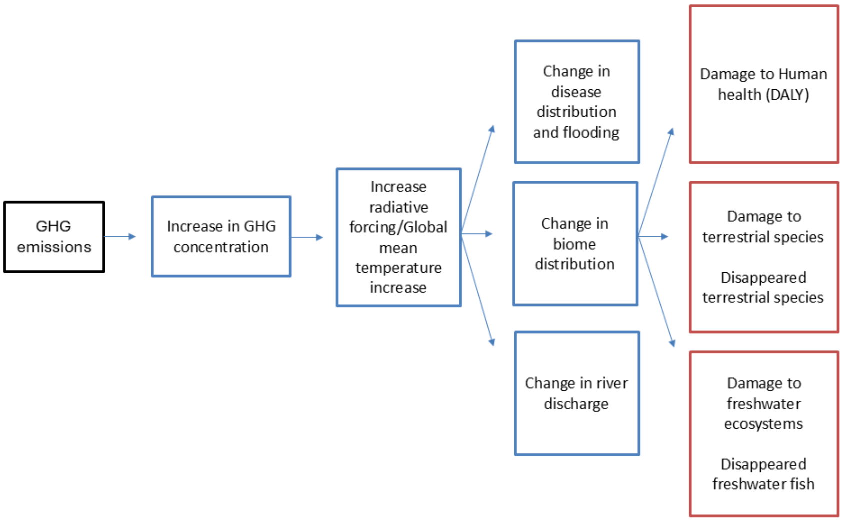
Climate Change

Climate change describes the gradual change in weather patterns, such as temperature and rainfall, over long-time spans. The term is usually used to refer to *anthropogenic climate change* even though the phenomenon can also have natural causes. Climate change is currently occurring due to an increasing concentration of greenhouse gases (GHGs) in the atmosphere due to human activity, a phenomenon known as the greenhouse effect that causes, among other things, global temperatures to rise. In the coming decades, this increase is expected to continue according to different emissions scenarios developed by the IPCC 2021 (IPCC, 2021[10]). Such temperature rise will have a major impact on humans, animals and ecosystems. The largest source of GHG emissions are fossil fuels. They stem mainly from energy production, with the combustion of coal, oil, and gas releasing carbon dioxide (CO_2), agriculture and waste, releasing methane (CH_4) and nitrous oxide (N_2O). Industrial processes add fluorinated gases including HFCs, PFCs, SF_6, and NF_3. According to the IPCC 2021 report (IPCC, 2021[10]), these sectors remain the dominant global sources of emissions.

Impact pathway

Fig. 2.1 shows the impact pathway for climate change, as modeled in ReCiPe 2016. GHGs emissions ultimately reflect in impacts on human health, terrestrial ecosystems, and freshwater ecosystems. The emission of GHGs (i.e. pressure) increases their atmospheric concentration (i.e. state), which enhances radiative forcing and drives a rise in global mean temperature. The midpoint indicator is the cumulative increase in infrared radiative forcing. Higher temperatures alter disease distribution, raising the probability of parasitic diseases such as malaria and dengue fever, and increase flooding frequency, together resulting in damage to human health. Rising temperatures also shift biome distribution, for example by expanding the range of pests and plant diseases or by intensifying drought and heat stress, which accelerates the disappearance of terrestrial species. In parallel, changing rainfall patterns and prolonged droughts alter river discharge, reducing freshwater availability and leading to the loss of freshwater fish populations.

Figure 6.1. Cause-effect chain for impact category Climate Change



Note: Impact pathway from GHGs emissions to damages to human health, ecosystem quality and resource availability.
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

Fine Particulate Matter formation (Air pollution)

Particulate matter (PM) is a major component of air pollution, consisting of a mixture of liquid and solid particles suspended in the atmosphere, collectively known as aerosols. Anthropogenic emissions include soot and smoke from combustion processes as well as dust from building materials. PM is usually classified into fractions by size: PM_{10}, $PM_{2.5}$, and PM_1, referring to particles with a diameter smaller than 10, 2.5, and 1 µm, respectively. WHO studies (WHO, 2013[24]) show that the mortality effects of chronic exposure are largely attributable to fine particles, particularly $PM_{2.5}$, which penetrate deeply into the lungs and bloodstream. $PM_{2.5}$ is also more directly linked to anthropogenic sources than PM_{10}, making it the most policy-relevant fraction.

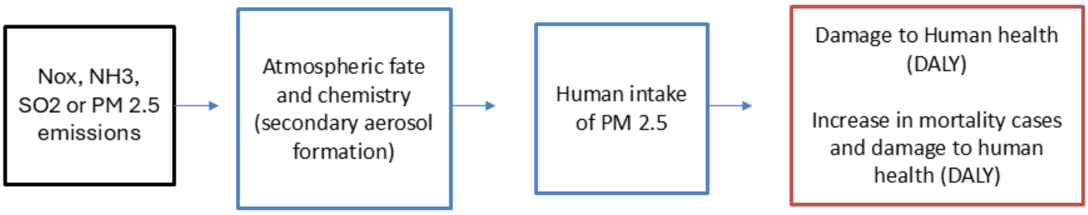
PM can be directly emitted as *primary particles* or formed in the atmosphere as *secondary aerosols* through reactions involving ammonia (NH_3), sulfur dioxide (SO_2), nitrogen oxides (NO_x), and organic compounds. Secondary aerosols, such as sulfates and nitrates, can travel further than primary particles and are influenced by meteorological conditions including wind, humidity, and temperature. Scientific debate remains regarding the relative toxicity of primary versus secondary inorganic aerosols (SIA), and whether secondary organic aerosols (SOA) contribute to comparable health damages. While ReCiPe 2016 assigns no burden to SOA, more recent studies (EEA, 2021a [9]; CE Delft, 2024[1]) suggest including SOA in damage cost assessments.

