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Valuing corporate environmental impacts

PwC methodology document

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Contents

For more information please contact:

William Evison

Tel: +44 (0)7718 864854
Email: william.j.evison@uk.pwc.com

Quiller Brooke

Tel: +44 (0)7808 035534
Email: quiller.g.brooke@uk.pwc.com

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Valuing corporate environmental impacts: Introductory paper

PwC methodology paper

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Contents

1. Using these papers

This short introductory paper is a preface to six methodology papers which present our latest thinking on the valuation of environmental impacts for Environmental Profit and Loss (E P&L) Accounts. The six papers cover impacts associated with: air pollution, greenhouse gases, land use, solid waste, water consumption, and water pollution. The methodologies were orignally developed for the E P&L, but are flexible to the objectives of the user and have since been applied in many corporate contexts.

All the papers follow a common structure. The first two chapters provide a summary of the impacts covered and a high level overview of the methodology, key variables and assumptions. The third chapter provides a brief overview of the data requirements, both in terms of metric data (emissions or resource use in biophysical units) and contextual data required to estimate the consequences of the emissions or resource use in a given location. The focus of the papers is on the valuation of emissions and resource use, and so they do not go into details of quantification approaches. The following chapters (valuation 'modules') present details of the valuation methodologies with one chapter devoted to each of the specific impact pathways identified in the first chapter. The final chapter explores the sensitivity of the results to the chosen approach. Appendixes provide references and selected supporting information.

Readers interested in an overview should read the first two chapters. Readers looking for more detail will find that material in chapters two and three is repeated and expanded upon in the subsequent methodology chapters and so they may prefer to read the first chapter and then skip to the detailed methodology chapters and appendices.

2. Introduction

Our growing population, decreasing stock of raw materials and increasingly fragile natural environment are changing the world we live in. The business models of today are not equipped to deal with this change. How business operates in the future will need to be transformed. At the same time, what customers, suppliers, employees, governments and society in general expect from business is already changing.

There is an understandable desire for growth – to lift people from poverty, create jobs and improve wellbeing. But, there is also a growing recognition that we need the right kind of growth – good growth that is real, responsible inclusive and lasting.

From a responsible business perspective, this means considering the broader environmental, social, economic and fiscal impacts on stakeholders, beyond just shareholders, and making business decisions which optimise the impacts, while continuing to grow shareholder returns.

Key amongst these are business impacts on the environment (on natural capital) and the consequences of these impacts for human wellbeing, many of which are not currently reflected in market prices. The Environmental Profit & Loss (E P&L) is a tool which businesses can use to value these impacts on current and future populations.

3. What is an E P&L?

Ever since PUMA published 'the world's first Environmental Profit & Loss account'[1] in 2010, E P&L has become a common shorthand for exercises which seek to estimate the value of environmental impacts associated with corporate activities.

PUMA's first E P&L quantified and valued the environmental impacts associated with its operations and entire supply chain. And subsequent product level E P&L's extended the scope to cover the use and disposal or re-use of their products.

The methods can be applied across sectors and to almost any scope – a whole enterprise and its value chain, a tier of the supply chain, a business unit, a product, an initiative or investment, a single production site, even a single material input.

The central purpose of any E P&L analysis is to provide more useful insight into environmental impacts than would otherwise exist. To be useful, this insight needs to be credible and easily understood by decision-makers, it needs to be timely and therefore practical to produce and it needs to be actionable. We, and organisations we have worked with, believe that E P&L results based on our methodologies deliver these attributes for many potential applications.

Given that the use of E P&L as a tool is still evolving, its suitability to inform specific business decisions needs to be evaluated on a case by case basis with particular reference to the quality and resolution of environmental metric data (which are the assumed starting point for these valuation methodologies). Linked to this, E P&L results need to be critically evaluated alongside other sources of decision support information. As suggested by the name, the E P&L only considers environmental impacts; it doesn't evaluate wider economic, fiscal and social impacts, and does not seek to provide a basis for truly holistic corporate decision making.[2]

[1] See PUMA's Environmental Profit and Loss Account for the year ending 31 December 2010: http://about.puma.com/wp-content/themes/aboutPUMA_theme/financial-report/pdf/EPL080212final.pdf

[2] See PwC's Total Impact Measurement and Management (TIMM) for such a holistic framework: http://www.pwc.com/totalimpact

4. What impacts does the E P&L value?

The E P&L seeks to value the impacts on people resulting from changes in the environment associated with corporate value chains. These impacts can be positive (profits) or negative (losses). The values generated by an E P&L, therefore, represent an estimate of the change in wellbeing (or in economic terms 'welfare') experienced by people as a result of corporate environmental impacts.[3]

We categorise impacts into six areas, a brief summary of each impact area is presented here:

- **Air pollution:** Release of pollutants such as particulate matter, sulphur dioxide and nitric oxides reduce the quality of air, with negative consequences on people's health, as well as on the natural and built environment.
- **Greenhouse gases (GHGs):** The causal link between anthropogenic emissions of GHGs and changes in global climate is now well established. The impacts are expected to be far-reaching and will affect our health, economy and the natural environment.
- **Land use and biodiversity:** Natural land areas provide essential services to society which regulate our environment, provide goods and services that support livelihoods, offer opportunities for recreation and provide cultural and spiritual enrichment. The conversion and degradation of natural areas results in a reduction of these services.
- **Waste:** The disposal of waste can drive a number of impacts including the release of GHGs and other air pollutants, leachate of pollution into water bodies and soils, and disamenity around disposal sites.
- **Water consumption:** Corporate water consumption can, in some circumstances, reduce the availability of clean water for local communities, resulting in increased consumption of dirty water, with associated impacts on people's health. Increasing water scarcity can also impact on agricultural productivity, and the quality of the natural environment, with associated reductions in ecosystem services.
- **Water pollution:** The release of toxins to waterways can lead to impacts on people's health if the pollutants are ingested via drinking water or through bioaccumulation in food. Excess nutrient pollution leads to eutrophication which reduces environmental quality and can adversely affect fisheries productivity and recreation opportunities.

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[3] It is worth noting that this is unlikely to be the same as the cost of reducing those impacts (the marginal abatement cost) which will depend on specific company circumstances and available technologies. It is also not intended to be an estimate of the amount that would be payable if the impact were regulated – for example through a cap and trade scheme or environmental tax – which would be determined by the nature and objectives of the regulatory instrument.

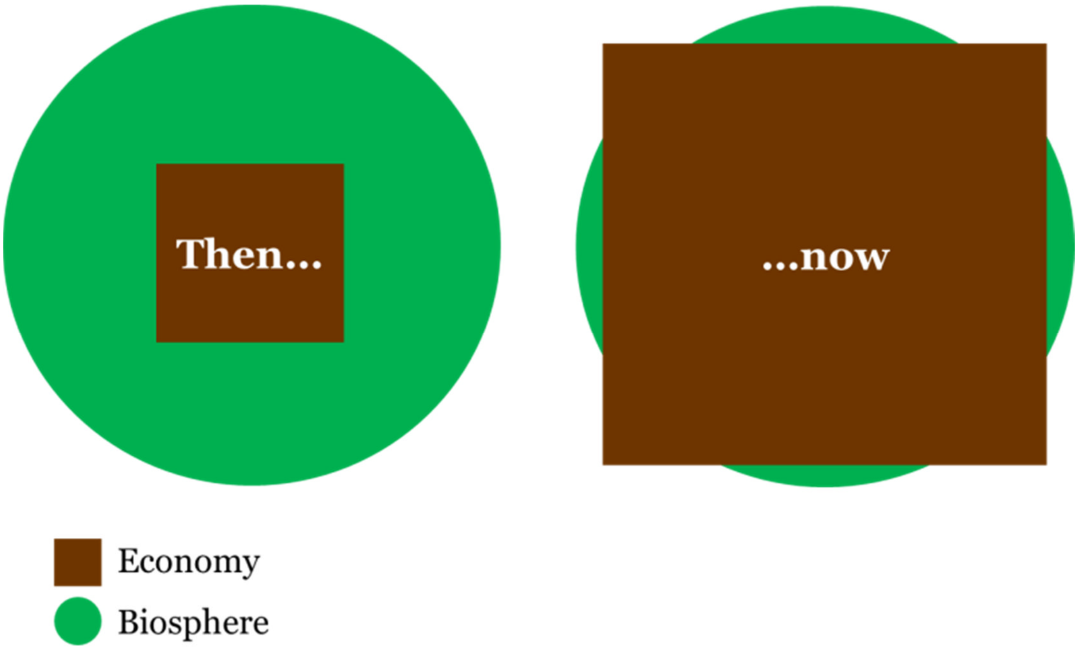
5. Why do we need to value corporate environmental impacts?

5.1. For the good of the planet

When the economic rules that govern private companies were first defined, economies were insignificant in relation to the seemingly limitless biosphere. Today that situation has changed; economies have grown and now demand vast quantities of resources (Figure 1). According to several scientific analyses we have already exceeded key ecological limits[4] and are operating beyond the carrying capacity of our planet[5].

In today's economic system, some environmental impacts are 'externalised' by companies – meaning that they affect society at large rather than those directly involved in the company's value chain. Putting a monetary value on these environmental impacts allows companies to take them into account in their decision-making and thus enables them to deliver better outcomes for the environment and society.

Figure 1: The changing relationship between the economy and the biosphere[6]



[4] See for example "Planetary Boundaries: Exploring the Safe Operating Space for Humanity" http://www.ecologyandsociety.org/vol14/iss2/art32/

[5] See for example "One Planet, how many people? A review of Earth's carrying capacity", UNEP (2012) http://na.unep.net/geas/archive/pdfs/GEAS_Jun_12_Carrying_Capacity.pdf

[6] Adapted from Global Footprint Network, Annual Report 2011: http://www.footprintnetwork.org/images/article_uploads/2011_Annual_Report_RF.pdf

5.2. *For the good of business*

Some business models already deliver environmental benefits hand-in-hand with shareholder returns. By valuing these positive impacts the E P&L provides a means to recognise and reward them, and an incentive for more businesses to follow suit.

On balance however, the environmental impacts associated with corporate value chains tend to be negative. At present, government policies do not always oblige companies to 'internalise externalities', but a range of factors are creating more pressure for them to do so.

In most developed economies, clean air and water laws mean that companies (and ultimately consumers) already pay for some of the costs of pollution; but increasing focus on enforcement coupled with new legislation in emerging economies and growing employee awareness are adding to these costs. Consumer pressure in relation to environmentally harmful products and production methods continues to drive changes in manufacturing and sourcing strategies. Local communities have successfully sued major corporations for unlawful dumping of waste, and shareholder attention has been raised by high profile environmental incidents and the associated compensation costs and punitive damages. Increasing incidence of droughts, floods, soil erosion and pests have caused disruption to operations and price volatility in agricultural commodities – imposing some of the costs of environmental decline onto company balance sheets and income statements.

These drivers are becoming more acute over time. So, although few of the costs estimated in an EP&L will currently hit the company's bottom line, they are strong indicators of future risks.

Monetary valuation of impacts provides a range of additional benefits to businesses, enabling them to:

- Simplify many complex environmental metrics into a single unit allowing for comparability, prioritisation and target setting;
- Improve cut-through and understanding with senior decision makers and provide a stronger basis for dialogue with other stakeholders; and,
- Identify material opportunities to reduce impacts or develop new environmentally positive products and services.

Companies we have worked with have found that the process of developing an E P&L is typically a valuable exercise in its own right, helping to:

- Connect different teams and data owners within the business and get new functions and decision makers to engage with environmental information;
- Broaden and deepen understanding of environmental impacts along the whole value chain; and,
- Establish or enhance comprehensive environmental datasets across a wide range of impact areas.

6. How do we value corporate environmental impacts?

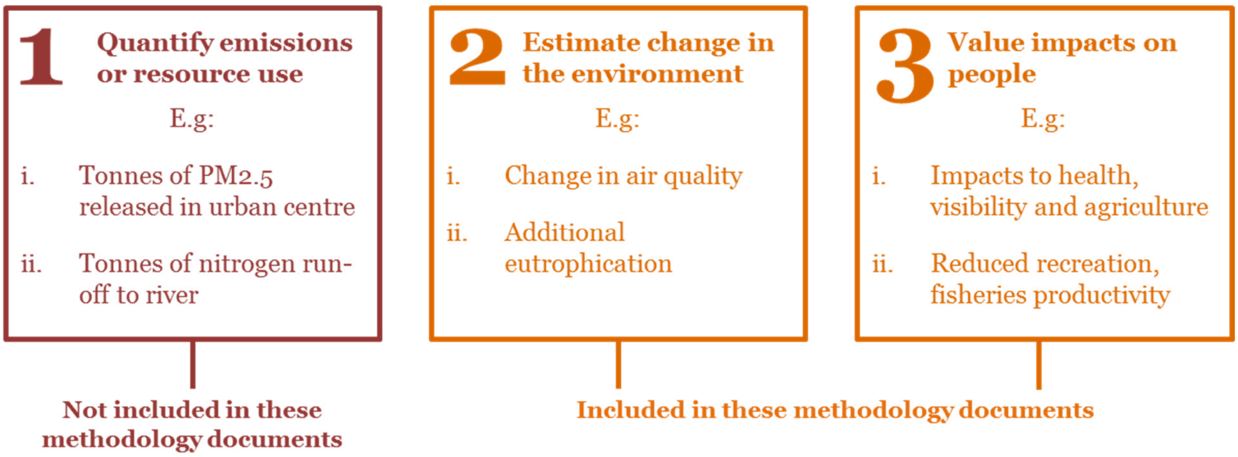
There are three steps to estimating the scale of corporate environmental impacts (Figure 2).

1. The first step is to quantify environmental emissions or resource-use in biophysical units (kilograms, litres, hectares etc.).
2. The second step is to understand how the corporate emissions or resource-use cause changes in the natural environment.
3. The final step is to value the impacts on people associated with these changes in the environment.

Traditional environmental reporting typically stops at the first step, providing an understanding of the magnitude of emissions and resource use; the E P&L goes further to also consider the consequences of these emissions and resource use for the environment and people.

The E P&L valuation methodologies address the second two steps. A wide range of methods exist to measure or estimate biophysical quantities of emissions or resource use and are the subject of separate documentation.

Figure 2: Three steps to estimating and valuing impacts



Each methodology paper explains steps two and three in some detail for the relevant impact area.

As we developed the methodologies we used a set of basic methodological principles (Box 1) to guide our decision-making.

Box 1: Methodological principles

Completeness: Each methodology should aim to cover more than 90% of the value of impacts, as identified in our impact pathways.
Consistency: Apply a consistent conceptual framework based on the theory of environmental and welfare economics; follow a consistent impact pathway approach to understand causality; apply common assumptions and datasets across different methodological areas.
Practical and 'fit for purpose': Directly linked to environmental metrics which corporates can feasibly measure; able to produce approximate results based on limited data; sophisticated enough to

produce more accurate results in more data rich situations.
***Location specific*: ** Taking into account the huge spatial variation in the value of ecosystem services and environmental impacts, the approaches are designed to be applicable at specific location, region or country level, dependant on company and contextual data. To deliver consistency in multi-scale assessments the approaches are designed to be 'nested' such that results for a specific location are compatible with results produced at a broader scale.
***Best available approaches*:** Assimilate and build on existing (peer-reviewed) methods wherever these exist and can reasonably be adapted for application to corporate impacts.
***Transparency*:** Provide clarity on sources and methods; highlight limitations and areas for further development.

Drafts of some of the methodologies underwent an academic review in 2011[7], and we have continued to draw on input from the academic and expert practitioner communities as we have refined them significantly over the last few years.

[7] See "An Expert Review of the Environmental Profit & Loss Account", PPR (2011).
http://www.kering.com/sites/default/files/e-pl-review_final-for_publicationwebsitefinal_final_1.pdf

7. Areas for further development

None of the E P&L methodologies presented here is perfect and all would benefit from further research and refinement. Specific limitations are identified in each of the papers and below we have summarised some more general areas for development. We welcome feedback as we continue to refine the methodologies.

7.1. Quality of underlying research

All of the methodologies assimilate and build on existing research. In some cases this research has limitations in either the scientific understanding of the impact pathway, or the economic valuation of the impacts on people, or both.

Table 1 below summarises our assessment of the overall 'robustness' of the methods employed in each impact area considering:

1. The extent and quality of the academic literature which underpins it;
2. The degree of consensus in this underlying literature; and,
3. The applicability of the underlying literature to the measurement or valuation of corporate environmental impacts.

We have split this assessment into two parts: The 'science' (step 2 in Figure 2) – understanding how emissions or resource use change the environment, including how these changes affect people. And the 'economics' (step 3 in Figure 2) – valuing the consequences of environmental changes for people. Brief notes on the rationale behind these ratings follow.

Table 1: Summary assessment of 'robustness' (considering extent of literature, degree of consensus and applicability)

Impact area	Science	Economics (Step 3, Figure 2)
Air pollution	5	4
Greenhouse gases	5	4
Waste	4	4
Land use	3	3
Water consumption	3	2
Water pollution	2	3

Legend
6 – most robust
5
4
3
2
1 – least robust

7.2. Notes on our robustness ratings

7.2.1. Air emissions

- Highly advanced scientific literature with clearly defined causal pathways from emission, through dispersion, to dose-response and specific health endpoints.
- Advanced economic literature on the valuation of health impacts, although variation in estimates produced. More limited research on non-health impacts.

