value balancing alliance

Methodology

Impact Statement

Focus: Environment

Version 0.1

VBA METHODOLOGY V0.1

Impact Statement

– Topic Specific Method Paper: Environment –

CONSULTATION DRAFT

March 2021

Note on this document

This is the first version of our Impact Statement methodology for environmental aspects. We piloted this version in 2020 and the learnings will inform the further development in 2021.

We are very aware that this is a work in progress. We are still discussing with third-party experts and our members important elements and we will be using a review panel and formal consultation and piloting process to test and improve the standardized approach.

In addition, we have developed a General Methodology that addresses aspects that are applicable across individual economic, environmental and social indicators. Moreover, we have developed a methodology for specific socio-economic aspects.

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About the Value Balancing Alliance

The Value Balancing Alliance e.V. (VBA) is a non-profit organization that aims to change the way how company performance is measured and valued. The alliance's objectives are to create a global impact measurement and valuation (IMV) standard for monetizing and disclosing positive and negative impacts of corporate activity and to provide guidance on how these impacts can be integrated into business steering.

VBA, which was founded in June 2019, represents several large international companies, including Anglo American, BASF, BMW, Bosch, Deutsche Bank, DPDHL, Kering, LafargeHolcim, Mitsubishi Chemical, Otto, Porsche, Novartis, SAP, Schaeffler and SK. The alliance is supported by the four largest professional services networks – Deloitte, EY, KPMG and PwC – and by the OECD and leading academic institutions, including the University of Oxford and the Impact Weighted Accounts Initiative at Harvard Business School. Furthermore, in partnership with the Capitals Coalition, the alliance receives funding from the EU through its LIFE programme for the Environment and Climate Action[1] and is member of the EU Platform Sustainable Finance.

A global IMV standard is needed not only to foster long-term thinking and performance comparability but also to consolidate the knowledge already available in this field. Therefore, the VBA is building on the work of leading universities and well-known organizations, such as the World Bank, the OECD, the Capitals Coalition, the WBCSD, the Impact Management Project, the GRI, SASB and the IIRC. The envisioned transformation and system change require the cooperation of all players in the business ecosystem. The alliance will make its work available to the public and encourages more companies to join along the way.

[1] The EU has provided the VBA with financial support to develop a first set of accounting principles and guidelines regarding environmental impacts for business. Over the next three years, the VBA (in partnership with the Capitals Coalition) will develop a standard for measuring and valuing companies' environmental impacts in monetary terms.

Table of contents

1. INTRODUCTION

The current economic system focuses on financial value and excludes many of the impacts of business on society, such as environmental and social impacts. These impacts are often referred to as externalities for this very reason. Many of these impacts are directly or indirectly linked to current and future business value, and to stakeholders' interests. Therefore, businesses are becoming increasingly interested in these impacts and ways of taking them into account in their strategies and business decisions.

There are two major perspectives on value. First, the stakeholder perspective focuses on the positive and negative impacts of corporate activities on the environment and, by extension, society. This is known as ***the value to society perspective***. Second, a financial view of how these impacts (and dependencies) affect the (longer-term) financial performance of corporations is known as ***the value to business perspective***. Both perspectives are inherently connected. As such, they have been widely acknowledged as "double materiality".[2]

The VBA aims to embrace both methodological streams – one focusing on impacts and the other on dependencies – as they are fundamental for understanding a company's long-term value creation.

Our aim is to work towards global standardization. Moreover, our methodology is not limited to environmental impacts – we believe that the same principles should apply to all sustainability impacts.

Value to society – Impact Statement

The General Method paper introduces the calculation methodology for monetary impact valuation and is followed by deep-dive topic papers on socio-economic and environmental impacts. Notably, these papers focus on topics that are already reasonably mature rather than a comprehensive set of impacts:

- General Method paper – sets out the overarching framework as well as the key concepts and process of methodology development,
- Environmental Method paper – explains the IMV details for specific environmental topics and specific sub-indicators, and
- Socio-economic Method paper – explains the IMV details for specific socio-economic topics as well as specific sub-indicators.

The General Method paper is the foundational document. It sets out the guiding objectives, outlines the methodology development process, explains the document's development process, and summarises key concepts and general choices that need to be made and that should be common for all economic, environmental, and social impacts.

[2] On double materiality see, e.g.: Accountancy Europe (2020): Interconnected Standard Setting for Corporate Reporting, https://www.accountancyeurope.eu/wp-content/uploads/191220-Future-of-Corporate-Reporting.pdf;
CDSB (2020): Falling short?, https://www.cdsb.net/sites/default/files/falling_short_report_double_page_spread.pdf; EU Commission (2019) Guidelines on reporting climate-related information, https://ec.europa.eu/finance/docs/policy/190618-climate-related-information-reporting-guidelinesen.pdf.
Natural Capital Protocol, p.15

The aim of the Environmental Method paper is to provide specific details for Natural Capital Accounting per impact driver and summarises key concepts.

All documents and described methodologies are in an interim state and will be finalized after the piloting and learning in 2023 (expected).

The method is being developed using an iterative process. The methods currently described in this document are version V0.1. The methodology has been piloted with international companies.

2. TOPIC-SPECIFIC DETAILS

2.1. GREENHOUSE GAS EMISSIONS

- *Topic description*
- *Impact pathway*
- *Quantification and monetary valuation*
    - *Measuring impact drivers*
    - *Environmental outcomes*
    - *Impacts on society and monetary valuation*
- *Sources*

2.1.1. TOPIC DESCRIPTION

The earth's atmosphere shields us from harmful radiation, provides us with air to breathe and traps enough heat from the sun to enable the planet to support complex forms of life. Scientists have long been aware of this essential "greenhouse effect". However, in recent decades, they have become increasingly concerned about changes in the composition of the Earth's atmosphere and the potential of these changes to increase the amount of heat trapped.

The data now conclusively show that the Earth is warming and has been for some time. Scientists are confident that the net effect of human activities – and the resulting increase in atmospheric greenhouse gas (GHG) concentration – has contributed to this warming. This is discussed in detail by the Intergovernmental Panel on Climate Change (IPCC)[3].

Even in the absence of humans, Earth has a naturally occurring carbon cycle in which carbon is exchanged between different living organisms and the environment through natural processes. Some processes (e.g. photosynthesis) remove GHGs from the atmosphere, while others (e.g. respiration or decomposition in the soil) emit carbon into the atmosphere. Since the industrial revolution, human activity has modified the carbon cycle by adding sources (e.g. burning fossil fuels) and removing sinks (e.g. changes in land use, especially deforestation).

This has led to an increasing concentration of GHGs in the atmosphere, which results in an increase in the greenhouse effect. This, in turn, changes the Earth's climate.

2.1.2. IMPACT PATHWAY

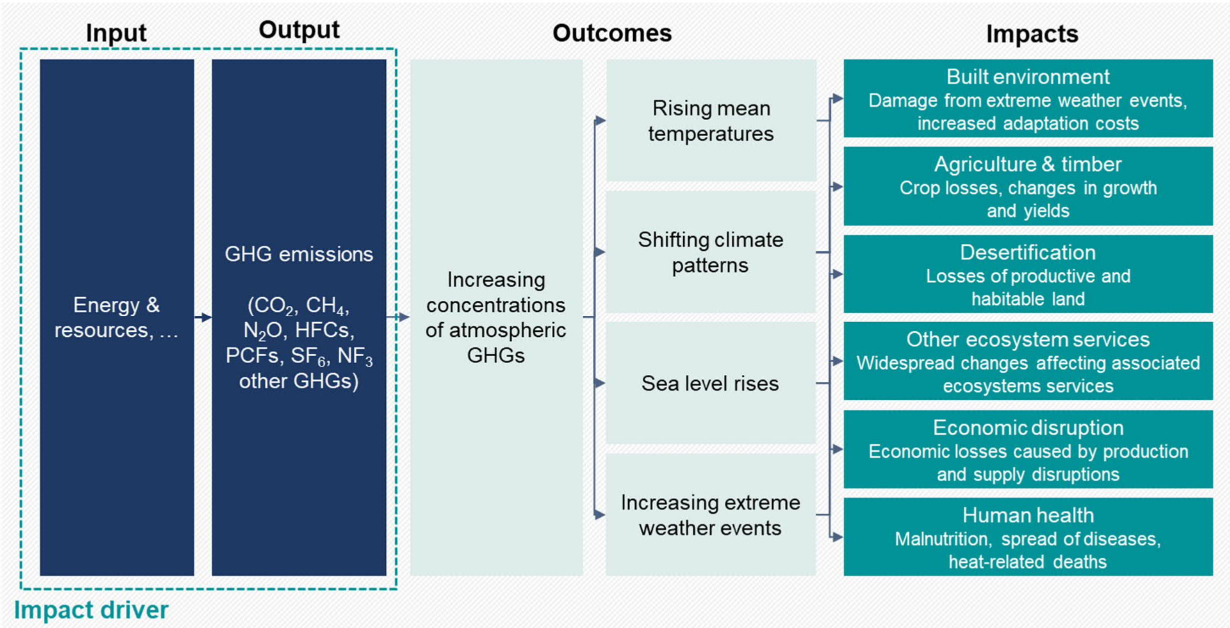

Figure 1: Simplified impact pathway GHGs

[3] IPCC (2014): Climate Change 2014 – AR5 Synthesis Report.

Climate change is driven by the total concentration of GHGs in our atmosphere, regardless of where they are emitted or removed.

GHGs are atmospheric compounds that absorb and re-emit infrared radiation emitted by the Earth's surface, the atmosphere and clouds. This property causes the greenhouse effect in which heat is trapped within the Earth's surface-troposphere system. There are both natural and anthropogenic GHGs. The Intergovernmental Panel on Climate Change (IPCC) lists 18 different GHGs. The seven principal classes of GHGs are carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), sulphur hexafluoride (SF_6), various hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and nitrogen trifluoride (NF_3).[4]

Contributions to climate change depend on the type of gas. These contributions can be normalised by calculating them relative to the effect of carbon dioxide as "CO_2 equivalents" (CO_2e) using their Global Warming Potential (GWP). As different gases have different lifetimes, a GWP is calculated over a specific time horizon. The GWP most widely used is GWP100.[5]

Typical corporate activities leading to greenhouse gas emissions include energy use (both directly through fossil fuel combustion, and indirectly through use of electricity, steam or cooling energy) as well as direct GHG emissions from chemical processes. For a detailed classification of sources and activities leading to GHG emissions, see the GHG Protocol.

The increasing concentration of GHGs in our atmosphere has a direct effect on our environment through:

- Shifting climate patterns,
- Rising sea levels,
- Increasing extreme weather events and
- Rising mean temperatures.

All this results in multiple societal impacts, including impacts on:

- Health,
- The built environment,
- Economic disruption,
- Agriculture & timber
- Desertification and
- Other ecosystem services.

Users of this methodology should:

- Use a valuation approach that covers all key drivers and impact areas.

[4] GHG-Protocol (2013): Required Greenhouse Gases in Inventories. Accounting and Reporting Standard Amendment. IPCC (2014): Climate Change 2014 – AR5 Synthesis Report.
[5] GHG-Protocol (2016): Global Warming Potential Values.

2.1.3. QUANTIFICATION AND MONETARY VALUATION

(i) Measuring impact drivers

The GHG Protocol is the most widely used (voluntary) standard for estimating corporate GHG emissions, and it is recommended as the key guiding document for calculating emissions.

Users should:

- Apply the rules outlined in the "General Method" paper of this methodology (include, e.g., all relevant value chain levels, etc.)
- Select appropriate data sources for calculating GHG emissions (incl. GWP) [6]
- Exclude GHG emission offsetting and
- Consider biologically sequestered carbon separately.

(ii) Environmental outcomes

The concentration of GHGs in the atmosphere is key, regardless of where the emissions originate. Therefore, users should:

- Use a valuation approach that reflects this (the global value for the social cost of carbon (SCC)).

(iii) Applying a monetary valuation for value to society

The SCC is most widely accepted approach in the literature, and it is also used by policy makers. It provides an estimate, in USD, of the economic damages that would result from emitting one additional ton of GHGs into the atmosphere. The SCC puts the effects of climate change into economic terms and, thereby, helps policy makers and other decision-makers to understand the economic impacts of decisions that will increase or decrease emissions (RFF 2019b).

The SCC depends on numerous assumptions and parameters, especially:

- The underlying scientific model and
- Other considerations (e.g. the discount rate).

The choice of parameters reflects political and ethical judgments (e.g. whether one values the well-being of future generations differently from one's own well-being). The full range of GHGs, outcomes and types of impact considered are implicitly included in the various models.

Depending on the assumptions, the SCC may vary by a factor of 10 or more[7].

[6] GHG-Protocol (2004, Revised); GHG-Protocol (2013): Required Greenhouse Gases in Inventories. Accounting and Reporting Standard Amendment; GHG-Protocol (2016): Global Warming Potential Values; IPCC (2014): Climate Change 2014 – AR5 Synthesis Report.
[7] Note that the Value Balancing Alliance is working with a team at RFF to determine a recommended value for an SCC. RFF (2019a): Comparing carbon Pricing Proposals. | RFF (2019b): Social Cost of Carbon 101

Users of this methodology should:

- Use a SCC based on the model families in Box 1 and apply a PRTP of 0 and a social discount rate of 3.5[8] and
- Use a globally consistent SCC.

Box 1: SCC Models

What is the social cost of carbon?

It is an estimate, in USD, of the economic damages that would result from emitting one additional ton of GHGs into the atmosphere. The SCC puts the effects of climate change into economic terms to help policy makers and other decision-makers understand the economic impacts of decisions that would increase or decrease emissions.

RFF's Social Cost of Carbon Initiative has gathered a team of distinguished economists and scientists to improve the science behind SCC estimates (including projections for economic growth, population and emissions as well as climate models and damage functions). This is an **ongoing process**. VBA's partnership with RFF helps ensure that our SCC value remains at the leading edge of academic research on the topic.

How is the SCC calculated?

The SCC is calculated in four steps using specialized computer models:

1. Predict future emissions based on population, economic growth and other factors.
2. Model future climate responses, such as temperature increases and sea level changes.
3. Assess the economic impact of these climatic changes on agriculture, health, energy use and other aspects of the economy.
4. Convert future damages into their present-day values and sum to determine total damages.

These four steps provide a baseline value for the damages caused by emissions. The modeling process is then repeated after including a small amount of additional emissions to determine the impact on the total cost of emission-related damages. The increase in damages from the additional emissions provides an estimate of the SCC.

SCC Models

Social cost of carbon: net present value of the damage caused by an additional ton of CO_2 emission today

Three major model families:

- *DICE/RICE – Bill Nordhaus*
- *PAGE – Chris Hope*
- *FUND – Richard Tool and David Anthoff*

Major sensitivities:

- *Model (scenario, climate science, damage function),*
- *Discount rate (weight factor applied to harm, discounting 100 years),*
- *Equity*

[8] Based on values chosen by governments like HM Treasury (2018): The Green Book – Central Government Guidance on Appraisal and Evaluation.
https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/938046/The_Green_Book_2020.pdf ;
Note that the Value Balancing Alliance is working with a team at RFF to determine a recommended value for an SCC.
RFF (2019a): Comparing carbon Pricing Proposals.; RFF (2019b): Social Cost of Carbon 101; EPA (2016a): EPA Fact Sheet – Social Cost of Carbon.; EPA (2016b) Technical Support Document.

2.1.4. SOURCES

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Topic-specific details

2.2. OTHER AIR EMISSIONS

- *Topic description*
- *Impact pathway*
- *Quantification and monetary valuation*
    - *Measuring impact drivers*
    - *Environmental outcomes*
    - *Impacts on society and monetary valuation*
- *Sources*

2.2.1. TOPIC DESCRIPTION

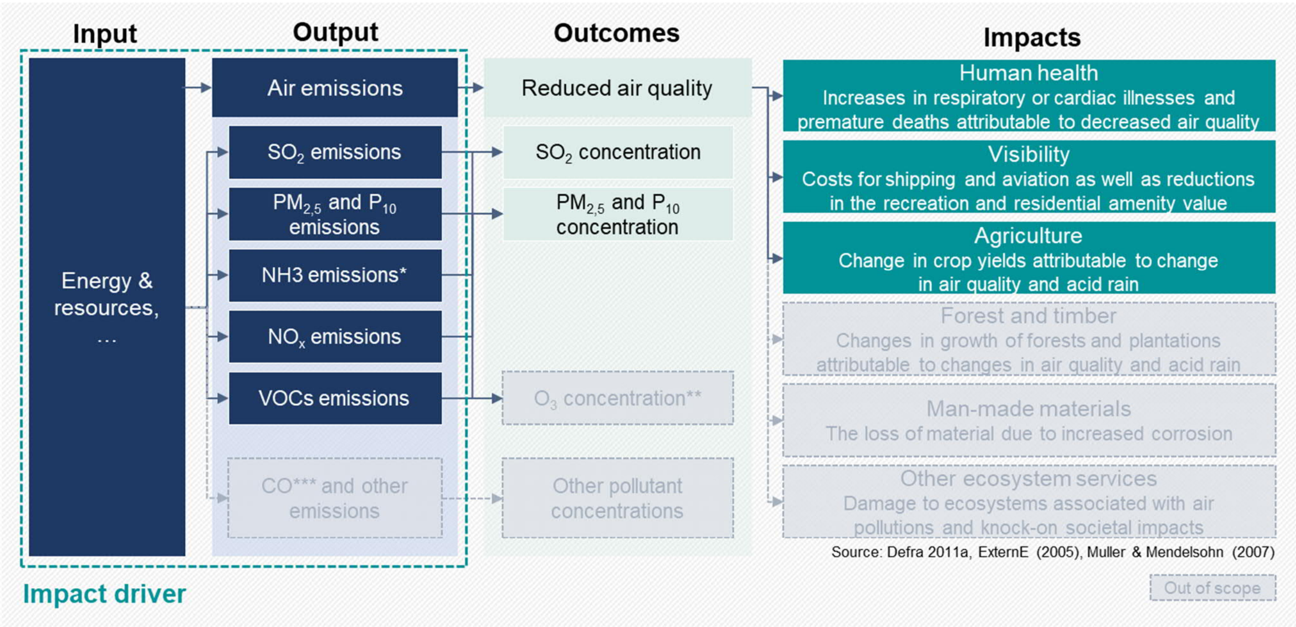
Economic activity in all sectors results in emissions of waste gases and suspended solids into the air (whether directly as a result of industrial processes or indirectly as a result of, for instance, energy or resource consumption). Changes in the concentrations of these emissions may have negative impacts on people (e.g. on their health) and on the natural and built environment. Therefore, these emissions carry a societal cost.

Unlike GHGs, which contribute to climate change on a global scale, the impacts of air pollution are principally local or regional. Moreover, local or regional factors, such as weather conditions and population density, influence the severity of the impact of air pollutants.

Air pollution can be subdivided into two types. “Primary pollutants” have direct, negative impacts on the environment and people. “Secondary pollutants” result from reactions between primary pollutants and other gases under certain conditions and, subsequently, have negative impacts on the environment and people.

2.2.2. IMPACT PATHWAY (OVERVIEW)

In order to value corporate environmental impacts on society, corporate emissions into the atmosphere must be linked to impacts on humans via environmental outcomes. This link is reflected in the impact pathway shown in Figure 2, which shows the three stages of the impact pathway for air pollution. Air pollution follows a complex pathway with multiple pollutants playing a role in multiple environmental and societal outcomes.



*Ammonia (NH_3) has a short lifetime in the atmosphere and most (by weight) is quickly deposited .While this process can have localized impacts on areas close to the emissions source, the impacts are minor compared to impacts on health. Given the low materiality, this secondary deposition in soil and water is omitted from this methodology.
** Ozone (O_3) is formed via a non-linear reaction between VOCs and NOx in the presence of sunlight.
***The close relationships between CO, NOx, and VOC pathways to O_3 formation make it difficult to avoid double counting of secondary impacts. Therefore, CO is excluded from Muller and Mendelsohn's (2007) analysis, Defra's (2011a) air emissions damage cost methodology and ExternE (2005) analyses.

Figure 2: Simplified impact pathway other air emissions

i. Impacts from air emissions are driven by the type and quantity of air emissions resulting from different business activities.
ii. Changes in the state of natural capital (or outcomes): businesses affect air quality through emissions of pollutants. These primary pollutants react with other elements in the air to produce secondary pollutants. Both primary and secondary pollutants can lead to specific environmental outcomes such as smog and acid rain.
iii: Impacts on society are principally related to health but also include impacts on agriculture and visibility.

Impact drivers

The most significant primary pollutants (in terms of societal costs) are listed below (in no particular order).

Primary air pollutants

- **Particulate matter (PM):** PM refers to a range of different types of solid particles that are suspended in ambient air. PM is produced by the burning of biomass and fossil fuels and the creation of dust from agriculture or industry. PM is classified according to particle size: PM_{10} refers to coarse particulate matter (particles with a diameter of 10 micrometres or less) and $PM_{2.5}$ refers to fine particulate matter (particles with a diameter of 2.5 micrometres or less). PM_{10} is expressed exclusive of $PM_{2.5}$ in this document (and associated analyses) to avoid double counting.

- **Volatile organic compounds (VOCs):** VOCs comprise a wide range of organic compounds that have a high vapour pressure under normal atmospheric conditions, for example benzene, aliphatic hydrocarbons, ethyl acetate, glycol ethers, and acetone. They are released in large quantities as a result of human activities (e.g. the use of solvents in industrial processes) and some natural processes.
- **Mono-nitrogen oxides (NO and NO_2, commonly referred to as NOx):** These are naturally present in the atmosphere but are also released in large quantities through the combustion of fossil fuels and transport fuels.
- **Sulphur dioxide (SO_2):** SO_2 is released through the processing of sulphurous mineral ores and through industrial processes that involve the burning of sulphurous fossil fuels. The vast majority of SO_2 in the atmosphere comes from human sources.
- **Carbon monoxide (CO):** CO is released through combustion of fuels. It is also a by-product of numerous industrial and agricultural processes.

Environmental outcomes

Emissions of air pollutants increases their concentration in the atmosphere. This directly reduces ambient air quality and causes secondary phenomena such as smog and acid rain. Primary air pollutants react with other elements in the air to form secondary air pollutants, the most important of which are listed below.

Secondary air pollutants

- **Sulphates (SO_4-) and nitrates (NO_3-):** These are formed from SO_2 and NOx respectively and are both types of $PM_{2.5}$.
- **Ammonium (NH_4+):** Ammonia production is mainly a result of agricultural activities. It is commonly associated with the waste of cattle and other livestock. Some nitrogen-based fertilisers can also result in NH_3 emissions. NH_3 is largely deposited into soil or water soon after emission, but a small portion may react with ambient air to form ammonium ions (NH_4+), which also contribute to the level of $PM_{2.5}$.
- **Ozone (O_3):** Ozone is formed via a non-linear reaction between VOCs and NOx in the presence of sunlight.

Impacts on society

Increased concentration of pollutants can adversely affect people in various ways:

Human health: Respiratory diseases caused by air pollution have significant societal costs. The damages include increased incidents of chronic diseases, such as asthma and bronchitis. In some cases, these pollutants can lead to premature mortality from cardiovascular diseases, pulmonary diseases and lung cancer.

Visibility: Air emissions, particularly PM and O_3 precursors, reduce visibility through the formation of smog. Reduced visibility affects various forms of navigation. It also lowers people's enjoyment of recreational sites and their local neighbourhoods (i.e. disamenity).

Agriculture: Changes in the atmospheric concentration of certain gases can affect crop growth, thereby reducing yields. Acid rain can directly damage crops. It can also acidify soils with impacts on future growth.

The followings are also impact areas, although studies show that they are not material relative to the above impact areas.

Forests and timber: Changes in the atmospheric concentration of air pollutants can cause visible physical changes in tree growth and affect metabolism at the cellular level. Prolonged impacts can severely affect forest health. Acid rain directly damages forests and soils and can result in reductions in timber production.

Built environment: Acidic components in the air and acid rain can corrode materials used in construction (e.g. limestone, certain metals) and may lead to structural damage over time. Particulates can discolour property leading to reductions in aesthetic and cultural quality.

Other ecosystem services: Reduced air quality and increased acid rain damage forests and bodies of water. This can lead to reduced recreational enjoyment of the natural environment.