Impact pathway

The modelling of air pollution impacts in ReCiPe 2016 follows a chain of processes from emission to health damage as illustrated in Figure 2.2. Emissions of NO_x, NH_3, SO_2, or primary $PM_{2.5}$ enter the atmosphere, where they undergo dispersion and chemical transformation (i.e. state). Gaseous precursors form secondary aerosols, which, together with primary particles, increase ambient concentrations of $PM_{2.5}$.

These fine particles are inhaled by the population, leading to increased mortality and morbidity, and ultimately to measurable impacts on human health.[43]

Figure 6.2. Cause-effect chain for impact category Fine Particular Matter formation



Note: Impact pathway from fine dust formatting emissions to damage to human health
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

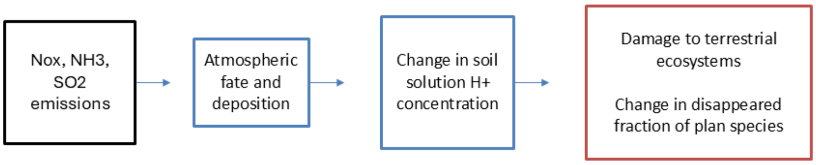
Terrestrial acidification (Air pollution)

The release and subsequent atmospheric deposition of inorganic compounds such as sulfates, nitrates, and phosphates can alter the acidity of soils. Most plant species thrive only within a specific pH range, and significant deviations from this optimal level can be harmful to their growth, this is process is known as acidification. When soil acidity changes, it can therefore lead to shifts in which plant species are able to survive or dominate in an area. The main contributors to acidifying emissions are nitrogen oxides (NO_x), ammonia (NH_3), and sulfur dioxide (SO_2).

Impact pathway

An emission of NO_x, NH_3 or SO_2 is followed by atmospheric transformation and transport processes before being deposited onto the soil surface. Once deposited, these substances leach into the soil, altering the concentration of hydrogen ions (H^+) in the soil solution. The resulting change in soil acidity negatively affects plant species, leading to their loss.

Figure 6.3 Cause-effect chain for impact category Terrestrial acidification



Note: Impact pathway from acidifying emissions to damages to ecosystem quality.

[43] According to (WHO, 2013[24]), $PM_{2.5}$ aggravates the severity of COPD (Chronic Obstructive Pulmonary Disease) and asthma, causes inflammatory reactions and arteriosclerosis leading to cardiovascular disease, reduces heart rhythm variability and increases the risk of arrhythmia and cardiac arrest, and contributes to lung cancer. Additional effects include DNA damage and allergic or inflammatory responses.

Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

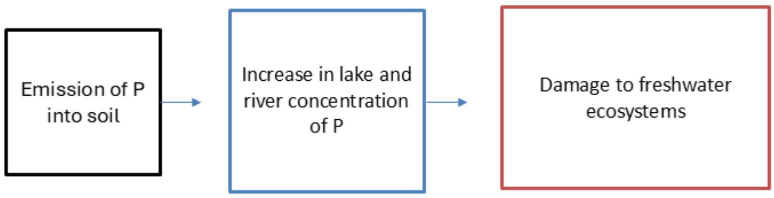
Freshwater eutrophication

Freshwater eutrophication occurs due to the discharge of nutrients such as nitrogen (N) and phosphorus (P) into soil or freshwater bodies, leading to elevated nutrient levels that disturb ecological processes and natural cycles. Within the EU, agriculture is the largest source of eutrophying emissions due to fertilizer application and livestock manure. Additional contributions arise from wastewater discharges and sludge dumping, which further increase nutrient loads to soil and water. Excessive nutrients eventually result in damages to ecosystem quality, with disappeared fraction of autotrophic and heterotrophic species.

Impact pathways

Environmental impacts related to freshwater eutrophication are numerous. They follow a sequence of ecological changes triggered by increasing nutrient emissions into freshwater. Elevated concentrations of phosphorus enhance nutrient uptake by autotrophic organisms such as cyanobacteria and algae, as well as heterotrophic species such as fish and invertebrates. This leads to algal blooms, reduced oxygen availability, and degraded habitat quality, ultimately resulting in a relative loss of species. The midpoint indicator is the increase in phosphorus concentration in freshwater, expressed as kg P eq × m^3 × yr. Fig. 2.3 illustrates the environmental cause-effect chain for freshwater eutrophication as illustrated in ReCiPe 2016, where emissions of phosphorus to water or soil increase nutrient concentrations in rivers and lakes, leading to damages to freshwater ecosystems. The latter are measured in disappeared fraction of autotrophic and heterotrophic species.