- International institutions (like the OECD and World Health Organisation) have published guidance on quantification and valuation of impacts, and many governments use estimates in policy making (e.g. see EU ExternE study, UK Defra damage costs, US EPA BenMAP model)[8]

7.2.2. Greenhouse gases

- Highly advanced literature on the science of climate change led by the International Panel on Climate Change (IPCC).
- Owing to the level of international policy attention the future costs of climate change have also been extensively studied. While there is significant variation in estimates, much of this variation revolves around points of theory and ethics (e.g. discounting) rather than the nominal values themselves.
- The use of a social cost of carbon (SCC) is now common place in policy analysis and many governments and some businesses now have an approved SCC for use in decision-making.

7.2.3. Wa*ste*

- The principal impacts of waste are associated with GHGs, and in some instances disamenity, leachate, and air pollution. GHGs and air pollution both have advanced scientific and economic literature.
- Disamenity and leachate are both relatively well studied in developed economies but they are highly context dependant – generalised models require significant simplifications.

7.2.4. Land use

- Advanced ecological literature on the impacts of land conversion and on-going use on the provision of ecosystem services. But more limited understanding of role of biodiversity in the delivery of ecosystem services.
- Valuation of ecosystem services is a rapidly developing field in academia. Consistent globally applicable assessments are hindered by the relatively limited body of peer-reviewed literature. There are also significant challenges in aggregation and generalisation given the degree of spatial variation in value estimates.
- The valuation of ecosystem services is increasingly being integrated into policy making. For example, the UK National Ecosystems Assessment9 considers the value of ecosystem services under different land use planning scenarios.

7.2.5. Water consumption

- The science is well understood and trends are observable in well-maintained global databases on water use and water-borne disease (e.g. UN Water and the FAO's AquaStat).
- The valuation of impacts draws largely on the valuation of health and life, which has an advanced literature underlying it and is used by policy makers and international institutions for decision-making.
- However, demonstrating causality between corporate water use and additional human impacts has not been the focus of work in this area to date, and is difficult given the number of context-specific variables influencing impacts. There are some useful studies in the Life Cycle Analysis literature, but even the most sophisticated benefit transfer is unlikely to be a good substitute for detailed primary research where site specific detail is important.

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[8] ExternE, (2005). "Externalities of Energy: Methodology 2005 Update"; Defra (2011). "Air Quality Appraisal – Damage Cost Methodology". http://archive.defra.gov.uk/environment/quality/air/airquality/panels/igcb/documents/damage-cost-methodology-110211.pdf ; "BenMAP Manual", US EPA (2012), http://www.epa.gov/airquality/benmap/docs.html

[9] "The UK National Ecosystem Assessment Technical Report". UNEP-WCMC, (2011), Cambridge.

7.2.6. Water pollution

- Epidemiological research into human toxicity impacts in controlled experimental conditions is good. However, outside controlled conditions impacts are highly uncertain due to difficulties in estimating emission-receptor-impact pathways.
- The valuation of impacts draws largely on the valuation of health and life, which has an advanced literature underlying it and is used by policy makers and international institutions for decision-making.
- The valuation of non-health impacts associated with eutrophication is mostly studied using willingness to pay analysis for improved water quality in developed countries. There has been limited work considering these impacts in developing countries.

7.2.7. Data availability

In the interests of making the approaches practical to apply and delivering the desired consistency across diverse locations we have favoured methods that can be applied with readily available data. As a result it has sometimes been necessary to compromise on points of theoretical purity or the granularity of analysis. For example, in some areas we have found it necessary to employ cost-based value estimates as proxies (in lieu of welfare derived alternatives), and some input datasets are only available in a consistent form at a state or country level making more granular analyses difficult.

7.2.8. Non marginal impacts

Many of the E P&L values reflect an implicit assumption that the environmental changes caused by any individual business are marginal relative to the current state of the environment. In reality, non-linearities and threshold effects mean that this condition may not hold and identifying ways to take this into account would be an important, if challenging, area for further work.

7.2.9. Known omissions

As noted in the first chapter of each paper, specific impact pathways are sometimes excluded. Generally this is done on the basis that they are expected (or known) to be immaterial for most applications.

In addition, some classes of environmental impact are not covered by these methodologies – for example, noise and light pollution, radiation, littering (of land, water and oceans) and indoor environmental impacts. Depending on the specific application it may be helpful to include some of these where they are likely to be material.

8. Conclusion

All of the E P&L valuation methodologies build on the existing body of peer reviewed literature and none of them are perfect. Given that the use of E P&L as a tool continues to evolve, its suitability to inform specific business decisions still needs to be evaluated on a case by case basis.

Nonetheless, the central purpose of any E P&L is to provide more useful insight into environmental impacts than would otherwise exist – and for most possible applications, this is what it does.

Organisations we have worked with identify a range of benefits; from the obvious, that it enables comparison and prioritisation between diverse impact areas and can be used to communicate the true environmental costs and benefits of business activities; to the less obvious – that it can improve environmental understanding across a business, and put environmental information on the boardroom agenda. In short, it helps to answer the essential question: Which of my environmental impacts matter most, and where?

We welcome feedback as we continue to develop the methodologies.

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Valuing corporate environmental impacts: Air pollution

PwC methodology paper

Version 1.5

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Contents

Table of Tables

Table of Figures

Table of Equations

Abbreviations and acronyms

Abbreviation	Full name
APEEP	Air Pollution Emissions Experiment and Policy Model
CH_4	Methane
CO	Carbon monoxide
CO_2	Carbon dioxide
CO_{2e}	Carbon dioxide equivalent
DOC	Degradable organic carbon
E P&L	Environmental Profit and Loss
EEIO	Environmentally Extended input-output modelling
ERQ	Environmental Regulatory Quality
EU	European Union
FAO	Food and Agriculture Organisation
GDP	Gross Domestic Product
GHG	Greenhouse gas
GNI	Gross national income
HARAS	Hazard Rating System model
HM Treasury	Her Majesty's Treasury (United Kingdom)
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
IQ	Intelligence quotient
kWh	Kilowatt hour
LCA	Life cycle assessment
LFGTE	Landfill-gas-to-energy
MCF	Methane correction factor
NH_3	Ammonia
NH_4	Ammonium
NO_x	Mono-nitrogen oxides: NO and NO_2
O_3	Ozone
OECD	Organisation for Economic Co-operation and Development
PM_{10}	Coarse particulate matter (diameter under 10µm)
$PM_{2.5}$	Fine particulate matter (diameter under 2.5µm)

PPP	Purchasing power parity
SCC	Societal cost of carbon
SO_x	Sulphurous oxides
SRTM	Source Receptor Transfer Matrix
US EPA	United States Environmental Protection Agency
VOC	Volatile Organic Compound
VSL	Value of Statistical Life
WTP	Willingness To Pay

1. The environmental impacts of air pollution

1.1. Introduction

Economic activity in all sectors results in some level of emissions of waste gases and suspended solids into the air (whether directly as a result of industrial processes, or indirectly – for example, as a result of energy or resource consumption). Changes in concentrations of these may have negative impacts on health, as well as on the natural and built environment. Emission of these pollutants, therefore, carries a societal cost. In this paper, we set out a methodology for identifying and valuing these costs in monetary terms.

The majority of air pollutants are covered in this paper, but several classes of pollutants are addressed in other papers. Greenhouse gas (GHG) emissions are considered separately in the PwC methodology paper: *Valuing corporate environmental impacts: Greenhouse gases*. Air pollution also results from waste incineration, and several key incineration pollutants are addressed in the PwC methodology paper: *Valuing corporate environmental impacts: Solid waste*. These overlaps are discussed further in the following chapters.

1.2. Overview of impact area

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

Air pollution can be subdivided into 'primary pollutants', which directly cause negative impacts on the environment and people, and 'secondary pollutants', which result from reactions between primary pollutants and other gases under certain conditions, and which subsequently also have negative impacts on the environment and people.

The most significant primary and secondary pollutants (in societal cost terms) are listed below (in no particular order).

1.2.1. 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 from 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); $PM_{2.5}$ 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 which 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 such as the use of solvents in industrial processes, as well as from some natural processes.
- **Mono-nitrogen oxides (NO and NO_2, commonly referred to as NO_x):** These are naturally present in the atmosphere but are also released in large quantities through the combustion of fossil fuels and particularly transport fuels.
- **Sulphur dioxide (SO_2):** SO_2 is released through the processing of sulphurous mineral ores and from many industrial processes which involve 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 and is also a by-product of numerous industrial and agricultural processes.

1.2.2. Secondary air pollutants

Major secondary pollutants include:

- **Sulphates (SO_4^-) and nitrates (NO_3^-):** These are formed from SO_2 and NO_x respectively and are both types of $PM_{2.5}$.
- **Ammonium (NH_4^+):** Ammonia production is mainly a result of agriculture, particularly from the waste of cattle and other livestock. Some nitrogen-based fertilisers can also result in NH_3 emissions to air. 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 $PM_{2.5}$.
- **Ozone (O_3):** Ozone is formed via a non-linear reaction between VOCs and NO_x in the presence of sunlight.

1.2.3. Environmental and societal outcomes

Emission of air pollutants increases their concentration in the atmosphere. This reduces ambient air quality directly and causes secondary phenomena such as smog and acid rain. These outcomes can adversely affect people in various ways:

- **Human health:** Respiratory diseases lead to large societal costs from air pollution. These damages include increased incidents of chronic diseases such as asthma and bronchitis and, in some cases, premature mortality from cardiovascular diseases, pulmonary diseases and lung cancer.
- **Visibility:** Air emissions, particularly PM and O_3 precursors, contribute to reduced visibility through the formation of smog. Reduced visibility affects various forms of navigation and also reduces people's enjoyment of recreational sites and the neighbourhoods where they live (i.e. disamenity).
- **Agriculture:** Changes in the atmospheric concentration of certain gases can negatively impact the growth of crops leading to reduced yields. Acid rain can damage crops directly and can also acidify soils with impacts on future growth.
- **Forests and timber:** Changes in the atmospheric concentration of air pollutants can cause visible physical changes in tree growth and also affect metabolism at the cellular level. Prolonged impacts can severely impact on 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 in 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 to forests and bodies of water can lead to reduced recreational enjoyment of the natural environment.

1.2.4. Factors affecting the impact on people of air pollution

Factors beyond the total mass of pollutants emitted materially affect the societal impacts from these emissions. For example, strong winds may disperse pollutants away from heavily populated areas or heavy rainfall may cause particulate matter to be rapidly deposited, giving only a small dose to each person. Table 1 presents the key variables known to influence the different potential environmental outcomes resulting from air pollution emissions.

Table 1: Key variables known to influence environmental outcomes from air pollution

Outcome	Variable
All outcomes	Meteorological conditions that influence dispersion (wind speed and direction, mixing height)
	Ambient concentrations of pollutants
Health	Population density of the region (i.e., number of people in contact with pollutant)
Agriculture, forestry and timber	Yields and types of crops and forests in the region
Visibility	Quality of vista, local preferences
Built environment	Extent and nature (e.g. building materials, cultural heritage, financial value) of buildings

1.3. Impact pathways

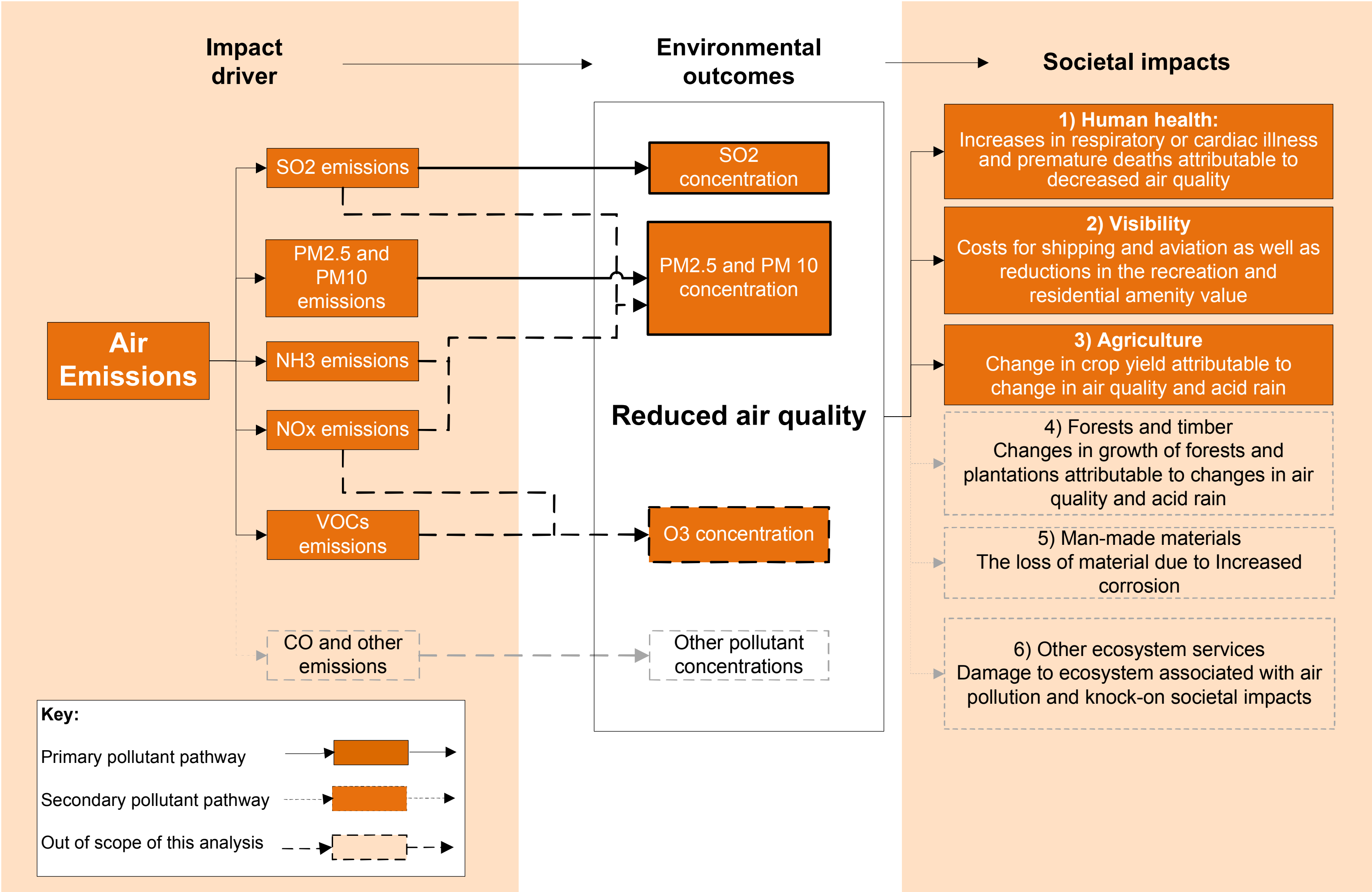
In order to value corporate environmental impacts, we need to understand how corporate emissions into the atmosphere affect humans. Therefore, we define impact pathways that describe the links between corporate activities, the environmental outcomes from those activities, and the resultant societal impacts. Our impact pathway framework consists of three elements:

- **Impact driver:**
  - *Definition:* These drivers are expressed in units which can be measured at the corporate level, representing either an emission to air, land, or water; or the use of land or water resources[1].
  - *For air pollution:* The type and quantity of air emissions resulting from different business activities.
- **Environmental outcomes:**
  - *Definition:* These describe actual changes in the environment which result from the impact driver (emission or resource use).
  - *For air pollution:* Businesses directly affect air quality through emissions of pollutants. These primary pollutants react with other elements in the air to produce secondary pollutants (see Figure 1). Both primary and secondary pollutants can lead to specific environmental outcomes such as smog and acid rain.
- **Societal impacts:**
  - *Definition:* These are the actual impacts on people as a result of changes in the environment (environmental outcomes).
  - *For air pollution:* The impacts are principally related to health but also include impacts via agriculture and visibility.

The three stages of the impact pathway are shown in Figure 1 overleaf. Air pollution exhibits a complex pathway, with multiple pollutants each playing a role in multiple environmental and societal outcomes. The reasons for any limitations of scope are explained at the end of this chapter.

[1] A note on language: In this report, the measurement unit for any 'impact driver' is an 'environmental metric.' Therefore, air pollution is the impact driver, and tonnes of $PM_{2.5}$, PM_{10}, SO_2, NO_x, etc. are the environmental metrics.

Figure 1: Impact pathways for air pollution



1.4. Prioritising which impacts to quantify and value

This section outlines the environmental outcomes and societal impacts from air pollution that we quantify and value in this methodology. The section also identifies those impacts which are beyond the scope covered by this paper.

We base our materiality assessment on previous large scale assessments in the EU and US. For example, the UK Government (Defra, 2011a) focuses its analysis only on health impacts from PM (from different industries), NOx, SOx and NH3.[2] ExternE, an EU wide study of externalities from industry, considered the same pollutants as well as VOCs. It included impacts on agriculture and buildings, as well as health in its valuation, although it notes health impacts are "by far the largest part of the total" (ExternE 2005).

One of the most detailed studies of the societal impacts of air pollution to date was conducted in the US by Muller and Mendelsohn (2007). A summary of their findings is presented in Table 2. The findings are consistent with those of other EU, US and OECD studies (including Pope et al. 1995, ExternE 2005, OECD 2009, Defra 2011a); mortality and morbidity effects dominate the societal costs (together representing 94.5% of the total). We also see these values transferred to other countries as proxies, including from developed to developing countries (Sengupta and Mandal, 2013; Pervin et al., 2008; World Bank, 2007).