Users of this methodology should:

- Include all material impact drivers and impacts listed above and
- Include other material impacts.

2.2.3. QUANTIFICATION AND MONETARY VALUATION

This section covers the three steps in more detail: (i) measuring impact drivers, (ii) measuring changes in the state of natural capital and (iii) valuing impacts. For guidance on actions, see the Natural Capital Protocol.

Given the state of research and the available studies, it is not feasible to standardise all aspects of quantification. In the following, detailed examples are given to illustrate how quantification may work in practice.

(i) Measuring impact drivers

As described in the impact pathway, the context in which the pollutants are released plays a key role in the outcomes and the ultimate impacts they will have on society, especially on human health.

Therefore, context needs to be taken into account when measuring impact drivers. For example, pollutants released in a densely populated area are likely to be inhaled by a greater number of people and therefore, have a greater impact on society.

Users of this methodology should:

- Apply the rules outlined in the "General Paper" of this methodology (e.g. include all relevant value-chain levels),
- Include the six key pollutants: PM_{10}, $PM_{2.5}$, SO_2, NOx, NH_3, and NMVOCs and
- Define measurement categories that take into account differences in location category (e.g. population density, type of source) to a suitable degree.

They may:

- Include additional pollutants if material.

Guidance on data sources:

- Air emissions are best measured on site using direct, in-line measurement. However, this is rarely practical across entire value chains. Instead, the drivers of air pollution can be measured in order to indirectly estimate emissions indirectly. For example, the quantity and type of fuel together with the type of combustion engine can be used to calculate emissions from fossil fuel-based energy generation or transport.
- If direct data on emissions or impact drivers (e.g. fuel use) is not available, modelling techniques, such as EEIO analysis or industry/lifecycle assessments, can be used. Such approaches provide different levels of data specificity. For example, LCA databases are typically rich in data on specific plastics, but government agencies or the IPCC database are likely to provide more up-to-date information on electricity emission factors. Similarly, EEIO data are only as specific as the sector and geographical resolution used in the model.
- The availability of metric data will differ according to the company's level of control over the producers and users of this information. This control is likely to vary across a company's value chain. Table 4 summarizes the likely metric-data availability across the corporate value chain and the implications for contextual information.

When collecting data on direct emissions, the location category can be based on the facility's location. When using data from an input-output model, the economic sector can be mapped to the location categories.

Table 1: Available metric data for other air pollutants per value chain level

Value chain level	Data
Own operations	Direct measurement of drivers of air pollution, such as electricity and fuel use, should be possible. Estimation techniques relevant for the supply chain can be used if direct data are unavailable.
Immediate / key suppliers	Supplier questionnaires can be directed to areas of high materiality or those with limited high-quality data from other sources. Most companies do not directly measure air pollution, but they generally have information on, for instance, air-pollution drivers, fuel use, electricity consumption and waste sent for incineration. Emission factors are required to convert these data into tonnes of different air pollutants. If this kind of data is not available, immediate suppliers may be captured together with the rest of the upstream supply chain using EEIO.
Upstream supply chain	EEIO can be used to derive an approximation of emissions (in tonnes of different types of gases) based on a company's purchase ledger. LCA databases can be used for more process-specific data where appropriate. Other data sources include government and industry reports. Reports from the IPCC and the IEA may be particularly relevant.
Downstream / use phase	The emissions associated with a company's product or service over its useful lifetime can also be calculated. For example, in the case of a vehicle, this would include impacts associated with fuel consumption, replacement parts and servicing. Use-phase estimates are often highly approximate, as they are based on limited data and sweeping assumptions. For some products, use-phase impacts can be highly material. In such cases, it is particularly important to be as accurate as possible.

The VBA methodology considers four location categories, each of which are adjusted to different countries:

- Urban,
- Peri-urban,
- Rural and
- Transport.

(ii) Environmental outcomes

The emission of air pollutants increases their concentration in the atmosphere. This directly reduces ambient air quality and causes secondary phenomena such as smog and acid rain. The dispersion of emissions depends on a range of factors, including (typical) meteorological conditions and whether the source is stationary. It needs to be modelled using suitable dispersion models.

Users of this methodology should

- Include all material outcomes and
- Use dispersion models that consider the local meteorological conditions as well as the persistence in air of pollutants to a suitable degree.
- In practice, it may not be possible to develop own models and secondary sources may be needed. In this case, users should review secondary data for limitations and suitability for impact valuation purposes.

(iii) Impacts on society and monetary valuation

The impact on society will depend not only on environmental outcomes but also on the type of impact category. Therefore, different impacts are addressed separately below.

I. Impacts on health from precursors of particulate matter formation

The impact drivers that must be considered when assessing these impacts are PM, NOx, SO_2, and NH_3.

An air dispersion model is needed to determine the change in primary and secondary pollutant concentrations over a specified area. The dispersion model should consider the local meteorological conditions as well as the persistence of pollutants in the air. The impact on human health depends on the concentration and should be modelled using a suitable dose-response function.

- Wind speed and direction: A six-hour moving average for the year is needed for the dispersion model.
- Mixing height: Two observations per day, one day per month for the year are required for the dispersion model.
- Rainfall: Hourly rainfall is required for the dispersion model.

An estimate of the number of people affected is produced by overlaying a population-density grid that describes the demographics in the location of interest. When modelling a specific location (i.e. a city), population-density information is used to create a dense urban grid i.e. (city centre), a peri-urban grid (i.e. city outskirts) and a rural grid (i.e. surrounding rural areas).

The air pollutant dispersion is modelled using Sim-Air ATMOS 4.0, which can account for local meteorological and demographic conditions. Sim-Air ATMOS 4.0 is a simplified version of a US

National Oceanic and Atmospheric Administration model that has been adapted for relatively rapid assessment. It has been widely used in Asia and is applicable globally.

When the location of the emission source is not precisely known (e.g. only country level data are available) but the nature of the economic activity is known (e.g. automotive parts manufacturing), the same approach can be applied to each of the major locations of the polluting activity within a country. Averages weighted by the proportion of industrial production in each location can then be produced.

II. Impacts on health from precursors of ozone formation

The impact drivers that must be considered when assessing these impacts are NOx and VOCs.

Environmental outcomes and societal impacts are evaluated in one step using a multivariate transfer function, which extends Muller and Mendelsohn's (2007) societal cost estimates beyond the US to give global coverage (subject to the availability of local contextual data).

Secondary pollutant formation is too complex to model directly. Therefore, expanding on existing damage cost estimates is more accurate and practical. To estimate the societal cost of air pollution, a transfer function based on one of the most comprehensive assessments of air pollution's societal costs to date and used as a substitute for a model of atmospheric chemistry has been developed.

$$\begin{aligned}\boldsymbol{ln}\,(\boldsymbol{societal\ cost})_{\boldsymbol{i}} &= \boldsymbol{\alpha} + \boldsymbol{\beta_1}\,\boldsymbol{ln}(\boldsymbol{population\ density}) + \boldsymbol{\beta_2}\,\boldsymbol{ln}(\boldsymbol{median\ income}) \\ &\quad + \boldsymbol{\beta_3}\boldsymbol{ln}\,(\boldsymbol{ozone\ concentration})\end{aligned}$$

The three explanatory variables cover the main drivers of ozone impacts on mortality and morbidity:

- Population density is a proxy for the number of people likely to be in contact with the pollutant.
- Median household income is a proxy for the impact of budget constraints on people's WTP to avoid the adverse health impact of air pollution.
- Ambient ozone concentration reflects the fact that the impact of additional ozone depends on the absolute level of ozone in the atmosphere at a given location. While this function does not directly account for meteorological factors that affect the dispersion of pollution, the ambient ozone concentration will provide an indirect indication. This is because in areas where meteorological factors are such that pollution disperses, the ambient concentration of ozone will be lower.

Although the transfer function is derived from physiological impacts in the US, it is applicable to the rest of the world because the effects of air pollutants on the health of a given population are driven by human physiology and are therefore relatively consistent between countries.

The societal cost varies with ambient O_3 levels and income levels. Both variables show significant variation in the US sample, providing a credible basis for estimation of societal costs elsewhere.

III. Impacts on visibility

The impact drivers that must be considered when assessing these impacts are PM, NOx, NH_3, SO_2 and VOCs.

Air emissions, especially PM and O_3 precursors, contribute to reduced visibility through the formation of smog. Reduced visibility affects various forms of navigation. It also reduces people's enjoyment of recreational sites and the neighbourhoods in which they live (i.e. disamenity).

The societal cost of air pollution's impact on visibility is estimated directly from emissions using function transfer (described below) in a single step without intermediate estimates of environmental outcomes. The use of a transfer function is the most practical method for estimating the societal cost of air pollution from reductions in visibility. This is because there are no consistent primary studies estimating WTP to avoid reductions in visibility for specific locations across the globe. In addition, visibility-impairing pollutant formation is difficult to model directly on a global scale. Therefore, existing damage cost estimates can be used to derive an approximate indication of impacts.

$$\begin{aligned} ln\ (societal\ cost)_i \\ &= \alpha + \beta_1\ ln(population\ density) + \beta_2\ ln(median\ income) \\ &+ \beta_3\ ln(annual\ rainfall) + \beta_4\ ln(average\ annual\ maximum\ temperature) \\ &+ \beta_5 ln\ (ambient\ ozone\ concentration) \end{aligned}$$

Table 2: Data sources

Data	Data source(s)
Population density	World bank (2014a)
Annual rainfall	Weather Base (2014)
Median income	World Bank (2014b)
Maximum temperature	Weather Base (2014a)
Maximum O_3 concentration	Baldasano et al. (2003)

Key assumptions

US median income is used and adjusted for GDP per capita to approximate median income in other countries. This assumes the proportion of GDP attributed to income in the US is comparable to that of other countries. This is imperfect at best but, it is considered acceptable given the low materiality of the impacts.

The relationships between visible distance and air pollutants implied by US data are applicable to the rest of the world. This is reasonable, as the chemical reactions in the atmosphere that form smog and reduce visibility are consistent around the world. The social costs of negative visibility effects will vary with ambient O_3 concentration, local income, population density, temperature and rainfall.

Each of these factors shows significant variation in the US sample, thereby providing a basis for estimating a function to describe how WTP changes based on these variables that can be applied elsewhere.

IV. Impacts on agriculture

The impact drivers that must be considered when assessing these impacts are NOx and VOCs.

Changes in the atmospheric concentration of certain gases can negatively affect the growth of crops, leading to reduced yields. Acid rain can directly damage crops and acidify soils, thereby affecting future growth.

Environmental outcomes and the impact on agricultural productivity are evaluated in one step using value transfer. The average of marginal damage costs from Muller and Mendelsohn's (2007) US dataset is taken and adjusted for purchasing power differences between countries. To transfer the damages per tonne for NOx and VOC for agriculture, the societal costs provided by Muller (2012) are adjusted for GNI (PPP) to account for differences in purchasing power. The econometric estimation of transfer functions was explored, but variables with enough explanatory power to explain how crop yields varied with ozone were not found. Therefore, a simpler value transfer approach was adopted.

The only variable used in this approach is country PPP data relative to the US. Key variables in the underlying analysis of agricultural damage costs (see table 3) include crop type, health, prices, productivity and ambient pollutant concentrations.

Table 3: Societal costs

Pollutant	**Societal cost per tonne (USD, 2011)**
NO_x	28.67
VOC	14.96

Source: Muller (2012)

The impacts of air pollution on agriculture are affected by variables that are difficult to directly model. It is not possible to adequately represent these variables using a multivariate transfer function from Muller and Mendelsohn's (2007) US dataset because the range of crops used in their analysis is not sufficiently representative of global crop types that are sensitive to air pollution. Therefore, this methodology opts for a simple and transparent value transfer approach that takes the average of marginal damage cost estimates from Muller and Mendelsohn (2007) and adjusts them internationally for purchasing power parity. Ascribing a value to impacts on agriculture acknowledges that an impact exists. Consistent with the study on which it is based, the impact tends to have very materiality in the results. However, this approach is highly approximate. If air pollution impacts on agriculture were identified as potentially significant during the project, then this approach should be revisited.

Users of this methodology should include:

- All material impacts.

For each of these, users should:

- Apply the rules outlined in the “general” section of this methodology and
- Select appropriate sources/studies (e.g. dose-response functions) to model these impacts, taking the application of impact valuation into account.

2.2.4. SOURCES

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Topic-specific details

2.3. WATER CONSUMPTION

- *Topic description*
- *Impact pathway*
- *Quantification and monetary valuation*
  - *Measuring impact drivers*
  - *Environmental outcomes*
  - *Impacts on society and monetary valuation*
- *Sources*

2.3.1. TOPIC DESCRIPTION

All corporate activity directly and indirectly relies on water availability. Water consumption is defined as the volume of water that is evaporated, incorporated into a product or polluted to such an extent that it is unusable (Mekonnen & Hoekstra, 2011). Water consumption reduces the amount of water available for other uses. Depending on the level of competition and the socio-economic context, this can have consequences for the environment and people. This methodology focuses on valuing the impacts of corporate water consumption.

Water is a fundamental requirement for life and the right to water is a basic human right. Other goods or services cannot serve as substitutes for the water required to sustain life. Consequently, its worth is infinite and beyond the boundaries of economics. However, after basic needs are met, the marginal value of water can be quantified. For example, we can distinguish between the value of water in locations where (and, at times, when) there is competition among users for water and those where there is a plentiful supply. The difference in impacts associated with water consumption in these locations provides useful information for companies seeking to minimise their negative impacts and their exposure to water risks in their value chain.

As demonstrated in the following discussion, water availability is typically not the sole or most significant driver of impacts of corporate water consumption. Areas in which the water-consumption impacts are the highest are often characterized by poor sanitation, inadequate water-supply infrastructure, basic public health care, poverty and high malnutrition. Responsibility for the impacts of water consumption is shared by corporate users, other water consumers and, most importantly, local and national governments. The methodology presented here estimates the impacts of corporate water consumption taking the local context as a given and it does not consider the level of responsibility for the prevailing socio-economic context[9].

2.3.2. IMPACT PATHWAY

In order to value corporate environmental impacts on society, the link between water consumption and impacts on humans via environmental outcomes must be established. This is reflected in the impact pathway shown in Figure 3.

[9] In some cases the socio-economic development associated with the corporates activities may actually reduce vulnerability to water stress within the community. These benefits are not considered here but could be measured through social and economic impact indicators.

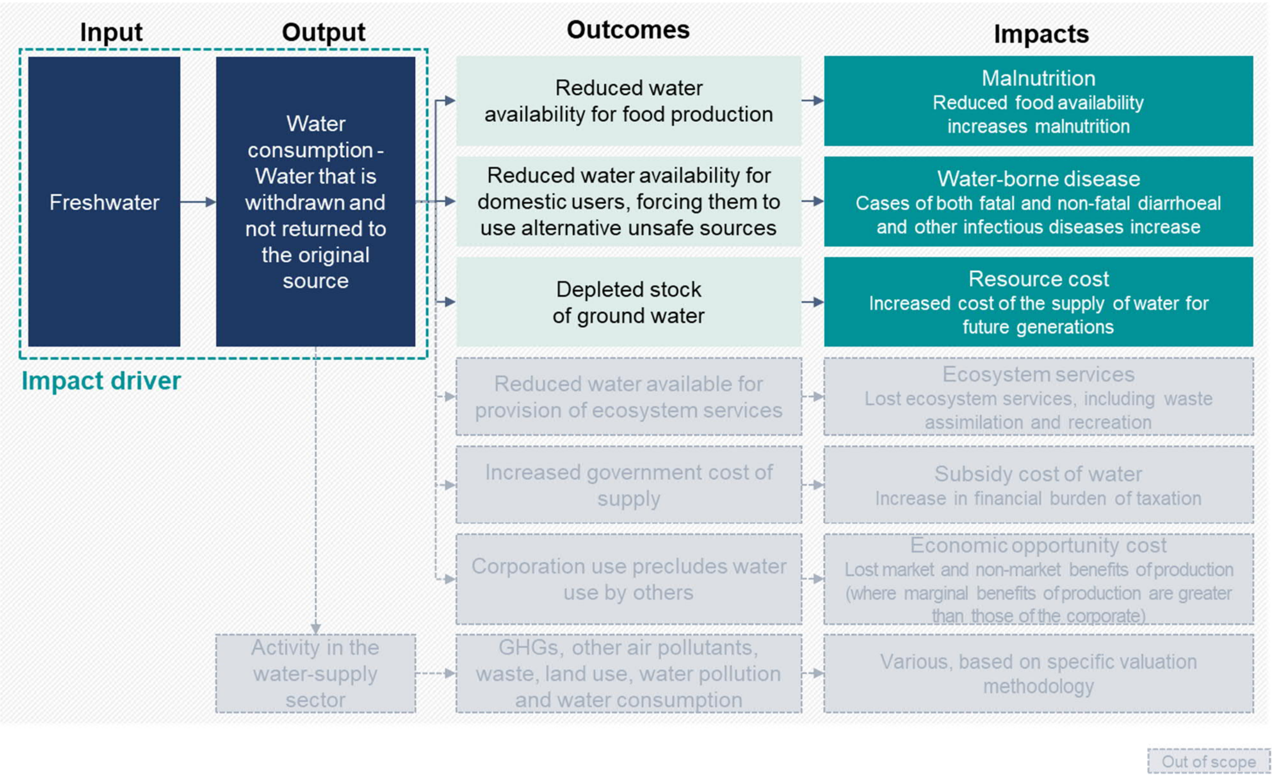

Figure 3: Simplified impact pathway water consumption

(i) Impact drivers: The volume and location of corporate water consumption
(ii) Reduced availability of water for other users, depletion of groundwater reserves at an unsustainable rate and the impact of the water supply sector.
(iii) Human health impacts, future costs to society of alternative water sources and impacts via GHGs, air pollution and waste from the water supply sector.

Impact drivers

It is important to distinguish between water use and water consumption. This methodology measures the impact of water consumption.

The WRI (2013) defines the two measures as follows:

- Water use “describes the total amount of water withdrawn from its source to be used. Measures of water usage help evaluate the level of demand from industrial, agricultural, and domestic users”.
- Water consumption “is the portion of water use that is not returned to the original water source after being withdrawn. Consumption occurs when water is lost into the atmosphere through evaporation or incorporated into a product or plant (such as a corn stalk) and is no longer available for reuse”.

Environmental outcomes

Water consumption depletes bodies of water, which may, in turn, lead to water stress or reduce the availability of water within the ecosystem. This depends on the location and type of water body (e.g. groundwater, surface water) as well as on the state of the local environment. Environmental outcomes are addressed in more detail in the quantification section.

Impacts on society

Where corporate water consumption reduces the clean water available to other users reliant on the same source, societal impacts could include the following:

Human health – malnutrition: In water-scarce areas, corporate water consumption may reduce the water available to agricultural users, thereby reducing yields. In areas dependent on local food production, this may lead to an increase in malnutrition.

Human health - infectious water-borne diseases: A reduction in clean water availability may force people to use other water sources. Depending on its quality, this may lead to cases of diarrhoea and other water-borne diseases. Although this impact is associated with polluted water, the primary corporate driver of impact is the reduction in clean water availability and is therefore considered under this Water Consumption methodology rather than in the Water Pollution methodology. Impacts associated with direct release of pollutants to water by corporate are considered in Water Pollution.

Resource depletion: Some communities depend on groundwater and extract it at an unsustainable rate, leading to groundwater depletion and an inflow of saline water. Over-exploitation of non-renewable water supplies will have future impacts associated with the increased scarcity and cost of supply unless other sources are secured.

The following are also drivers. However, studies show that they are not material relative to the above impact areas or there are no robust data sets that allow for the estimation of their impacts on a global level.

Other ecosystem services: Removal of fresh surface water can reduce the functioning of ecosystems, especially in riparian areas. The associated loss in ecosystem services may lead to a reduction in other ecosystem services and associated impacts for the local population, including, for instance, market- and non-market-related losses in fishing and recreation.

Subsidy cost of water: Water pricing rarely reflects the financial cost of the water supply and it is frequently subsidised. Therefore, corporate use increases the financial burden for taxpayers.

Economic opportunity costs of water: Where there is direct competition for water and the corporation using the water is not the most economically productive user (based on the marginal private and public benefits of production), then water use has an opportunity cost.

Environmental impacts of the water-supply sector: The supply of water to corporations requires energy and raw materials, which have other environmental impacts, including GHG emissions, air emissions, waste, water pollution and land use.

2.3.3. QUANTIFICATION AND MONETARY VALUATION

This section covers the three steps in more detail: (i) measuring impact drivers, (ii) measuring changes in the state of natural capital and (iii) valuing impacts. For guidance on actions, see the Natural Capital Protocol.

Given the state of research and available studies, it is not feasible to be prescriptive on all aspects of quantification. In the following, detailed examples are provided to illustrate how quantification may work in practice.

(i) Measuring impact drivers

Measurement of corporate water consumption is best done at the point of the water consumption. This may be possible for a company to do within its own operations and to gain this information from its direct suppliers. However, when assessing more distant parts of the value chain, estimation techniques may be required to quantify the volume of water consumed as an indirect result of the company's activities.

Modelling techniques, such as lifecycle analysis (LCA) and environmentally extended input-output (EEIO) tables, can be used. Such approaches offer different levels of data specificity depending on the application. For example, LCA databases are typically rich in data on plastics, while the statistical agencies of industry bodies are likely to have more up-to-date information on the intensity of industrial water use. Similarly, EEIO data are only as specific as the country and sector resolution provided in the model.

Table 4: Available metric data for water consumption per value chain level

Value chain level	Metric data
Own operations	Direct measurement of water consumption. Estimation techniques relevant for the supply chain can be used if direct data are unavailable.
Immediate/ key suppliers	Supplier questionnaires can be directed to areas of high materiality or those with limited high-quality data from other sources. Most companies can quantify their water consumption based on their water bills. Any water that is released unpolluted back into the same fresh water source should be excluded from this measure of water consumption. If this kind of data is not available, immediate suppliers may be captured together with the rest of the upstream supply chain using EEIO.
Upstream/ supply chain	EEIO can be used to derive an approximation of corporate water consumption based on a company's purchase ledger. LCA databases can be used for more process-specific data where appropriate. Other data sources include government and industry reports. Reports from the IPCC may be particularly relevant.
Downstream/ use phase	It is necessary to estimate the probable water consumption associated with the product or service over its life. For instance, for an item of apparel, this is the expected amount of water used to wash the item over its lifetime.

Given the close links between specific environmental outcomes and societal impacts, they are presented together for the key impacts identified above.

I. Malnutrition

(ii) Quantifying the environmental outcomes

The environmental outcome of water consumption is reduced water availability for other users. For the malnutrition impact pathway, we focus on reduction in the fresh water available to the agricultural sector (Pfister et al., 2009). This reduction is calculated by considering the following variables:

- The volume of corporate water consumption (m^3) per capita and
- The level of water stress in the focal watershed using a water-stress index (WSI).

This approach assumes that increased corporate water consumption directly reduces the water available to agricultural users (based on the WSI), which is likely to be the case if the water infra-structure is linked to the same source for both users.

Step 1: Calculate the water depravation factor

The water deprivation factor (WDF) estimates the amount of water that the agricultural sector is deprived of as a result of water consumption by others, in relation to the sector's total water consumption ($m^3_{deprived}/m^3_{consumed}$).

The WSI_i applied by Pfister et al. (2009) "indicates the portion of consumptive water use that deprives other users of freshwater". Therefore, multiplying the WSI by the proportion of water used by the agricultural sector in the same watershed indicates the amount of water that the agricultural sector is deprived of when water is consumed by a different user.

The WDF in watershed "i" is calculated by multiplying the water stress index (WSI_i) by the fraction of water consumption by agriculture in that watershed, $WU_{\%, agriculture, i}$.

Equation: Water depravation factor

$$\boldsymbol{WDF_i = WSI_i \times WU_{\%,agriculture,i}}$$

(iii) Estimating the impacts on society

Step 2: Calculate the effect factor

The effect factor (EF) is the number of malnourishment cases caused each year by the deprivation of one cubic metre of freshwater (in capita.year/$m^3_{deprived}$). It is a function of the water required to avoid malnutrition ($WR_{malnutrition,i}$) and the human development factor related to vulnerability to malnutrition ($HDF_{malnutrition,i}$).