Figure 6.4. Cause-effect chain for Freshwater Eutrophication



Note: Impact pathway from phosphorus emissions (P) to damage to ecosystem quality.
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

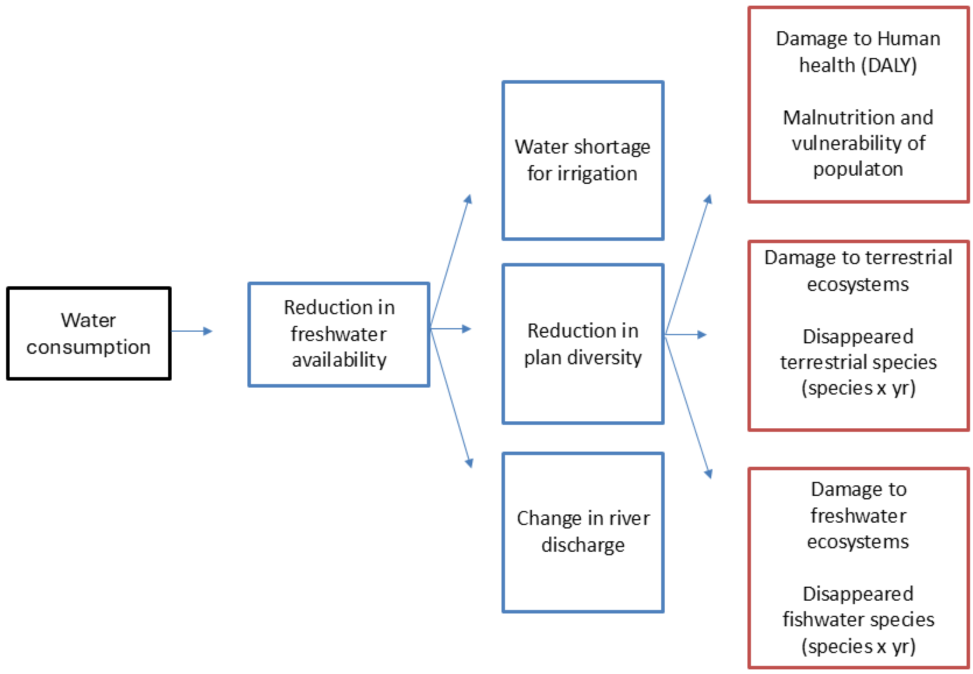
Water Use

Freshwater is an essential resource for both human well-being and functioning of ecosystems. Its availability underpins food production, industrial processes, energy generation, and biodiversity. However, increasing demand, combined with uneven geographical distribution and the effects of climate change, places growing pressure on global freshwater resources. In ReCiPe 2016, water consumption is defined as the use of water in such a way that it is no longer available in the watershed of origin, for example through evaporation, incorporation into products, transfer to other watersheds, or discharge into the sea. Such consumption reduces the availability of freshwater for humans and ecosystems within the affected watershed ultimately resulting in damages to human health and ecosystem quality.

Impact pathway

Fig. 2.4 summarizes the cause-effect chain for water use, where increased water consumption reduces freshwater availability and results in damages at three endpoints: human health (via malnutrition and vulnerability), terrestrial ecosystems (via loss of terrestrial plant species), and freshwater ecosystems (via loss of aquatic species). The modeling of water use impacts begins with the quantification of the reduction in freshwater availability, expressed per $m3$ of water consumed, which represents the midpoint indicator for this impact category. Reduced availability creates competition between water uses. For humans, a shortage of water for irrigation reduces crop yields and can increase malnutrition in vulnerable populations[44]. Impacts on terrestrial ecosystems are modeled through reduced vegetation and plant diversity, based on the assumption that reduced blue water (lakes, rivers, aquifers, precipitation) also lowers green water (soil moisture), thereby constraining net primary production (NPP) and driving terrestrial species loss. For freshwater ecosystems, reduced river discharge caused by water consumption leads to habitat loss, which in turn drives the disappearance of freshwater fish and invertebrate species.

Figure 6.5. Cause-effect chain for Water Use



Note: Impact pathway from water consumption to impacts on human health and ecosystem quality.
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

[44] The extent of this effect depends on socioeconomic context: countries with lower Human Development Index (HDI) values are more exposed, while industrialized countries (HDI > 0.88) can typically mitigate the risk through food imports, resulting in limited health damage.