Table 2: Illustrative study of estimated societal costs of air pollutants in the US (USD billion per year)

	Health - Mortality	Health - Morbidity	Agriculture	Forestry and timber	Visibility	Built environ-ment	Recreation	Total	Share of societal costs
$PM_{2.5}$	14.4	2.6			0.4			**17.4**	**24%**
PM_{10}		7.8			1.3			**9.1**	**12%**
NO_x	4.4	0.8	0.7	0.05	0.2		0.03	**6.2**	**8%**
NH_3	8.3	1.5			0.2			**10.0**	**14%**
SO_2	16.1	2.9			0.4	0.1		**19.5**	**26%**
VOC	9.6	1.8	0.5	0.03	0.2			**12.1**	**16%**
Total	**52.8**	**17.4**	**1.2**	**0.08**	**2.7**	**0.1**	**0.03**	**74.3**	
Share of societal costs	**71.1%**	**23.4%**	**1.6%**	**0.1%**	**3.6%**	**0.1%**	**0.0%**		

Source: Muller and Mendelsohn (2007)

On the basis of these studies we therefore focus our detailed assessment primarily on health impacts, and also include a more basic assessment of impacts on agriculture and visibility – discussed in more detail below.

The relative costs of each impact area and each pollutant may vary across locations, or for specialist processes (such as incineration which releases dioxins – see PwC methodology paper: *Valuing corporate environmental impacts: Solid waste.*). If, in a specific context, other impacts or pollutants are likely to be important, a methodology will be required to estimate and value these.

1.4.1. Impacts covered by this methodology paper

1.4.1.1. Health impacts

Our priority focus in this analysis is the impact of air pollution on health. The literature is consistent that health impacts are by far the most important (Muller and Mendelsohn, 2007; Pope et al, 1995; ExternE, 2005). However, the type and extent of damages caused by air pollution are regionally specific. Of the main pollutants,

[2] Defra Damage Costs are intended for use in cost-benefit analysis as per the HM Treasury's Green Book guidance, see: https://www.gov.uk/air-quality-economic-analysis

PM (both direct emissions and secondary PM from NO_x, SO_2 and NH_3) has the most significant impact on health (Pope et al., 1995). Direct PM emissions and the contribution of other gases to PM (SO_2, NO_x, NH_3) as secondary pollutants are therefore addressed in detail using an air dispersion model as the main focus of this methodology. Direct health impacts of SO_2 are also considered in the dispersion model.

Health impacts of low level O_3, formed from VOCs and NOx, are also important.[3] The chemical relationship between VOCs, NOx and the formation of O_3 is non-linear and considered too complex to model without detailed location-specific information (Ostro, 1994). In the absence of available datasets, we address it here using a multivariate transfer function derived from Muller and Mendelsohn's (2007) results from the US.

1.4.1.2. Visibility and agriculture

The societal costs of reduced visibility and agricultural losses are significantly smaller in magnitude than those of human health due to air pollution. They contribute just 3.6% and 1.6% of the societal cost of air pollution respectively in Muller and Mendelsohn's (2007) study of the U.S. However, in rural contexts and in more agriculturally dominated economies, they may have greater relative significance, hence their inclusion here.

Without adequate inventories to characterize exposure to pollutants and local preferences it is difficult to estimate the societal cost of impacts on agriculture and impaired visibility. However, benefit transfer of values between locations gives an indication of potential impacts. Recognising that this is a more uncertain approach, but acceptable given the low materiality, we calculate the country-specific impacts of reduced visibility using a multivariate transfer function and calculate the impacts on agriculture using an adjusted value transfer approach.

1.4.2. Impacts covered by other PwC methodology papers

1.4.2.1. Dioxins and heavy metals

Airborne emissions of dioxins, arsenic, chromium, cadmium and nickel can cause cancer, while lead and mercury can have neurotoxic effects. These emissions are primarily associated with waste incineration; therefore, our approach to quantifying and valuing them is covered in the PwC methodology paper *Valuing corporate environmental impacts: Solid waste.*

1.4.2.2. Greenhouse gas emissions

Greenhouse gas emissions are considered separately in the PwC methodology paper *Valuing corporate environmental impacts: Greenhouse gases.*

1.4.3. Limitations of scope

1.4.3.1. Forests and timber, built environment, and recreation

These impacts are considered immaterial relative to the impacts described above. Together, they represent less than 0.5% of the total societal cost in Muller and Mendelsohn's (2007) analysis[4]. They are, therefore, omitted from this methodology. If they are thought to be material in a particular context (for example due to local stakeholder pressure) the impacts should be estimated and valued on a case by case basis.

1.4.3.2. Chemical deposition in soil and water

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 small compared to impacts on health. Given the low materiality, this secondary deposition in soil and water is omitted from this methodology. However, impacts associated with NH_3 in wastewater are considered separately in the PwC paper *Valuing corporate environmental impacts: Water pollution.*

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3 This methodology does not attempt to quantify any potential indoor health impacts of VOCs. These are considered to be an aspect of employee working conditions and would therefore be addressed when considering the social impacts of a business.

4 Our category 'built environment' is equivalent to Muller and Mendelsohn's (2007) category 'man-made materials'.

1.4.3.3. Carbon monoxide

CO is a toxic gas which, if inhaled in sufficient quantities, can be fatal. It can have societal impacts via inhalation indoors and outdoors and through its contribution to O_3 formation. However, we exclude it from our methodology on three counts:

- CO is particularly dangerous in indoor environments, which are outside the scope of this methodology. Indoor air quality would be considered as part of employee working conditions when evaluating the social impacts of a business.
- Regulations requiring vehicles to be fitted with catalytic converters have significantly reduced the dangers from carbon monoxide in the urban environment in many countries, such that emissions are now quite low.
- The close relationships between CO, NO_x, and VOC pathways to O_3 formation make it difficult to avoid double counting of secondary impacts. Reflecting this, CO is excluded from Muller and Mendelsohn's (2007) analysis, Defra's (2011a) air emissions damage cost methodology, and ExternE (2005) analyses.

2. Summary of methodology

2.1. Introduction

The impact pathway presented in Chapter 1 identifies how emissions can lead to different types of impacts. Our valuation framework is structured to follow this pathway, at each stage demonstrating the causal links between corporate activities (which result in air pollution) and societal costs.

To understand the value of environmental impacts associated with corporate activities, it is necessary to:

1. Obtain environmental metric data: The starting point for each of our methodologies is data on emissions or resource use. These metric data are based on an understanding of the corporate activities which they result from. Data can come from a variety of sources, some of which (e.g., life cycle inventories (LCI) or environmentally extended input-output modelling (EEIO)) are subject to their own distinct methodologies[5]. The assumed starting points for this methodology are metric data in the form specified in Table 3 below.

Table 3: Metric data for air pollution

Impact driver (emission or resource use)	Environmental metric data
Six main pollutants (PM_{10}, $PM_{2.5}$, NO_x, NH_3, SO_2, VOC)	Mass of emissions from corporate activities (tonnes)

2. Quantify environmental outcomes: We quantify biophysical changes in the environment resulting from corporate emissions or resource use (as measured by the metric data). This is discussed further in Table 4, left-hand column.

3. Estimate societal impacts: We estimate the societal cost (impact on people) resulting from environmental changes which in turn are the result of corporate activities. This is discussed further in Table 4, right-hand column.

It is not always necessary or appropriate for economic valuation of the environment to go through each of these steps explicitly. A single methodological step may cover some or all steps at once. However, developing each E P&L valuation methodology by following a clearly defined impact pathway helps to retain a causal link and ensure consistency.

2.2. Summary of methodology

Environmental metric data on air pollution are the starting point for this methodology and, hence, the methods for collecting or estimating these data are not exhaustively covered. However, for the purposes of valuation it is important to understand how other factors - such as meteorological and demographic variation - influence the consequences of emissions.

How these factors are taken into account in the valuation methodology is summarised in Table 4 which describes the methodologies for each of the major valuation modules in turn:

1. Primary pollutant health impacts (PM, NO_x, SO_x, NH_3)
2. Secondary O_3 health impacts

[5] Information on the likely sources of metric data is provided in Chapter 3.

3. Visibility impacts
4. Agricultural productivity impacts

Table 4 summarises our methodology for each of these valuation modules, showing:

- The key methods and steps;
- The key variables for which data must be collected at each step;
- The key assumptions and justifications underlying each methodological choice.

Table 4: Summary of air pollution societal impacts calculation methodology, key variables and assumptions

	Quantify environmental outcomes	Estimate societal impacts
	Health valuation module from SO_2 and primary and secondary PM (primary pollutants: PM, NO_x, SO_2, and NH_3)	
Methods	• Lagrangian puff air dispersion model determines change in primary and secondary pollutant concentrations over a specified area. • Dispersion model considers local meteorological conditions, as well as the persistence in air of pollutants in estimating the dosing. • An estimate of the number of people affected is produced by overlaying a population grid describing the demographics in the location of interest.	• Dose-response functions estimate health outcomes for populations exposed to pollutants. • To value specific morbidity health outcomes Willingness to Pay (WTP) estimates from peer reviewed literature are used. For mortality, the OECD estimate of the value of a statistical life (VSL) is used.
Key variables	• Meteorological conditions: wind speed, precipitation, mixing height. • Demographics: population density and distribution.	• Population density and baseline mortality rate. • Value of Statistical Life (VSL).
Assumptions and justification	• Air dispersion is modelled using Sim-Air ATMOS 4.0 which can account for local meteorological and demographic conditions in its modelling. Sim-Air ATMOS 4.0 is a simplified version of a US National Oceanic and Atmospheric Administration model, adapted for relatively rapid assessment. It has been widely used in Asia and is applicable globally. • If the emission source location 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 manufacture) the same approach can be individually applied to major locations of the polluting activity within a country. Averages weighted by the proportion of industrial production in each location can then be produced. • As a minimum we develop average coefficients for application to three types of stationary source: inner city, urban industrial and rural; and two types of mobile source: urban transport and rural transport. • We assume pollutant concentration changes can be described as a linear function of emissions. This linear 'source-receptor' modelling technique is well established in the literature.	• We model health impacts using linear dose response functions for pollutant exposure. • A linear function assumes that emission concentrations are already above any damage threshold, such that any addition of pollution in the environment causes an impact. Linear functions are widely applied (ExternE, 2005, World Bank, 2008) and are the most appropriate for globally applicable approaches because determining whether pollutants are below any damage threshold requires data on ambient concentration and biogenic emissions which are not widely available.

Health valuation module from secondary O3 formation (primary pollutants: NO_x and VOCs)	
Methods	• Environmental outcomes and societal impact are evaluated in one step using a multivariate transfer function, which extends Muller and Mendelsohn's (2007) societal cost estimates beyond the U.S. to give global coverage, subject to the availability of local contextual data. • The transfer function estimates the societal cost of air pollution as a function of ambient O3 concentration, local income, and local population density.
Key variables	• Environmental data: ambient O3 concentration. • Demographic data: local or country level income and population density.
Assumptions and justification	• Secondary pollutant formation is too complex to model directly, and therefore expanding on existing damage cost estimates is more accurate and practical. • A transfer function based on one of the most comprehensive assessments of air pollution societal costs to date is used as a substitute for a model of atmospheric chemistry. • Extrapolating a transfer function from U.S. based societal costs assumes: – The physiological impacts derived from US data are applicable to the rest of the world. This is reasonable, 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 O3 levels and income levels. This is reasonable, because both of these variables show significant variation in the US sample, providing a credible basis for estimation of societal costs elsewhere.
Visibility valuation module (primary pollutants: PM, NO_x, NH_3, SO_2, VOCs)	
Methods	• Environmental outcomes and societal impacts associated with WTP to reduce visibility impairment from air pollution are evaluated in one step using a multivariate transfer function, which extends Muller and Mendelsohn's (2007) US societal values to give global coverage subject to the availability of local contextual data. • The transfer function provides an estimate of the societal cost of reduced visibility as a function of ambient O_3 concentration, local income, local population density, temperature, and rainfall.
Key variables	• Environmental data: ambient O_3 concentration, temperature, rainfall. • Demographic data: local/country income and population density.

Quantify environmental outcomes	Estimate societal impacts
Assumptions and justification	• Visibility-impairing pollutant formation is complex to model directly globally, and therefore drawing on existing damage cost estimates is preferred to give an approximate indication of impacts. • Extrapolating using the transfer function from U.S. based societal costs assumes: – 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 which form smog and reduce visibility will be consistent around the world. – The social cost of visibility harms 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, providing a basis for estimating a function to describe how WTP changes based on these variables that can be applied elsewhere.
Agricultural productivity valuation module (primary pollutants: NO_x and VOCs)	
Methods	• Environmental outcomes and the impacts on reduced agricultural productivity are evaluated in one step using value transfer. • We take the average of marginal damage costs from Muller and Mendelsohn's (2007) US dataset and adjust this for purchasing power differences between countries.
Key variables	• The only variable used in our approach is country PPP data relative to the US. Key variables in the underlying analysis of agricultural damage costs are: crop type, health, prices and productivity, as well as ambient pollutant concentrations.
Assumptions and justification	• The impacts of air pollution on agriculture are affected by a large number of variables which are complex to model directly. • It was also 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 which are known to be sensitive to air pollution. • We therefore opt for a simple and transparent value transfer approach, taking the average of marginal damage cost estimates from Muller and Mendelsohn's (2007) and adjusting these internationally for purchasing power parity. • Ascribing a value to impacts on agriculture acknowledges that an impact exists, and consistent with the study on which it is based, the impact tends to have very low materiality in our results. • However, this approach is highly approximate, and if air pollution impacts on agriculture were identified as potentially significant during project scoping then this approach should be revisited.

3. Data requirements

3.1. Introduction

Gathering appropriate data is a precursor to valuing the environmental impacts from air pollution. The availability of high quality input data is a key determinant of the accuracy of impact quantification and valuation.

Three broad categories of data are required for quantification and valuation:

- **Environmental metric data:** Quantities of air pollutants released into the atmosphere.
- **Contextual data:** Provides additional relevant information about the basic metric data. For example, describing the context in which pollutants are released (e.g. location, surrounding population density, local weather patterns). The availability of useful contextual data will depend to an extent on the source of the metric data. For example, in the case of directly collected data, location and location characteristics should be known. Whereas in the case of data sourced from an environmentally-extended input-output (EEIO) model or life cycle inventory (LCI), it is likely that only the country and perhaps the industrial sector will be known.
- **Other coefficients**: Typically numerical values derived from the academic literature or other credible sources which are required in calculations to convert metric and contextual data into value estimates.

While methods for the collection or estimation of basic metric data are not the subject of this paper, the data generation methods used are nonetheless relevant to the likely availability of contextual data and therefore the viability of different potential valuation approaches. This chapter therefore has two purposes: firstly, it describes the most likely sources of metric data across a typical corporate value chain and the implications for contextual data availability; secondly, it sets out key contextual and other coefficient data requirements and the preferred sources for these.

3.2. Environmental metric data

To estimate impacts, we require the mass of each air pollutant emitted from a given source location in a given year. Our methodology considers emissions of six key pollutants: PM_{10}, $PM_{2.5}$, SO_2, NO_x, NH_3, and VOCs.

Measurement of air emissions is best done on site using direct in-line measurement. However, this is rarely practical across entire value chains and, instead, the drivers of air pollution can be measured to 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) are not available, modelling techniques such as EEIO analysis or industry / lifecycle assessment databases can be used. Such approaches give different levels of data specificity depending on the application. 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 for electricity emission factors. Similarly, EEIO data are only as specific as the sector and geographical resolution provided in the model.

The availability of metric data will vary according to the company's level of control over the producers and users of this information. This is likely to vary across a company's value chain. In Table 5 (overleaf), we summarize the likely metric data availability across the corporate value chain and implications for appropriate contextual information.

Table 5: Likely data availability across a corporate value chain

	Metric data	Implications for contextual data
Own operations	Direct measurement of drivers of air pollution, such as electricity and fuel use should be possible. The other estimation techniques detailed for the supply chain can also be used if direct data are unavailable.	Based on knowledge about the location of the company and supplier, it should be possible to source contextual information from public sources, if not from the company and suppliers themselves.
Immediate/ key suppliers	Supplier questionnaires can be directed to areas of high materiality or those with limited quality data from other sources. Most companies tend not to measure air pollution gases directly but will have information on air pollution drivers, fuel use, electricity consumption, waste to incineration, etc. Emission factors are required to convert these data to tonnes of different air pollutants.	
Upstream/ supply chain	EEIO can be used to give 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 this is deemed appropriate. Other data sources include government and industry reports, those from the IPCC and IEA may be particularly relevant.	Depending on the level of the company's visibility up its supply chain and the length and complexity thereof, specific location information may be available for some suppliers, supplier groups or commodities. Multi-region EEIO models and trade-flow data bases, can provide generalised indications of location. Tracing raw material flows can be another method for determining the location of different activities in the supply chain.
Downstream/ use phase	It is often desirable to estimate the emissions associated with a company's product or service over its useful lifetime. In the case of a vehicle for example, this would include impacts associated with fuel consumption, replacement parts and servicing. Use phase estimates are often highly approximate, based on limited data and sweeping assumptions. For some products, use phase impacts can be highly material and in such cases it is particularly important to be as accurate as possible.	Depending on the product, the location of sale or warranty registrations may provide a good indication of the use and disposal phase location. For intermediary products it may be necessary to consider subsequent trade flows as well. Customer use data, either directly reported by products or collected through customer surveys can also provide valuable information on the location and intensity of use. National and regional statistics on waste treatment rates may be useful and are considered further in the PwC methodology paper *Valuing corporate environmental impacts: Solid waste.*
End of life/ re-use impacts	Different products are disposed of in different ways. Some may be down-cycled, recycled or up-cycled (in each case the allocation of emissions from the associated processes, to product users, needs consideration). Others will be sent to landfill or incineration. Again, actual data are often limited and assumptions based on the location of sale, major materials, and the design of the product may be the primary source of waste treatment emissions estimates.	