The HDF uses the relationship between the human-development index and the malnutrition rate expressed in DALYs to provide an indication of malnutrition vulnerability. The factor ranges from 0 to 1.

$WR_{malnutrition}$ is the minimum per capita water requirement of water for the agricultural sector to avoid malnutrition. Therefore, its inverse represents the number of cases of malnutrition caused by each cubic metre of water deprived (capita.year/m^3). Pfister et al. (2009) derive it from a country-level dataset and it is consistent with the values found by Yang et al. (2003) and FAO (2003). The equation Effect factor summarises the calculation of the EF.

Equation: Effect factor

$$\boldsymbol{EF_i = WR_{malnutrition}{}^{-1} \times HDF_{malnutrition,i}}$$

Step 3: Calculate the damage factor

The damage factor (DF) estimates the amount of harm per case of malnutrition. It is derived from a linear regression of the malnutrition rate ($MN_{\%}$) and the DALY malnutrition rate ($DALY_{malnutrition, rate}$) at a country level to arrive at a conversion from cases of malnutrition to DALYs. The regression gives a damage factor of 0.0184 DALYs/capita.year.

Equation: Damage factor

$$\boldsymbol{DF_{malnutrition}} = \frac{\boldsymbol{DALYs}}{\boldsymbol{capita.year}}$$

Step 4: Calculate the human health factor

The human health factor (HHF) brings together the outputs of Steps 1 to 3. The HHF describes the DALYs per unit of water consumed. It is a product of the WDF, the EF and the DF (see equation Calculating the human health damage factor of malnutrition with units in parentheses).

Equation: Calculating the human health factor of malnutrition (units in parentheses)

$$\boldsymbol{HHF_i}\left(\frac{\boldsymbol{DALYs}}{\boldsymbol{m^3}_{consumed}}\right) = \boldsymbol{WDF_i}\left(\frac{\boldsymbol{m^3}_{deprived}}{\boldsymbol{m^3}_{consumed}}\right) \times \boldsymbol{EF_i}\left(\frac{\boldsymbol{capita.year}}{\boldsymbol{m^3}_{deprived}}\right) \times \boldsymbol{DF_{malnutrition}}\left(\frac{\boldsymbol{DALYs}}{\boldsymbol{capita.year}}\right)$$

The methodology does not explicitly include the ability to import food from other regions when the local agricultural sector does not provide enough food. This is indirectly accounted for by including the prevailing malnutrition rate, which is lower in countries where it is easier to import food from alternative sources.

As described above, the number of cases of malnutrition is estimated in disability adjusted life years (DALYs) using a regression of country-level malnutrition cases and DALYs associated with malnutrition per m3 of consumed water. A monetary value for each DALY is calculated based on the value of a statistical life (VSL) and the lost DALYs associated with the VSL estimate to produce an estimate of the welfare impacts. See section 3.1.3 for more details on the valuation of DALYs.

II. Infectious water-borne disease

(ii) Quantifying the environmental outcomes

This impact pathway estimates the volume of water that could be withdrawn by domestic users if it were not consumed by corporate users using the WSI for the location of consumption. Therefore, the WSI is a measure of the competition for water between corporate users and other users (Pfister et al., 2009). The volume of water not available to domestic users is assumed to be equal to total corporate consumption adjusted by the WSI. This assumption holds when conditions (e.g. infra-structure) are such that when a corporate user reduces its water consumption, a domestic user can access that water. This may not be the case when corporate users do not consume water from the same infrastructure or source as domestic users. However, the alternative approach of modelling water consumption infrastructure is constrained by data availability and may increase associated errors.

(iii) Estimating the impacts on society

An econometric approach is taken to assess the influence of corporate water consumption on the prevalence of water-related disease in different countries. Quantile regression analysis is used to explain the variation in the observed DALYs per capita rate associated with water-borne infectious diseases. The results from selected quantiles can be applied to other locations using the level of water-borne disease to assign the most appropriate coefficients.

Step 1: Construct a regression model

The explanatory variables explain the socio-economic drivers of water-borne disease. They are:

- Domestic water use,
- Health expenditure,
- Prevalence of undernourishment,
- Government effectiveness and
- Water stress level using a water stress index (WSI).

Equation: Water-borne disease regression model

$$\ln \boldsymbol{Dalys} = \boldsymbol{\alpha} + \boldsymbol{\beta_1} \ln \boldsymbol{dww} + \boldsymbol{\beta_2} \ln \boldsymbol{undernour} + \boldsymbol{\beta_3} \ln \boldsymbol{healthexp} + \boldsymbol{\beta_4} \ln \boldsymbol{wsi} + \boldsymbol{\beta_5} \ln \boldsymbol{govteff}$$

The derived relationship is used to predict the decline in the prevalence of water-borne diseases if the water that corporations deprive domestic users of (based on the WSI) is reallocated to domestic users. The resultant change in DALYs per capita is valued and allocated across total corporate water use to give a welfare impact per m^3.

If we apply an ordinary least squares (OLS) analysis, we would overestimate impacts for countries with low levels of water-borne disease and underestimate impacts for countries with high levels. Therefore, we use a quantile regression to better reflect the impacts in different locations. Where locations have levels of disease close to the cut-off between the quantiles, we suggest using a sensitivity analysis to explore the potential impacts using the coefficients in both quantiles.

Step 2: Predict how water-borne disease would change if corporate use decreased

In this step, we use the relationships derived from our regression analysis to estimate how the prevalence of disease would change if the portion of water used by corporations that deprives other users of water is reallocated to domestic users. While the relationship is derived at a country level, it can be applied to estimate impacts at a more location-specific level if data are available.

Step 2.1: Predict the baseline prevalence of disease

Our regression analysis shows that for locations in which the prevalence of disease is below a certain level (0.0016 DALYs/capita/yr for diarrhoea and 0.0009 DALYs/capita/yr for non-diarrhoea), the level of domestic water use does not influence the prevalence of disease. Therefore, for these locations, corporate water use has no impact on the prevalence of water-borne diseases. While these values are empirically derived, the cut-off is somewhat arbitrary, as it is based on observed changes in significance. For locations close to these cut-off values, we recommend conducting sensitivity analyses to understand the impacts using the fiftieth-percentile relationship.

For locations with disease levels above these values, we predict the DALYs per capita per year for each group of diseases using our model. We use this predicted value in the following calculations because it provides a “fairer” estimate.

In order to predict the values, we use the equation above. We input the values for domestic water use, health expenditure, undernourishment, government effectiveness and the WSI, and multiply them by the appropriate coefficient depending on whether the actual prevalence of disease falls between the thirtieth and sixtieth percentiles or between the sixty-first and one hundredth.

Step 2.2: Re-estimate the prevalence of disease with corporate water use reallocated to domestic users

The second part of this calculation involves re-predicting the prevalence of disease after including the water used by corporations. We multiply the region’s total corporate and industrial water use[10] by the WSI to derive the portion that deprives other users of water. We then reallocate that quantity of water to domestic users in order to hypothesise how much lower DALYs per capita per year would be if that water was available.

The estimated reduction in DALYs per capita per year is multiplied by the region’s population and allocated to the total corporate water use (not just the portion that is depriving others). This gives a DALY per m^3 of corporate water consumption in a given year.

Step 3: Apply the value of a DALY

To value the impacts of disease, we assign locally-specific DALY values to the DALY/m^3 estimates generated in the previous step. The monetary value of each DALY is based on the value of a statistical life (VSL) and the lost DALYs associated with the VSL estimate to produce a welfare estimate of the impacts.

Valuation of DALYs

Having established the number of DALYs lost as a result of water consumption owing to malnutrition and waterborne disease, we assign a monetary value to those DALYs to estimate the societal cost of water consumption.

[10] The calculation is non-linear. Therefore, in order to calculate an average impact per unit of corporate water consumption, the data must represent the total industrial and corporate water for the same geographical region as the other data inputs. If 1 m^3 is used, this would give the marginal impact.

Health economists and policy makers typically use DALYs to understand the relative severity of health conditions. They often use them to compare the cost effectiveness of investments (cost savings per avoided DALY). Lvovsky et al.'s (2000) publication for the World Bank builds on this to present a method for estimating the welfare value of DALY savings.

Lvovsky et al. (2000) derive the DALY value from the value of a statistical life (VSL) based on the number of lost DALYs associated with that lost life (*equation Value of a DALY*). This approach has been applied in a government policy context by Pearce et al. (2004) to help evaluate the EU's REACH policy (Registration, Evaluation and Authorisation of Chemicals). The following discussion presents our application of this approach. The VSL values used are consistent with those used in the other environmental-impact methodologies.

Equation: Value of a DALY

$$\textit{Value of DALY} = \frac{VSL}{\textit{Number of DALYs lost}}$$

The OECD nations VSL estimate of US$3.4m (2011, inflated from 2005) (OECD, 2012) is the basis of our DALY valuation. The OCED estimate is based on a meta-analysis of studies which consider acceptance of risks to life and extrapolate to give a VSL (e.g. wage premiums to accept working in riskier environments). The median age of individuals in the studies is 47 years old, with a life expectancy is 78, such that the resulting estimate of VSL is associated with 31 years of lost life.

In order to estimate the value, the number of years lost is converted to DALYs. A year of disability-free life does not hold the same number of DALYs for all ages. People place a higher value on avoiding disability between the early teens and the mid-50s (age weighting for DALYs). The DALYs are therefore age weighted (Prüss-Üstün et al., 2003).

Prüss-Üstün et al. (2003) provide a formula and suggested coefficients for calculating the relative weighting of each year of life (X_w), which is set out in the following equation.

Equation: Age-weighting formula for calculating DALYs

$$X_w = Cx^{-\beta x}$$

where x is the age in years. The suggested coefficients are C= 0.1658 and β = 0.04. This formula is used to calculate the relative weight applied to each of the 78 years of life expectancy associated with the OECD's VSL estimate.

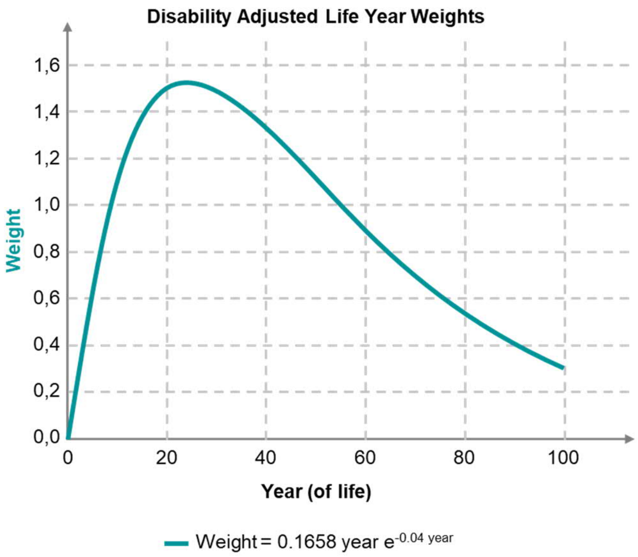

Figure 4: Age weighting for DALYs

People are willing to pay more to avoid disability today than to avoid it the future. Therefore, a discount rate of 3% (as per the social discount rates used in the other methodologies) is applied to future years beyond the age of 47. The discounted age weighting is calculated as per the following equation.

Equation: Discount age weighting for DALYs

$$X_{wd} = \begin{cases} Cx^{-\beta x} \; when \; x < 47 \\ Cx^{-\beta x}/\left(1 + 0.03^{x-47}\right) \; when \; x \geq 47 \end{cases}$$

The discounted, age-adjusted proportion of life lost (PLL_{wd}) is calculated using equation age adjusted years of lost life. This represents the proportion of life lost for a person who expected to live to 78 but died prematurely at 47.

Equation: Age-adjusted years of lost life

$$PLL_{wd} = \left(\sum_{x=47}^{78} X_{wd}(x) \middle/ \sum_{x=0}^{78} X_{wd}(x) \right)$$

To calculate the number of DALYs, PLL_{wd} is multiplied by life expectancy. Table 8 contains the steps in the calculation that result in the DALY value of USD 185,990 (in 2011 USD).

Table 5: Steps in the calculation that results in the DALY value of USD $185,990 (in 2011 USD)

Age at time of premature death	Life expectancy	Proportion of life lost (PLL_{wd})	DALYs lost ($PLL_{wd} \times$ $life\ expectancy$)	VSL	DALY value $\left(\frac{VSL}{Number\ of\ DALYs\ lost}\right)$
47	78	23.4%	18.3	USD 3.4m	USD 185,990

When adjusting the DALY value, which is derived from the VSL, we need to control for socio-economic conditions. As the underlying willingness-to-pay studies are from specific countries, the income of the respondents in the study places an implicit limitation on the value. If the study is conducted in a relatively high-income country, it will elicit a higher VSL than if it is conducted in a relatively low-income country. Theoretically, when adjusting the value to other countries, controls for the income level should be included. The OECD suggests that a transfer factor using PPP-adjusted GNI per capita with an income elasticity of health between 0.4 and 0.8 should be applied. Some arguments suggest that adjustments should be made to account for differing appetites in different countries and contexts. However, we have not identified a reliable dataset to control for this aspect.

If an income adjustment is included, then we can account for differences between income per capita adjusted for PPP. An income elasticity of 0.6 is recommended as a central estimate of the values presented by the OECD (2010).

Equation: Income-adjustment transfer function

$$Transfer\ function = \left(\frac{GNI_a}{GNI_b}\right)^e$$

where:

GNI_a = Gross National Income per capita of the new policy site, adjusted for purchasing power parity

GNI_b = Gross National Income per capita of the reference site, adjusted for purchasing power parity and

e = Income elasticity of the willingness to pay for health or life.

The ethical concern is whether it is appropriate to adjust the value of life or health between countries because, by doing so, one values the lives of those in lower-income countries less than the lives of those in higher-income countries.

A suggested approach to addressing this issue is:

Situation 1 – aggregating studies from different countries in order to build a health-impact model. These values should be comparable and, therefore, expressed in "international" dollars (as suggested by the OECD).

Situation 2 – transferring a health-impact value to different countries.

2.1) If the analysis is meant to cover various countries, the value should still be comparable and, therefore, not adjusted.

2.2) If the analysis is meant to only cover one country, it should be adjusted in order to reflect local constraints.

Situation 3 – presenting results locally or internationally.

Create two sets of coefficients:

- The global consolidation of results: use an international set of values in which adjustments are made to use international dollars as the basis for the values.
- For presenting in-country impacts to local stakeholders: use a local set with adjustments for local income.

III. Groundwater depletion

Some communities depend on groundwater and extract it at an unsustainable rate. This leads to groundwater depletion and an inflow of saline water. Over-exploitation of non-renewable water supplies will have future impacts associated with the increased scarcity and cost of supply unless other sources are secured.

(ii) Quantifying the environmental outcomes

The rate of groundwater depletion and the expected time to depletion are used to estimate the future annual shortfall in the water supply. This assumes that new groundwater reserves with the same cost of extraction will not be discovered. This holds for most locations where enough hydrological data are present to identify depletion.

We utilise a geographic information system (GIS) AQUASTAT dataset (FAO, 2012) to estimate the national average groundwater-scarcity ratio. This is the ratio of the depletion rate to the replenishment rate of groundwater resources.

(iii) Estimating the societal impacts

To estimate the societal impact of groundwater depletion, we calculate the replacement cost as a lower-bound estimate of the likely societal impacts of groundwater depletion.

When the ratio is greater than 1 (i.e. groundwater is being depleted faster than it is being replenished), we take the following steps to estimate the societal cost per m^3 of corporate water consumption:

- Calculate the percentage reduction in groundwater abstraction required to achieve a sustainable groundwater-scarcity ratio.
- Multiply the required reduction in groundwater abstraction by the percentage of the national water supply derived from groundwater using the AQUASTAT dataset (FAO, 2012) to estimate the percentage reduction in the national water supply required to achieve a sustainable groundwater-extraction rate.
- Apply the current cost per m^3 of water supply to the percentage as a cost-based approach to estimate how much it would cost today to avoid the unsustainable depletion of groundwater resources. The cost of the water supply for the US is based on data from the International Benchmarking Network and is PPP adjusted for other countries.
- The previous step results in an estimate of the cost per m^3 of the current supply that would have to be applied to reduce the impact of unsustainable groundwater consumption. This is adjusted by the ratio of water withdrawal (supply) to water consumption, as our valuation methodology values the impact per m^3 of water consumed.
- The impacts are then projected 50 years into the future and discounted to present day using a 3.5% social discount rate. See the VBA Methodology V0.1 – General Method paper for detail on the discount rate applied.[11]

The projected cost of supply is an appropriate proxy for the societal costs. While this is a likely to represent a lower bound of potential impacts, replacement costs are deemed an acceptable proxy when better information is not available. Given that these costs are used to represent a societal cost and are not intended to be an accurate assessment of technological solutions, we use a consistent measure to avoid arbitrary bias. This also assumes that profits from groundwater extraction are not ring-fenced as funding for the future supply of water.

As with any projection, there is significant uncertainty regarding the future societal impacts of groundwater depletion occurring today.

[11] Based on values chosen by governments like HM Treasury (2018): The Green Book – Central Government Guidance on Appraisal and Evaluation. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/938046/The_Green_Book_2020.pdf

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Topic-specific details

2.4. WATER POLLUTION

- *Topic description*
- *Impact pathway*
- *Quantification and monetary valuation*
  - *Measuring impact drivers*
  - *Environmental outcomes*
  - *Impacts on society and monetary valuation*
- *Sources*

2.4.1. TOPIC DESCRIPTION

Economic activity in all sectors results in direct or indirect discharges of substances into water (i.e. directly as a result of industrial processes and agriculture, or indirectly through the consumption of energy or resources). Despite improvements in some developed countries, water pollution is on the rise globally. Pollution and the degradation of water bodies can adversely affect human wellbeing and, thereby, carry a societal cost. In this paper, we present a methodology for identifying and valuing the costs of water pollution in monetary terms.

The impacts of water pollution are principally local or regional and highly dependent on the physical environment and local demographic exposure. For example, changes in the concentration of arsenic following a release depends on the size of the water body and the flow rate. The extent of its subsequent impact on people depends on the likelihood that local populations will come into contact with the polluted water.

2.4.2. IMPACT PATHWAY

In order to value corporate environmental impacts on society, the link between effluents in the water and impacts on humans via environmental outcomes must be established. This is reflected in the impact pathway shown in Figure 11. Water pollution follows a complex pathway, with multiple pollutants playing a role in multiple environmental and societal outcomes.

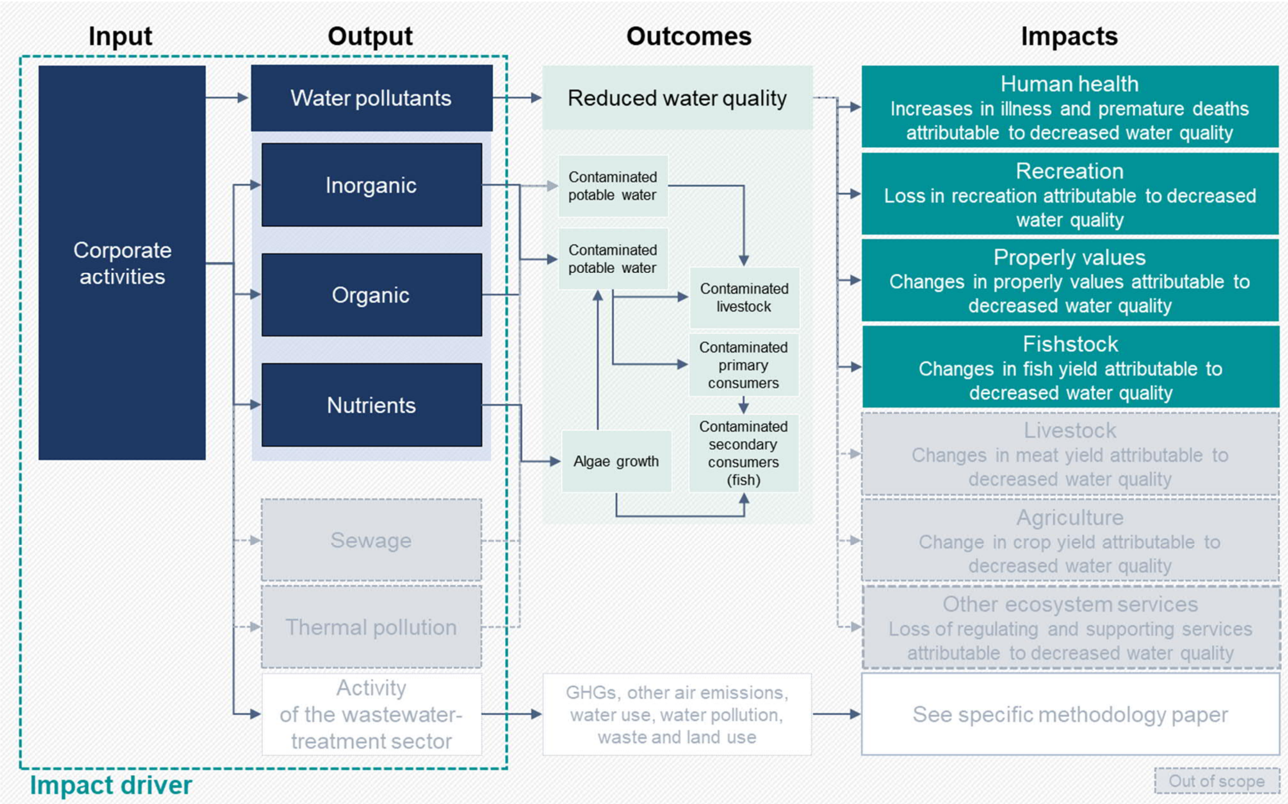

Figure 5: Simplified impact pathway water pollutants

i. Impacts are driven by the release of different types of chemicals and compounds into the water
ii. Changes in the state of natural capital (or outcomes): These are primarily identifiable as increased concentrations of pollutants and associated reductions in water quality. Secondary effects include the bioaccumulation of pollutants in the food web.
iii: Impacts on society are principally related to health but also include impacts on amenity values, recreation and the market economy.

Impact drivers

The most significant water-pollutant categories (in societal cost terms) are listed below, where they are sub-divided into toxic pollutants, nutrient pollutants, pathogens and thermal. Numerous individual pollutants can be categorised into these key areas.

Toxic pollutants: Organic and inorganic toxic substances, including heavy metals and chemical compounds that may persist or cause undesirable changes in the natural environment, bioaccumulate in the food web and have adverse effects on human health.

Nutrient pollutants: Nitrogen (N) and phosphorus (P) are basic building blocks of plant and animal proteins. In elevated concentrations, they can cause a range of negative effects, such as algal blooms that lead to a lack of oxygen in the water.

Pathogens: Coliforms are a broad class of bacteria, some of which are harmful, disease-causing organisms, such as Escherichia coli (E. coli). These can be released or encouraged to grow through discharges of inadequately treated sewage.

Thermal pollution: Discharges of water above or below the ambient temperature of natural water bodies can change the ecological balance.

Environmental outcomes

The discharge of pollutants into water bodies increases their concentration in those bodies of water, thereby directly reducing water quality and causing secondary phenomena, such as eutrophication.

Impacts on society

These changes can adversely affect people in several ways:

Human health impacts: The build-up of toxins in the human body through the prolonged ingestion of contaminated water or food can cause acute illness, cancer and a range of other conditions.

Impaired recreation value: The nutrient enrichment of bodies of water can cause excessive macrophyte growth, leading to eutrophication. This can affect the recreational use of water bodies due to health impacts from toxic blooms, water congestion from excessive vegetative growth, unfavourable appearance and/or unpleasant odours.

Property values: The eutrophication of water bodies can affect the value of local property (Krysel et al. 2003). Research also suggests that leisure and residential property can be devalued by as much as 20% as a result of consistently poor physical water quality (Wood and Handley, 1999).

Fish stocks: Eutrophication reduces the oxygen content of water and can lead to economic losses due to decreased fish yields as well as changes in species composition. In the US, annual losses suffered by the commercial fishing and shellfish industry as a result of nutrient pollution (attributable to lower yields from oxygen-starved waters and fluctuations in consumer confidence owing to tainted seafood) are estimated at more than USD 40 million (Hoagland & Scatasta, 2006).

Users of this methodology should:

- Include all material impact drivers and impacts listed above and
- Include other material impacts.