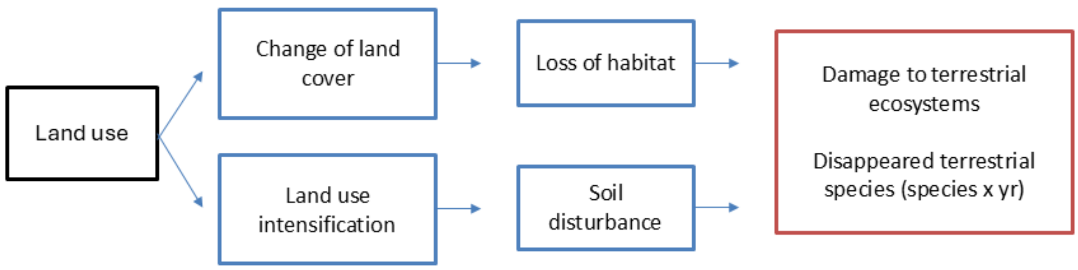
Land Use

Land use is a major driver of biodiversity loss and ecosystem degradation worldwide. Human activities such as agriculture, forestry, and urban development alter natural habitats, disrupt ecological processes, and reduce the capacity of ecosystems to sustain species diversity. Biodiversity itself is a multidimensional concept, encompassing variation at the genetic, species, and ecosystem levels, and its response to land-use change is complex and context-dependent. Because ecosystems and species react differently to disturbances, the impacts of land use are highly variable across regions and land-use types, making their quantification a particular challenge in impact assessments.

Impact pathway

Land use refers to the complete cycle of land transformation, occupation, and relaxation, as explained in Box 2.1. The impact pathway, illustrated in Fig 2.6, considers both the change of land cover, and the actual use of the new land, such as agricultural or urban activities, which lead respectively to alteration in the original species composition and in the natural habitat, and to a further disqualification of the land as a suitable habitat for species. Eventually, this results in relative species loss in terrestrial ecosystems.

Figure 6.6. Cause-effect chain for Land Use

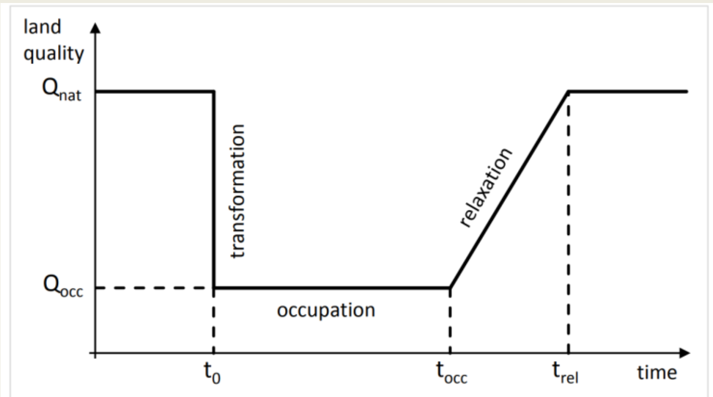


Note: Impact pathway from land use to damages to ecosystem quality.
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

Box 6.1. Three phases of land use

The process of land use can be modelled in three phases: transformation, occupation and relaxation. As shown in Fig. 2.5, the first phase consists of transformation, where land is converted to make it suitable for a new function, leading to an immediate reduction in land quality. This is followed by the phase of occupation, during which the land remains under anthropogenic use and biodiversity levels remain altered. Finally, a restoration phase may occur after land use ends, allowing ecosystems to recover either to their original state or to a new, altered state. It is assumed that during the period of relaxation, the land still has (some) negative impact on species richness, given that it is not immediately returned to primary habitat or will not return to the original habitat, but rather to a different state. Three types of land use impacts can therefore be distinguished: transformation impacts, caused by land cover change; occupation impacts, occurring during the period of land use; and permanent impacts, which occur when ecosystems cannot fully recover after disturbance.

Figure 6.7. Schematic overview of the three phases of land use and their impact on land quality



Note: Land transformation and occupation occurs between t_0 and t_occ, and relaxation occurs between t_occ and t_rel. Qnat shows the original, natural land quality and Q_occ is the land quality after land transformation.
Source: ReCiPe 2016 (Huijbregts et al., 2016[7]), ReCiPe 2016.

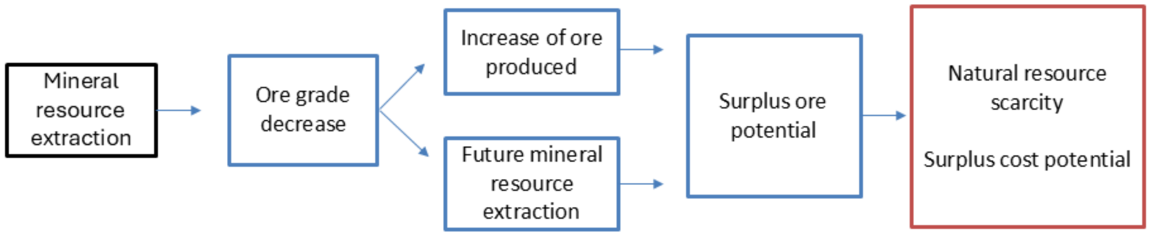
Mineral Resource Scarcity

Mineral resources are essential inputs for modern economies, providing the raw materials for construction, energy systems, transportation, and digital technologies. However, continuous extraction leads to declining ore grades, meaning that more material must be mined and processed to obtain the same amount of useful resource. This not only increases energy use and environmental pressures but also reduces the availability of high-quality deposits for future generations. In LCIA, mineral resource scarcity is considered in terms of the additional effort and costs imposed on future extraction. As higher-quality ores become depleted, society must turn to lower-grade deposits, which require more energy, water, and land to access. This cumulative burden is captured through the concept of *surplus cost potential*, reflecting the long-term economic implications of current resource use.