3.3. Contextual data and other coefficients

Meteorological conditions and population densities are particularly important determinants of the impacts on people from air pollution. For a company's own operations, or that of its closest suppliers, specific locations may be known. However, emissions data from the rest of the value chain may only be available at a country level. This methodology is designed to be applicable at different geographical scales; the preference should always be for location-specific estimates; country level data are considered to be the minimum acceptable level[6].

Where only country level data are available, we recommend calculating the impacts for several specific in-country locations and taking an average (which could be appropriately weighted by population or economic output, depending on the emitting activity) to estimate the likely impacts. Contextual information provided alongside metric data will often help to identify probable locations. For example, emissions from agricultural activities are likely to be rural, emissions from certain industrial sectors are likely to be concentrated in key industrial clusters. Table 6 presents the required contextual data, and suggests some publically available data sources.

Table 6: Contextual data requirements

Data	Explanation
Meteorological data	
Wind speed and direction	A six hour moving average for the year is needed for the dispersion modelling.
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.
Temperature	Used in the visibility equations, temperature affects ozone formation.
Population data	
Population density	Used in the dispersion model and other equations to estimate the number of people affected in given area.
Baseline mortality	Used in the dispersion model, dose-response functions output a percentage increase from the baseline.
Median income	Used in the ozone related equations to represent differences in willingness to pay. Median is preferred to mean because it is less influenced by income inequality and large or small outliers.
Atmospheric data	
Ground level ozone concentration	Used in the ozone related equations, ambient levels affect the impact of additional emissions.

[6] Caution is required in interpretation of country level estimates, acknowledging that in-country variation may be greater than variation across country averages – this should be tested with sensitivity and uncertainty analysis.

Data	**Explanation**
Other coefficients	
$PM_{2.5}$ response coefficient	Provides the change in mortality rate per 10 microgram/m3 increase in concentration.
SO_2 response coefficient	Provides the change in mortality rate per 10 microgram/m3 increase in concentration.
Values for mortality and morbidity endpoints	Provides the value of relevant health endpoints including chronic bronchitis, cardiac hospital admission, restricted activity day and mortality.
Societal cost estimates for each metric across US districts for secondary health, visibility and agriculture	Provides a large heterogeneous sample of estimates from which to derive international transfer functions with the aid of additional contextual data.

4. Detailed methodology: Primary SO_2 and primary and secondary PM health impact valuation module

This chapter covers the valuation of human health impacts from pollutants modelled via dispersion modelling. These pollutants include PM, NO_x, SO_2, and NH_3. Other pollutants that harm human health such as ozone formed from VOCs and NO_x are valued in a separate module (see Chapter 5). This valuation approach traces the pollutant from its environmental outcomes to its societal impacts. This addresses the pollutant emission through dispersion to inhalation by humans (i.e. dose) to health harms (i.e. response) and values those health harms. A summary of the valuation approach can be seen in Table 7.

Table 7: Summary of human health from dispersion methodology (from Chapter 2)

	4.1 Quantify environmental outcomes	4.2 Estimate societal impacts
Health impact module from SO_2 and primary and secondary PM (primary pollutants: PM, NO_x, SO_2, and NH_3)		
Methods	• Lagrangian puff air dispersion model determines change in primary and secondary pollutant concentrations over a specified area. • Dispersion model considers local meteorological conditions, as well as the persistence in air of pollutants in estimating the dosing. • An estimate of the number of people affected is produced by overlaying a population grid describing the demographics in the location of interest.	• Dose-response functions estimate health outcomes for populations exposed to pollutants. • To value specific morbidity health outcomes Willingness to Pay (WTP) estimates from peer reviewed literature are used. For mortality, the OECD estimate of the value of a statistical life (VSL) is used.

4.1. Quantify environmental outcomes

In order to evaluate the impacts of air pollution on people, we model the change in concentration of the pollutant in the air via dispersion modelling[7] to determine human exposure to the pollutant.

Dispersion models use meteorological conditions and other contextual information to predict changes in concentrations of air pollutants at receptor locations. They capture the geographic extent of the impact of the emissions (US EPA, 2012). There are a number of different model types available including box, Gaussian, Lagrangian, and Eulerian. The choice of model type largely depends on a trade-off between ease of use and the

[7] Dispersion modelling is not the only method available to estimate human exposure to air pollutants. However, we feel it is the most appropriate for our purposes. Some alternative modelling approaches include using the Air Pollution Emissions Experiment and Policy Model (APEEP) or assuming all impacts are local. Our chosen model is a better fit than APEEP for company-level applications which we are conducting, and using dispersion modelling creates more accurate outcomes than assuming all impacts are local. Refer to Appendix I for a deeper discussion.

level of detail in the outputs it provides. For more information on dispersion modelling and model types, please see Appendix I.

Our chosen dispersion model is an open-source, simplified air pollution dispersion model called ATMOS 4.0. ATMOS is 'a meso-scale three-layer forward trajectory Lagrangian Puff-transport model'[8], designed for relatively rapid assessments of air pollution impacts (Heffter, 1983; Guttikunda, 2009). It is a simplified version of the US National Oceanic Atmospheric Administration, Branch Atmospheric Trajectory (BAT) model. It feeds into another model, called the Sim-Air Model, which quantifies the number of health impacts based on the change in concentration calculated by ATMOS. It has been used for numerous assessments of air pollution impacts, principally in Asia. For example, Afrin *et al.* (2012) used it to assess PM damages in Bangladesh in association with US EPA International Emissions Conference and Guttikunda (2010) used it to assess air pollution in Delhi, India. However, the model can be applied more widely subject to data availability (Guttikunda *et al.*, 2005, Guttikunda, 2009) which is useful for our global modelling needs.

We have selected this model because it can provide rapid assessment using a peer-reviewed dispersion methodology. The simplified model is designed to provide credible outcomes which match the level of data that companies have access to. While the model is simpler in nature than other air quality models, it is appropriate in scope for this application because it is rarely feasible or practical to run more complex and demanding models to cover a company's entire supply chain.

Our approach using ATMOS 4.0 represents a significant improvement over simpler benefit transfers of existing estimates. It allows us to explicitly address the spatial aspects of air pollution and to take into account more variables and a far more detailed datasets. And crucially, in cases where location specific data are available, it enables us to generate credible localised estimates based on local conditions and demographics. Figure 2 outlines the process for modelling changes in air pollutant concentrations driven by corporate activity.

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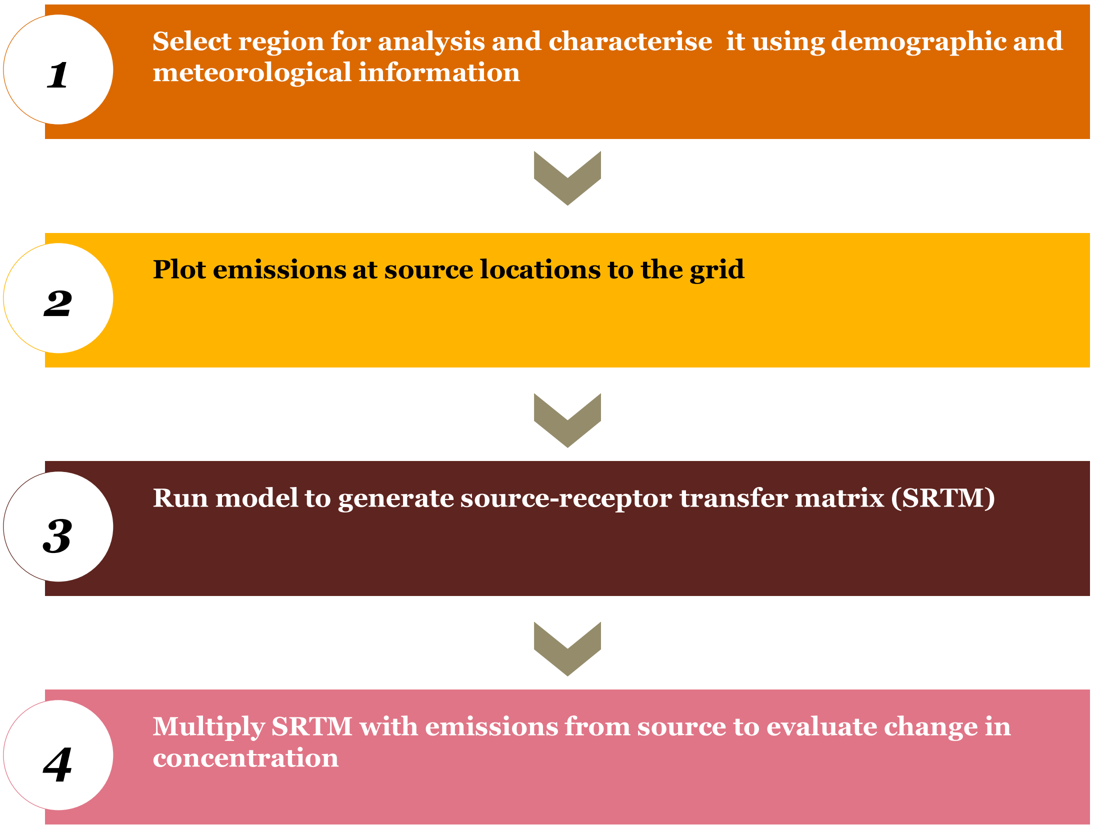
[8] ATMOS is run in the programming language FORTRAN. This methodology makes no direct modifications to the underlying program. It can be accessed and downloaded here: http://www.urbanemissions.info

Figure 2: Steps required to model changes in air pollutant concentrations driven by corporate activity



The following sections present the calculation steps to estimate impacts on human health using the dispersion model. The methodology is designed for location-specific analysis, but where precise locations are not known country level impacts can be estimated by calculating several location specific coefficients and taking an average (locations should ideally be weighted be to best represent the probable location of the emitting activity, for example by population or industrial activity).

4.1.1. Step 1: Select region for analysis and characterise it using population and meteorological information

The first step in modelling the local impacts of air pollution is to represent the local region (e.g., its population density and meteorological conditions) in the model. Air pollution is a local impact - primarily affecting those near the pollution site, so representing the region near the pollution source as accurately as possible is essential to achieve accurate results.

We represent the modelled location using a grid which corresponds to the geographical extent of the analysis. The maximum geographical region that ATMOS can model is 2.5 degrees latitude (approximately 250 km) by 2.5 degrees longitude (ranging from 0km at the poles to 250km at the equator), with grid squares of a maximum size of 0.1 degrees latitude by 0.1 degrees longitude.

For the analysis presented here a default of 50 km by 50 km is used[9]. This allows a ‘square area’ to be consistently selected for most cities (up to about 600 degrees north, equivalent to Oslo, and 600 degrees south, equivalent to the southern tip of Patagonia). However, this can be adjusted to select the most appropriate scale

[9]The model was tested with larger parameters (up to 250km squared) with little observed differences in results

for a specific application, and where necessary multiple grids can be stacked together - for example to estimate longer range / trans boundary impacts.

The level of detail in characterising local conditions depends on the specificity with which we can describe the source of the emissions. Wherever possible, locally-specific information should be sought, such that the grid can be populated with data which accurately reflects the local context. However, where emissions can only be estimated at a more aggregated scale (e.g. country level), it is generally necessary to estimate impacts for multiple locations within the geographic area of interest and then produce an appropriately weighted average.

4.1.1.1. Meteorological conditions

The dispersion of pollutants is highly dependent on the weather conditions surrounding the pollutant source. The ATMOS model is used to estimate how pollutants disperse and how concentrations of the pollutant change in the areas surrounding the pollutant source. The model uses four data records per day for 365 days: midnight; 6am; midday; and 6pm. Each data record includes mixing height, wind speed, wind direction and precipitation, all of which influence pollutant dispersion.

Mixing height - also referred to in the literature as planetary boundary layer height, is the maximum height above the surface up to which a plume of gases in the air will vertically disperse over a given time period. The science for accurately measure mixing height continues to evolve and there is currently no universally accepted way of determining the mixing height at a location (Seidel et al., 2009). To avoid data intensive calculations, a relatively simple approach is used here, as described by Seibert (2000). Radiosonde data are used to identify the vertical location of the first discontinuity in the temperature profile of the troposphere, thus giving an approximation of the mixing height at a given location.

Meteorological data are collected on a consistent basis globally for a range of scientific applications. The data required are available for most global locations through public sources online and are described in Table 8. Data from the nearest monitoring station to the point of release should be selected.

Table 8: Point measurement meteorological data for the dispersion model (ATMOS)

Data input	Description
Wind speed (m/s)	The model uses four data points a day for 365 days: midnight, 6am, midday and 6pm. Hourly wind speed readings are used to produce a six hour moving average for each data point. Where there is a gap in the data and a reading is not available, the equivalent reading from the same time the previous day is used. The sensitivity analysis (Chapter 8) shows that wind speed is the most influential variable in the analysis. However, it is also the variable with the most detailed data available.
Wind direction (degrees)	Hourly wind directions readings are used. Where a reading is not available, the equivalent reading from the same time the previous day is used. The wind direction at the relevant hour is used for each data point.

Data input	Description
Precipitation (mm)	Average monthly precipitation and average number of days of precipitation per month are used for each location. To estimate hourly rainfall, the average monthly rainfall is divided by the number of days of precipitation in that month and then divided by 24. Average hourly precipitation is assigned to each data point for the number of days of precipitation in each month .For example, if there are 15 days of precipitation in January then the average hourly precipitation will be assigned for all 4 data points for the first 15 days of January with the remaining 16 days are assigned a zero. This gives the most realistic account of the precipitation profile of a month where daily precipitation data are not available.
Mixing height (m)	Two observations of mixing height each month are taken: one at midnight and one at midday. The midnight reading is also used for the 6am data point and the midday reading is used for the 6pm data point. These values are repeated for each day in the month. This approximation is used because of the lack of readily available hourly data that can be used to derive mixing height.

4.1.1.2. Population

To assess the impact on the human population of each pollutant, the number and distribution of people living around the pollutant source is required. Local population statistics can be used to represent the distribution of people within the grid. Figure 3 below illustrates a generic schematic for a city, showing the number of people in each grid square, in applying the method a specific city can be modelled.

Some studies have indicated that certain age groups, and in particular the elderly and young, may be more susceptible to the health impacts of air pollution. Our method does not break down the exposed population by age demographics because consistent international data are not available. This is aligned with the approaches described by Industrial Economics (2011) in their analysis of the US Clean Air Act and in Ostro's (1994) methodology review.

Figure 3: Generic urban/city population density schematic including population in each 5km² square

78	389	389	389	389	389	389	389	389	78
389	389	17267	17267	17267	17267	17267	17267	389	389
389	17267	17267	19184	19184	19184	19184	17267	17267	389
389	17267	19184	19184	23019	23019	19184	19184	17267	389
389	17267	19184	23019	30675	30675	23019	19184	17267	389
389	17267	19184	23019	30675	30675	23019	19184	17267	389
389	17267	19184	19184	23019	23019	19184	19184	17267	389
389	17267	17267	19184	19184	19184	19184	17267	17267	389
389	389	17267	17267	17267	17267	17267	17267	389	389
78	389	389	389	389	389	389	389	389	78

4.1.2. Step 2: Plot emissions at source locations on the grid

Tonnes of emissions per year should be entered into the appropriate grid cell to represent the source location of the emissions. Where exact locations are known the grid should cover the major population centres around the emission source, taking into account prevailing winds.[10]

In the event that the precise location of emissions is not known multiple population grids can be created to represent relevant population centres or industrial areas. For example, twenty representative sites within a country could be modelled in this way, with results weighted by population or industrial production and averaged to produce a value estimate which can be applied at a country level. Producing value estimates for a range of potentially relevant sites also provides a basis for uncertainty analysis with upper and lower bounds represented by the highest and lowest values within the sample of sites.

However, an estimate produced in this way is likely to exhibit very significant uncertainty because the impact of emissions in a remote coastal area (high wind, low population) will be far lower than emissions in a densely populated inland city (lower wind, high population). Even in the absence of location information it is usually possible to produce a significantly more accurate value estimate with limited additional information. For example, if the nature of the emitting activity is known more accurate identification of representative sites should be possible. Heavy industrial activities are frequently located on the outskirts of major population centres, high end retail is generally found in town and city centres, and finance and consulting services tend to cluster in commercial districts. Agricultural activities and certain other primary industries are almost exclusively performed in lightly populated rural areas.

We therefore produce a number of country-level estimates based on a significant sample of population and industrial centres; differentiated by the type of activity being undertaken. As a minimum we create differentiated estimates for three types of stationary source: inner city, urban industrial and rural; and two types of mobile source: urban transport and rural transport. Our preferred approach is to base this differentiation on land zoning / planning maps for the reference areas identified. If these are unavailable, appropriate emission source locations can be identified with reference to population densities or satellite images.

By estimating value estimates in this way it is possible to significantly reduce the uncertainty caused by a lack of location specific emissions data. In some cases it is possible to produce activity specific value estimates to further reduce uncertainty. For example, if it is known that a specified quantity of emissions relates to automotive components suppliers in Germany; a reliable average can be produced by modelling a handful of industrial automotive agglomerations in the country.

Three illustrative source locations are presented in Figure 4 to demonstrate potential applications to the generic city population distribution shown in Figure 3.

[10] For example, the principle locations of interest for a factory on the east side of a city with strong westerly winds may actually be smaller population centres to the east of the city.