2.4.3. QUANTIFICATION AND MONETARY VALUATION

This section covers the three steps in more detail: (i) measuring impact drivers, (ii) measuring changes in the state of natural capital and (iii) valuing impacts. For guidance on actions, see Natural Capital Protocol.

Given the state of research and available studies, it is not feasible to be prescriptive on all aspects of quantification. The following provides detailed examples that illustrate how quantification may work in practice.

(i) Measuring impact drivers

As described in the impact pathway, the context in which the pollutants are released plays a key role in the outcomes and, ultimately, in the impact they will have on society, especially human health. This needs to be taken into account when measuring impact drivers. For example, the fate of pollutants released into water varies depending on geophysical parameters, while their impacts on people depend on local demographic characteristics.

Users of this methodology should:

- Apply the rules outlined in the “general” section of this methodology (e.g. include all relevant value chain levels),
- Include the key pollutants: antimony, arsenic, barium, benzene, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, nitrogen, phosphorus, selenium, vanadium and zinc
- Define measurement categories that take differences in location category (e.g. population density, type of water body) into account to a suitable degree.

They may:

- Include additional material pollutants.

Guidance on data sources:

- Effluent discharges to water are best measured on-site using direct, in-line measurement.
- However, aside from large, regulated facilities in developed countries, this is rarely a practical approach. Therefore, the drivers of water pollution can be measured in order to indirectly estimate discharges. For example, the quantity and type of chromium together with specifics on the tanning process can be used to calculate the load and toxicity of the discharges that result from the tanning of a hide. Similarly, typical loading factors can be used for phosphorous runoff associated with pastoral agriculture.
- If direct data on discharges or drivers of discharges are not available, modelling techniques such as LCA and EEIO analysis can be used. Such approaches offer different levels of data specificity depending on the application.

- The availability of metric data will vary according to the company's level of control over the producers and users of that data. This is likely to vary across a company's value chain. Table 9 lists examples of likely metric-data availability across the corporate value chain and implications for appropriate contextual information.

Table 6: Available metric data for water pollutants per value chain level

	Metric data
Own operations	For manufacturing facilities, information on effluents released into the water may be available in the company's management information, especially if the company is regulated. The estimation techniques relevant for the supply chain can be used if direct data are unavailable.
Immediate/ key suppliers	Supplier questionnaires can be directed to areas of high materiality or those with limited high-quality data from other sources. Most companies do not measure pollutants released into the water by substance unless regulations require them to do so. If regulated, wastewater-discharge figures can be found in the company's management information. The estimation techniques relevant for the supply chain can be used if direct data are unavailable.
Upstream/ supply chain	EEIO can be used to derive an approximation of effluent discharges to water based on a company's purchase ledger. LCA databases can be used for more process-specific data where doing so is deemed appropriate.
Downstream/ use phase	It is necessary to estimate the probable emissions associated with a product or service over its lifetime. For a laundry detergent, for instance, this may relate to direct chemical emissions to water after use. For other products, it may relate to indirect water pollution as a result of electricity consumption.
End of life/ re-use impacts	Different products are disposed of in different ways. Some may be recycled or upcycled (in which case, the allocation of emissions must be considered). Others will be sent to landfill or incinerators. These aspects need to be estimated based on the type of product and the disposal location.

(ii) Environmental outcomes

Effluents increase the concentration of chemical compounds in bodies of water. This directly reduces water quality and causes secondary phenomena, such as eutrophication and bioaccumulation in fish/biota. Substances may also be transferred from soil to plants and from air to plants. Given that the environmental outcomes and the impacts on society depend on the type of pollutant, they are addressed by pollutant below.

Users of this methodology should

- Include all material outcomes
- In practice, it may not always be possible to develop one's own models and secondary sources may be needed. In this case, users should review secondary data for limitations and suitability for impact-valuation purposes.

(iii) Impacts on society and monetary valuation

The impact on society will depend not only on environmental outcomes but also on the impact category. Therefore, different impacts are addressed separately below.

I. Toxic pollutant (organic and inorganic substances) valuation

The valuation module for toxic pollutants traces the pollutant from release to ingestion to induced health effects. It ultimately places a value on those health effects.

Pollutants can enter humans via a number of pathways including direct ingestion (e.g. drinking), indirect ingestion (e.g. via bioaccumulation in fish) and direct inhalation (e.g. of evaporated pollutants that were initially released into water). Once ingested (or inhaled), the health effects depend on the pollutant and its dose. We assign values to those health effects using published data on what individuals would pay to avoid them, which enables us to derive a total societal cost of water pollution.

(ii) Quantifying the environmental outcomes

The potential impacts of effluents on human health are modelled based on the chemical fate as they travel through different media (water, soil, air, food products) and the likelihood of human exposure. In order to evaluate the impacts of water pollution on people, we model the pollutant's movement through the environment, humans' exposure to the pollutant and the human health outcomes. The model's output is the pollutant-specific characterisation factor, which gives the number of health effects per unit of emitted pollutant.

Our preferred model for calculating characterisation factors is USEtox (Rosenbaum et al., 2011). Of the model options, it offers the largest substance coverage with more than 1,250 substances, and it encompasses more up-to-date knowledge and data on effect factors than other approaches. It was specifically designed to determine the fate, exposure and effects of toxic substances. In addition, it has the ability to consider spatial differences through the addition of country-specific parameters.

USEtox has been adopted for regulatory assessments in, for instance, the European Union's EUSES and for persistence-screening calculations as recommended by such bodies as the OECD (Klasmeier et al., 2006). This model is already widely used in Life Cycle Impact Assessment (LCIA), and it has been recommended by the UNEP and the SETAC (Jolliet et al. 2006). It was developed by a team of researchers from the Task Force on Toxic Impacts under the UNEP-SETAC Life Cycle Initiative to include the best elements of other LCA models.

We have built on the USEtox model in two relevant ways: increasing geographic specificity using country-level data from GLOBACK and limiting the model to emissions to water (to avoid double-counting with our other valuation methodologies, such as the air-pollution methodology). These modifications do not change any of the model's underlying calculations.

In USEtox, substances that have the potential to increase human disease have a characterisation factor (CF) (Rosenbaum et al., 2008). In LCIA, the mass of each chemical emitted is multiplied by a CF to provide the impact indicators. CFs are obtained using characterization models – in this case, USEtox – that represent the cause-effect chain starting from an emission followed by environmental fate, human exposure and the resulting effect on the exposed population. The CF in the USEtox model includes a fate factor (FF), an exposure factor (XF) and an effect factor (EF).

Equation: Characterisation factors for human health

$$\boldsymbol{CF = FF \times XF \times EF}$$

- The fate factor (FF) describes the amount of the contaminant in the air, water and soil (termed "environmental compartments"). It is calculated based on the substance's mobility and persistence in the environment.
- The exposure factor (XF) describes the contaminant intake of the human population due to the mass of substance in the environment. Essentially, it represents a substance's likelihood of interacting with a receptor.
- The effect factor (EF) describes the substance-specific dose-response to determine the change in lifetime disease probability due to changes in the lifetime intake of a pollutant.

Fate, exposure and effect factors are represented by individual matrices, which are multiplied to obtain a final characterisation factor. These characterisation factors represent the environmental outcomes for human toxicity.

Step 1: Calculate the fate factor

The fate factor estimates the amount of pollutant available for eventual intake by humans. It assesses the residence time of a substance in the water – the longer the pollutant remains, the more of it is available for ingestion (and inhalation) over a given period of time. Fate factors are expressed in residual mass per unit of emission. The outputs of this step are substance- and country-specific fate factors for fresh water and for marine water.

Four processes affect the available mass of a substance in water: adsorption/sedimentation, volatilisation, degradation and advective transport out of the water compartment (Henderson et al., 2010). Intermediate transfer rates and removal rates depend on the conditions in an environmental compartment and on the properties of the substance. A fate matrix calculates the intermediate transfer rates and removal rates against the substance-specific parameters and context-specific parameters. For example, substances that are easily transformed by micro-organisms have high degradation rates in water, while substances that are not susceptible to biodegradation persist in water. The combination of environmental conditions, such as temperature, with substance properties, such as degradation, predicts the amount of substance available for eventual ingestion.

Fate factors can be defined for an individual water system if data are available. If individual water-system data are not available, fate factors can be defined at a country level.

Step 2: Calculate the exposure factor

The exposure factor is the rate of a substance's intake (directly or indirectly) by humans (i.e. the dose). The exposure factor estimates the number of people exposed to a pollutant, and the amount and extent of that exposure.

Exposure to water pollutants can take place through direct ingestion (e.g. drinking water), direct inhalation (e.g. of evaporated water pollutants), indirect ingestion through bio-concentration in animal tissues (e.g. in meat, milk and fish) and dermal contact. This methodology covers ingestion (direct and indirect) and direct inhalation, as dermal contact is not currently covered by the USEtox model.

The outputs of this step are substance- and country-specific exposure factors for freshwater and marine water.

Direct ingestion
In our model, direct ingestion occurs via the drinking of water. The model assumes that the population at risk for drinking contaminated surface water is comprised of those people without access to improved water sources (as determined by the World Bank). Access to an improved water source refers to reasonable access to an adequate amount of water from an improved source, such as a household connection, a public standpipe, a borehole, a protected well or spring, or rainwater collection. This metric is used as a proxy to determine the percentage of the population likely to drink untreated surface water.

The model also includes variables to cover the amount of polluted water ingested by the population at risk.

An important limitation of our methodology is that direct ingestion is limited to consuming contaminated surface water. The amount and source of groundwater for drinking are not considered in the current version of USEtox due to a lack of scientific consensus on the topic.

Direct inhalation
In our model, direct inhalation occurs via the human intake of polluted air. In order to avoid the double counting of impacts considered in the air emissions valuation model, we only consider the inhalation of pollutants initially released into water that subsequently evaporate and become airborne.

Indirect ingestion
In our model, indirect ingestion occurs via the human consumption of produce, meat, dairy products and fish. Each pollutant has a unique bioaccumulation/biotransfer profile for each type of product consumed, which is incorporated into the model.

For produce, the model considers the transfer of substances from soil to plants and from air to plants. However, in our use of the model, we only consider the mass of substances transferred from the freshwater or marine water environmental compartments to air or soil.

Ingestion through meat and milk is estimated using the Travis and Arms (1988) biotransfer factor models for cows, which we adapt for animal fat content and respective animal intake rates. The models consider the consumption of contaminated plants and drinking water. For plants, we only consider the masses of substances transferred from the freshwater or marine water environmental compartments.

Ingestion through fish is measured using bioaccumulation factors (BAFs) when these measurements are available in the literature. Otherwise, the Arnot and Gobas (2003) model in the Estimation Programs Interface (EPI) Suite for the upper trophic level is used to estimate the steady-state BAF for non-dissociating substances and substances.

Food-consumption patterns vary by country. We use underlying assumptions from the GLOBACK data. Dietary habits used in the model include the amount of water drunk daily as well as the amount of leaf and root crops, meat, dairy, freshwater fish and marine fish eaten daily. No distinction is made between subpopulations (e.g. age groups or gender), with averages applied over the entire population.

The USEtox model uses a production-based intake scenario, which tracks long-range substance transport via food (Pennington et al. 2005). For the production-based intake scenario, the contaminant levels in food and drinking water are associated with the location of food production (and contamination) rather than the population's location. This differs from a subsistence scenario, which is more often adopted in substance screening and reflects exposure for an individual who eats, drinks and lives within the region of an emission (Pennington et al. 2005).

Step 3: Calculate the effect factor

The effect factor determines the quantitative relationship between the dose of a substance received and the incidence of adverse health effects in the exposed population. It reflects the change in lifetime disease probability due to changes in lifetime intake of a pollutant (cases/kg) (Rosenbaum et al., 2011).

The effect factor for each substance is based on a linear dose-response function. Dose-response functions describe how the number of health outcomes (responses) change with increasing concentrations of water pollutants (doses). Although there are a variety of approaches to modelling dose-responses, we believe the linear model is the most appropriate for our purposes.

The outputs for this step are substance- and country-specific effect factors for freshwater and marine water. USEtox calculates separate effect factors for non-carcinogenic effects and carcinogenic effects using the same equation.

Equation: Effect factor for cancer and non-cancer

$$EF = \frac{0.5}{N * LT * BW * ED_{50h}}$$

where:

ED_{50h} is the effective dose inducing a response over background of 50% for humans [mg/kg-day]

0.5 is the response level corresponding to the ED_{50h} [individual lifetime risk of cancer or non-cancer]

BW is the average body weight of humans (70 kg)

LT is the average lifetime of humans (70 years)

N the number of days per year

Step 4: Calculate characterisation factor using FF, XF, and EF

The output of the USEtox model is the characterisation for each substance in each country, which describes the number of incidences (cancer or non-cancer) per kilogram of substance released. The basic calculation is given in the following equation.

Equation: Characterisation factors for human health

$$CF = FF \times XF \times EF$$

To move from the number of disease cases to the potential consequences of a chronic toxicological effect, additional information on the severity or the damage caused by incidences is required.

The key assumptions underlying this step are listed in Table 7. Data types required for the model are listed in Table 8.

Table 7: Assumptions required to determine environmental outcomes

Assumptions	Comment on purpose and reasonableness
Simplified fate and exposure modelling using the USEtox parameters at a country level	Geophysical data are defined at a country level but they can be defined locally when the exact location of the emission source is known. It is therefore necessary to simplify geophysical conditions. USEtox was developed by the Task Force on Toxic Impacts under the UNEP-SETAC Life Cycle Initiative to include the best elements of available LCA multi-media models.
Steady-state conditions assumed when calculating substance fate	This modelling technique is well established in the literature.
Linear dose-response function assumed when determining ED50	A linear function assumes that emission concentrations are already above a damage threshold, such that any addition of pollution to the environment has an impact. Determining whether pollutants are below a damage threshold requires data on ambient concentrations and biogenic emissions, which are not globally available. Therefore, linear functions are the standard in academic and government analyses.

Table 8: Data required to determine environmental outcomes

Data input	Description	Source
Substance		
Molecular weight	Sum of atomic weights of all atoms in the compound's molecule.	g/mol (USEtox database)
Partitioning coefficients	Defines the equilibrium distribution of a substance between two solvent phases separated by a boundary. It is used to determine the amount of substance remaining in water. For example, substances with high air-water partition coefficients also have low residence times and low fate factors in water due to rapid volatilisation.	l/kg (USEtox database)
Degradation rate in water, air, soil and sediment	Defines the rate of degradation of the substance in the different environmental media. It is used to determine the amount of substance that persists in the environment. For inorganics, degradation rates are set at 1.10-20/s, indicating no degradation of inorganics in the environment.	/s (USEtox database)
Bioaccumulation factor in fish/biota	Ratio of the chemical concentration in fish to the chemical concentration in the water body where the fish are exposed.	l/kgfish, l/kgbiota (USEtox database)
Dose response	Quantitative relationship between the dose of a chemical received and the incidence of cancerous or non-cancerous health impacts. The model uses the lifetime dose of pollutant that causes an adverse health effect (cancer or non-cancer) with a probability of 50% to determine the number of cases.	IRIS and CPDB databases
Context		
Land, freshwater and coastal area	Defines the area within which the pollutant could disperse. Typically set at a country level but can be defined locally.	GLOBACK parameter set, CIA World Factbook, World Resources Institute
Temperature, wind speed, average precipitation	Weather conditions influence the amount of substance remaining in the water. Conditions typically set at a country level but can be defined at a local level.	
Immediate destination of emission	Defines the type of water (fresh or salt) to which the pollutant is directly emitted. Typically based on coastal population concentration at a country level but can be defined locally.	Earth Institute – Columbia University

Data input	Description	Source
Exposed population	For indirect ingestion of pollutants, the exposed population is based on a production-based intake scenario. For direct ingestion, the inverse of calculations on access to treated freshwater determines the number of people drinking contaminated water within a region.	World Bank, CIA World Factbook
Water consumption	The amount of water consumed per day influences the amount of pollutant intake by humans. Daily intake is set at a country level.	GLOBACK
Consumption of meat, dairy, fruits, vegetables, grains, fish and seafood	Dietary habits influence the amount of pollutant intake by humans. Daily intake is set at a country level.	GLOBACK, CIA World Factbook, Food and Agriculture Organisation of the United Nations

(iii) Estimating the societal impacts

Step 1: Estimate DALYs for each health harm

To determine the DALY for each substance, we used the documented critical effects (associated with substance-specific ED50s from the IRIS and CPDB databases). For pollutants with multiple critical effects, we applied a weighted average. Average values for cancerous and non-cancerous effects (11.0 and 2.7, respectively) were used when critical effects were not identified in the reference databases. These average values were calculated in Huijbregts (2005) and weighted by incidence cases.

Huijbregts (2005) suggests that using the average cancer DALY per incidence is appropriate because the uncertainty factors are low when compared with the uncertainty reported for the toxic potencies of the majority of the carcinogenic substances. However, applying an average is some-what more difficult for non-carcinogenic effects, as the uncertainty factors are much higher for a myriad of reasons, including the fact that standard toxicological-response variables in test species are not specific for disease genesis in humans and, therefore, cannot be properly translated to real-life conditions (De Hollander et al. 1999). Moreover, DALYs are not available for all potential non-carcinogenic health effects resulting from chemical exposure. Despite these concerns, we propose the use of an average non-carcinogenic DALY of 2.7 as an interim solution, as there are no other published average DALYs available for non-carcinogenic effects.

Table 9 shows a small selection of the DALYs associated with individual pollutants' negative health effects. It includes the average values (e.g. 2.7 for non-cancer) when other published data are not available.

Table 9: Sample of DALYs for health harms from pollutants

Pollutant	Cancer		Non-cancer	
	Critical effect	DALY	Critical effect	DALY
Antimony	None	NA	Longevity, blood glucose, and cholesterol	2.7
Arsenic	Skin	9	Hyperpigmentation, keratosis and vascular	9
Barium	None	NA	Nephropathy	2.7
Benzene	Leukaemia	19	Decreased lymphocyte count	2.7
Cadmium	Kidney	11.5	Proteinuria	2.7
Copper	None	NA	Accumulation in the liver, kidney and spleen	16
Lead	Kidney	11.5	Mental development	15.9
Mercury	Stomach	10	Neurological changes, liver	17.6
Molybdenum	None	NA	Increased uric acid levels	2.7
Nickel	None	NA	Decreased body and organ weight	2.7
Selenium	None	NA	Clinical selenosis	2.7
Vanadium	None	NA	Decreased hair cystine	2.7
Zinc	None	NA	Decreases in erythrocyte Cu, Zn-superoxide dismutase (ESOD) activity	2.7

Step 2: Assigning a monetary value to a DALY

Health economists and policy makers typically use DALYs to understand the relative severity of health conditions and to compare the cost effectiveness of investments (cost saving per avoided DALY). Lvovsky et al.'s (2000) World Bank publication presents a method for estimating the welfare value of DALY savings.

Lvovsky et al. (2000) derive the DALY value from the value of a statistical life (VSL) based on the number of lost DALYs associated with that lost life. This approach has subsequently been applied in a government policy context by Pearce et al. (2004) to help evaluate the EU's REACH policy (Registration, Evaluation and Authorisation of Chemicals). In the following, we present our application of this approach. The values used are consistent with those used in the other environmental-impact methodologies.

Equation: DALY value

$$\boldsymbol{DALY\ value}\ \ \frac{\boldsymbol{VSL}}{\boldsymbol{Number\ of\ DALYs\ lost}}$$

The OECD's VSL estimate of USD 3.4m (2011, inflated from 2005) (OECD, 2012) serves as the basis of our DALY valuation. The OECD estimate is based on a meta-analysis of studies that consider the acceptance of risks to life and extrapolate to give a VSL (e.g. wage premiums for working in riskier environments). The median age of individuals in the studies is 47 with a life expectancy of 78, such that the resulting VSL estimate is associated with 31 years of lost life.

In order to estimate the value, the number of years lost is converted to DALYs. A year of disability-free life does not hold the same number of DALYs for all ages. People place a higher value on avoiding disability between their early teens and mid-50s (Figure 6). Therefore, the DALYs are age weighted (Prüss-Üstün et al., 2003).

Prüss-Üstün et al. (2003) provide a formula and suggested coefficients for calculating the relative weighting of each year of life (X_w), which is set out in the following equation. In order to estimate the value, the number of years lost is converted to DALYs. A year of disability-free life does not hold the same number of DALYs for all ages. People place a higher value on avoiding disability between the early teens and the mid-50s (age weighting for DALYs). The DALYs are therefore age weighted (Prüss-Üstün et al., 2003).

Prüss-Üstün et al. (2003) provide a formula and suggested coefficients for calculating the relative weighting of each year of life (X_w), which is set out in the following equation.

Equation:

Equation: Age-weighting formula for calculating DALYs

$$\boldsymbol{X_w = Cx^{-\beta x}}$$

where x is the age in years, and the suggested coefficients are C= 0.1658 and β = 0.04. This formula is used to calculate the relative weighting applied to each of the 78 years of life expectancy associated with the OECD's VSL estimate.

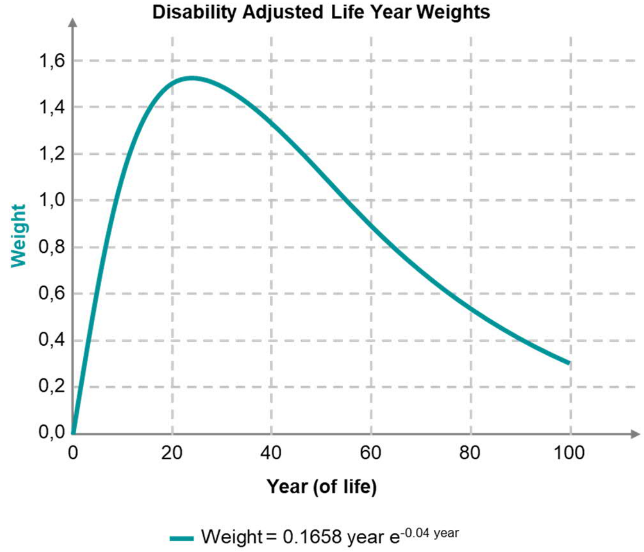

Figure 6: Age weighting for DALYs

People are willing to pay more to avoid disability today than to avoid it the future. Therefore, a discount rate of 3% (as per the social discount rates used in the other methodologies) is applied to future years beyond the age of 47. The discounted age weighting is calculated using the following equation.

Equation: Discount age weighting for DALYs

$$X_{wd} = \begin{cases} Cx^{-\beta x} \text{ when } x < 47 \\ Cx^{-\beta x}/(1+0.03^{x-47}) \text{ when } x \geq 47 \end{cases}$$

The discounted, age-adjusted proportion of life lost (PLL_{wd}) is calculated using the following equation. It represents the proportion of life lost for a person who expected to live to 78 but died prematurely at 47.

Equation: Age-adjusted years of lost life

$$PLL_{wd} = \left(\sum_{x=47}^{78} X_{wd}(x) \Big/ \sum_{x=0}^{78} X_{wd}(x) \right)$$

To calculate the number of DALYs, PLL_{wd} is multiplied by the life expectancy. The following table presents the steps in the calculation that result in the DALY value of USD 185,990 (in 2011 USD).

Table 10: Calculation that results in the DALY value of USD 185,990 (in 2011 USD)

Age of premature death	Life expectancy	Proportion of life lost (PLL_{wd})	DALYs lost ($PLL_{wd} \times$ $life\ expectancy$)	VSL	DALY value $\left(\frac{VSL}{Number\ of\ DALYs\ lost}\right)$
47	78	23.4%	18.3	USD 3.4m	USD 185,990

When adjusting the DALY value, which is derived from the VSL, we need to control for socio-economic conditions. The underlying studies on the willingness to pay cover specific countries. Therefore, the income of the respondents in the study places an implicit limit on the value. If a study is conducted in a relatively high-income country, it will elicit a higher VSL than if it is conducted in a relatively low-income country. Theoretically when adjusting the value to other countries, we should control for the income level. The OECD suggests that a transfer factor using PPP-adjusted GNI per capita with an income elasticity of health between 0.4 and 0.8 should be applied. There is also an argument that adjustments should be made to account for differing appetites in different countries and contexts. However, we have not identified a reliable dataset to control for this aspect.

If an income adjustment is to be included, then we can account for differences between income per capita adjusted for PPP. An income elasticity of 0.6 is recommended as a central estimate of the values presented by the OECD (2010).