Impact pathway

The modeling of mineral resource scarcity is subdivided into several steps (Fig. 2.7). It begins with the extraction of mineral resources, which gradually lowers the global average ore grade (OG), meaning the concentration of the target resource within ores declines over time. As ore grades decrease, a greater quantity of ore must be mined and processed to obtain the same amount of useful mineral, thereby increasing the ore produced per kilogram of extracted resource (OP). When combined with projections of future resource demand, this leads to the calculation of the surplus ore potential (SOP), which serves as the midpoint indicator of this impact category. A higher SOP reflects a greater long-term pressure on mineral extraction and signals that lower-grade deposits will need to be exploited. This in turn increases the surplus cost potential (SCP), which represents the endpoint and quantifies the additional extraction costs imposed on future resource users. Both indicators are based on the principle that high-quality, low-cost deposits are exploited first, while lower-quality, higher-cost resources are left for future generations. The higher the SCP, the greater the damages associated with natural resource scarcity.

Figure 6.8. Cause-effect chain for Mineral Resource Scarcity



Note: Impact pathway from mineral resource extraction to damage to natural resources availability.
Source: Authors elaboration based on ReCiPe 2016 (Huijbregts et al., 2016[7])

6.4. Revealed and stated preference methods for measuring non-market value

Revealed Preference (RP) methods use observed market behaviour in related markets to infer the economic value of non-market environmental goods or impacts. By analyzing actual choices made by individuals facing budget constraints, these methods indirectly estimate Willingness-To-Pay (WTP) in markets where no direct transactions exist. Common applications include travel cost methods for recreational sites, hedonic pricing of property attributes, and defensive expenditures to avoid environmental harms. While increasingly sophisticated econometric techniques and improved data have enhanced these methods, challenges remain. These include isolating the effect of specific variables, dealing with ‘missing-variable bias’ (e.g., noise nuisance valuations affected by correlated factors like spoiled views), and the risk of inaccurate damage estimates when individuals lack full information about environmental impacts, such as health effects from pollution. Due to these limitations, especially regarding incomplete information, stated preference methods may be preferred in some contexts. In the context of cost benefit analysis of public policies and social impact analysis in general, damage costs are typically preferred over abatement costs as a measure of environmental and health impacts.[45]

Stated Preference (SP) methods rely on surveys and interviews to directly elicit individuals’ WTP for

[45] 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 social cost benefit analysis (CE Delft, 2024[6]).

environmental changes, capturing both use and non-use values. In theory, if respondents are well-informed, honest, and rational, these methods provide the most reliable measure of people's preferences. However, in practice, challenges include hypothetical bias (where respondents overstate their WTP due to lack of actual budget constraints) and sensitivity to survey design and perceived outcomes. Despite these issues, stated preference methods are particularly valuable for valuing public goods and non-market impacts that revealed preference techniques cannot adequately capture. Stated preference methods do not reveal exchange values directly and hence require adjustment for use in accounting.[46]

6.4.1. Revealed preference methods

Market prices for similar markets

The most straightforward way to measure prices and estimate values for accounting purposes is by directly observing transactions involving non-market goods, when such exchanges occur. For instance, if a wetland offers water purification services and its owners or managers charge the water company that extracts water for municipal use, this transaction represents a measurable exchange of a non-market good (ecosystem services). Similarly, stumpage fees charged to timber companies and rental prices for agricultural land (where markets exist for leasing land for crops or grazing) are also examples of directly observed values. These rental prices can be used to determine prices for accounting related to biomass provisioning services. In all these cases, there is a clear connection to benefits recognized in the System of National Accounts (SNA). Although using directly observed values is the preferred method, the prices recorded may sometimes underestimate the true value of non-market goods. This outcome is well-recognized and considered to reflect current institutional arrangements.

Residual value

Residual value and resource rent methods estimate the value of a non-market good by calculating the gross value of the final marketed product that depends on the amenity and subtracting the costs of all other inputs, such as labour, produced assets, and intermediate inputs.[47] Changes in value due to public or private sector actions may be estimated by evaluating differences in residual values and resource rents between alternative and counterfactual scenarios. Applying these methods is challenging for several reasons. First, because residual values may include a mix of other unpaid or indirect inputs, isolating the contribution of the non-market good to such values can be difficult. Second, the accuracy of estimates depends on correctly calculating the value of all "paid" inputs. Third, the magnitude of the residual value is influenced by institutional arrangements, such as market size and type. Finally, as residual value methods are often better suited to broad, industry-level data, price estimates may lack the detail needed for location-specific monetary valuations. Nevertheless, insofar as these approaches rely on observed data, the values they produce reflect current institutional contexts and can offer a useful broad perspective on monetary values in a given context.

Productivity change method

This method treats non-market goods as an input in the production of a market good. Assuming other factors remain constant, changes in the non-market good (e.g. an ecosystem service) are considered to

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[46] For more, refer to material on Simulated exchange value in Section 2.3.2.