Figure 4: Three illustrations of applying source locations: stationary inner city (upper left); stationary urban industrial (upper right); mobile source – urban transport (lower left)

0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.08	0.08	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.08	0.08	0.08	0.08	0.00	0.00	0.00
0.00	0.00	0.00	0.08	0.08	0.08	0.08	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.08	0.08	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00
0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.13	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.13	0.00	0.00
0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01

4.1.3. Step 3: Run model to generate source-receptor transfer matrix (SRTM)

A source-receptor transfer matrix (SRTM) is the output of the ATMOS model. The SRTM specifies the change in air pollution concentration in each grid square based on the emissions from the source square(s). The model calculates this based on the meteorological conditions as well as deposition rates (wet and dry) of particulates. Any increase in emissions in a particular source cell increases the concentration in a receptor cell linearly – we calculate the SRTM for one kg of pollutant and multiply the results by the quantity of emissions in a given location.

Dispersion of primary pollutants $PM_{2.5}$ and PM_{10} is calculated based on the quantity of emissions, their location and meteorological information. For the secondary pollutants of NO_x, SO_2 and NH_3, the model first calculates the amount of secondary pollutant formed and then models the dispersion in the same way as $PM_{2.5}$.

The quantity of secondary pollutants from NO_x and SO_2 is determined by their reaction with available oxygen. This is a non-linear photochemical process which is highly dependent on local conditions and is subject to seasonal variations (Khoder, 2002). As recommended by the Sim-Air model, we apply a constant reaction rate from primary to secondary pollutants and rates for dry and wet deposition. This is also in line with the approach taken by ExternE (2005), Muller and Mendelsohn (2007) and others.

For NH_3, the main pathway to human health impacts is through NH_4 (which contributes to $PM_{2.5}$) formation, through reactions with SO_x and NO_x already in the atmosphere. The model first estimates NH_3 deposition and the reactions to form NH_4, before calculating the dispersion in the same way as $PM_{2.5}$. The rate of deposition and fraction which forms NH_4 varies by region, local canopy cover and weather conditions. A range of deposition values have been recorded: Loubet et al. (2006) estimates between 40% and 98% are deposited within 2km. We use a 70% deposition rate as the default value, and where the local average deposition rate is known this can be used.

4.1.4. Step 4: Multiply SRTM with emissions from source to evaluate change in concentration

To calculate the change in concentration of pollutant (i.e. dose), measured in $^{mg}/_{m^3}$ across the grid, the SRTM is multiplied by the emissions within the grid outlined in Step 2. Figure 5 shows an illustrative output of this process.

Figure 5: Concentration changes as a result of emissions

0.0001	0.0001	0.0002	0.0003	0.0005	0.0007	0.0006	0.0004	0.0003	0.0002
0.0002	0.0002	0.0004	0.0006	0.0006	0.0009	0.0014	0.0010	0.0004	0.0003
0.0002	0.0003	0.0005	0.0008	0.0010	0.0013	0.0017	0.0011	0.0005	0.0003
0.0001	0.0002	0.0003	0.0005	0.0009	0.0013	0.0013	0.0011	0.0006	0.0003
0.0001	0.0001	0.0003	0.0004	0.0004	0.0006	0.0011	0.0013	0.0009	0.0005
0.0001	0.0002	0.0004	0.0004	0.0003	0.0010	0.0020	0.0017	0.0010	0.0005
0.0001	0.0002	0.0003	0.0003	0.0010	0.0029	0.0031	0.0019	0.0010	0.0005
0.0001	0.0001	0.0004	0.0005	0.0015	0.0036	0.0034	0.0016	0.0008	0.0004
0.0001	0.0001	0.0004	0.0006	0.0009	0.0025	0.0030	0.0014	0.0005	0.0002
0.0001	0.0001	0.0003	0.0005	0.0005	0.0013	0.0018	0.0010	0.0005	0.0002

4.2. Estimate societal impacts

To value the impact of the environmental changes for this impact pathway, we first assess the harm to health from increased pollutant concentrations and then value those harms.

We use dose-response functions from published academic literature to estimate how the changes in pollutant concentrations, as calculated in the previous section, affect human health. Dose-response functions describe how the number of health outcomes (responses) change with increasing concentrations of air pollutants (doses), such as those calculated above in Figure 5.

Although there are a variety of approaches to modelling dose-response relationships, there is academic support for a linear relationship between ambient pollutant concentrations and the number of health incidents (Ostro, 1994; ExternE, 2005; World Bank, 2008). This is particularly prevalent in policy analysis because of the ease of computation and broad geographic applicability. We have chosen the linear model as most appropriate for our purposes. For more discussion of dose-response functions generally and linear dose response functions in particular, see Appendix II.

The societal cost of the health outcomes quantified above are valued using estimates of willingness to pay (WTP) for health and life[11]. There are alternative approaches to this valuation including the Cost Approach, but

[11] On occasion estimates of willingness to accept compensation (WTA) for ill-health or death are also cited in the literature. In theory, WTP and WTA should be approximately the same, but in practice, estimates of WTA, particularly from stated preference surveys, are often found to be higher. One reason is that WTA is not bound by a person's income, but other reasons for the divergence have also been reviewed extensively in the literature. Consistent with the literature we rely on we refer to WTP values throughout.

we feel that WTP is the most complete valuation approach. For more information on WTP and its comparison to Cost Approach, see Appendix III.

WTP values for health and life are widely used in public policy decision-making. We select WTP valuations from the OECD, which were derived from a meta-analysis of studies in many countries.

Figure 6 outlines the process for estimating the value of the health outcomes from corporate air emissions. Each of the steps is discussed in more detail below.

Figure 6: Steps for estimating the value of the health outcomes from corporate air emissions

1. **Estimate mortality and morbity from pollutant doses**
2. **Select WTP estimates for mortality and morbidity outcomes**
3. **Apply benefit transfer and test for income elasticity sensitivity**
4. **Compute the total cost of human health impact for each air pollutant**

4.2.1. Step 1: Estimate mortality and morbidity from pollutant doses

To estimate the health impacts of relevant air pollutants, we use the 'doses' (changes in pollutant concentration) and the number of people affected - estimated through dispersion modelling - and apply an appropriate linear dose-response function for each pollutant. The output is an estimate of the number of episodes of different health outcomes by location as a result of the original emission.

4.2.1.1. Selection of health endpoints

Between two thirds and three quarters of the health costs of air pollution are attributable to premature mortality (OECD 2009, Muller and Mendelson 2007). Of the six emissions considered in this paper, two are directly associated with premature mortality: SO_2 and $PM_{2.5}$. $PM_{2.5}$ consists of both primary $PM_{2.5}$ and secondary $PM_{2.5}$ formed from reactions of NO_x NH_3 SO_2 and other gases in the atmosphere. Hence the contribution to premature mortality of $PM_{2.5}$ NO_x and NH_3 can be quantified using a response coefficient for $PM_{2.5}$, as can part of the contribution of SO_2. SO_2 is also directly associated with premature mortality independent of its role in $PM_{2.5}$ formation and this affect is quantified using an SO_2 specific response coefficient. Consistent with recent research (Muller and Mendelson, 2007) PM_{10} is assumed not to contribute directly to premature mortality.

For morbidity impacts, we focus on chronic bronchitis (PM_{10}), restricted activity days (RAD) ($PM_{2.5}$) and chest discomfort (SO_2). Chronic bronchitis is typically the most material of these, while RADs and chest discomfort are fairly broad categories to capture many of the short term health ailments associated with these pollutants. While the literature notes there is uncertainty as to the precise cause-effect relationships between each pollutant and specific health outcomes, these outcomes are typical (see, for example, Ostro (1994), and ExternE's 2005 review of approaches).

4.2.1.2. Selection of response coefficients

We use linear dose-response coefficients originally published in peer reviewed academic literature which have subsequently been applied by international institutions such as the WHO, World Bank, US EPA, UK Defra, OECD and EU. In each case we have sought to identify the response coefficient which are cited most frequently and/or is recommended by the most recent comprehensive and authoritative study related to that emission and health endpoint.

Table 9: Response coefficients required to quantify health outcomes

Pollutant	Health endpoint	Unit
SO_2	All-cause mortality	Percentage change in mortality rate per 10 ug/m^3 change in pollutant concentration
$PM_{2.5}$	All-cause mortality	Percentage change in mortality rate per 10 ug/m^3 change in pollutant concentration
$PM_{2.5}$	Restricted activity days	Change in incidence of outcome per exposed person as a result of a 10 ug/m^3 change in the concentration of $PM_{2.5}$
PM_{10}	Chronic bronchitis	Change in incidence of outcome per exposed person as a result of a 10 ug/m^3 change in the concentration of PM_{10}
SO_2	Chest discomfort	Change in incidence of outcome per exposed person as a result of a 10 ug/m^3 change in the concentration of SO_2

The key assumptions associated with the application of dose-response functions in this methodology are outlined in Table 10.

Table 10: Assumptions required for estimating the change in health outcomes arising from change in pollutant concentration

Assumption	Explanation
Linear dose-response function	Linear functions are derived from epidemiological studies. They assume that emission concentrations are already above any damage threshold such that any addition of pollution in the environment causes an impact. This can over-estimate impacts at low levels if the ambient level is below the threshold at which they are harmful. However, in most industrial locations this is not the case (ExternE, 2005). Linear functions are widely applied in the literature because location specific functions would require detailed data on ambient concentrations around the world.
Response coefficients can be applied on a global basis	Dose-response functions are mostly estimated based on data from the United States, United Kingdom and Canada. These are routinely applied internationally and at different geographical scales where more precise data are unavailable. ExternE (2005) notes that there are too few studies outside of Europe and North America to derive a robust relationship, and therefore recommend European and North American values are used elsewhere.

4.2.2. Step 2: Select WTP estimates for mortality and morbidity outcomes

Sources for mortality and morbidity endpoints are summarised in Table 11. The first three are meta-analyses performed by the OECD. If income adjustments are to be made these provide a consistent basis for this (see Section 4.2.3.1).

Table 11: Selected sources for mortality and morbidity endpoints

Source (study year) / base country	Health endpoint
OECD (2012) / Nations	Mortality (i.e., VSL)
OECD / Switzerland	Chronic bronchitis
OECD / EU	Cardiac hospital admission
ExternE (2005) citing Ready et al. (2004) EU	Restricted activity day

4.2.3. Step 3: Apply benefit transfer and test for income elasticity sensitivity

Conducting primary research on WTP is expensive and time-consuming, particularly at the global scale of a typical E P&L. We therefore base our estimates on existing studies from the OECD.

The discussion that follows considers the equity considerations of applying WTP estimates for health and life. This section presents an approach for transferring those estimates between countries, if these are to be included.

Benefit Transfer involves applying estimates of WTP from existing studies to different, but sufficiently similar, contexts. Equation 1 presents a method for adjusting these values based on income (see Equity Considerations below) (Markandya, 1998; OECD, 2012).

Equation 1: Income adjustment transfer factor

$$Transfer\ factor = \left(\frac{GNI_A}{GNI_B}\right)^{\varepsilon}$$

GNI_A is Gross National Income per capita of new site (to which the value is being transferred); GNI_b is Gross National Income per capita of reference site (for which the value as originally calculated, shown in

Table above), and both are adjusted for purchasing power parity (PPP)[12]. ε is the income elasticity of WTP for health or life. This transfer factor is then multiplied by the WTP values for each country and each mortality and morbidity type described above. This particular functional form is widely used by academics and policy-makers, including the OECD and World Bank, to transfer WTP values to avoid mortality or morbidity from developed countries to other countries.

It is difficult to identify which variables affect WTP. However, these are broadly encapsulated by an individual's preferences regarding risks to health and life and their budget constraints. Research shows that while WTP changes with income, the relationship is not directly proportional. This means that it is appropriate to apply an income elasticity of between one and zero. The OECD guidelines recommend an elasticity in the range 0.4-0.8 for benefit transfer of WTP for mortality and morbidity. We use a central estimate of 0.6 (OECD, 2012). Other approaches are presented in Box 1.

*Box 1: **Other approaches to income elasticity***

An income elasticity of 1 implies WTP is directly proportional to income, an elasticity of 0 implies WTP is the same irrespective of income.

The recommended income elasticity differs across institutions and academics. An income elasticity of 0.32 was calculated by Alberini *et al.* (1995) in transferring US values for acute illness to Taiwan. This study was subsequently used as a basis of transfers to other developing countries by Heintz and Tol (1996) and Quah and Tay (2002).

EPA analyses have typically applied a range of estimates with a low end of 0.04, a central value of 0.8, and a high end of 1.0 (US EPA, 2010). The OECD recommends a sensitivity analysis in the range of 0.4 and 0.8 (OECD, 2012). This is consistent with the methods to transfer health values across all E P&L methodologies.

4.2.3.1. Equity considerations

Most countries operate a principally market-based economy, where the allocation of resources is determined largely by the forces of supply and demand, which also establish prices in the economy. In this context, an individual's income determines the quantity of marketed goods that they can obtain. When estimating the monetary value of goods (or 'bads') which are not currently traded in markets, the income constraint must therefore be considered.

As peoples income changes, their level of demand for a good usually changes, and the amount they would pay for each unit of the good also changes. Empirical evidence for environmental goods (or avoidance of 'bads') suggests that this 'income effect' is positive – people are prepared to pay more as their income increases (Pearce, 2003). For this reason, if values estimated in one location are to be used in a different location, they need to be adjusted to take account of differences in the income constraints of people in each location.

This is best illustrated using an example. Suppose a survey of people living beside a lake in the USA finds that they value the leisure time they spend around the lake at $1,000 per year. This represents about 2% of their average annual income. Combining this with the number of people who live in close proximity to the lake allows for an estimate of the value of the lake for leisure purposes to be produced. This non-market value estimate can be taken into account when decisions which might affect the future of the lake (e.g. new developments) are considered.

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[12] This ensures that the values represent people's real purchasing power in different countries. For example, without this adjustment, $1 tends to buy more in a developing country than in the USA. After PPP adjustment, $1 buys the same in each country.

Now suppose we wish to estimate the value of a similar lake in Uganda. Resources to conduct a new survey aren't available but the number of people living near to the lake can be estimated, and it is known to be a popular recreation area. However, the average per capita income in Uganda is 1/100th of the average per capita income in the USA[13]. So assigning the same value of $1,000 per person in the Ugandan context would clearly be inappropriate; suggesting that local people would pay twice their average annual income for a year's worth of leisure at the lake. In order to estimate the value that local people place on the lake, relative to their other priorities, it is necessary to adjust for the differences in income constraints.

This central concept of income effects in non-market valuation of environmental goods is relatively uncontroversial, as is the practice of adjusting for differences in income and purchasing power when transferring value estimates between countries. However, when valuing goods (and bads) relating to human health, equity considerations become more apparent.

As with environmental goods, empirical evidence demonstrates that the amount individuals' would pay to maintain good health and to reduce risks to life increases with income (Viscusi and Aldy, 2003; Scotton and Taylor, 2010; OECD, 2010). This is reflected in estimates of the Value of a Statistical Life (VSL)[14]. When applying a VSL estimate calculated in one location to health outcomes in another location, it is common practice in the health literature (see for example: OECD, 2012; Hammitt and Robinson, 2011) to adjust the VSL to reflect the income differential between those locations, as described above.

These differences in preferences for life and health between locations may reflect a genuine acceptance of greater health risks, particularly in the context of other priorities such as economic development or employment. However, because preferences of this nature are often considered to be constrained by the limited choices available in low income contexts, the use of differing VSLs is contentious where decisions may relate to inter-regional resource allocations. In recognition of these concerns, the OECD (amongst others) recommend that where decisions may relate to allocations between regions a single VSL estimate should be used in policy analysis across those regions.

Given the range of possible decision-making contexts where E P&L results may be considered[15] it is important that the decision maker is aware of this potential issue and is in a position to make an informed decision. Whether the primary presentation includes or excludes income adjustments to health related values is therefore a decision for the ultimate user.

Either way we suggest that the effect of differing income levels on the results of an EP&L is assessed through sensitivity analysis.

Where the decision context has implications for inter-regional allocations, two sets of results should be presented: one which reflects equity concerns without any income adjustment to health related values, and a second which does take into account income differentials.

The decision maker will still need to consider a range of factors beyond pure environmental or health impacts. For example, a study which does incorporate income adjustments across a range of countries could provide incentives to shift polluting activities to lower income countries where the implied cost of impacts would be lower – this may be undesirable. However, a similar study which does not adjust for differences in income may deter foreign investment in lower income countries; investment which could have created improvements in well-being in excess of any health related losses.

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[13] Even after accounting for differences in purchasing power the ratio is 1/40th.

[14] *"Value of a Statistical Life (VSL), … represents the value a given population places ex ante on avoiding the death of an unidentified individual. VSL is based on the sum of money each individual is prepared to pay for a given reduction in the risk of premature death, for example from diseases linked to air pollution."* OECD, 2012

[15] For example, some decision contexts will be confined to a single country and could involve comparing environmental values to other factors (outside the E P&L) determined by prices or incomes within that country; while others could require prioritisation of impacts across many countries.

For this reason decision makers may also wish to consider a more holistic decision making framework such as PwC's Total Impact Measurement and Management (TIMM) which values environmental impacts alongside economic, fiscal and social impacts[16].

4.2.4. Step 4: Compute the total cost of human health impact for each air pollutant

The number of incidents of premature mortality and morbidity due to corporate air pollution is calculated in Step 1. The change in the number of health effects arising from changes in ambient pollutant concentrations is multiplied by the relevant PPP-adjusted WTP value (from Step 2 and 3) to give the total cost associated with the emissions. The calculation for the change in mortality for pollutant *i* is shown in Equation 2.