Equation: Income-adjustment transfer function

$$Transfer\ function = \left(\frac{GNI_a}{GNI_b}\right)^e$$

where:

GNIa = Gross National Income per capita of the new policy site, adjusted for purchasing power parity,

GNIb = Gross National Income per capita of reference site, adjusted for purchasing power parity and

e = Income elasticity of willingness to pay for health or life.

The ethical concern is whether it is appropriate to adjust the value of life or health among countries because, in doing so, one values the lives of those in lower-income countries less than the lives of those in higher-income countries.

One suggested approach to addressing this is the following:

Situation 1: When aggregating studies from different countries in order to build a health impact model, these values should be comparable and, therefore, expressed in "international" dollars (as suggested by the OECD).

Situation 2: When transferring a health-impact value to different countries:

2.1) If the analysis is meant to cover various countries, values should still be comparable and, therefore, not adjusted, 2.2) If the analysis is meant to only cover one country, it should be adjusted in order to reflect local constraints.

Situation 3: When presenting results locally or internationally, create two sets of coefficients:

- Global consolidation of results: Use an international set of values in which adjustments are made to use international dollars as the basis for the values.
- Presenting in-country impacts to local stakeholders: Use a local set with adjustments for local income.

For the VBA pilot valuation coefficients, we follow situation 1.

Step 3: Compute the total cost of human health impact for each toxic pollutant

After we have determined the characterisation factor (which establishes the number of negative health outcomes), the DALY impact per health outcome and the DALY value, computing the total value of pollutants becomes a simple arithmetical matter.

For each water pollutant, the change in the number of health effects arising from a release of pollutant into the water is multiplied by the relevant GNI PPP-adjusted DALY value to give the total costs associated with such emissions in the country. The cost of water pollution globally is the sum of substance-specific costs.

Equation: Country-specific pollutant cost for human toxicity

$$\begin{aligned}\boldsymbol{Impact_{c1,fw,mw,z}} &= \boldsymbol{Metric\ quanitity_{c1,fw,z}} \times \boldsymbol{Characterization\ factor_{c1,fw,z}} \times \boldsymbol{DALYs_z} \times \boldsymbol{DALY\ value_{c1}} \\ &+ \boldsymbol{Metric\ quanitity_{c1,mw,z}} \times \boldsymbol{Characterization\ factor_{c1,mw,z}} \times \boldsymbol{DALYs_z} \times \boldsymbol{DALY\ value_{c1}}\end{aligned}$$

where:

$\text{Metric quanitity}_{\text{c1,fw,z}}$ is the mass of the substance released into freshwater in a given country,

$\text{Characterization factor}_{\text{c1,fw,z}}$ is the number of disease incidences per kilogram of substance of a substance released into freshwater in a given country,

$\text{Metric quanitity}_{\text{c1,mw,z}}$ is the mass of the substance released into marine water in a given country,

$\text{Characterization factor}_{\text{c1,mw,z}}$ is the mass of the substance released into marine water in a given country,

DALYs_z is the number of DALY associated with the critical cancerous and non-cancerous effects of the substance, and

$\text{DALY value}_{\text{c1}}$ is the PPP-adjusted DALY value in monetary terms.

Equation: Global pollutant cost

$$\textit{Global impact}_z = \sum(\textit{impact}_{c1,fw,mw,z}, \textit{impact}_{c2,fw,mw,z}, \textit{impact}_{c3,fw,mw,z}, \ldots \textit{impact}_{cn,fw,mw,z})$$

II. Nutrient valuation

The valuation module for nutrients estimates the eutrophication potential of nutrients in fresh and marine water. It then estimates the value based on published data on what individuals would pay to avoid those negative effects.

(ii) Quantifying the environmental outcomes

In this methodology, we calculate the eutrophication potential of excessive nutrients released into the watercourse. We consider only P for emissions to freshwater, and both N and P for marine water due to the limiting nutrient theory.

Algal growth is limited by different nutrients in different environments. If more of a limiting nutrient is introduced into the system, it will promote an increase in growth. However, the introduction of other, non-limiting nutrients will have no effect on growth. In freshwater, P is often considered the limiting nutrient (Schindler 1977, Sharpley et al. 1994). When salinity increases, N's contributions to eutrophication increase. In temporal zones, N is probably the major cause of eutrophication in most coastal systems. However, P can limit primary production in other systems. Therefore, both N and P are treated as contributors to eutrophication in marine water (Howarth & Marino 2006). In impact assessments, most models adopt these general rules while acknowledging that they are a simplification, as other nutrients can be limiting in certain conditions (Finnveden & Potting 1999).

Step 1. Determine the environmental outcomes of phosphorus in freshwater

To determine the eutrophication potential of P in freshwater, we use a model developed by Helmes et al. (2012). This is the only P model we identified that can derive spatially explicit fate factors for P emissions to freshwater on a worldwide scale. It was created as part of Life Cycle Impact Assessment Methods for improved sustainability characterisation of technologies (LC-IMPACT), which aim to improve life cycle impact-assessment methods, characterisation factors and normalisation factors in a coherent and scientifically sound way. Its development was led by the EC as part of the 7th Framework Program. This model has been peer reviewed and published in the International Journal of Life Cycle Assessment.

Helmes et al.'s (2012) phosphorus model

The fate factor calculated by the Helmes et al. (2012) model essentially reflects the eutrophication potential of releasing one kilogram of P into freshwater. A higher fate factor means a higher cumulative persistence, implying that P will be available longer for algae growth, thereby increasing the negative effects of that growth.

Fate factors of P emissions to freshwater were derived for a 0.5° x 0.5° grid (50km) covering the globe and then averaged within a country. Aggregation to the country level was done using a weighted average of fate factors.

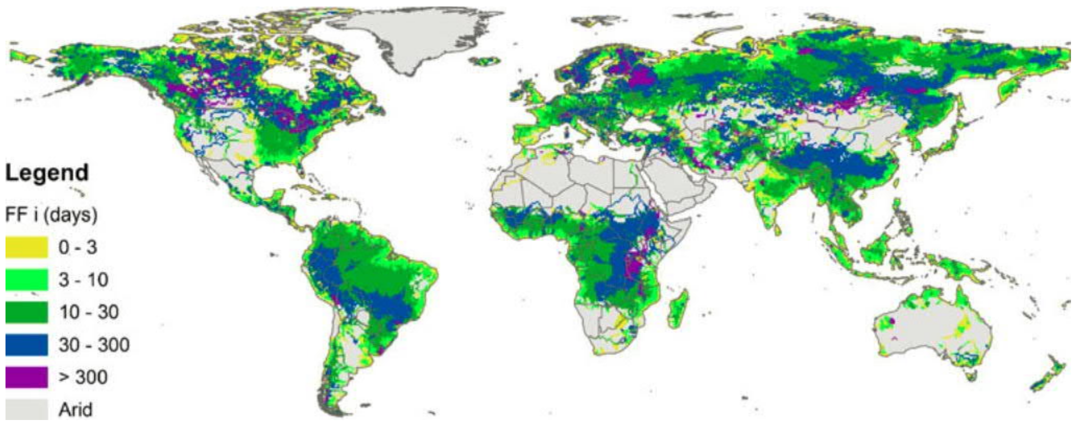

Figure 7: Fate factors for phosphorus emissions to freshwater

Source: Helmes et al. (2012)

The Helmes et al. (2012) model traces the persistence of one kilogram of P through grids. P's persistence is based on three processes: use, advection and retention.

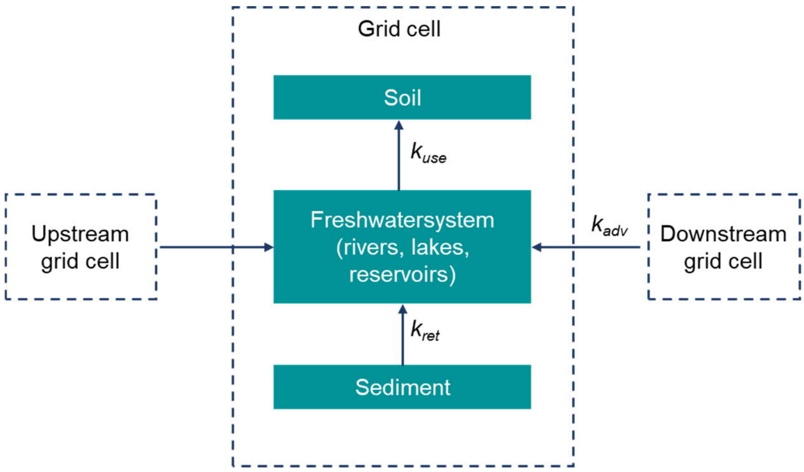

Figure 8: Freshwater phosphorus fate model

where:

> k_{use} is use – the removal of phosphorus from the system when water is taken for domestic, industrial or agricultural purposes,
>
> k_{ret} is retention – the uptake of phosphorus by biomass and its adsorption to suspended solids and
>
> k_{adv} is advection – the flow of water out of the grid.

In the societal outcomes section, we outline how WTP estimates are applied to emissions of P. The fate factors must be scaled to the countries on which the valuation estimates are based because these studies measure WTP to avoid the eutrophication associated with emission of one kilogram of P. This implicitly includes the eutrophication potential of P in that location.

Step 2: Determine the environmental outcomes of nitrogen (N) and phosphorus (P) in marine water

No equivalent to Hermes et al.'s (2012) model is available for eutrophication potential in marine water. In the absence of a detailed model, we apply a simplification to assess eutrophication in marine water.

According to the standard set in the *Handbook on Life Cycle Assessment*, which is the operational guide to the ISO LCA standards (Guinée, 2002), one kilogram of P has seven times more eutrophying potential in marine water than one kilogram of N. This relationship concurs with the Redfield ratio (Redfield, 1963). These weights were used to assess the eutrophication potential of nutrients to marine waters.

In the future, we may be able to utilise a marine-specific eutrophication model. A spatially differentiated fate factor model is currently being developed through the LC-IMPACT programme and tested on industry cases. This model examines marine eutrophication caused by N emissions spatially differentiated on the country level. It explores the potential increase in total nitrogen concentration (in the photic zone) or total marine N loading, weighted by residence time in the 64 marine ecosystems grouped into climate zones. However, due to limited testing and a lack of peer review, we do not use this methodology to model the fate of N in marine waters. As spatially differentiated global fate models for marine eutrophication are peer reviewed, the results should be considered in terms of replacing the allocation method for nutrients.

The key assumptions underlying this method are listed in Table 11.

Table 11: Assumptions required for determining environmental outcomes from excess nutrients

Assumption	Comment on purpose and reasonableness
Fate factor calculations are used to scale WTP figures based on eutrophication potential.	The effect-factor calculations (on ecosystems) from the life cycle assessment model that would bring the model from the midpoint to the endpoint were not included, as the EC deemed them immature. The EC recommends no end-point calculations based on life cycle assessments for eutrophication.
Use of the Redfield ratio to scale eutrophication potential of N and P.	The Redfield ratio is considered the standard in the *Handbook on Life Cycle Assessment*, the operational guide to the ISO standards (Guinée, 2002).

(iii) Estimating the societal impacts

Excessive nutrients have numerous impacts on various aspects of society, including recreation, property values and fish stocks. We use a welfare-based approach to calculate generic damage values for these impacts. Our methodology is adapted from Ahlroth (2009), who uses WTP to estimate damage values per kilogram of N or P. This approach makes the best use of the somewhat limited literature on the valuation of eutrophication impacts. We convert the published values to cover other countries using the benefit-transfer approach.

Step 1: Value eutrophication in freshwater

Ahlroth (2009) presents an approach that uses WTP estimates for reduced eutrophication impacts to calculate a generic damage value per kilogram of P released into freshwater in Sweden. Studies in other parts of the world are limited. The benefit-transfer approach presented below is based on Ahlroth's (2009) values, but it can be applied to other source data where available. When applying values from a benefit-transfer approach, it is important to consider the applicability of the values to other areas. For example, whether values derived from a study in Sweden could be applied to developing countries is questionable.

Ahlroth (2009) analysed existing valuation studies that estimated the value of improving water quality in a lake or watercourse. The author constructed a generic damage value per kilogram of P in Sweden using a structural benefit-transfer approach from eight studies to calculate total WTP and the annual deposit amount. For additional details on Ahlroth's (2009) work and the structural benefit-transfer method, see Appendix IX.

The underlying studies were similar in design and valued a quality change. Respondents were presented with different water-quality scenarios, which were described using a water-quality ladder. The ladder presented five incremental improvements in water quality based on the water's suitability for drinking, bathing, irrigation, recreational fishing and boating (Norwegian State Pollution Control Agency, 1989). Respondents provided their WTP for moving between the scenarios. An average WTP per unit of emission was calculated based on the reduction in nutrient loading necessary to move between water-quality scenarios.

Ahlroth (2009) assumes a constant marginal WTP, which results in a price of USD 136 per kilogram of P. To transfer this value from Sweden to other countries, we adjusted the WTP values by PPP. For a further discussion of benefit transfer and WTP, see Box 2.

Box 2: Benefit transfer of WTP

Conducting primary research on WTP is expensive and time-consuming, especially when such research is conducted on a global scale. A time- and cost-effective alternative to primary valuation studies that is widely used in policymaking is benefit transfer. This involves applying WTP estimates from existing studies to different but sufficiently similar contexts. These values are adjusted to account for the differences in context. The breadth of applicability of benefit transfer generally rises in line with the sophistication of the adjustment technique, as shown in Table 15.

Table 12: *Types of benefit transfer*

Method	Description
Value transfer	• The value from the primary study is adjusted for PPP and inflation so that it accurately reflects the real value of money.
Function transfer	• Values are estimated based on a number of other characteristics using econometric analyses of the determinants of WTP in the primary study so that the econometric function, rather than simply the value, is transferred. • Allows for greater adjustments for context and improves accuracy and reliability.
Meta-analysis	• Involves econometric analyses of several primary studies to estimate a function that can be applied in the same way as for function transfer. • Shares the advantages of function transfer relative to value transfer and is appropriate when there is no clear single candidate for function transfer.

In the context of a globally applicable methodology, primary research on WTP values across cities and countries is limited. Moreover, available studies often use inconsistent approaches. Benefit transfer can help overcome this lack of consistent primary work by providing a single value or set of values that can be consistently applied and adjusted to different geographical and socioeconomic contexts.

In this methodology, we select Ahlroth's (2009) base values and adjust them to account for income. In the longer term, a more sophisticated benefit transfer function could be developed to allow adjustments for local contexts and preferences. Insufficient primary data on the characteristics of participants in the underlying studies was available to support such an approach at this time. If the valuation approach is to be applied to a more focused geographical area, it may be possible to find or collect such data.

Step 2. Value eutrophication in marine water

Our approach to valuing marine water nutrients is similar to our approach for freshwater nutrients.

For coastal areas, Ahlroth (2009) analysed existing valuation studies that estimated the value of improving the quality of marine water. As in the approach for freshwater, Ahlroth (2009) calculated a per kilogram WTP value for phosphorus and nitrogen using a structural benefit-transfer method.

The price of per kilogram of phosphorus in marine water is USD 68, while the price of nitrogen is USD 9. To transfer these values from Sweden to other countries, we adjust the WTP values by PPP.

Ahlroth (2009) constructed generic damage values for phosphorus, nitrogen, ammonia and nitrogen oxide (NO_x). The scope of our water-pollution methodology does not cover emissions to air that lead to eutrophication. Therefore, only the generic damage values for phosphorus and nitrogen are used for the E P&L. However, the aerial eutrophication emissions are likely to be trivial given the results of general research on the amount of eutrophying nutrients emitted to air versus water.

Step 3. Sum to societal impacts of all excess nutrients

After we have established the eutrophication potential and the damage value (via WTP) for N and P in fresh and/or marine water, calculating the total societal cost of excess nutrients requires straightforward arithmetic.

For N and P, the change in eutrophication potential arising from a release of N or P into the water course is multiplied by the relevant PPP-adjusted WTP value to give the total costs associated with excessive nutrients emissions in the country. EquationCountry-specific pollutant cost for eutrophication summarises the matrix multiplication used to derive the societal cost figure for each country.

Equation: Country-specific pollutant cost for eutrophication

$$\begin{aligned}\boldsymbol{Impact}_{c1.fw,mw,N,P} &= \left(\boldsymbol{Metric\ quanitity}_{c1,fw,P} \times \boldsymbol{Eutrophication\ potential}_{c1,fw,P} \times \boldsymbol{WTP}_{c1,fw,P,}\right) \\ &+ \left(\boldsymbol{Metric\ quanitity}_{c1,mw,N} \times \boldsymbol{Eutrophication\ potential}_{c1,mw,N} \times \boldsymbol{WTP}_{c1,mw,N}\right) \\ &+ \left(\boldsymbol{Metric\ quanitity}_{c1,mw,P} \times \boldsymbol{Eutrophication\ potential}_{c1,mw,P} \times \boldsymbol{WTP}_{c1,mw,P}\right)\end{aligned}$$

where:

$\text{Metric quanitity}_{c1,fw,P}$ is the mass of phosphorus released into freshwater in a given country,

$\text{Metric quanitity}_{c1,mw,N}$ is the mass of nitrogen released into marine water in a given country,

$\text{Metric quanitity}_{c1,mw,P}$ is the mass of phosphorus released into marine water in a given country,

$\text{Eutrophication potential}_{c1,fw,P}$ is the eutrophication potential of phosphorus released into freshwater in a given country,

$\text{Eutrophication potential}_{c1,mw,N}$ is the eutrophication potential of nitrogen released into marine water in a given country,

$\text{Eutrophication potential}_{c1,mw,P}$ is the eutrophication potential of phosphorus released into marine water in a given country,

$\text{WTP}_{c1,fw,P}$ is the PPP adjusted willingness to pay for one kilogram of phosphorus in freshwater in any given country,

$\text{WTP}_{c1,mw,P}$ is the PPP adjusted willingness to pay for one kilogram of phosphorus in marine water in any given country and

$\text{WTP}_{c1,mw,N}$ is the PPP adjusted willingness to pay for one kilogram of nitrogen in marine water in any given country.

The key assumptions underlying this method are listed in 13.

Table 13: Assumptions required for determining societal impacts from excess nutrients

Assumptions	Comment on purpose and reasonableness
The WTP per kilogram is derived from a number of studies using transfer functions. Values are adjusted to account for income but not potential differences in environmental preferences by country.	WTP for eutrophication may vary. However, in the absence of better data that would allow for development of a more sophisticated function that includes preferences for the environment, this approach is considered an acceptable approximation.
The Redfield ratio is appropriate for scaling the eutrophication potential of N and P in marine water.	The Redfield ratio is considered the standard. It was set in the *Handbook on Life Cycle Assessment* – the operational guide to the ISO standards (Guinée, 2002).
Fate factor calculations are used to scale WTP figures based on eutrophication potential.	The use of the effect factor calculations (on ecosystems) from the LCA model would bring the model from the midpoint to the endpoint. However, these calculations were deemed immature by the EC. The EC recommends that no LCA-based endpoint calculations be used for eutrophication.

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Topic-specific details

2.5. LAND USE

- *Topic description*
- *Impact pathway*
- *Quantification and monetary valuation*
    - *Measuring impact drivers*
    - *Environmental outcomes*
    - *Impacts on society and monetary valuation*
- *Sources*

2.5.1. TOPIC DESCRIPTION

Natural land areas, which are often rich with biodiversity, provide essential services to society. For instance, they regulate our environment, provide goods and services that support livelihoods, offer opportunities for recreation, and provide cultural and spiritual enrichment. The Millennium Ecosystem Assessment estimated that 63% of these ecosystem services are already degraded, with important social and economic implications for current and future generations (MEA, 2005). A subsequent analysis requested by the G8+5 environment ministers entitled *The Economics of Ecosystems and Biodiversity* (TEEB) estimated that the economic cost of the degradation and loss of biodiversity and ecosystem services each year is between USD 2 trillion and USD 4.5 trillion (TEEB COPI, 2008).

Ecosystem services are a valuable resource for society. The degradation of natural land areas is reducing the availability of these ecosystem services each year. The restoration of natural land areas can lead to resumed provision of ecosystem services. The principal cause of ongoing losses of biodiversity and declines in associated ecosystem services is the conversion of natural land areas for agriculture. Agricultural land now covers 25% of the Earth's terrestrial surface, and estimates suggest another 10% to 20% of natural grassland and forestland may be converted by 2050 (MEA, 2005).

The objective of this methodology is to estimate the economic value of lost ecosystem services associated with the conversion and occupation of natural land areas. These values are associated with the use benefits that society gains from ecosystems, such as climate regulation, bioprospecting, food and fuel. These benefits also include non-use value from, for example, cultural experiences or education, as well as option values that reflect the fact that these areas might have future use values.

An important consideration for this methodology is the temporal dimension. Many natural areas were converted long ago and have changed uses and ownership many times since. Ecosystem services are flows, such that if their provision is reduced, that reduction is felt every year until the land is restored. This methodology values the ecosystem service reduction in the current year relative to the natural state and assigns this reduction in value to the current occupant of the land, irrespective of whether that occupant was directly responsible for the land's conversion.

This approach (i.e. valuing and attributing in-year losses to the current occupant) is appropriate for impact valuation for three reasons:

- It reflects the flow of impacts that are created as a result of occupation and are dependent on the management practices that the current occupier employs (even if others are responsible for the pre-conditions). For example, the 2013 UK floods were exacerbated by poor land management, which reduced farmland water interception and retention on land converted hundreds of years ago.
- It incentivises current land occupiers to minimise the loss of ecosystem services through, for example, the use of sustainable land-management practices.
- It avoids making highly uncertain assumptions regarding the future extent of lost ecosystem services or the date of past conversions.

2.5.2. IMPACT PATHWAY

In order to value corporations' environmental impacts on society, the link between land use and humans via environmental outcomes must be established. This is reflected in the impact pathway shown in Figure 9.

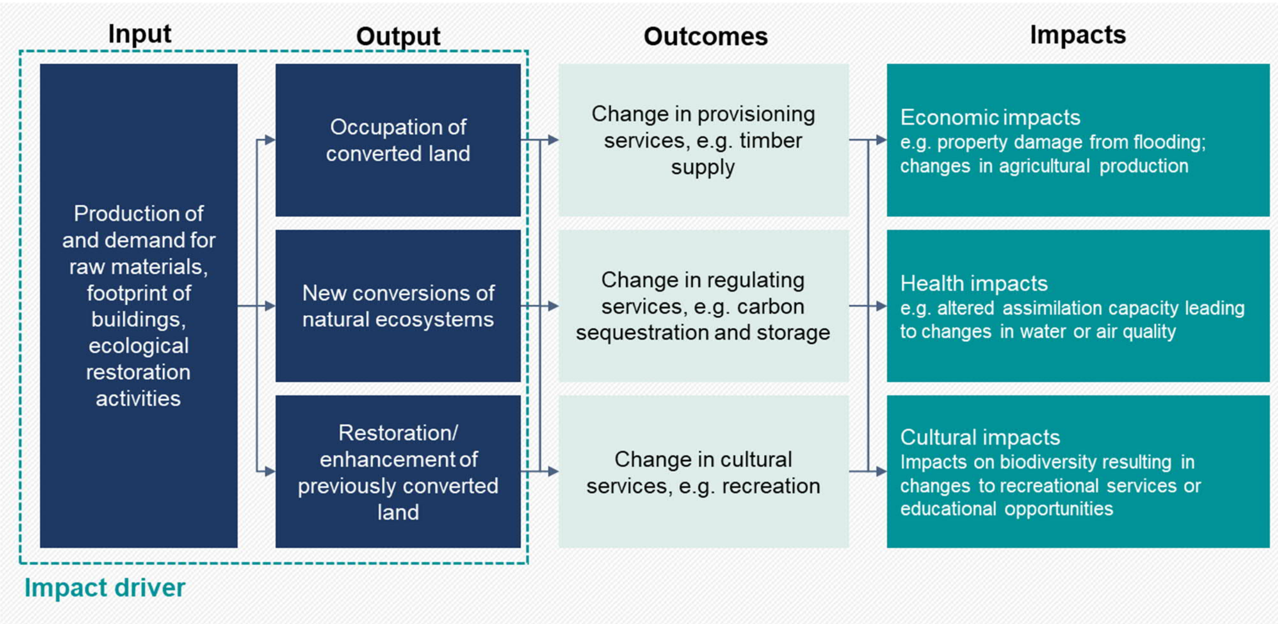

Figure 9: Simplified impact pathway land use

Table 14 presents a classification of different ecosystem services that can be affected by the conversion and occupation of land. These services create use, non-use and option values for society. The driver of the impact in our land-use methodology is the loss of the flow of these ecosystem services when land is not in its natural state.