[47] Resource rent measures the surplus value to the extractor of using a natural resource in production, while residual value measures the (subsequent) payment of part of the resource rent to government.

affect the output of the marketed good. The method for estimating the value of a non-market good via the productivity change method is conducted in three steps. First, the marginal product (i.e. the change in production value from a small change in the provision of a non-market good) is estimated. Second, the marginal product is multiplied by the price of the market good to calculate the marginal value product of the non-market good. Third, the marginal value product is then multiplied either by the physical quantity of the non-market good to derive its total value (in national capital accounting contexts) or by the change in the physical quantity of the non-market good expected from a given activity to derive the value of nonmarket impact (in cost-benefit analysis and environmental profit and loss contexts). These relationships are estimated for a specific accounting period, acknowledging they may vary over time.

The productivity change method has been applied to value non-market goods such as water and pollination in agriculture, particularly in areas with detailed data to estimate production functions. It is especially useful for valuing non-market goods such as ecosystem services that serve as inputs to existing SNA outputs. However, when multiple market and non-market goods interact, specifying the production function and isolating the marginal product of one non-market ood can be challenging due to the complexity of factors involved. Additionally, this method can be data-intensive and scaling it up to a national level may be difficult.

Hedonic price method

The Hedonic Price Method (HPM) estimates the value of non-market goods by analyzing how they are implicitly traded in related markets, typically housing and labour markets. Originating from Rosen (1974[63]), the method assumes that the price of goods (e.g. houses or jobs) reflect a bundle of characteristics, and consists of using statistical techniques to isolate the implicit contribution of each attribute to the value of the good as reflected in the marketplace.[48] In housing, HPM uses property transaction data to infer the value of features like air quality, proximity to green space, or noise levels. For example, the value of a quieter environment can be estimated by comparing otherwise identical homes in noisy and quiet areas. In labor markets, HPM estimates the value of risk by examining wage differences across jobs with varying levels of danger. The appeal of HPM lies in its basis on observable market behavior rather than stated preferences.

Like other methods, the limitations of HPM should be kept in mind. First, as with other revealed preference methods, HPM only measures use values. Second, consumers may lack knowledge about risks (e.g., flood zones in residential areas), which means that their purchasing decisions should not be considered to reflect this information. For example, many people are poorly informed about how environmental pollution relates to human health. In revealed-preference methods this results in pollution impacts being undervalued. In addition to assuming perfect information, HPM also assume competitive markets and free mobility,

[48] For example, in the housing market, the HPM uses housing market transactions to infer the implicit value of the house's underlying characteristics, including it's structural characteristics (e.g. the number and size of rooms, the presence and size of a garden), location/accessibility (e.g. proximity to schools, shops, roads), neighbourhood characteristics (e.g. crime rate) and the local environment and nearby amenities (e.g. air quality, proximity to green spaces). The price of a house is considered to be determined by the particular combination of these characteristics, such that properties possessing more and better desirable characteristics command higher prices and those with larger quantities of bad qualities command lower prices, all else constant. The HPM seeks to unbundle the contributions of each determinant of house prices in order to identify marginal willingness to pay for each housing characteristic. In job markets, the HPM is used to estimate the value of avoiding risk of death or injury by examining wage differentials that exist across jobs with different exposures to physical risk, assuming that work in jobs entailing higher risk is remunerated more highly in exchange for undertaking this risk. This risk premium provides an estimate of the market value of small changes in injury or mortality risks (OECD, 2018[37]).

conditions that are not often met in real world markets. Econometric issues exist surrounding the selection of functional forms, multicollinearity, heteroscedasticity, omitted variable bias, and defining the correct spatial extent of the relevant market. Additionally, most HPM studies use cross-sectional data and only estimate marginal implicit prices, rather than estimating full demand functions and the value of non-marginal and non-localised changes, due to theoretical and practical complexity. Despite its challenges, however, HPM remains a widely used and theoretically grounded method for valuing non-market goods in environmental and labor economics.

Travel cost method

The Travel Cost Method (TCM) estimates recreational use values of non-market goods, mainly outdoor natural areas. Natural areas like parks, woodlands, beaches, and rivers typically have no market price, so TCM uses the cost of travel and related expenses as proxies for their value. TCM recognizes that recreational experiences depend on several inputs, including the recreational site, travel, and sometimes overnight stays. While the site itself is unpriced, travel costs and other expenses are market-priced and thus used to estimate site value.[49]

TCM's applicability is narrow, focusing on recreational use and requiring large data sets, including GIS analysis for Random Utility Models (RUM). Single-site TCM models face limitations like ignoring substitute sites, which RUM can address. However, multiple-purpose trips remain challenging, especially for international tourists visiting multiple destinations. A common solution is to survey visitors on the proportion of trip enjoyment from the specific site, adjusting travel costs accordingly. Another challenge is valuing travel time, with results sensitive to assumptions. Researchers often assume travel time's value as a fraction of wage rates, though this is ad hoc and hard to validate empirically. Critics note this approach undervalues non-wage earners like students or homemakers, for whom marginal utility of time is not zero.