Equation 2: Calculating the total societal cost of mortality from air pollution

$$\textit{Total societal cost of mortality}_i = \textit{VSL} \times \textit{Change in mortality}_i$$

For morbidity, the total societal cost is calculated in the same way with WTP for each type of morbidity j applied to each pollutant i, as shown in Equation 3.

Equation 3: Calculating the total societal cost of morbidity from air pollution

$$\textbf{Total societal cost of morbidity}_{ij} = \textbf{WTP}_j \times \textbf{Change in morbidity}_{ij}$$

The total cost is obtained by summing the cost across all pollutants for mortality and all types of morbidity.

The assumptions on which this method relies are summarised in Table 12, and the data required are summarised in Table 13.

Table 12: Assumptions required for estimating WTP for impacts to human health

Assumption	**Explanation**
It is better to use a single value for mortality and for each type of morbidity than to use location specific values where available	There is considerable variation in methods and results from primary analysis conducted in different countries. This analysis draws on estimates from a meta-analysis conducted by the OECD. If income adjustments are to be made (see Section 4.2.3.1), this ensures that the underlying WTP is equal for all countries and that benefit transfer is applied consistently and based on a consistent methodology. The OECD study is chosen because it is drawn from a broad range of countries across developed and middle-income countries (there are fewer estimates in developing countries) and is therefore more appropriate for this global methodology than selecting an estimate from a particular country as the basis.
Income elasticity is 0.6	The recommended income sensitivity differs across institutions and academics. 0.6 is chosen as a midpoint of the OECD recommendations.

[16] See "Measuring and managing total impact: A new language for business decisions", PwC 2013: http://www.pwc.com/gx/en/sustainability/publications/total-impact-measurement-management/assets/pwc-timm-report.pdf and: http://www.pwc.com/totalimpact for more information.

Table 13: Data required for estimating WTP for impacts to human health

Data
Willingness to pay to avoid mortality and morbidity
Income elasticity of willingness to pay
Gross national income per capita, adjusted for PPP
Inflation

5. Detailed methodology: Secondary air pollutants health impact module

This chapter covers the impacts on human health from the formation of Ozone (O_3). Our valuation module traces the pathway from emission of NO_x and VOCs (and NH_3) to the formation of O_3 to the health harms from inhaled O_3 to the societal cost of those health harms. Due to the complexity of the pathway that forms O_3 and the availability of robust damage values, we estimate societal costs in one step using benefit transfer of those damage values. A summary of the methodology can be seen in Table 14.

Table 14: Summary of secondary pollutants health methodology (from Chapter 2)

5.1.Quantify environmental outcomes		5.2. Estimate societal impacts
Health impact module from secondary O_3 formation (primary pollutants: NO_x and VOCs)		
Methods	• Environmental outcomes and societal impact 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. • The transfer function provides an estimate of the societal cost of air pollution as a function of ambient O_3 concentration, local income, and local population density.	

5.1. Quantify environmental outcomes

Environmental outcomes of secondary pollutants are evaluated jointly in one single step using a robust OLS multivariate regression to define a benefit-transfer function.

Secondary pollutants are not subject to dispersion modelling in our approach. Ozone is not typically emitted directly by companies. Rather, it is formed through a photochemical reaction between VOCs, NO_x, and CO. This is highly dependent on specific atmospheric and weather conditions. It is beyond the scope of the ATMOS and Sim-Air models to simulate secondary pollutant formation as it only models linear secondary pollutants and physical dispersion.

Instead, we estimate the societal cost of ozone formation per tonne of VOC and NO_x emission using a multivariate transfer function estimated using data obtained from Muller (2012). As this provides a direct link between emissions and societal costs, it is not necessary to quantify environmental outcomes.

5.2. Estimate societal impacts

We use robust OLS multivariate functions to explain the variation in health impacts of VOCs and NO_x via ozone formation. The results are used to create a transfer function for other countries.

The underlying principles of the valuation of mortality and morbidity are the same as indicated above for primary pollutants. However, the difficulty of modelling ozone formation directly makes it necessary to directly estimate societal costs from the quantity of pollutants emitted. We therefore draw on societal cost estimates from Muller and Mendelsohn's (2007, 2009, 2011) peer reviewed studies and derive multivariate functions from their data to transfer these to other countries.

Muller and Mendelsohn's analysis provides US county-level societal cost estimates by multiplying the proportion of people at risk by the predicted ambient concentrations to estimate exposures. Per tonne values for the secondary ozone effects from NO_x and VOCs provided from the APEEP model are used to estimate transfer functions for mortality and morbidity.

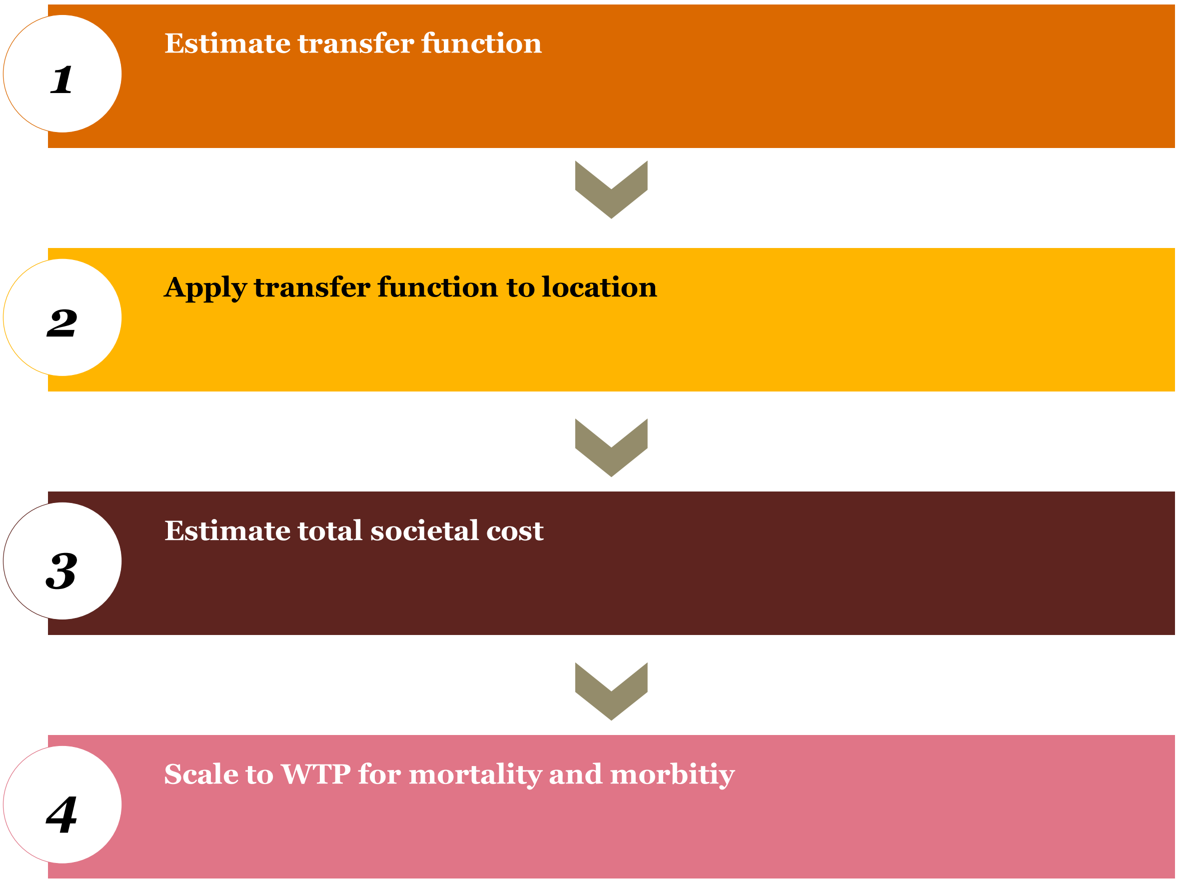
The lead author of these analyses, Nick Muller, has provided us with an update of the marginal societal cost values from his model, as summarised in Table 15. These values are based on a random sample of 346 (ten percent of the counties used in his original model).

Table 15: Societal cost per tonne (USD, 2011)

Pollutant	Mortality	Morbidity
NO_x	321.72	5.09
VOC	145.70	2.27

We use this dataset of cost estimates and contextual data by US county to estimate functions to transfer the societal costs of these impacts to different countries.

Figure 7: Process steps for estimating societal costs for secondary pollutants



5.2.1. Step 1: Estimate transfer function

Guided by Muller's input and using his dataset (Muller, 2012), we use multiple robust OLS regressions to estimate societal cost from morbidity and mortality per tonne of each pollutant as a function of population density, median income, and ambient ozone concentration for the US counties covered. All variables are log-transformed to allow easy interpretation of the coefficients by taking the natural logarithm (ln) of each. This means that each coefficient approximates the percentage change in the dependent variable (societal cost) associated with a one percent change in each independent variable (population density, median income, and ambient zone concentration). This is shown for each pollutant i in Equation 4.

Equation 4: Robust OLS transfer functions for the health impacts of secondary pollutants

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

α, β_1, β_2, and β_3 are coefficients estimated by the regression (see Table 16, below).

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 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 which affect the dispersion of pollution, the ambient ozone concentration will provide an indirect indication of this because where meteorological factors are such that pollution disperses the ambient concentration of ozone will be lower.

5.2.2. Step 2: Apply transfer function to location

To apply the transfer function, the coefficients in Table 16 should be input to the general functional form given in Equation 4 above. Then, data on the population density, median income, and ambient ozone concentration should be input to this function to calculate the societal cost of morbidity and mortality per tonne of each pollutant.

5.2.3. Step 3: Estimate total societal cost

To calculate the total societal cost, the marginal societal costs calculated in Step 2 are multiplied by the quantities of each pollutant released at each location for both mortality and morbidity. The sum of these gives the total societal cost.

The equations can be tested against the average US values in the full Muller and Mendelsohn dataset. Appendix VI shows that our model (which can value emissions globally) creates similar outputs in the US to Muller and Mendelsohn's US-only figures

5.2.4. Step 4: Scale to WTP for mortality and morbidity

The underlying WTP estimates of health and life are different in the Muller and Mendelsohn study. This is because they use US derived estimates of 'cost of illness' rather than WTP. To ensure consistency with the WTP values applied for the VSL and morbidity events in the other parts of our air pollution valuation approach, an adjustment is made to these figures.

For mortality, the VSL estimate from the OECD, used in our study, transferred to the US is 1.98 times larger than the figure used by Muller and Mendelsohn in their analysis (Table 16). For morbidity, the Muller and Mendelsohn estimates are lower and are scaled using the chronic bronchitis ratio of 0.92. It is not possible to disaggregate the bronchitis and cardiac admission impacts in Muller and Mendelsohn's per tonne estimates so the more conservative option is chosen (smaller adjustment giving higher values).

Table 16: Adjustment to account for differences in the underlying estimates of WTP for health and life

Value (USD, 2011, '000)	Muller and Mendelsohn estimates for U.S.	PwC analysis for U.S.	Ratio PwC to M&M estimate
Mortality (VSL)	2,125	4,200	1.98
Morbidity:			
Chronic bronchitis	343.50	321	0.92
Cardiac hospital admission	18.82	11.58	0.62

The main assumptions and data required for this approach are outlined in Table 17 and Table 18.

Table 17: Assumptions required for estimating health impacts of NOx and VOCs via O3

Assumptions	Explanation
Population density, median income and ozone concentration are the primary determinants of health impacts from NO_x and VOCs via ozone	These three variables represent differences in the propensity for emissions to have impacts (due to ambient pollution levels), the probable scale of the impact (based on the number of people that could come into contact with the pollution) and the value of impacts (which is affected by income). The impacts of weather on dispersion will also be represented within the ambient pollution level variable.
It is appropriate to transfer a relationship derived in the US to other countries	While the variation within the US is less than that between the US and other countries, it is considered an acceptable approximation in the absence of better data. The relatively low materiality of these health impacts compared with those calculated in the previous section further justify this approach.

Table 18: Data required to execute this valuation

Data	Explanation
Population density	Used in estimating the transfer function
Median income	Used in estimating the transfer function
Ambient ozone concentration	Used in estimating the transfer function
WTP values for mortality and morbidity	Used to scale the values to align with the rest of the model

6. Detailed methodology: Visibility module

This chapter covers the impacts on visibility from air pollution. The impacts include reduction in the quality of views (e.g., mountain vistas) as well as difficulties in aviation. Due to the existence of robust damage values per tonne of emissions, we estimate societal costs in one step to cover the process from emission to visibility degradation to societal costs. A summary of the methodology can be seen in Table 19.

Table 19: Summary of visibility methodology (from Chapter 2)

6.1. Quantify environmental outcomes	**6.2.Estimate societal impacts**
Visibility module (primary pollutants: PM, NO_x, NH_3, SO_2, VOCs)	
Methods	Environmental outcomes and societal impact are evaluated in one step using a multivariate transfer function, which extends Muller and Mendelsohn's (2007) US societal values to give global coverage subject to the availability of local contextual data. The transfer function provides an estimate of the societal cost of air pollution as a function of ambient O_3 concentration, local income, local population density, temperature, and rainfall.

6.1. Quantify environmental outcomes

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.

6.2. Estimate societal impacts

The methodology to estimate the impacts on visibility is also based on Benefit Transfer of Muller and Mendelsohn's (2007) US based analysis. The updated average of the marginal damage costs provided by Muller (2012) is presented in Table 20. Reductions in the level of air pollutants - particularly fine particulate matter, improve visibility, leading to physical and economic benefits in both recreational and residential settings (Industrial Economics, 2011).

Table 20: Societal costs of air pollutants due to reduced visibility per tonne emitted

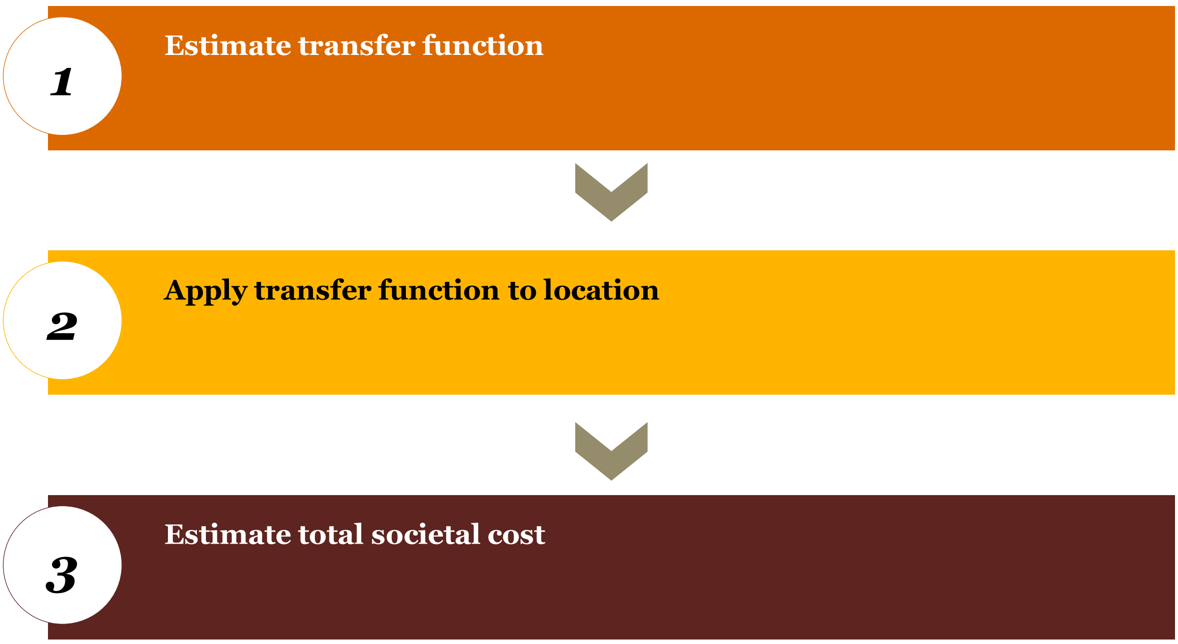
Pollutant	Societal cost per tonne (USD, 2011)
$PM_{2.5}$	21.2
PM_{10}	63.60
NO_x	10.43
NH_3	68.74
SO_2	37.83
VOC	7.71

Source: Muller (2012)

Muller and Mendelsohn (2007) use a model which describes the visual range in each county as a function of climatic and geographical factors and ambient concentrations of PM_{10}. They use a regression model to estimate the relationship between visual range while controlling for temperature, precipitation, latitude and altitude, Muller and Mendelsohn use contingent valuation (Chestnut and Rowe, 1990; Loehman and Boldt, 1990; McClelland *et al.*, 1990). Muller and Mendelsohn use estimates of household WTP for incremental changes in visibility associated with recreation experiences and regional estimates from Chestnut and Dennis (1997) and McClelland *et al.* (1990).

Benefit Transfer is the most practical method for estimating the societal cost of air pollution from reductions in visibility[17]. This is because there are no consistent primary studies estimating WTP to avoid reductions in visibility for specific locations across the global scale an E P&L is likely to cover, and conducting a new, primary study on this scale is not feasible (see above). We therefore derive and apply multivariate transfer functions, based on the key factors outlined by Muller (2012). We follow the same approach as for secondary pollutant health impacts (above).