Table 14: Ecosystem services

Service class	Specific ecosystem service	Potential relevance of impact to people
Provisioning services	Food from natural/semi-natural ecosystems	Local
	Fibre, other raw materials	Local
	Domestic and industrial water	Regional
	Bio-prospecting and medicinal plants	Global
	Ornamental products	Regional

Service class	Specific ecosystem service	Potential relevance of impact to people
	Air purification	Global
Cultural services	Recreation	Regional
	Spiritual and aesthetic	Regional
	Cognitive and learning opportunities	Regional
Regulating services	Stable climate	Global
	Pollution control and waste assimilation	Regional
	Erosion control	Regional
	Disease and pest control	Regional
	Flood control and protection from extreme events	Regional

In this approach, supporting services are excluded. Supporting services include those services that are necessary for all other ecosystem services to function, such as nutrient cycling, soil formation and water cycling. If included, these values would double-count with provisioning services, which are underpinned by supporting services.

Several factors influence the extent to which people around the world are affected by the loss of specific services. The extent to which they are affected by these losses depends on the geographical level at which these services operate. For example, the harvesting of food and fibre from natural areas tends to be local, while climate regulation is global (Table 14). This geographical scope defines the population that could be affected by the loss of these ecosystem services from an area. The actual extent to which people are affected depends on how vulnerable they are to the loss of a specific service.

Similarly, the ways in which people are affected is highly context specific. The loss of carbon sequestration anywhere in the world will contribute to climate change, which will affect everyone globally but not equally or in the same way. The loss of soil fertility and associated provisioning services could lead to malnutrition and displacement for subsistence farmers. However, in developed countries, the impacts are more likely to be a loss in revenue or profitability, or the loss of recreational opportunities.

2.5.3. QUANTIFICATION AND MONETARY VALUATION

This section covers the three steps in more detail: (i) measuring impact drivers, (ii) measuring changes in the state of natural capital and (iii) valuing impacts. For guidance on actions, see the Natural Capital Protocol.

Given the state of research and available studies, it is not feasible to standardise all aspects of quantification. Detailed examples are given in the following to illustrate how quantification may work in practice.

(i) Measuring impact drivers

The ideal dataset would specify the land area that is occupied, its current use and the country in which those areas are located across the value chain. In practice, the production of raw materials, especially agricultural raw materials, is likely to represent most of the land use for most companies outside the services sector. However, in many cases, only companies that are directly associated with the production of raw materials will have data on the area used. Therefore, it will need to be estimated.

To estimate the impact of the land use in a company's value chain, the indicator should be measured in hectares of occupied land and the type of occupation. The type of occupation is important because it may be used as a proxy for the proportion of ecosystem services that are lost.

Land use can be estimated using productivity modelling for agricultural processes or such techniques as environmentally extended input output modelling (EEIO) or life cycle assessment (LCA).

When land occupation is associated with the production of more than one type of product, the impacts need to be allocated to the different products (e.g. an agricultural process that rears cows for both milk and leather production). Typically, the allocation of impacts occurs on a per mass basis or using an economic allocation. When selecting an allocation method, companies should consider how it might affect decision making.

Table 15: Available metric data for land use per value chain level

	Metric data
Own operations	Land-use footprint of buildings should be available in the company's management information.
Immediate/ key suppliers	Land-use footprint of buildings may be available from suppliers. When this information is unavailable, gaps in metric data can be filled using modelling techniques, such as EEIO.
Upstream/ supply chain	Land-use footprints of buildings can be estimated using EEIO and LCA (or inferred from other suppliers). Land-use footprints for raw materials can be estimated using production models based on data on raw-material demand from the company and its manufacturing suppliers. The source location of these materials may be known by the company. If this is not the case, suppliers may be able to provide the information or trade data can be used to identify the most likely sources.

Downstream/ use phase	Land-use area is highly dependent on the product in question. For instance, cars require car parks and garages. However, many products, such as clothing or cosmetics, have no direct land use requirements. Indirect land use (e.g. rubber production for tyres) can be estimated using production models based on assumptions regarding the quantity of raw material used, which may be available from customer surveys or industry information. EEIO and LCA can also be used to estimate indirect land use where appropriate.
End of life/ reuse impacts	Land-use area can be modelled using EEIO or LCA techniques. This information may be supplemented by customer surveys or industry information.

This methodology considers the following land-use occupation types:

- Agriculture subtypes
  - Wheat,
  - Vegetables, fruit and nuts,
  - Cereal grains,
  - Oilseeds,
  - Sugarcane and sugar beets,
  - Plant-based fibres,
  - Crops n.e.c.,
  - Animal rearing and
  - Paddy rice.
- Forestry
- Paved (land fully converted; no ecosystem services provided)

(ii) Environmental outcomes

The objective of this step is to estimate the extent of ecosystem-service loss associated with land-use management practices relative to the natural eco-region. The estimates are based on changes in biomass, soil organic carbon (SOC) and species richness in the eco-region.

This methodology is used to estimate the average impact per hectare of land occupied and the type of land occupation at a national level. The first step focuses on understanding the original, natural ecosystems at a national level.

Step 1: Understand the eco-regions at a national level

Land in the desert provides different services than land on the coast. Therefore, it is necessary to assign distinct eco-region types to each area of land use in order to accurately assess the extent of ecosystem-service loss.

Arguably, the most complete dataset for defining eco-regions on a global scale is the WWF Wild-finder, which presents the distribution of 16 biomes (based on a more detailed set of 867 ecosystem types). The WWF's ecoregions shown in figure 10 are classified using biogeographical data collated through extensive collaboration with more than 1,000 biogeographers, taxonomists, conservation biologists and ecologists from around the world. Estimates of ecosystem valuations are not available in sufficient quantity at this level of detail. Therefore, we map the 16 biomes to six eco-regions, presented in Table 16. While these six groups are broad, increased differentiation is introduced when they are valued using other location-specific information.

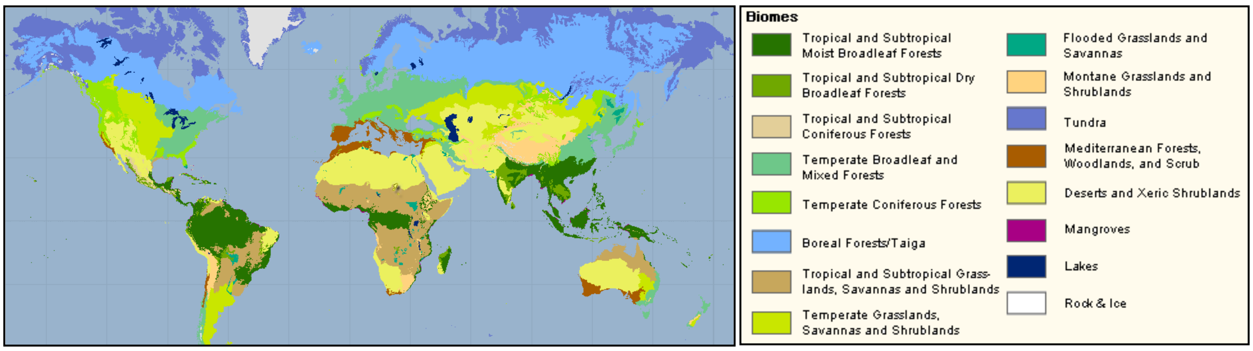

Figure 10: WWF Wildfinder biomes

Source: WWF Wildfinder database, https://www.worldwildlife.org/pages/wildfinder-database

Table 16: Mapping biomes to eco-regions

WWF biome	Eco-region used in this methodology
Tropical and subtropical moist broadleaf forests	Tropical forests
Tropical and subtropical dry broadleaf forests	Temperate and boreal forest
Tropical and subtropical coniferous forests	
Temperate broadleaf and mixed forests	
Temperate coniferous forest	
Boreal forests/taiga	
Mediterranean forests, woodlands and scrubs	
Temperate grasslands, savannas and shrublands	Grasslands
Tropical and subtropical grasslands, savannas and shrublands	
Montane grasslands and shrublands	
Tundra	
Deserts and xeric shrublands	Desert/arid grassland
Flooded grasslands and savannas	Inland wetlands
Mangroves	Coastal wetlands

Step 2: Estimate the extent of ecosystem-service loss in each eco-region in each location

The objective of this step is to calculate the reduction in ecosystem services at the national level. Changes in ecosystem-service provision depend on the type of natural ecosystem displaced and the land-use activity. The extent of ecosystem-service loss, expressed as a percentage, can vary significantly according to the type of land-use change.

When examining impacts across extensive global value chains, it is impossible to measure specific changes in service provision for a range of different eco-regions. In such cases, the relative biomass, SOC and species richness expected from the natural eco-region and associated with the current land-management regime are used as proxies.

There is ecological support for a relationship between these variables and the ecosystem's functioning at a general level (Hooper *et al*., 2005). However, we recognise that this method offers a crude approximation of the complexity of different ecological systems globally, as many other important interactions are present in nature.

Table 17 identifies the proxy variables used for each ecosystem service. They are intended for use when minimal details about land-use practices are known. In this case, we make the conservative assumption (leading to higher impacts) that there is intensive industrial production on the site. As a result, we assume that some services are completely lost (e.g. opportunities to gather food or fibre).

Table 17: Proxies for estimating the relative change in ecosystem services for use when specific data are unavailable

Ecosystem service		Extent of loss – proxy
Provisioning services	Food from natural/semi-natural ecosystems	Total loss – N/A
	Fibre, other raw materials	Total loss – N/A
	Domestic and industrial water	Total loss – N/A
	Bioprospecting and medicinal plants	Total loss – N/A
	Ornamental products	Total loss – N/A
	Air purification	Partial loss – biomass and SOC
Cultural services	Recreation	Partial loss – biomass, SOC and species richness
	Spiritual and aesthetic	Partial loss – biomass, SOC and species richness
	Cognitive and learning opportunities	Partial loss – biomass, SOC and species richness
Regulating services	Stable climate	Partial loss – biomass and SOC
	Pollution control and waste assimilation	Partial loss – biomass and SOC
	Erosion control	Partial loss – biomass and SOC
	Disease and pest control	Partial loss – biomass and SOC
	Flood control and protection from extreme events	Partial loss – biomass and SOC

To calculate the extent of ecosystem-service change for eco-region *e* in location *l*, we use data for the current use, *u*, relative to the typical ecosystem associated with the eco-region in question.

Equation: The extent of ecosystem-service loss using a proxy

$$\textit{Extent of ecosystem service loss } (\%)_{el} = \frac{\textit{proxy for current land use } (\textit{unit})_{ul}}{\textit{expected proxy of natural } (\textit{unit})_{el}}$$

When more than one proxy is used, the average of the percentages is used to infer the extent of ecosystem-service loss. Table 18 presents an example of estimated ecosystem services loss for an area of pasture land use in an Australian grassland.

Table 19 presents the data requirements and sources for estimating environmental outcomes and Table 20 presents the key assumptions.

Table 18: Example output from estimating ecosystem-service loss for an Australian grassland

Country	Eco-region and conversion	Area of attributable land use	Ecosystem service	Extent of service loss
Australia	Grasslands to crops (cereal/ grains)	12,300 ha	Food from natural/semi-natural ecosystems	100%
			Fibre, other raw materials	100%
			Domestic and industrial water	100%
			Bio-prospecting and medicinal plants	100%
			Ornamental products	100%
			Air purification	58%
			Recreation	72%
			Spiritual and aesthetic	72%
			Cognitive and learning opportunities	72%
			Stable climate	58%
			Pollution control and waste assimilation	58%
			Erosion control	58%
			Disease and pest control	58%
			Flood control/protection from extreme events	58%

Table 19: Data required to estimate ecosystem-service loss

Variable	Suggested data source(s)
Eco-region distribution	WWF Wildfinder
Biomass	IPCC (2006) provides generic biomass estimates for different land uses and approved methods for estimating the change following conversion. IPCC (2016) and Więski, Guo, Craft and Pennings (2008) provide biomass estimates for generic land use.
SOC	IPCC (2006) provides rates of SOC change for different land uses dependent on climate and land-management practices.
Species richness	Mehmeti, Demaj and Waldhardt (2009) and Tracy and Sanderson (2000) estimate species richness associated with different land uses. Kier et al. (2005) and Więski, Guo, Craft and Pennings (2008) estimate species richness in different biomes. Other location-specific data on species richness are available from ecological research.

Table 20: Key assumptions for estimating ecosystem-service loss

Assumption	Explanation
Mapping of WWF biomes to six eco-regions is appropriate	The eco-regions are used as the starting point for the valuation if the underlying valuation studies are representative of the ecosystems classified within the eco-region. These broad groups are considered appropriate because the principle drivers of value are the nature of the ecosystem service itself and the characteristics of the benefiting population rather than the type of ecosystem from which the service was derived. For example, both mangroves and coastal marshes provide coastal protection, which is valued in a similar way despite the significant differences in ecology.
Mapping of global climates to six IPCC climate categories is appropriate	The six IPCC climates are determined using Koppen-Geiger data provided by Portland State University based on Strahler and Strahler (1992). The four non-montane IPCC categories (i.e. “tropical moist”, “tropical dry”, “temperate/boreal moist/wet” and “temperate/boreal dry”) are mapped to the Koppen-Geiger categories to determine the percentage composition of each country. As these categories do not consider elevation, a map of IPCC climate zones is used to estimate the tropical montane area, which is then split between “tropical montane moist/wet” and “tropical montane dry”. The prevailing climate of a country is determined to be the IPCC climate category with the highest area of land within a country.
Changes in biomass, SOC and species richness pre-/post-conversion are suitable indicators for changes in ecosystem-service provision	This adjustment is considered an acceptable approximation for applications on a global scale in the absence of data for other commonly applicable indicators. We recognise that many other factors affect ecosystem functioning and that changes in SOC, species richness or biomass will not necessarily lead to proportional changes in functioning due, for example, to keystone species or functional duplication across multiple species.

(iii) Estimating the societal impacts

To estimate the societal impacts of land-use change, per hectare valuation estimates are calculated for different ecosystem services at a national level. 1,500 estimates of ecosystem services are classified into eco-regions and medians are taken across each ecosystem service to estimate the current marginal value of ecosystem services.

Step 1: Standardise the ecosystem-service valuation estimates

The estimates in our database of ecosystem-service valuations are presented in different currencies for different years. In order to bring these estimates together in our meta-analysis, it is necessary to standardise the units. Therefore, we make a number of adjustments to the individual published estimates contained within the database in order to express them in 2016 USD per hectare per year (2016 is the base year for our analysis; it can be inflated to reflect the year of study):

- Converting values to per hectare per year: Some values in the original publication are expressed as per household, as total values for a larger area or as a net present value (NPV). Where possible, these values are converted based on other information provided by the authors (e.g. number of households, area, discount rate and years over which NPV is calculated).
- Converting values to current USD: Exchange rates from the date of the estimate are used and US inflation rates are applied. This ensures that inflation is applied consistently and it avoids potentially large fluctuations resulting from variations in inflation.
- Adjusting income to correct for purchasing power parity (PPP) differences across estimates: We apply current PPP adjustments (based on the ratio of local gross national income (GNI) to US GNI) to local and national ecosystem services. We do not apply income adjustments to global services, as these are typically based on a global (or at least international) WTP (includes a classification of ecosystem services by scale of the service delivery – local, national or global.)

At this stage, 462 values must be excluded from the database either because it is not possible to express them as per hectare per year values or because they are not associated with a specific eco-region. This reduces our sample size, even though every effort is made to include values where possible. For example, estimates of grouped ecosystem services (e.g. labelled total economic value (TEV) or total of provisioning services) are retained as are estimates covering several countries (with exchange rates based on the relevant basket of economies).

Step 2: Estimate the current marginal value of ecosystem services by eco-region and country

The current marginal value of ecosystem services represents the impacts associated with losing another hectare of natural ecosystem today given the prevailing level of ecosystem services and the scarcity of natural ecosystems. The methodology aims to estimate the current marginal value of the ecosystem services provided by each of the six eco-regions in different contexts.

Remove outliers by ecosystem service and eco-region

To estimate the current marginal value of ecosystem services, we take the average across the estimates in the database. The estimates are split by ecosystem services within each of the six eco-regions. We do not segregate the estimates by country or geographical region. We believe averaging across eco-regions is a better approach than averaging estimates by country or region because: (i) there is more similarity among ecosystem services across eco-regions in different countries than among different eco-regions within the same country and (ii) there is insufficient data to provide reliable and comparable estimates by country or region. The TEEB valuation database takes a similar approach, as it emphasises the commonality of ecosystem types rather than country borders, which are largely arbitrary from an ecosystem point of view.

There is significant variation across estimates for most ecosystem services. In general, the data display a long tail with most estimates at the lower end of the range and a few high values. As a result, the mean values tend to be higher than the medians, with large standard deviations.

This analysis calculates an average value in order to provide an indication of the central tendency within the distribution of values of ecosystem services in a given eco-region. In nature, there is variation within eco-regions. In addition, there is considerable variation in the way human society interacts with (and, therefore, gleans value from) ecosystems. Therefore, it is not unreasonable to expect significant variation in our sample of values and we should retain most outliers. However, some values are several orders of magnitude higher than most and disproportionately skew the results (even if the median is used). We therefore exclude estimates that are two standard deviations higher or lower than the mean.

Figure 12 provides an illustration of the results before and after the exclusion of two outliers. 462 estimates are excluded across all ecosystem services and eco-regions, leaving 1,126 estimates. An alternative approach would be to retain all values in our pool for calculating marginal values, but we feel this would expose the mean and median values to bias, and reduce reliability and comparability across eco-regions.

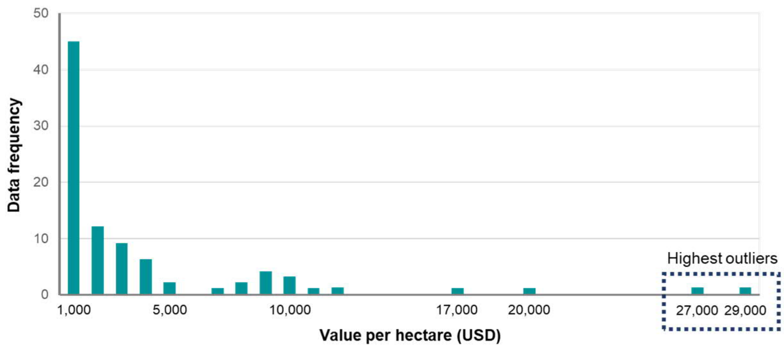

Figure 11: Example of distribution of estimates for food services from coastal wetlands (USD/ha/yr)

Calculate average values for ecosystem services by eco-region

After excluding the largest outliers, we calculate the mean and median values for each ecosystem service by eco-region.

The sum of the value of ecosystem services from an area represents the TEV. We can therefore sum across the estimates for each ecosystem service to derive the average TEV of each of the six eco-regions. In so doing, we exclude ecosystem services, for which we only have one value estimate. 13 estimates are excluded across all ecosystem services and eco-regions.

Some estimates in the database do not represent individual ecosystem services but refer to a TEV or the total value of provisioning, cultural or regulating services. These estimates cannot be included in the calculation of the average value of individual ecosystem services, but they can be included at the point of aggregation to a TEV for each eco-region. They are weighted by the number of estimates in order to give all data points equal weight in the average TEV (e.g. if 100 individual ecosystem-service values went into the aggregated TEV, then an estimate of TEV is given a 1/101 weighting and averaged with the aggregated TEV).

Calculate the values of ecosystem services by country

The objective of this step is to approximate the demographic differences that influence the extent to which ecosystem services provide value contexts. In particular, we seek to reflect the extent to which people depend on different services in different contexts. For example, rural communities tend to be more reliant on ecosystem services (directly or indirectly). As such, they are generally more vulnerable should those services be reduced. In addition, the number of beneficiaries is important – where there are more people, the value at risk is higher.[12] Similarly, if people are more affluent, they will have a higher willingness to pay, such that the total impact of losses will be higher.

The adjustments presented here are made using national data. As we discussed earlier, services accruing at the international level are not adjusted for local conditions because the underlying estimates already reflect international preferences. Local and national services are adjusted by income and population in order to transfer the median eco-region estimates to different countries.

Income adjustments
Adjustments for income are applied using current GNI ratios. This converts the standardised database figure from US purchasing power to local currency purchasing power. All values are expressed in USD/ha/yr.

Population dependency and distribution
The proportion of the population living in rural areas and the concentration of the urban population are used to adjust country-specific values, such that countries with higher proportions of rural population have higher valuation estimates. A population adjustment factor between 0 and 1 is calculated based on the country-level population density and the urban-rural population concentration relative to the global average. This adjustment is applied as a scale multiplier to each country-level estimate of local and regional ecosystem services. Global ecosystem services are not adjusted.

Table 21 presents the value of ecosystem services from tropical forests in a number of countries.

[12] The total change in societal welfare given a change in provision of services is the sum of all individuals' marginal willingness to pay for the change in service (Samuelson, 1954).

Table 21: Ecosystem-service values of tropical forests in different countries

USD/ha/yr		Brazil	Colombia	Dem. Rep. of the Congo	Indonesia
Total		**1,377**	**1,159**	**649**	**845**
Food from natural/semi-natural ecosystems	Local	264	209	76	200
Fibre, other raw materials	Local	844	669	243	638
Domestic and industrial water	Regional	84	67	24	64
Bioprospecting and medicinal plants	Global	81	81	81	81
Air purification	Global	1,091	1,091	1,091	1,091
Recreation	Regional	739	586	213	558
Stable climate	Global	622	622	622	622
Pollution control and waste assimilation	Regional	621	492	179	469
Erosion control	Regional	1,849	1,466	533	1,397
Flood control and protection from extreme events	Regional	45	36	13	34

The principle data requirements and key assumptions required for the calculations are presented in Table 22 and Table 23, respectively.

Table 22: Data required for estimating the current marginal value of ecosystem services

Variable	Suggested data source(s)
Primary estimates of ecosystem-service values	The TEEB database provides an excellent starting point. Other published estimates have been added from, for instance, the *Journal of Environmental Economics and Management*, *Ecological Economics*, the *American Journal of Agricultural Economics*, *Land Economics*, *Environmental and Resource Economics*, *Environment and Development Economics* and the *Journal of Environmental Economics and Policy*.
Unit and currency conversions	Exchange rates, inflation and GNI for income adjustments are sourced from the World Bank.
Population density and distribution	The World Bank provides data on average in-country population density and the distribution of populations between urban and rural areas.

Table 23: Key assumptions for estimating the current marginal value of ecosystem services

Assumption	Explanation
Underlying estimates provide a representative sample of the ecosystem services supplied by each of the six eco-regions.	The database is arguably the most comprehensive repository of primary estimates available. The data are distributed across 14 ecosystem services and 6 eco-regions, and the number of values for each range from 2 to 90 (after all exclusions). Although there is less confidence at the lower end of this range, we believe the data give a good indication of the likely scale of value that can be delivered by different ecosystem services.
Estimates more than two standard deviations from the mean should be excluded.	The objective of this valuation is to estimate the average value of different ecosystem services in a generalised eco-region. The largest outliers are special cases that do not provide a fair representation of the average values (e.g. strawberry crops in deserts). Therefore, they are inappropriate for this calculation.
The median provides a better estimate of the central tendency.	Statistically, the mean is a more efficient estimate (the variance of the mean of multiple random samples from the population will be lower), while the median is more robust (outliers have less influence on the result). Efficiency and robustness must typically be traded off against each other. Given the data distribution and the study's objective of identifying the most likely impact, we believe the median is more appropriate in this case.
Income and rural population concentration factor provide appropriate adjustments to reflect differences in the level of benefits and value delivered to people by ecosystem services in different countries.	Income accounts for differences in ability to pay (WTP is bounded by income) and it can serve as a proxy for the appetite for trading off environmental goods (some of which could be considered luxury goods in the short term) for other economic gains. Population adjustments reflect the number of people likely to benefit from ecosystem services and their reliance on those services.