Averting behaviours and preventative expenditures method

Methods based on averting behaviour assume individuals and households can reduce exposure to non-market "bads" by adopting costlier behaviours or purchasing market goods (defensive expenditures) that substitute for these non-market goods. For example, installing double-glazed windows reduces road traffic noise exposure by substituting peace and quiet. If noise decreases, households spend less on such defensive goods, reflecting their valuation of policies reducing pollution. Many applications focus on health impacts, such as improved water quality or bicycle helmet safety for children. Beyond market goods, costly behavioural changes, such as spending more time indoors to avoid air pollution, also represent averting behaviour. Although this time-use is unobservable and itself a non-market good, it can be valued using wage analogues (e.g. in relation to the risks from air pollution). From a policy perspective, accounting for these behaviours is vital because ignoring them leads to biased (typically underestimated) assessments of environmental risks and pollution damages.

Applying this method in practice is associated with several challenges. First, defensive expenditures often

[49] The basis of the TCM is the recognition that individuals produce recreational experiences through the input of a number of factors. Amongst these factors are the recreational area itself, travel to and from the recreational area and, in some cases, staying overnight at a location and so on. Typically, while the recreational area itself is an unpriced good, many of the other factors employed in the generation of the recreational experience do command prices in markets, such as travel costs. Travel costs could therefore be used as a proxy for the value of accessing the site (OECD, 2018[37]).

only partially capture the value of avoiding the bad.[50] Second, defensive behaviours often generate additional benefits or costs.[51] While the net cost after accounting for such joint benefits should be measured, separating these effects is difficult. Assigning monetary value to behavioural changes is also a challenge, especially for groups like children where wage-based valuations don't apply. Finally, it is difficult to disentangle the effects of the non-market bad and averting behaviour on outcomes due to unobserved confounding factors, leading to inaccurate estimates. These issues limit the practical impact of averting behaviour and defensive expenditure methods compared to other revealed preference approaches.

Simulated exchange value

The simulated exchange value method estimates the price and quantity of a non-market good as if it were traded in a hypothetical market, providing a direct value based on the exchange value concept (UN et al., 2024). It combines demand function derived from either revealed or stated preference valuation methodologies with a supply function and an appropriate market structure. The method then calculates a simulated price to estimate the expected value of the good if it were to be exchanged in a market. Although the simulated exchange value method can be adapted to various complexities and market contexts, it has seen less application compared to other valuation methods.

Abatement cost method

The mitigation or abatement cost approach values environmental or health damages by estimating how much it costs to reduce or mitigate the harmful emissions or pollutants causing those damages. For this reason, the abatement cost method is less suitable for use in the calculation of value factors because it takes the value of avoiding damages as a proxy for the value of damage rather than estimating the value of the damages themselves. It assumes the value of preventing or avoiding damage is roughly equal to the cost of abating (i.e., controlling, reducing, or eliminating) the pollutant or hazard. The abatement cost method notably depends on the environmental target being set, as marginal abatement costs increase with the stringency of such targets. Although damage cost approaches are typically preferred over abatement cost approaches for measuring environmental and health impacts, uncertainties in damage costs as well as the implementation of target-based policies have led a number of countries to employ the abatement cost approach to value the impact of GHG emissions.

Replacement cost

The replacement cost method estimates the cost of replacing a non-market good with a substitute that provides the same benefits. Because it is generally used to value the total value of an asset, it is less suited for use in calculating value factors. Substitutes can be consumption items (e.g. air filtration units that replace air filtration provided by trees), input factors (e.g. sorghum replacing non-priced rangeland forage) or capital factors (e.g. water treatment plants). The price of the non-market good equals the cost of using the substitute to deliver identical benefits per unit. The method relies on several assumptions, including that the substitute must perform exactly the same function as the non-market good, it must be the least-cost alternative, and there must be a willingness to pay for the substitute if the non-market good in question is no longer supplied. Relatedly, the restoration cost method estimates the cost of restoration costs needed to return the environment to a specified condition. As avoidance and abatement costs may already have been incurred when accounting for the costs of environmental degradation in monetary terms, this method

[50] For example, double-glazing improves indoor quietness but not outdoor noise exposure, so it underestimates total willingness-to-pay.

[51] For example, time indoors can be used productively and double-glazing also saves energy.

provides a measure of unaccounted environmental quality changes. Core assumptions method include that cost estimates represent the least cost option and that there is consensus on the environmental quality target for restoration.

Avoided damage cost

The avoided damage costs method values non-market goods by estimating the costs that would arise if nonmarket goods were lost. Like the replacement cost method, as it is generally used to value the total value of an asset, it is less suited for use in calculating value factors. The validity of this method relies on conditions similar to those required for the replacement cost approach. This method is especially effective for valuing regulating services such as soil erosion control, flood prevention, air filtration, and global climate regulation. It estimates the economic units expected to avoid damage costs due to the non-market good. For example, governments may avoid healthcare costs due to air filtration. However, these economic units are not necessarily direct users of the services; rather, avoided damage cost estimates serve as a way to value those services. In some cases, prices based on both replacement costs and avoided damage costs can be calculated. When both are available, the lower of the two should be used, which in most cases tends to be the replacement cost-based price.