Figure 8: Process steps for estimating societal costs for visibility impacts



6.2.1. Step 1: Estimate transfer function

We estimate a transfer function in the same manner as for secondary pollutant health impacts (see above), but using a different set of explanatory variables. This log-log form equation (see above for discussion) is shown in Equation 5 for each pollutant *i*.

Equation 5: Robust OLS function to estimate the marginal societal cost per tonne of emissions from reduced visibility due to air pollution

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

[17] ExternE (2005) reports that "In the absence of a specific contingent valuation study for Europe aiming to elicit the average willingness-to-pay measure to improve visibility, some adjustment in the US numbers may be done to account for lower concern about visibility effects".

The same explanatory variables are used for each visibility transfer function to provide a simple and consistent approach, which also reduces the data burden of the analysis. The ozone measurement used is the fourth-highest daily maximum 8-hour average concentration for ozone in the year, which aligns with the approach taken by Muller and Mendelson and the US Environmental Protection Agency in their APEEP model.

6.2.2. Step 2: Apply transfer function to location

To apply the transfer function, the coefficients in Table 22 are input to the general functional form given in Equation 5 above. Then, data for each variable (population density, etc.) are input to this function to calculate the societal cost of reduced visibility per tonne of each pollutant.

6.2.3. Step 3: Estimate total societal cost

To calculate the total societal cost, the marginal societal costs calculated in Step 2 are multiplied by the quantities of each pollutant released at each location. The sum of these gives the total societal cost.

The estimates are fairly consistent with the source data, which builds confidence in the approach. For quantitative comparison of them models outputs with the training set of source data, please see Appendix VI.

The key assumptions underlying this method are listed in Table 21. The data requirements are listed in Table 22.

Table 21: Assumptions required for estimating health impacts of air pollution on visibility

Assumptions	**Comment on purpose and reasonableness**
Population density, medium income, annual rainfall, average annual maximum temperature, ambient concentration of O_3 are the primary determinants WTP for impacts on visibility	These variables represent: • Differences in the propensity for emissions to have impacts (due to ambient pollution levels, temperature, rainfall); • The number of people affected and the value of impacts according to people's budget constraints. The impacts of wind on dispersion are also represented within the ambient pollution level variable. Given this and the relatively high R^2 values for most pollutants, it is considered to be a fairly good set of explanatory variables. The R^2 is lower for NO_x and NH_3, perhaps due to the nature of the chemical relationships. Given the low materiality of the absolute values, we do not deem this to be a cause for concern.
It is appropriate to transfer a relationship derived in the US to other countries	The chemical reactions which lead to smog formation and reduce visibility will be the same across countries. In contrast, WTP for reduced visibility impairment may vary. ExternE (2005) noted that WTP in the EU was much lower than in the US. However, in the absence of better data to develop a more sophisticated function, this approach is considered an acceptable approximation, particularly given the materiality of the impacts relative to the societal costs of health impacts (see above).

Table 22: Data required for estimating visibility impacts

Data	Use
Population density	Used in estimating the transfer function
Annual rainfall	Used in estimating the transfer function.
Median income	Used in estimating the transfer function
Maximum temperature	Used in estimating the transfer function.
Maximum O_3 concentration	Used in estimating the transfer function

7. Detailed methodology: Agricultural productivity module

This chapter covers the impacts on agriculture from air pollution. The impacts cover reduction in agricultural productivity. Due to the existence of robust damage values per tonne of emissions and the complexity of primary estimation, we calculate societal costs in one step to cover the process from emission to environmental degradation to human harms to societal costs. A summary of the methodology can be seen in Table 23.

Table 23: Summary of agriculture methodology (from chapter 2)

7.1.Quantify environmental outcomes	**7.2.Estimate societal impacts**
Agricultural productivity** module **(primary pollutants: NO_x and VOCs)	
Methods	• Environmental outcomes and societal impact are evaluated in one step using a simple value transfer. • We take the average of marginal damage costs from Muller and Mendelsohn's (2007) US dataset and adjust this for purchasing power differences between countries.

7.1. Quantify environmental outcomes

The societal cost of air pollution's impact on agriculture is estimated directly from emissions using value transfer.

7.2. Estimate societal Impacts

Tropospheric ozone inhibits plant growth. As a result, VOC and NO_x emissions can result in reduced agricultural productivity.

Our methodology for estimating the impacts of these emissions on agriculture are based on Muller and Mendelsohn's (2007) US analysis, as for other impact areas introduced above. The updated average marginal damage costs provided by Muller (2012) are presented in Table 24.

Table 24: Societal costs of air pollutants due to reduced agricultural productivity per tonne emitted

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

Source: Muller (2012)

Muller and Mendelsohn's (2007) analysis uses a dose-response function from the National Crop Loss Assessment Network (Lesser et al. 1990). This expresses the change in yield as a proportion of the baseline yield. To derive the yield loss in absolute terms, they multiply the estimated response function by the baseline yield.

They show that not all air pollutants covered in this methodology paper are harmful for crop production. The economic impact for agriculture is determined by the loss of crop output. This is measured as the change in production caused by a one tonne increase in the level of pollutant relative to a baseline production level, multiplied by the average market price for the associated crops.

Our approach to transfer the damages per tonne for NO_x and VOC for agriculture is to adjust the societal costs provided by Muller (2012) for GNI (PPP) to account for differences in purchasing power, as described above. We explored econometric estimation of transfer functions but were unable to find variables with sufficient explanatory power to explain how crop yields varied with ozone. Therefore, we adopt the simpler value transfer approach, as opposed to Benefit Transfer (see above).

Figure 9: Process steps for estimating societal costs for agriculture impacts

1. **Estimate transfer values**
2. **Estimate total societal cost**

7.2.1. Step 1: Estimate transfer values

We leverage Muller and Mendelsohn's damage values for the impacts of a tonne of air pollutants on agricultural productivity. The values are summarized in Table 25. For individual countries, we adjust the transfer value based in PPP using the ratio Gross National income.

Table 25: Transfer values for agriculture

Pollutant	**Societal cost per tonne (USD, 2011)**
NO_x	20.03
VOCs	16.06

7.2.2. Step 2: Estimate total societal cost

Once the metric data on emissions and country-adjusted damage values are calculated, estimating societal cost becomes straight forward arithmetic. The assumptions upon which this relies are shown in Table 26.

Table 26: Assumptions required for estimating impacts on agriculture

Assumptions	**Explanation**
It is appropriate to transfer values derived in the US to other countries	The impacts on agriculture are affected by a large number of variables which cannot be adequately represented by a simple function. Given the low materiality of agriculture impacts on overall societal costs, a more detailed analysis is not considered appropriate at this stage and these values are accepted as an approximation.

8. Sensitivity analysis

8.1. General approach to sensitivity analysis

Sensitivity analysis refers to a process of testing the robustness of a methodology, and its outputs, to changes in the inputs. This is in order to identify those parameters with the greatest potential to drive the results, and to then focus attention towards those drivers.

There is no single approach to conducting sensitivity analysis, and the approach can vary based on the needs of the analysis. Our approach focuses on understanding the inputs which have greatest influence on the results and which we consider to have the most uncertainty surrounding them. It does not consider the outputs (i.e. what would the input need to be to give a pre-defined conclusion) because this depends on the context within which the approaches are being applied.

8.2. Module-specific sensitivity analysis

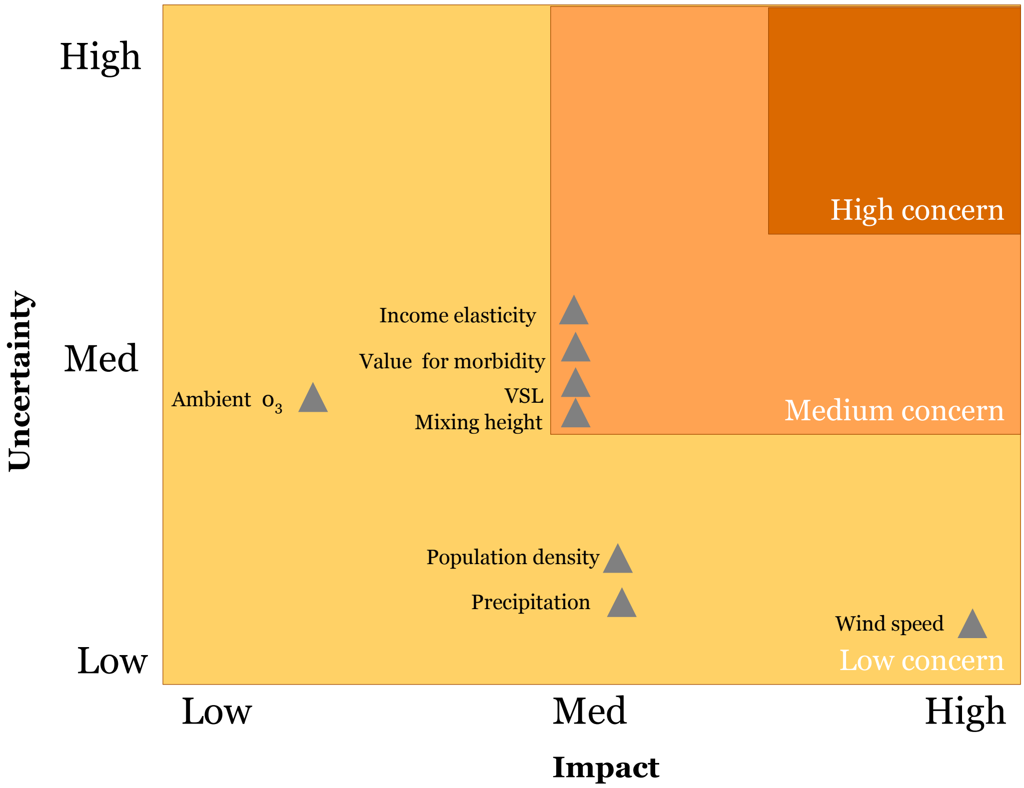
8.2.1. Overall summary and considerations for model use

This section presents a summary of the findings of our sensitivity analysis, more detailed discussion on the parameter influence on results and uncertainty follows.

The key parameters tested in our sensitivity analysis are mapped in Figure 10 maps on an influence/uncertainty matrix. The parameter with the highest impact on the results is wind speed, however this also has the most detailed data and so is considered to have low uncertainty.

Parameters relating to the valuation itself are of medium concern, based on medium impact and uncertainty scores. The sensitivity and uncertainty around these estimates is well documented in the literature; we feel by aligning our methodology with approaches used in international policy making (such as by the OECD) we are presenting the most appropriate methods for global supply chain analysis.

Figure 10: Impact/uncertainty matrix summarising the sensitivity assessment summary for key variables



8.2.2. Materiality

Our analysis corroborates the findings of the studies presented in Section 1.4; health impacts (mortality and morbidity) dominate the results. Figures 11 and 12 present the contribution of different impacts to the total per kg impacts of emitting NH3 and SOx across different countries. Based on this materiality assessment we focus the sensitivity analysis on parameters which influence the calculation for health impacts.

This sensitivity analysis considers the influence of these parameters on the coefficients for China, Nigeria and the US, giving a cross section of different country contexts. It is calculated for a central point emission in each of the capital cities within these countries. The parameters have a similar influence on the results irrespective on whether the analysis is carried out for non-point sources (transport) or in urban or rural settings.

Figure 11: Analysis of the relative contribution of visibility, agriculture, mortality and morbidity to the societal cost of a tonne of NH3 emissions in different countries

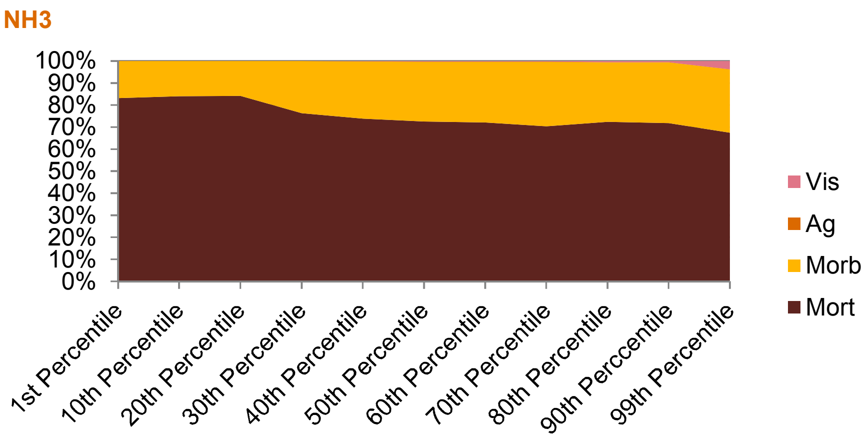
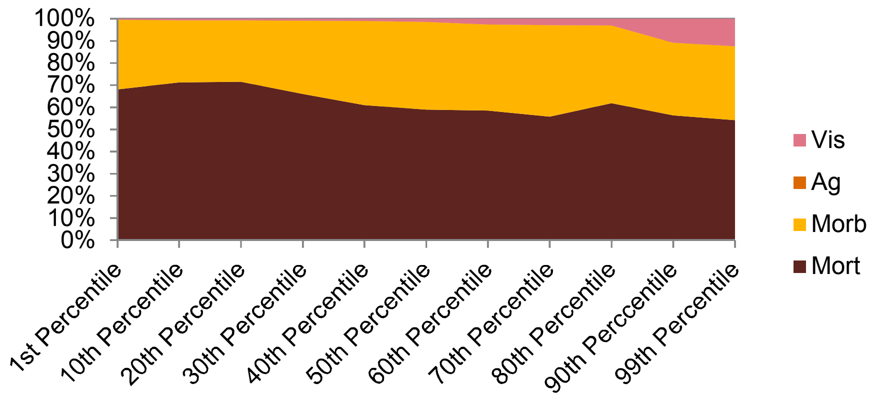


Figure 12: Analysis of the relative contribution of visibility, agriculture, mortality and morbidity to the societal cost of a tonne of SOx emissions



8.2.3. Parameter impact

The parameters included in our sensitivity analysis, and their influence on the results are provided in Table 27.

Changes in the wind speed data have the highest influence on the results. A stronger wind spreads the pollutant further, giving each person a smaller dose and resulting in fewer health outcomes, explaining why the societal cost goes down as the wind speed increases. The model uses four data points per day: midnight, 6am, midday and 6pm, based on a six hour moving average. Whilst wind speed has an important influence on the results, this granularity of data gives us confidence in the results.

Several other environmental variables also have notable impact including precipitation and mixing height. These also affect the dispersal of pollutants and how many people receive a harm-inducing dose.

The underlying valuation of WTP for mortality and morbidity are important as well. They function as scalars on the total number of health endpoints.

Table 27: Assessing the change in the overall societal cost per unit of emission by varying key parameters and decisions

Variable	Flex	Impact rating [18]	US (% change to module)	US (% change to overall cost)	China (% change to module)	China (% change to overall cost)	Nigeria (% change to module)	Nigeria (% change to overall cost)
Health valuation modules								
Mixing height	10%	Med	-8.7%	**-7.8%**	-2%	**-2%**	-9.2%	**-9.2%**
Wind speed	10%	High	-52%	**-46.3%**	-60%	**-60%**	-23%	**-22.8%**
Precipitation	10%	Med	-2.9%	**-2.7%**	-1 %	**-1%**	-0.5%	**-0.1%**
Income elasticity	10%	Med	1.2%	**1.1%**	-9%	**-8.9%**	-9.4%%	**-9.4%**

[18] Low = average response for overall cost for three countries is less than 1%
Med = average response for overall cost for three countries is 10% or less
High = average response for overall cost for three countries is greater than 10%

Variable	**Flex**	**Impact rating** [18]	**US** (% change to module)	**US** (% change to overall cost)	**China** (% change to module)	**China** (% change to overall cost)	**Nigeria** (% change to module)	**Nigeria** (% change to overall cost)
Values for morbidity	10%	Med	8.8%	**7.9%**	3.9%	**3.9%**	1.3%	**1.3%**
VSL	10%	Med	6.4%	**5.7%**	6.1%	**6.0%**	7.3%	**7.3%**
Population density	10%	Med	3.7%	**2.6**	10%	**10%**	10	**9.9%**
Ambient O3	10%	Low	9.5%	**0.0%**	9%	**0.0%**	9.8%	**0.0%**

8.2.4. Parameter uncertainty

The uncertainty of parameters is driven by the relative difficulty of measurement. Table 28 presents a qualitative assessment of uncertainty for each of the parameters. Measured parameters, such as wind speed, have a lower uncertainty relative to calculated parameters, like mixing height, or parameters such as WTP which are estimated.

The primary limitation of measured parameters is dependent on how close the location of measurement is to the location of the emissions. As this will depend on the location in question, and the quality of the underlying metric data on emissions, we do not attempt to represent this uncertainty here.

Table 28: Assessing the uncertainty of key parameters based on the reliability of the measurement and the variance in attempts to measure the parameter

Variable	**Uncertainty rating**	**Reliability/quality of measurement**	**Variance of the number measured**
Mixing height	Med	Calculated based on radiosonde data, but no consensus on best method in literature	<25%
Wind speed	Low	Measured with good degree of precision	<10%
Precipitation	Low	Measured with good degree of precision	<10%
Values for mortality	Med	Estimated, method used is peer reviewed and broadly accepted	<50%
Value for morbidity	Med	Estimated, method used is peer reviewed and broadly accepted	<50%
Population density	Low	Measured with good degree of precision	<10%
Ambient O3	Med	Measured and estimated depending on location	<25%

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Appendices

Appendix I: Background on dispersion modelling, our chosen model, and alternative approaches

Background on dispersion modelling

All air quality modelling includes consideration of physical dispersion. Some models also consider the photochemical reactions that lead to the formation of secondary pollutants such as ozone, as well as topography. These models are computationally intensive and require a number of input variables based on local observations. See below for discussion of model selection and addressing photochemical reactions.