Step 3: Estimate the average of marginal values of ecosystem services in each eco-region

The estimates calculated above represent the current marginal value of ecosystem services by eco-region and country, which correspond to the impact of additional land conversions today. This value is applied to new, in-year conversions.

It would be inappropriate to apply this current marginal value to land that was converted in the past. This is because impacts associated with additional losses in ecosystem services increase as more natural areas are converted over time. Two factors contribute to this situation: the increasing scarcity value and the increasing marginal-damage costs associated with cumulative environmental degradation. This is particularly true because ecosystems display threshold effects, whereby the damages increase exponentially after a particular point of loss of function.

Rather than applying the current marginal value, the appropriate measure for land converted in the past is the average of the marginal values resulting from increases in scarcity. This is because at any given point in time, each hectare of cleared ecosystem contributes equally to the prevailing lack of service provision (all else equal). Box 3 illustrates this point with an example.

Box 3: Why the average of marginal costs should be applied to occupation of previously converted land

In a hypothetical country, there are eight similar plots of forest ecosystem. The cost of lost ecosystem services from the first plot is $1. Each time an additional plot is converted, the scarcity increases and the value of the subsequent plot (equivalent to the cost of losing it) increases by $1 (i.e. a linear relationship between scarcity and value).

Year 1	**$1**								
Year 2		**$2**							
Year 3			**$3**						
Year 4				**$4**					
Year 5									

In the fifth year, four plots have been converted. At the time of conversion of each plot, the additional loss of ecosystem services increased. However, now that they have been converted, each of these plots contributes equally to the lack of ecosystem services delivered to the population. Indeed, if any of these plots were restored to forest, the benefit would be $4. They should therefore be assigned equal values. In year 5, the total cost of lost ecosystem services is $10 and the average marginal value over time is $2.5. As more plots are converted, the average cost associated with each cleared plot increases.

Year 1	$1							
Year 2	$1	$2						
Year 3	$1.5	$1.5	$3					
Year 4	$2	$2	$2	$4				
Year 5	$2.5	$2.5	$2.5	$2.5				
Key	Average marginal value – previous conversion		Marginal value – New in-year conversion		Intact natural ecosystems			

In order to calculate the average marginal value, we need to assume a relationship between the extent of natural land areas lost (over time) and the corresponding value loss associated with converting an additional hectare. Figure 13 illustrates a number of possible relationships. The graph demonstrates that if the current marginal value (y) is applied to all areas of land use, the impact given by the area under line i would be a gross overestimate. Three different curves are shown to illustrate the possible relationship. In curve A, costs increase linearly. In curves B and C, the incremental costs increase slowly at first and then more rapidly as a greater total area is lost. While one of these relationships may hold true, the actual relationship will differ across ecosystem services in different contexts.

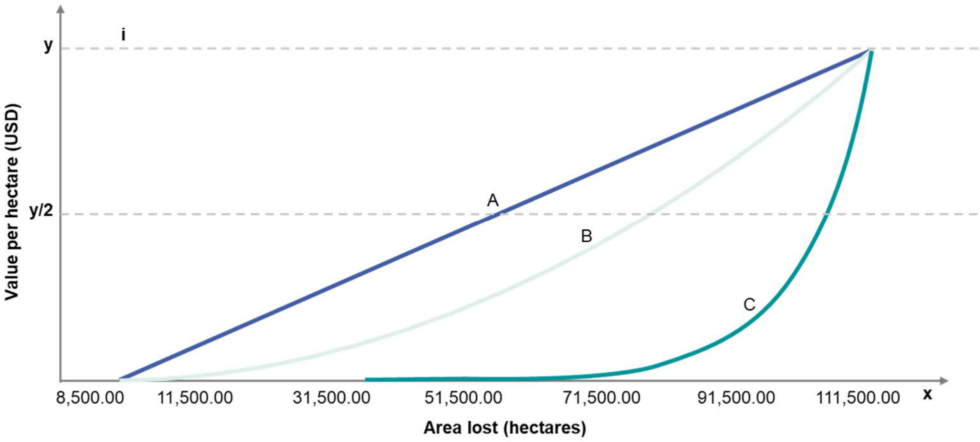

Figure 12: Ecosystem services have increasing marginal value as more natural areas are lost

Therefore, we assume a linear relationship (curve A) in our calculations. This conservative approach leads to higher estimates of potential impacts (as any other convex[13] relationship would suggest that the impacts of past conversions are lower). In this instance, the average marginal cost is half the current marginal cost.

Step 4: Estimate the societal cost by applying marginal values to environmental outcomes

After we have calculated the area of land use, the extent of ecosystem-services loss and the value of ecosystem services, calculating the overall societal cost of land use is straightforward arithmetic. The results of this step give the estimate of the lost value as a result of ecosystem-services reductions associated with land use.

[13] A concave relationship is improbable, as such a relationship would suggest that the value of ecosystems declines as they become scarcer. An S curve in which incremental costs increase slowly at first and then more rapidly (but as a greater total area is lost, the damage costs level off) may be possible. In such a relationship, *A* would still be conservative if the upward convex arc was longer than the levelling-off concave arc (i.e. costs increase over a long period of time and then level-off sharply as degradation passes a damage threshold). The fact that more recent ecosystem valuation studies result in higher marginal ecosystem values than earlier studies supports the assertion that we have not reached the concave portion of a sigmoidal curve (at least at the level of aggregated ecosystem types). This indicates that *A* represents a conservative approximation of the relationship.

In cases where services are only reduced, we calculate the appropriate portion of lost value based on the percentage change in service provision.

Equation: Calculate the lost ecosystem service value, per hectare per eco-region

$$\begin{aligned} &\boldsymbol{Lost\ ecosystem\ service\ value}\ (\$/\boldsymbol{ha})_{\boldsymbol{el}} \\ &\qquad = \boldsymbol{Extent\ of\ ecosystem\ service\ loss}\ (\%)_{\boldsymbol{el}} \\ &\qquad \times\ \boldsymbol{Ecosystem\ service\ value}\ (\$/\boldsymbol{ha})_{\boldsymbol{el}} \end{aligned}$$

where:

The extent of ecosystem-service loss represents the ratio of the specific land use type relative to the national weighted average natural-state ecosystem services.

The ecosystem service value represents the weighted national average of ecosystem services based on the proportion of each biome within the national borders.

2.5.4. SOURCES

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Topic-specific details

2.6. WASTE

- *Topic description*
- *Impact pathway*
- *Quantification and monetary valuation*
  - *Measuring impact drivers*
  - *Environmental outcomes*
  - *Impacts on society and monetary valuation*
- *Sources*

2.6.1. TOPIC DESCRIPTION

Corporate activities in all sectors result generate solid waste. The disposal of this solid waste can lead to a range of environmental outcomes that adversely affect human wellbeing, thereby carrying a societal cost. In this paper, we set out a methodology for identifying, quantifying and valuing that cost in monetary terms.

Most material impacts associated with solid waste are covered in this paper, but two classes of related impacts are partially addressed in other papers. In terms of GHG and air-pollution outcomes, waste disposal is an intermediate step. The approaches to quantifying these outcomes as they relate to waste disposal are defined in this methodology. We believe that this increases the accuracy of societal impact estimates and increases the applicability of the results to companies, which tend to treat waste as a discrete environmental issue. This comprehensive approach adds some complexity but is important because GHGs and air pollution make up a significant proportion of the societal cost of a tonne of waste.

Importantly, this methodology is concerned with the impacts of waste disposal. It does not attempt to evaluate the costs associated with design or production inefficiencies that may be indicated by the presence of waste.

For solid-waste disposal, the type of waste and the method of its disposal are key factors that dictate the environmental outcomes. Common types of waste, disposal approaches and environmental outcomes are discussed below. The impact pathway describes how these factors influence environmental outcomes and, subsequently, affect people.

Types of Waste

Solid waste is typically classified as either hazardous or non-hazardous.

Hazardous waste: This is waste that is defined as particularly dangerous or damaging to the environment or human health, usually through inclusion on official lists by regulators.

Non-hazardous waste: This covers all types of waste not classified as hazardous. In other contexts, it may cover all waste not otherwise classified.

The type of waste influences the type and extent of impacts associated with different disposal techniques. More specific classifications may be required to ensure that the right impacts are allocated to the waste produced by a given activity. For example, inert waste is a subclass of non-hazardous waste that is chemically unreactive and does not decompose. Therefore, it does not release GHGs.

Approaches to waste disposal

The method of treating solid waste influences the type and severity of environmental outcomes. The most common treatment approaches are listed in the following.

Incineration: The combustion of solid waste. This produces various flue gases, residual fly ash and disamenity from the undesirable aesthetic qualities of waste incinerators (see below). Fly ash may be disposed of in landfill sites or used as a construction aggregate. The heat produced by incineration may be recovered to produce electricity.

Landfill: The disposal of solid waste in specially designated areas. Waste (except inert waste) decomposes in landfill sites, producing GHGs and leachate (liquid released from landfill sites, principally due to infiltration by rainfall). The presence of a landfill also has a disamenity impact on surrounding residents and visitors to the vicinity. Landfill quality varies dramatically. We use the term to cover everything from unmanaged dumpsites in which leachate and GHGs can escape unabated into the environment to carefully managed, impermeably lined, sanitary landfills in which these emissions are collected, processed and, in some cases, used to generate electricity.

Recycling: The disassembly and processing of solid waste to constituent materials for reuse. This requires energy and results in production-grade materials. The use of recycled raw materials avoids the consumption of energy and materials that would otherwise be required for extracting and processing virgin raw materials. The principle methodology issues associated with recycling relate to the quantification of emissions rather than the valuation of those emissions (which is done in the same way as for any other industrial process). The impacts of recycling should be allocated between the company demanding the recycled raw material and the company producing the waste. Therefore, recycling does not have a dedicated section in this report.

Specialist processing: Local regulations may mandate or recommend specialised treatment of some solid-waste products, especially hazardous waste (e.g. hydrocarbons and radioactive waste). As the nature of the treatment and the resulting impacts will be highly specific to each situation, we do not present a generalised methodology here.

2.6.2. IMPACT PATHWAY

In order to value corporate environmental impacts on society, we must establish a link between waste and impacts on humans and via environmental outcomes. This is reflected in the impact pathway shown in Figure 13.

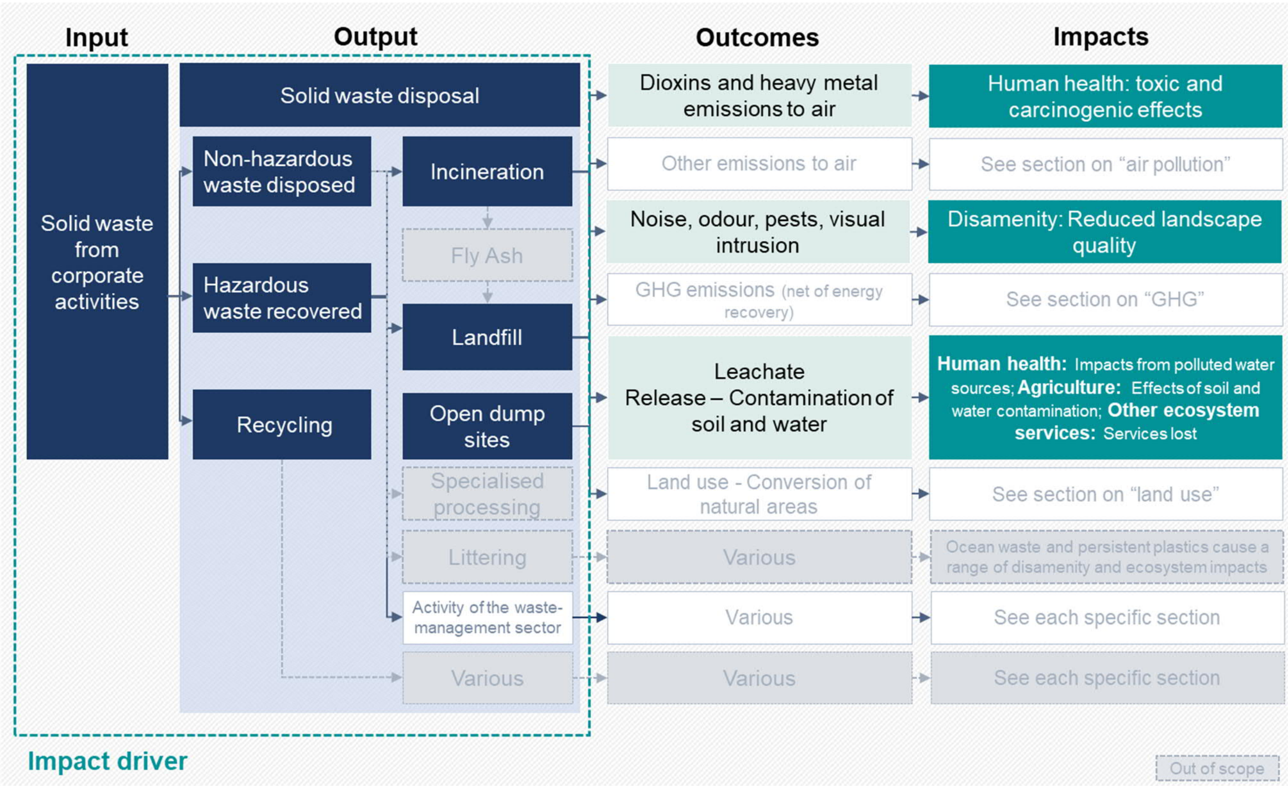

Figure 13: Simplified impact pathway waste

2.6.3. QUANTIFICATION AND MONETARY VALUATION

This section covers three steps in more detail: (i) measuring impact drivers, (ii) measuring changes in the state of natural capital and (iii) valuing impacts. For guidance on actions, see the Natural Capital Protocol.

Given the state of research and available studies, it is not feasible to standardise all aspects of quantification. Detailed examples are given in the following to illustrate how quantification may work in practice.

(i) Measuring impact drivers

The amount of waste produced by business activities is typically calculated in tonnes using either direct, on-site measurements or estimations. If direct information is not available, techniques such as LCA and/or environmentally extended input-output (EEIO) models can be used.

Different types of waste will have different environmental outcomes in certain circumstances. Therefore, they are often recorded separately. This distinction is particularly relevant for the impact of GHGs and leachate from landfills, and for the impact of GHGs and air pollution from incineration. Despite inconsistencies in the definitions of the two categories between countries, the approaches that we have developed or adapted from the literature in each of these areas take this distinction into account.

The most influential factor in determining the environmental outcomes associated with the disposal of solid waste is the mode of treatment. Therefore, it is important to understand how much waste is disposed of through landfills or incineration. If the mode of treatment is not known, it can be approximated using country- or state-level information, although this will only provide a picture of the average impacts in each region. This information can be obtained from publicly accessible sources, such as sources provided by governments' environment ministries or international databases maintained by the OECD or the World Bank.

The availability of actual (rather than modelled or estimated) metric data will vary according to the company's level of control over the producers and users of that data. This control is likely to vary across a company's value chain.

The key indicators that should be measured in metric tonnes are:

- Hazardous solid-waste disposal,
- Non-hazardous solid-waste disposal and
- Waste characteristics (e.g. fossil carbon percentage) and composition (e.g. principle materials) if known.

The first step is to estimate the environmental metric, which represents the waste flows by composition (i.e. hazardous or non-hazardous) and disposal method (landfill or incineration). These can be estimated directly using information provided by companies, or indirectly using such techniques as LCA or EEIO analysis. When a direct approach is taken, waste data should be apportioned to landfill and incineration using actual data where available. Otherwise, general trends at a country or sub-national level can be used.

Table 24: Available metric data for waste per value chain level

	Metric data
Own operations	Information on waste tonnage broken down by waste type and composition should be available in the company's management information. Estimation techniques used for supply chains can also be utilized if direct data are unavailable.
Immediate/ key suppliers	Information on waste tonnage broken down by waste type and composition may be available from some suppliers. Where such information is unavailable, gaps in metric data can be filled using modelling techniques, such as EEIO.
Upstream/ supply chain	Reliable metric data on waste tonnage, type and composition are unlikely to be available from indirect suppliers.

	Metric data can be modelled using EEIO techniques, which can be supplemented with data from customer surveys or industry information.
Downstream/ use phase	Reliable metric data on waste tonnage, type and composition are unlikely to be available from users. Metric data can be modelled using EEIO techniques, which can be supplemented with data from customer surveys or industry information.
End of life/ reuse impacts	Some metric data can be derived using physical production characteristics, such as the masses of constituent materials. Other metric data can be modelled using EEIO techniques, which can be supplemented with data from customer surveys or industry information.

I. *Greenhouse gas emissions from landfills and incineration*

(ii) Quantifying the environmental outcomes and *(iii) Estimating the societal impacts*

The environmental outcomes (contributions to climate change) and the societal impacts associated with those outcomes are evaluated by applying the societal cost of carbon (SCC) to net GHG emissions (see the GHG valuation methodology for more detail).

GHG emissions from waste disposed at landfill sites are estimated as follows:

- GHG emissions (principally CH4) from each tonne of waste sent to a landfill are estimated over 90 years using the Intergovernmental Panel on Climate Change (IPCC, 2000a) Waste Model based on the mass and type of waste, and the conditions of the landfill.
- The social cost of carbon is applied to the total GHG emissions released each year.
- The present value of the associated impacts is then calculated by applying a social discount rate of 3.5%.[14]

Key assumptions:

- Methane emissions are the only GHG emissions considered significant from landfills. CO_2 emissions from landfill sites – emitted either directly as landfill gas or indirectly through the flaring of CH_4 – are non-anthropogenic. Therefore, they do not contribute to human-caused climate change (IPCC, 2000a).
- Hazardous and non-hazardous waste can be considered together. The organic carbon content of waste varies based on its composition, which primarily changes as a result of the industry type rather than on the basis of whether the waste is hazardous or non-hazardous.

[14] Based on values chosen by governments like HM Treasury (2018): The Green Book – Central Government Guidance on Appraisal and Evaluation. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/938046/The_Green_Book_2020.pdf

- The methane-capture rate of landfill sites is assumed to be 0% unless specific capture rates are reported. Where the proportion of sites with capture is known but their capture efficiency is unknown, we assume 20% efficiency. The methane-capture rate significantly influences the final GHGs estimated using the IPCC model. It is therefore prudent to assume that this is 0% unless rates are known, a view that is consistent with the IPCC's guidance (2006c).
- Methane emissions are considered over 90 years, which is the maximum duration for the landfill gas-generation phase recommended by the IPCC (IPCC, 2000a). This is considerably longer than the 25-30 years after which COWI (2000a) suggests that landfill gas reaches insignificant amounts. As the methane emissions estimated by the IPCC model follow a first-order decay function and, hence, trend towards insignificant amounts in later years, we believe taking the uppermost limit is prudent in order to capture all impacts.
- The rate of methane production from waste decomposition can be affected by the country's climate conditions. The IPCC model uses a first-order decay function to model methane produced by deposited waste. This function is fully explained by the IPCC (2006c), which presents several reaction constants based on the climatic conditions of the country of interest.
- All sites are classed as "uncategorised solid-waste disposal sites" unless details about the specific site are known (e.g. depth and moisture content). The IPCC advise that the characteristics and assumptions for uncategorised solid-waste disposal sites should be used "if countries cannot categorise their solid waste disposal site[s]" (IPCC, 2006c). This is consistent with a situation in which a company cannot provide specific information on the relevant waste-disposal site(s).

GHG emissions from waste sent to incineration sites are estimated as follows.

- CO_2 emissions per tonne of waste are estimated by applying the carbon intensity of the incineration process to the volume of waste sent to incineration. Table 25 presents the variables influencing CO_2 emission per tonne of incinerated waste (IPCC, 2000c).

Table 25: Variables influencing CO_2 emission per tonne of incinerated waste

Variable	Default values: Non-hazardous	Default values: Hazardous
The carbon content of wet waste	40%	50%
Fossil carbon fraction	40% (of total carbon)	90% (of total carbon)
Efficiency of combustion (depends on incinerator type)	95%	99.5%
Tonnes fossil CO_2 per tonne incinerated waste[15]	**0.557**	**1.642**

[15] Calculated by multiplying the three proportions in Table 25 by each other and a factor to convert carbon in CO_2 mass. For example, for non-hazardous waste, this is: 1 tonne × (0.4 × 0.4 × 0.95) × $^{44}/_{12}$ = 0.557.

If the carbon content or fossil carbon percentage of waste is known, or the efficiency of combustion at a specific site is known, then these figures should be used instead.

- Unlike GHG emissions from landfill sites, which are emitted for many years after the waste is disposed of, GHG emissions from the incineration of waste are instantaneous.

Key assumptions:

- Along with the large quantities of CO_2 that are released into the atmosphere, smaller quantities of nitrous oxide (N_2O) and CH_4 are also released. According to the IPCC (2000c), CO_2 is the most significant GHG resulting from waste incineration by at least two orders of magnitude. For this reason, only CO_2 emissions are considered further in this methodology.

Calculating net GHG emissions by adjusting for energy recovery

The principal environmental benefit of energy recovery is the avoidance of emissions. When landfill gas or incinerated waste is used to generate electricity, there is no need for that electricity to be generated by other means. Therefore, the potential emissions associated with that generation are avoided.

A similar methodology is used to estimate the avoided emissions from landfill and incineration. The only divergence is the variable used for the energy potential of waste, which is explained below.

GHG emissions can be avoided through energy recovery from both landfill sites and waste incineration. Landfill gas to energy (LFGTE) involves capturing landfill gas and burning it to produce energy, usually using a gas engine that runs an electric generator (Willumsen, 2004). Similarly, when waste is incinerated, it can be used to produce electricity, commonly known as waste-to-energy (WTE) (Kaplan et al., 2009). Therefore, this methodology considers GHGs avoided through both LFGTE and incineration WTE.

Landfill gas to energy

In some countries, a proportion of waste sent to landfills each year is used to generate electricity (LFGTE). As such, it displaces GHGs that would otherwise have been produced had that electricity been generated through other means. Where there is no evidence that LFGTE is present at the site or in the country of interest, then the default assumption should be that LFGTE does not occur and that no adjustment is necessary.

If LFGTE is present, then the GHG emissions avoided through energy recovery per tonne of waste should be estimated using the following equation.

Equation: Avoided emissions from LFGTE

$$\boldsymbol{Avoided\ GHG\ emissions\ from\ LFGTE\ (tCO2e)}$$
$$= \boldsymbol{waste\ sent\ to\ LFGTE\ (t) \times energy\ potential\ of\ waste\ \left(\frac{kWh}{t}\right)}$$
$$\boldsymbol{\times\ grid\ carbon\ intensity\ \left(\frac{tCO2e}{kWh}\right)}$$

Table 26 explains the variables required to estimate GHG emissions per tonne of waste avoided through LFGTE.

Table 26: Variables required to estimate GHG emissions per tonne waste avoided through LFGTE

Variable	Explanation
Tonnes of waste sent to LFGTE site	The World Bank estimates the number of LFGTE plants worldwide (Willumsen, 2004) and reports the tonnage of waste processed by plants in each country. More recent national industry statistics should be sought wherever possible.
Energy potential of waste, kWh/tonne of waste	The energy potential of waste depends on the type of waste and the technology used to collect and convert it. For example, Mendes et al. (2004) calculate a value of 166 kWh/tonne, assuming that 50% of the CH_4 is captured and that it is burnt in a gas engine with 30% energy-recovery efficiency.[16] Site-specific values should be sought wherever possible.
Carbon intensity of national or local electricity grid, CO_2e/kWh	The International Energy Agency provides the CO_2 intensities of national and regional electricity grids around the world.

Depending on the available data, it may be necessary to perform different calculations to reach the result shown in equation Avoided emissions from LFGTE. In particular, if the tonnage of waste sent to LFTGE sites is not known, the calculation can be modified based on the national prevalence of LFGTE (as a percentage).

Incineration waste to energy (WTE)

The same approach should be used to estimate the GHGs avoided when energy is recovered through waste incineration. The variables set out Table 25 can be used with the exception of the energy potential of waste, which should be replaced with a variable covering the energy recovered per tonne of waste incinerated. In practice, this value will vary based on waste composition and incinerator/generator specification. However, unless location-specific data are available, we assume that where energy-recovery technology is available, it is common between countries. The energy potentials shown in Table 27 have been derived for use in European policy making (COWI, 2000b).