6.4.2. Stated preference methods

Contingent valuation

The contingent valuation (CV) method is a stated preference approach that uses surveys to directly ask individuals about their willingness to pay or willingness to accept compensation for a hypothetical change in the availability or quality of a non-market good. This technique constructs a hypothetical market, defining the good, the institutional setting in which it would be provided, and how it would be financed, allowing respondents to express their preferences as if they were making real market decisions. CV is notably flexible and can be applied to a wide range of contexts, including future scenarios, non-use values, and situations where no actual market exists. It remains one of the few methods capable of capturing the total economic value of non-market goods, including both use and non-use benefits.

A large body of literature exists offering guidance on best practices for survey design and valuation using CV. This is especially important because debates about the validity of the method continue, often focusing on specific biases and limitations. Recent developments in behavioural economics have helped shed light on some of these challenges, while the growing use of online surveys has expanded the method’s reach and enabled further testing of potential biases and techniques for reducing them.

Discrete choice experiments

Discrete choice experiments (DCEs) are a stated preference method grounded in Lancaster’s theory of value, where goods are valued for their attributes. Developed in the 1980s, DCEs have become widely used in environmental economics, surpassing contingent valuation in popularity. They present respondents with sets of alternatives, each defined by different attribute combinations, including a baseline or status quo option. Based on the random utility model, DCEs infer values from observed choices rather than direct questions, making them suitable for estimating both total and marginal values of goods. By varying the levels of attributes, researchers can estimate willingness to pay and model trade-offs among attributes. DCEs are particularly consistent with welfare theory and are capable of measuring non-use values.

DCEs offer several advantages over CV, especially for valuing multi-dimensional changes where trade-offs between components matter. They allow for the estimation of marginal values for each attribute, providing more detailed information for policy design and management decisions. This makes DCEs especially useful in complex settings such as water service provision where multiple service dimensions must be valued together. Their structure also supports value transfer between contexts, due to the detailed, attribute-level data they generate. Compared to CV, DCEs impose an internal scope test through repeated choice tasks, often resulting in more consistent responses. Additionally, because DCEs rely on indirect inference rather than direct willingness-to-pay questions, they may reduce issues like protest responses or strategic bias, although evidence on this remains mixed.

Despite these strengths, DCEs face several limitations. One major challenge is the cognitive burden on respondents, who must evaluate complex trade-offs across multiple attributes in several repeated choice sets. This can lead to fatigue, inconsistent responses, or reliance on simplifying heuristics rather than full utility-maximizing behavior. There are also statistical challenges with handling repeated responses and potential learning or fatigue effects. Another limitation is that DCEs may not be ideal for valuing whole programmes or sequential policy elements, since they rely on the assumption that total value equals the sum of the parts, as some studies have shown that summed attribute values can sometimes overstate the value of the whole compared to CV results.

Additional issues include a common tendency for respondents to choose the status quo or opt-out option, which may reflect status quo bias caused by inertia, uncertainty, or task complexity. This can distort valuation outcomes if not properly addressed. As with any stated preference method, DCE results are highly sensitive to survey design, including how attributes are framed, the levels chosen, and the mode of presentation. While DCEs offer detailed and flexible data for policy analysis, their success depends on careful experimental design, pre-testing, and appropriate statistical modeling. In sum, DCEs provide powerful tools for valuing complex goods, but require thoughtful application to mitigate their cognitive and methodological challenges.

Subjective wellbeing

Subjective Well-being (SWB) valuation is an alternative to traditional non-market valuation methods by linking self-reported well-being data (such as life satisfaction) to policy impacts, thus capturing experienced rather than hypothetical utility. One major advantage is that SWB values are derived from actual lived experiences, allowing for the assessment of how real-world changes, such as environmental quality or health conditions, affect people's lives. This method avoids common biases in stated preference methods, such as hypothetical or strategic bias, and does not rely on strong rationality assumptions. It is also particularly suited to valuing complex or non-marginal changes, such as those involving spiritual or community benefits, or health conditions that are difficult to assess through willingness-to-pay frameworks. Moreover, when national SWB datasets are available, this approach can be highly cost-effective, eliminating the need for primary data collection. It may also reduce issues like focusing illusion and priming effects that skew results in traditional surveys.

However, SWB valuation has several notable limitations. Estimating accurate income coefficients remains a major challenge, often leading to inflated welfare values due to issues like endogeneity and reverse causality. SWB data typically cannot capture non-use values effectively and are limited in forecasting the impacts of future or marginal policy changes, as it relies on past or present experiences. The method also struggles with choosing between different SWB dimensions (e.g. life satisfaction vs. momentary happiness), each of which may respond differently to policy changes. Measurement issues also persist insofar as respondents may be influenced by context, mood, or survey design, and narrow response scales may fail to capture the full richness of human experience. Furthermore, due to hedonic adaptation, people often adjust quickly to both positive and negative changes, potentially masking long-term welfare effects. These challenges, along with the need for more robust causal inference methods, mean that while SWB valuation is a valuable complement to existing methods, it is not yet a complete substitute.