[16] This is broadly consistent with other "default" values in the literature. For example, Willumsen (2004) uses values of 50% and 37% for CH_4 capture and energy-recovery efficiency, respectively.

Table 27: Energy potential based on type of energy recovery in facility

Type of energy recovery in facility	Energy potential (kWh/tonne waste)
Electricity only (assume 25% recovery rate)	625
Electricity plus heat (assume 83% recovery rate)	2,075

The lower energy potential (625 kWh/t) should be used as a default for countries in which energy recovery occurs but the type of recovery technology is unknown. For countries in which energy recovery is commonly used to produce both heat and electricity, 2,075 kWh/t should be used.

II. *Disamenity (landfill and incineration)*

(ii) Quantifying the environmental outcomes and (iii) Estimating the societal impacts

Environmental outcomes (i.e. increases in odour, noise and changes to visual amenity) and societal impacts are evaluated in one step using a hedonic pricing model. This model uses price information from a surrogate market (i.e. the housing market) to measure the implicit value of a non-market benefit or disbenefit (in this case, the disamenity associated with living near a waste-management site). We have developed a multivariate hedonic transfer function based on a meta-analysis of hedonic pricing studies in the academic literature. This function is used to estimate the willingness to pay (WTP) (to avoid disamenity) based on local average house prices, household density and the housing-market discount rate.

The societal cost of disamenity is then expressed in terms of the estimated amount per tonne of waste based on site lifetime and waste-flow data.

House-price differentials (at given distances from the site) relative to house prices not near waste-management sites are assumed to reflect the societal costs of the disamenity of waste facilities after controlling for other factors that affect house prices. This is the standard approach used by academics and governments. The hedonic transfer factor is derived from six primary studies from five countries[17] on how proximity to waste-management facilities affects house prices for a given average house price and household density. Adjusting for these two variables is considered an acceptable approximation of disamenity for any given country given the limited global coverage of existing primary estimates.

[17] A review of the literature identified six such functions from primary studies of landfill sites in the UK, Israel, South Africa, Uganda and Nigeria (Cambridge Econometrics et al., 2003; Eshet et al. 2007; Du Preez & Lottering, 2009; Nahman, 2011; Isoto & Bashaasha, 2011; Akinjare et al., 2011).

III. ***Leachate release (from landfill)***

(ii) Quantifying the environmental outcomes

The likelihood and severity of potential environmental outcomes associated with leachate from landfill are estimated on a scale of 1 to 1,000 using the hazard rating system (HARAS) leachate risk model (Singh et al., 2012). This model is based on source-pathway-receptor relationships.

The HARAS model is available in two forms. The most detailed form is applicable to site-specific analysis with significant data-input requirements. The simplified version can be used for high-level assessments based on five variables:

- Proportion of hazardous waste,
- Climatic conditions and precipitation,
- Presence of a liner at landfill sites,
- Geology and soil permeability, and
- Population density in proximity to sites.

The HARAS leachate risk model is peer reviewed and widely used to evaluate the leachate risk. The simplified version of the HARAS model is considered appropriate where the data requirements of the more complex version cannot easily be met.

(iii) Estimating the societal impacts

Societal impacts are assessed by first identifying a worst-case estimate of leachate clean-up costs as a proxy for the worst-case societal impact and subsequently adjusting that estimate by multiplying it by the HARAS risk score (expressed as a fraction between 0 and 1). Adjustments should be made for local purchasing power parity (PPP) relative to the US (from which the leachate clean-up cost estimates are sourced).

Clean-up costs are widely used as a proxy to estimate the value of non-market impacts when damage costs are unavailable. In practice, they are likely to be a lower-bound proxy for the societal cost of leachate impacts when data on the latter are unavailable. The selection of a worst case is equivalent to the worst-case criteria from the HARAS model (risk score = 1,000). The scaling of the worst-case damage costs according to a risk factor is appropriate because the impacts of leachate in any individual case are uncertain. This is consistent with the approach taken in national studies (e.g. CSERGE, 1993).

Aside from the factors that influence the HARAS risk score, we only adjust for PPP, assuming an income elasticity of 1, because there is insufficient evidence for other systematic preference adjustments.

value balancing alliance

IV. ***Air pollution (from incineration)***

(ii) Quantifying the environmental outcomes

Dioxin and heavy-metal emissions are calculated using incineration-emission factors. We estimate the change in the incidence of cancer and intelligence quotient (IQ) points by multiplying emissions by linear dose-response functions. Dose-response functions are based on epidemiological studies at given ambient concentrations and emission levels. The air pollutants traditionally considered are:

- NO_x,
- SO_x,
- NH_3,
- $PM_{2.5}$,
- PM_{10} and
- VOCs.

The environmental outcomes (i.e. increased ambient concentration of pollution) of traditional air pollution are considered in the VBA air pollution methodology paper. Avoided emissions are estimated as per avoided GHG emissions from incineration, with the air-emission intensity of electricity and heat generation replacing carbon intensity.

(iii) Estimating the societal impacts

To estimate the societal impacts, the increased incidence of cancer and lost IQ points are multiplied by the weighted societal cost of cancer (VSL and of non-fatal cancer) and the WTP to avoid loss of IQ points. VSL estimates are representative of the welfare loss associated with health endpoints. The welfare values associated with the health, agriculture and visibility impacts of air pollutions are considered in the VBA air pollution methodology paper.

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3. APPENDICES

3.1. ENVIRONMENTAL INDICATORS

Table 28: Environmental indicators

Indicator	Sub-indicator	Context	Frameworks defining measurements e.g.[18]
Environmental			
GHG	GHGs	n/a	GRI (305-1), CDP (C6, C7), CDSB (R03, R04), SASB (110a.1), GHG Protocol, WEF IBC Greenhouse gas (GHG) emissions
Other air emissions	NH_3	Urban, peri-urban, rural, transport	GRI (305-7), SASB (120a.1), WEF IBC Fine particulate matter
	NOx	Urban, peri-urban, rural, transport	
	PM_{10}	Urban, peri-urban, rural, transport	
	$PM_{2.5}$	Urban, peri-urban, rural, transport	
	SOx	Urban, peri-urban, rural, transport	
	NMVOC	Urban, peri-urban, rural, transport	
Water consumption	Water consumption	n/a	GRI (303-3), CDP (W1), CDSB (R04), SASB (140a.1), WEF IBC Fresh water consumption in water stressed areas
Water pollution	Nitrogen	Freshwater, marine water, unspecified	GRI (303-4)[19], WEF IBC Nutrients
	Phosphorus	Freshwater, marine water, unspecified	
	Antimony	Freshwater, marine water, unspecified	
	Arsenic	Freshwater, marine water, unspecified	
	Cadmium	Freshwater, marine water, unspecified	
	Chromium	Freshwater, marine water, unspecified	
	Copper	Freshwater, marine water, unspecified	
	Lead	Freshwater, marine water, unspecified	
	Mercury	Freshwater, marine water, unspecified	
	Nickel	Freshwater, marine water, unspecified	
	PAHs	Freshwater, marine water, unspecified	
	Zinc	Freshwater, marine water, unspecified	
Land use	Agriculture	Wheat, vegetables, fruit and nuts cereal grains, oilseeds, sugarcane and sugar beet, plant-based fibres, crops nec, animal rearing, paddy rice	GRI (304-1, 304-3, 304-4), CDP (F1), WEF IBC Land use and ecological sensitivity
	Forestry	n/a	
	Paved	n/a	
Waste	Hazardous	Incineration, landfill	GRI (306-5)[20], SASB (150.a)
	Non-hazardous	Incineration, landfill	

[18] Note that these frameworks define indicators in physical terms (mass, volume, etc.). Valuation is not addressed in them.
[19] Based on GRI 306 (2020)
[20] Based on GRI 303 (2018)

3.2. LIST OF FIGURES AND TABLES

Tables:

Figures:

3.3. LIST OF ACRONYMS

A4S	Accounting for Sustainability
CC	Capitals Coalition
CDSB	Climate Disclosure Standard Board
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
CH4	Methane
CV	Contingent valuation
DALYs	Disability-adjusted life years
DPSIR	Drivers, pressures, states, impacts, responses
ESG	Environmental, social and governance
GHG	Greenhouse gas
GRI	Global Reporting Initiative
GVA	Gross value added
GWP	Global warming potential
HALYs	Health-adjusted life years
HFCs	Hydrofluorocarbons
HPM	Hedonic pricing method
IASB	International Accounting Standards Board
IIRC	International Integrated Reporting Framework (<IR>)
IMP	Impact-management project
IO	Input output
ISO	International Organization for Standardization
IVR	Impact Valuation Roundtable
IWAI	Impact-Weighted Accounts Initiative
NCP	Natural Capital Protocol
NF_3	Nitrogen trifluoride
NFRD	Non-Financial Reporting Directive
N_2O	Nitrous oxide/laughing gas
OECD	Organisation for Economic Co-operation and Development

OEF	Organisation environmental footprint
OLS	Ordinary least squares
PCFs	Perfluorocarbons
PEF	Product environmental footprint
QALYs	Quality-adjusted life years
SASB	Sustainability Accounting Standards Board
SCC	Social Cost of Carbon
SDR	Social discount rate
SF_6	Sulphur hexafluoride
TCFD	Task Force on Climate-Related Financial Disclosures
TEV	Total Economic Value
UNPRI	United Nations Principles for Responsible Investments
VBA	Value Balancing Alliance
VSL	Value of a statistical life
WBCSD	World Business Council for Sustainable Development
WSI	Water Stress Index
WTA	Willingness to accept
WTP	Willingness to pay
YLL	Years of lost life

3.4. GLOSSARY

Term	Definition	Source
Activity	Actions taken or work performed through which inputs, such as funds, technical assistance and other types of resources, are mobilized to produce specific outputs.	DAC/OECD (2010)
Altruistic value	The value an individual places on knowing that a good exists, so that others alive today are able to enjoy it.	ISO 14008 (2019)
Amortisation	Accounting definition for intangible assets according to IAS 38.8. Amortisation is the systematic allocation of the depreciable amount of an intangible asset over its useful life.	IFRS Foundation (2010)
Asset	Definition according to the IFRS Conceptual Framework (rev. 2018), Par. 4.3: An asset is a present economic resource controlled by the entity as a result of past events (an economic resource is a right that has the potential to produce economic benefits).	IFRS Foundation (2010)
Bequest value	The value an individual places on knowing that a good will continue to exist so that individuals born in the future will be able to enjoy it.	ISO 14008 (2019)
Capital	Stocks of value on which all organizations depend for their success that serve as inputs to their business models, and which are increased, decreased or transformed through the organization's business activities and outputs. The capitals are categorized in this Framework as financial, manufactured, intellectual, human, social and relationship, and natural.	IIRC (2013)
Carbon Cycle	The term used to describe the flow of carbon (in various forms, e.g., as carbon dioxide (CO2)) through the atmosphere, ocean, terrestrial and marine biosphere and lithosphere. In this report, the reference unit for the global carbon cycle is GtCO2 or GtC (Gigatonne of carbon = 1 GtC = 1015 grams of carbon. This corresponds to 3.667 GtCO2).	IPCC (2014)
CO_2e	The universal unit of measurement to indicate the global warming potential (GWP) of each greenhouse gas, expressed in terms of the GWP of one unit of carbon dioxide. It is used to evaluate releasing (or avoiding releasing) different greenhouse gases against a common basis.	GHG Protocol (2015)

Contingent valuation	In contingent valuation, the good to be valued is presented in its entirety as a bundle of its attributes. The respondents are asked for their WTP to avoid a deterioration in quality or quantity of the good, or to ensure an improvement. Alternatively, respondents are asked for their WTA to tolerate a deterioration or to forgo an improvement.	ISO 14008 (2019)
Cradle-to-gate	See "upstream".	ISO 14044 (2006)
Cradle-to-gate	Life-cycle stages from the extraction or acquisition of raw materials to the point at which the product leaves the organization undertaking the assessment.	PAS (2011)
Cradle-to-gate inventory	A partial life cycle of an intermediate product, from material acquisition through to when the product leaves the reporting company's gate (e.g., immediately following the product's production).	GHG Protocol (2011)
Cradle-to-grave	LCA addresses the environmental aspects and potential environmental impacts (e.g. use of resources and environmental consequences of releases) throughout a product's life cycle from raw-material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave).	ISO 14044 (2006)
Cradle-to-grave	Life-cycle stages from the extraction or acquisition of raw materials to recycling and disposal of waste.	PAS (2011)
Cradle-to-grave inventory	Removals and emissions of a studied product from material acquisition through to end-of-life.	GHG Protocol (2011)
Depreciation	Depreciation is the systematic allocation of the depreciable amount of an asset over its useful life.	IFRS Foundation (2010) - IAS 16.6
Direct GHG emissions	Emissions from sources that are owned or controlled by the reporting company.	GHG Protocol (2004)
Direct use value	Value arising from the use of a good, which might or might not have a market price.	ISO 14008 (2018)
Disability-adjusted life years	A burden-of-disease measure based on the number of years lost due to premature death, disease or disability. The loss of one healthy year of life due to death or illness is equal to one DALY. DALYs were developed by the World Bank and World Health Organization in 1993 to quantify disease and disability burdens globally, and to determine intervention priorities. Instead of a scale of health (like QALYs), DALYs are related to a degree of disability for a specific disease or disability and range from none (0) to death (1).	Gold et al. (2002)
Discounting	Process of calculating the present value of future monetary values.	ISO 14008 (2019)

Discount rate	Definition that must be used for calculating the amount of provisions: pre-tax rate (or rates) that reflect(s) current market assessments of the time value of money and the risks specific to the liability. The discount rate(s) shall not reflect risks for which future cash-flow estimates have been adjusted.	IFRS Foundation (2010) – IAS 37.47
Downstream	GHG emissions or removals associated with processes that occur in the life cycle of a product subsequent to the processes owned or controlled by the reporting company	GHG Protocol (2011)
Driver (direct and indirect)	Any natural or human-induced factor that directly or indirectly causes a change in an ecosystem.	TEEB (2010)
Effects	Intended or unintended change directly or indirectly due to an intervention.	DAC/OECD (2010)
Environmentally extended input-output (EEIO) models	Traditional input-output (IO) tables summarize the exchanges between major sectors of an economy (Miller and Blair 2009). For example, output from the footwear manufacturing sector results in economic activity in associated sectors from cattle ranching to accounting services. Environmentally extended input-output models integrate information on the environmental impacts of each sector within IO tables (Kitzes 2013; Leontief 1970; Tukker et al. 2006).	Natural Capital Coalition (2016)
Existence value	The value an individual places on knowing that a good will continue to exist regardless of the individual's use of that good now or in the future. This includes biodiversity, and many cultural, aesthetic and spiritual aspects of human life (e.g. the deep seas, which might never be experienced by humans).	ISO 14008 (2019)
Externality	Consequence of an activity that affects interested parties other than the organisation undertaking the activity, for which the organization is neither compensated nor penalized through markets or regulatory mechanisms.	ISO 14007 (2019)
Gate-to-gate	Product's life cycle starting with production.	ISO 14007 (2019)
Gate-to-grave	Product's life cycle including use, end-of-life treatment, recycling and final disposal.	ISO 14007 (2019)
Greenhouse gases (GHG)	For the purposes of this standard, GHGs are the seven gases covered by the UNFCCC: carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), sulphur hexafluoride (SF_6), and nitrogen triflouride (NF_3).	GHG Protocol (2015)
Global Warming Potential (GWP)	A factor describing the radiative forcing impact (degree of harm to the atmosphere) of one unit of a given GHG relative to one unit of CO2.	GHG Protocol (2004)
Gross value added (GVA)	Output (at basic prices) minus intermediate consumption (at purchaser prices). This is the balancing item of the national accounts' production account. GVA can be broken down by industry and institutional sector. The sum of GVA across all industries or sectors plus taxes on products minus subsidies on products gives gross domestic product. By subtracting the	Eurostat (2020)

	consumption of fixed capital from GVA, the corresponding net value added (NVA) is obtained. The concepts of "GVA at market prices", "GVA at producer prices" and "GVA at basic prices" are not used in ESA 2010.	
Health-adjusted life years	A summary of population health measurements that combines death and morbidity impacts.	Gold et al. (2002)
Hedonic pricing method (HPM)	Uses statistical methods to isolate the implicit "price" of each of these characteristics. Hedonic pricing can be applied to the labour market, as wages also reflect how occupational risks to human health (morbidity and mortality) affect the monetary value of labour. Hedonic pricing can be used to explain variations in wages across occupations after taking the different risks into account and, thereby, to estimate the risk premium paid. This information can be used to estimate the monetary value of changes in mortality risks (also referred to as the value of a statistical life).	ISO 14008 (2019)
Human capital	The knowledge, skills, competencies and attributes embodied in individuals that facilitate the creation of personal, social and economic well-being.	Social & Human Capital Coalition (2019)
Human well-being	Concept prominently used in the Millennium Ecosystem Assessment. It describes elements largely agreed to constitute a “good life”, including basic material goods, freedom and choice, health and bodily well-being, good social relations, security, peace of mind, and spiritual experience.	TEEB (2010)
Impact	Positive and negative, primary and secondary long-term effects produced by a development intervention (directly or indirectly, intended or unintended).	DAC/OECD (2010)
Impact management	No clear, authoritative definition currently exists.	IMP (2020)
Impact measurement	The measurement and management of the processes of creating social and environmental impacts in order to maximize and optimize them.	IMP (2020)
Impact pathway	Describes how, as a result of a specific business activity, a particular impact driver results in changes in natural capital and how those changes affect different stakeholders.	Natural Capital Coalition (2016a)
Impact value chain	A logic model used for impact investing first described in 2004 in the Double Bottom Line Project Report: Assessing Social Impact in Double Bottom Line Ventures. Also referred to in the 2014 report: Measuring Impact: Subject paper of the Impact Measurement Working Group of the G8 Social Investment Impact Taskforce.	IMP (2020)

Indirect use value	The benefits that humans derive from ecosystems' services without direct intervention (e.g. the erosion or flood-risk protection of a forest).	ISO 14008 (2019)
Input	The financial, human and material resources used for a development intervention.	DAC/OECD (2010)
Internalisation	Act of taking externalities into account in decision-making.	ISO 14007 (2019)
Life-cycle assessment	Also known as life-cycle analysis. A technique used to assess the environmental impacts of a product or service through all stages of its life cycle from material extraction to end-of-life (disposal, recycling or reuse). The International Organization for Standardization (ISO) has standardized the LCA approach under ISO 14040 (UNEP 2015). Several life-cycle impact assessment (LCIA) databases provide a useful library of published estimates for different products and processes.	Natural Capital Coalition (2016a)
Logical framework	Management tool used to improve the design of interventions, most often at the project level. It involves identifying strategic elements (inputs, outputs, outcomes, impacts) and their causal relationships, indicators, and the assumptions or risks that may influence success and failure. It thus facilitates the planning, execution and evaluation of a development intervention.	DAC/OECD (2010)
Materiality	A matter is material if it can substantively affect the organization's ability to create value in the short, medium or long term. -- In the Protocol, an impact or dependency on natural capital is material if consideration of its value, as part of the set of information used for decision-making, has the potential to alter that decision (adapted from OECD 2015 and IIRC 2013). -- In the conceptual framework for IFRS (rev. 2018), Par. 2.11, the definition is as follows: Information is material if omitting, misstating or obscuring it could reasonably be expected to influence decisions that the primary users of general purpose financial reports (see paragraph 1.5) make on the basis of those reports, which provide financial information about a specific reporting entity. In other words, materiality is an entity-specific aspect of relevance based on the nature or magnitude (or both) of the items to which the information relates in the context of an individual entity's financial report. Consequently, the Board cannot specify a uniform quantitative threshold for materiality or predetermine what could be material in a particular situation.	IIRC (2013) ------------------- Natural Capital Coalition (2016a) ------------------- IFRS Foundation (2010)

Measurement	In the Protocol, the process of determining the amounts, extent and condition of natural capital and associated ecosystem and/or abiotic services in physical terms.	Natural Capital Coalition (2016a)
Monetary valuation	Procedure for determining monetary value.	ISO 14008 (2019)
Monetary value	Amount of money representing willingness to pay (WTP) or willingness to accept compensation (WTA).	ISO 14008 (2019)
Natural capital	An economic metaphor for the limited stocks of physical and biological resources found on earth, and for the limited capacity of ecosystems to provide ecosystem services.	TEEB (2010)
Outcome	The likely or achieved short-term and medium-term effects of an intervention's outputs.	DAC/OECD (2010)
Output	The products, capital goods and services that result from a development intervention. May also include changes resulting from the intervention that are relevant for the achievement of outcomes.	DAC/OECD (2010)
Own operation	Gate-to-gate: environmental aspects and potential environmental impacts throughout a product's life cycle starting with production (LCA addresses the environmental aspects and potential environmental impacts throughout a product's life cycle from raw-material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave; e.g. use of resources and environmental consequences of releases)).	ISO 14044 (2006)
Quality-adjusted life years	A health measure that incorporates quality of life and life expectancy based on average samples of health ratings from groups of people and/or professionals. One year in full or perfect health is equal to one QALY. Health-related quality of life (HRQL) is plotted on a scale of 0 (death) to 1 (full health). The QALY was developed in the late 1960s primarily for cost-effective analyses (CEA) to determine the effectiveness of different medical treatments, technologies and interventions.	Gold et al. (2002)
Revealed preference	Monetary value placed by an individual on a market good from which the individual`s valuation of a non-market good is inferred.	ISO 14008 (2019)

Social capital	The networks of relationships among people who live and work in a particular society, enabling that society to effectively function.	Social & Human Capital Coalition (2019)
Social cost of carbon	A measure of the economic harm from emitting one additional ton of carbon into the atmosphere (see also method paper on GHG).	RFF (2019b)
Stated preference	Monetary value expressed by an individual through survey-based responses for a good in a constructed or hypothetical market.	ISO 14008 (2019)
Total economic value (TEV)	A framework for considering various constituents of value, including direct use value, indirect use value, option value, quasi-option value and existence value.	TEEB (2010)
Upstream	Cradle-to-gate: environmental aspects and potential environmental impacts throughout a product's life cycle from raw material acquisition (LCA addresses the environmental aspects and potential environmental impacts (e.g. use of resources and environmental consequences of releases) throughout a product's life cycle from raw-material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave)).	ISO 14044 (2006)
Upstream	GHG emissions associated with processes that occur in the life cycle of a product prior to the processes owned, operated or controlled by the organization implementing this PAS	PAS (2011)
Upstream	GHG emissions or removals associated with processes that occur in the life cycle of a product prior to the processes owned or controlled by the reporting company.	GHG Protocol (2011)
Valuation	The process of estimating a value for a particular good or service in a certain context in monetary terms.	TEEB (2010)
Value of a statistical life	Represents the value a given population places ex ante on avoiding the death of an unidentified individual.	OECD (2012)
Value to business	The costs and benefits to the business, also referred to as internal, private, financial or shareholder value.	Natural Capital Coalition (2016a)
Value to society	The costs and benefits to wider society, also referred to as external, public or stakeholder value (or externalities).	Natural Capital Coalition (2016a)

Willingness to accept compensation	Minimum amount of money an individual is prepared to accept as compensation to forgo an environmental improvement or to tolerate an environmental loss.	ISO 14008 (2019)
Willingness to pay	Maximum amount of money an individual is prepared to pay to secure an environmental improvement or to avoid an environmental loss.	ISO 14008 (2019)
Years of life lost (YLL)	Years of life lost (YLL) take into account the age at which deaths occur by assigning a greater weight to deaths at a younger age and a lower weight to deaths at an older age. The years of life lost (percentage of total) indicator measures the YLL due to a cause as a proportion of the total YLL lost in the population due to premature mortality. YLLs are calculated from the number of deaths multiplied by the standard life expectancy at the age at which death occurs. The standard life expectancy used for YLL at each age is the same for deaths in all regions of the world and is the same as that used for the calculation of disability-adjusted life years (DALY). In addition, 3% time discounting and non-uniform age weights, which give less weight to years lived at young and older ages, are used (as with the DALY). With non-uniform age weights and 3% discounting, a death in infancy corresponds to 33 YLL, while deaths at ages 5 to 20 correspond to around 36 YLL.	WHO (2020)

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3.6. ACKNOWLEDGEMENTS

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