The mathematical techniques used in air dispersion modelling vary in scope and complexity. The main types of model are shown in Table.

Table 29: Types of dispersion models

Model type	Description	Complexity and detail
Box	Assumes homogeneous distribution of emissions over an air-shed modelled in the shape of a box. These are easy to operate, but the assumption of a homogeneous distribution of emissions is not realistic.	Simplest and least detailed
Gaussian plume	Based on algorithms that assume pollutant dispersion is normally distributed in the direction of the wind from the source at a particular stack height.	
Lagrangian	Models the movement of a 'parcel' or 'puff' of emissions from its source over time, using meteorological conditions in the region. This allows for the characterization of air quality dispersion based on factors including wind, precipitation, and daily and seasonal variance in these factors.	
Eulerian	Complex models that have the ability to model dispersion at a detailed level and to include variation in terrains. They also have the ability to model photochemical reactions.	Most complex and detailed

As Table 29 shows, the choice of model type largely depends on a trade-off between ease of use and the level of detail in the outputs it provides.

Alternative modelling approaches

Air Pollution Emissions Experiment and Policy Model (APEEP)

Mueller and Mendelsohn's (2007) APEEP model is a possible alternative to the methods set out above:

- It is publically available;
- It estimates both air pollution dispersion and the associated societal cost.

However, it also has a number of drawbacks which make it less suitable than our chosen methodology:

- It is very data intensive, such that collecting the required data on a global scale is not feasible;
- The referenced data sets provided with the model only apply for the USA;
- The model focuses on economy-wide changes in emissions, and is not easily amenable to isolating the impact on individual companies.

Assume all impacts are local

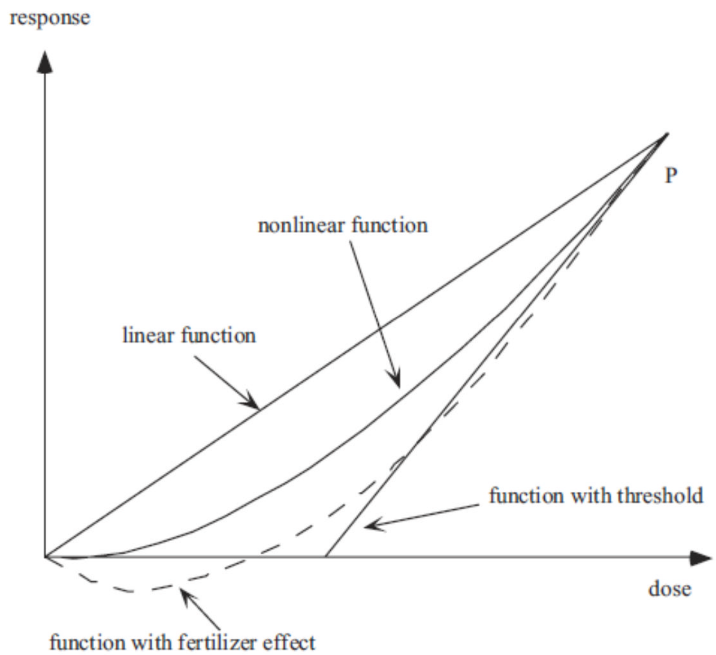
Another alternative to our chosen approach is to ignore air quality modelling and assume that all impacts are local. However, without the use of a dispersion model, the geographic extent of the pollution is not captured. In addition, if dispersion were not modelled and pollutants were considered only in their local area, the concentrations – and therefore the derived exposures to pollutants - would be unrealistically high.

Appendix II: Background on linear dose-response functions

There is academic support for a linear relationship between ambient pollutant concentrations and the number of health incidents. This is particularly prevalent in policy analysis because of the ease of computation and broad applicability. For example, ExternE (2005) cite a meta-analysis of 107 studies by Zmirou *et al.* (1997) whose results suggests that the dose-response function appears linear in the range of observed concentration for PM, SO_2 and NO_x. Figure 13 shows the possible behaviour of dose-response functions.

However, there is also extensive academic literature modelling the health function as a nonlinear relationship and commonly as a log-linear one. These tend to be advanced distributed-lag Poisson regression models, which are specific in terms of locality and population[19]. These models also control other local factors such as weather conditions, seasonality and long-term trends (Bell et al., 2004).

Figure 13: Forms of dose-response functions (Source: ExternE (2005))



The simplest functional form for the dose-response relationship is linear, whereby the number of cases of mortality or morbidity increases in proportion to an increasing concentration of air pollutants.

[19] For example, Abbey *et al.* (1991) examine the impact of ambient particular matter pollution on Seventh Day Adventist non-smokers living in California.

Equation 6: Linear dose-response function

$$dH = b \times P \times dC$$

where dH is the change in the number of estimated health incidents such as hospital admissions, death, and emergency room visits, b is a response coefficient that describes the change in number of incidents per unit change in concentration, P is the exposed population, and dC is the change in ambient pollutant concentration.

Using linear dose-response functions has a number of limitations:

- It is only applicable to specific air pollutants: NO_x, SO_2 and PM (ExternE, 2005).
- Linear functions gain broad applicability at the expense of local level specificity. Different countries have different baseline incidence rates, access to healthcare and populations. These all affect the likelihood of increased illness (Ostro, 1994).
- Linear dose-response functions are mostly drawn from the United States, United Kingdom and Canada. As such, any transfer of the dose-response function to other countries implicitly assumes that the relationship between ambient levels of pollution and health effects can be extrapolated across countries (Ostro, 1994).

However, linear dose-response functions also have several advantages which make them the most suitable approach for this methodology:

- Lower data requirements than alternatives. This is particularly important for an EP&L methodology which needs to be applicable globally. This is because the data needed for more complex functions, such as ambient concentrations and local illness probability, are only available for a small number of countries.

EP&L principally deals with small changes in concentrations which limits the impact of functional form on the final results.

Appendix III: Willingness To Pay and comparison to Cost Approach

Short-term and long-term exposures to air pollutants are consistently associated with ill-health effects (Defra, 2011b), which are typically divided into two categories:

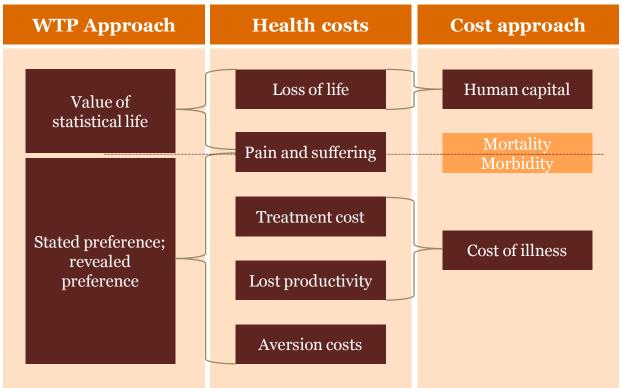
- Morbidity: increased incidence rates of illness. These are measured, for example, by cases of chronic asthma and chronic bronchitis, respiratory hospital admissions and emergency room visits for asthma;
- Mortality: premature death.

There are two broad approaches to estimating the social cost associated with morbidity and mortality:

- Cost Approach:
  - *Mortality:* The value of the lost contribution to economic activity due to premature death, known as the Human Capital approach;
  - *Morbidity:* The cost of treatment and lost productivity, a lower-bound proxy for WTP, known as the Cost of Illness approach;
- Willingness to pay:
  - *Mortality:* Stated and revealed preferences for avoided or reduced risk of death, known as the value of statistical life (VSL);
  - *Morbidity:* Stated and revealed preferences for avoided or reduced illness.

While both approaches are used in policy making, WTP approach is a more complete measure. WTP values for morbidity encompass direct (medical costs) and indirect costs of illness (i.e. lost productivity) as well as intangible aspects (e.g. pain and suffering). They therefore offer a better representation of individual preferences regarding the likelihood of illness or premature mortality *ex ante* (OECD, 2006). Figure 14 shows the difference between cost of illness estimates and WTP estimates.

Figure 14: Types of costs covered by Willingness To Pay Approach and Cost Approach



Under the WTP approach, the shadow price of mortality is termed the value of a statistical life (VSL). VSL is an individual-specific value, defined as the marginal rate of substitution between mortality risk and income, i.e. the individual's WTP for a small reduction in mortality risk divided by the risk change (Hammitt, 2002). Similarly, WTP for life encompass values of the loss of life and the intangible aspects of pain and suffering.

Appendix IV: Comparison of transfer functions with external values

Human health impacts from secondary pollutants

In the example below, the estimates are fairly consistent, with a difference of +17 to +32% compared with the dataset average[20]. The smallest difference between the estimated and actual value is for NO_x mortality, which has the highest value of damages per tonne. The largest difference in percentage terms is for morbidity damages arising from VOCs, where there is a 32% difference, but the difference in absolute terms is only 71 cents. This table provides a check on the results of the estimated transfer functions. It also suggests that, if there is any bias in the estimation, it is on the conservative side.

Table 30: Comparison of published values for secondary pollutants and values estimated using transfer functions

Pollutant	Actual Average Value for USA (USD/tonne)[21]	Estimated Average Value from transfer function (USD/tonne)	Difference (USD/tonne)	% Difference
Mortality				
NO_x	319.82	375.56	+55.74	+17.4
VOC	143.79	180.55	+36.76	+25.6
Morbidity				
NO_x	5.07	6.11	+1.04	+20.4
VOC	2.24	2.95	+0.71	+31.6

Visibility impacts

The equations can be tested by inserting the US values for median income, population density, maximum temperature, annual rainfall and ozone concentration and comparing them with the dataset provided by Muller (2012). This is shown in Table 31. The estimates perform well, using the population averages show a difference of -46% to +2% compared to the dataset average. The estimates for NO_x and NH_3 show underestimates. However, given the relatively low materiality of the damages, we do not see this as a cause for significant error.

[20] Calculated as the mean of the damages for each variable provided by Nick Muller to PwC in September 2012 for a sample of 346 US counties (Muller, 2012).

[21] Mean value of sample of 346 US counties provided by Nick Muller on 20 September 2012 from APEEP model.

Table 8: Comparison of published visibility impact values and values estimated using transfer functions for the US

Pollutant	Published average value for US (USD/tonne)[22]	Estimated average value from transfer function (USD/tonne)	Difference (USD/tonne)	% Difference
PM_{10}	62.90	63.58	0.68	+1
$PM_{2.5}$	21.00	19.12	-1.88	-9
SO_2	37.72	38.47	0.75	+2
VOC	7.63	7.72	0.09	+1
NO_x	10.47	5.63	-4.84	-46
NH_3	71.90	50.33	-21.57	-30

[22] Mean value of sample of 346 US counties provided by Nick Muller on 20 September 2012 from APEEP model.

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Valuing corporate environmental impacts: Greenhouse gases

PwC methodology paper

Version 4.3

pwc

Contents

Table of tables

Table of figures

Abbreviations and acronyms

Abbreviation	Full name
C	Carbon
CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
DECC	UK Department of Energy and Climate Change
DEFRA	UK Department for Food, Environment and Rural Affairs
E P&L	Environmental Profit and Loss
EEIO	Environmentally-extended input-output modelling
EU ETS	European Union Emissions Trading Scheme
GCM	General Circulation Models
GDP	Gross domestic product
GHG	Greenhouse gas
GWP	Global warming potential
HFC	Hydrofluorocarbon
IPCC	Intergovernmental Panel on Climate Change
LCA	Lifecycle assessment
MAC	Marginal abatement cost
MOSAICC	Modelling System for Agricultural Impacts of Climate Change
N_2O	Nitrous oxide
^oC	Degrees Celsius
PFC	Perfluorocarbon
PPP	Purchasing power parity
PRTP	Pure rate of time preference
SCC	Societal cost of carbon
SDR	Social Discount Rate
SF_6	Sulphur hexafluoride
SRES	Special Report on Emissions Scheme
t	Tonnes

1 The environmental impacts of greenhouse gases

1.1 Introduction

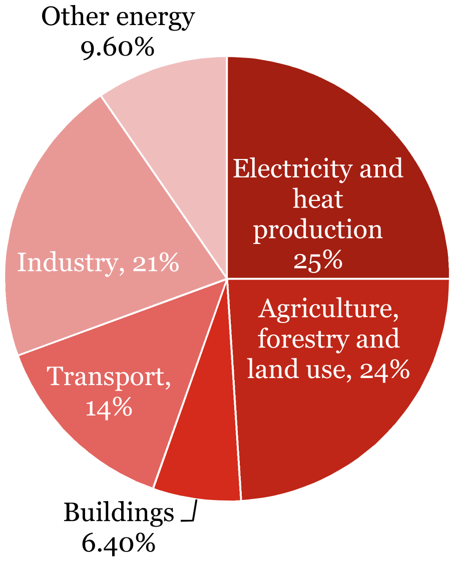
The earth's atmosphere shields us from harmful radiation, provides us with air to breathe, and traps enough heat from the sun to enable the planet to support complex life. Scientists have long been aware of this essential 'greenhouse effect,' but, in recent decades, they have become increasingly concerned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

Data now show conclusively that the earth is warming and has been for some time. In the last 100 years, global average surface temperatures have increased by 0.89 degrees Celsius (IPCC, 2013), and scientists have 'very strong confidence' that the net effect of human activities (and the resulting increase in atmospheric greenhouse gas (GHG) concentration) has contributed to this warming (IPCC, 2007). In this paper, we set out a methodology for valuing these costs in monetary terms that can be applied to each tone of GHG produced.

1.2 Overview of impact area

Greenhouse gases (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, where 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 six principal classes of GHGs are carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), sulphur hexafluoride (SF_6), various hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs) (IPCC, 2007). Figure 1 summarises the key sources of GHG emissions.

Figure 1: GHG emissions by sector in 2010 (IPCC, 2014)



According to the IPCC's Fifth Assessment Report on Climate Change (IPCC, 2014), there is 'high agreement and much evidence' that global GHG emissions will continue to grow over the next few decades. Under a range of scenarios, the IPCC's Fifth Assessment Report projects that the increase in global surface temperatures will be between 2.6 and 4.8 degrees Celsius by the end of the 21st century. The physical impacts (and resultant societal impacts) of this climate change are as diverse as its causes. Examples of the projected impacts are listed in Table 1.

Table 1: Projected impacts of climate change (IPCC, Climate Change 2007: Impacts, Adaptation and Vulnerability, 2007).

Impact areas	Examples of projected impacts
Freshwater resources and their management	• Drought-affected areas will likely increase in extent and heavy precipitation events, which are very likely to increase in frequency, will augment flood risk. • In this century, water supplies stored in glaciers and snow cover are projected to decline. This will reduce water availability in regions supplied by meltwater from major mountain ranges, which is where more than one-sixth of the world's population currently live.
Ecosystems	• Resilience of many ecosystems is likely to be exceeded this century by an unprecedented combination of climate change, associated disturbances (e.g. flooding, drought, wildfire, insects, ocean acidification) and other global drivers of change (e.g. land-use change, pollution, over-exploitation of resources). • Approximately 20-30% of plant and animal species (assessed so far) are likely to be at increased risk of extinction if increases in global average temperature exceed 1.5-2.5°C.
Agriculture	• Globally, the potential for food production is projected to increase with increases in local average temperature over a range of 1-3°C. Above 3°C, it is projected to decrease. • Increases in the frequency of droughts and floods are projected to affect local crop production negatively, especially in subsistence sectors at low latitudes.
Coastal systems and low-lying areas	• Coasts are projected to be exposed to increasing risks, including coastal erosion, due to climate change and sea-level rise. The effect will be exacerbated by increasing human-induced pressures on coastal areas. • Many millions more people are projected to be flooded every year due to sea-level rise by the 2080s. The numbers affected will be largest in the mega-deltas of Asia and Africa, while small islands are also especially vulnerable. • Adaptation for coasts will be more challenging in developing countries than in developed countries due, in particular, to the high costs of many forms of adaptation.
Industry, settlement and society	• Costs and benefits of climate change for industry, settlement and society will vary widely by location and scale. In the aggregate however, net effects will tend to be increasingly negative, the larger the change in climate. • Poor communities can be especially vulnerable, in particular those concentrated in high-risk areas. They tend to have more limited adaptive capacities and are more dependent on climate-sensitive resources such as local water and food supplies.
Health	• Projected climate change-related exposures are likely to affect the health of millions of people, particularly those with low adaptive capacity. Particular causes include increases in malnutrition, increasing deaths due to floods, heat-waves, storms, fires and droughts; and altered spatial distribution of some infectious disease vectors. • Studies in temperate areas have shown that climate change is projected to bring some benefits, such as fewer deaths from cold exposure. Overall however, it is expected that these benefits will be outweighed by the negative health effects of rising temperatures worldwide, especially in developing countries.

1.3 Impact pathways

In order to value corporate environmental impacts, we need to understand how corporate emissions into the atmosphere affect humans. Therefore, we define impact pathways that describe the links between corporate activities, the environmental impacts from those activities, and the resultant societal outcomes. Our impact pathway framework consists of three elements:

1. **Impact drivers:**

- Definition: These drivers are expressed in units which can be measured at the corporate level, representing either an emission to air, land, or water, or the use of land or water resources[1].
- For GHGs: The type and quantity of GHG emissions resulting from different business activities.

2. **Environmental outcomes:**

- Definition: These describe actual changes in the environment which result from the impact driver (emission or resource use).
- For GHGs: Changes in environment as a result of climate change including sea level rise and increased incidence of extreme weather.

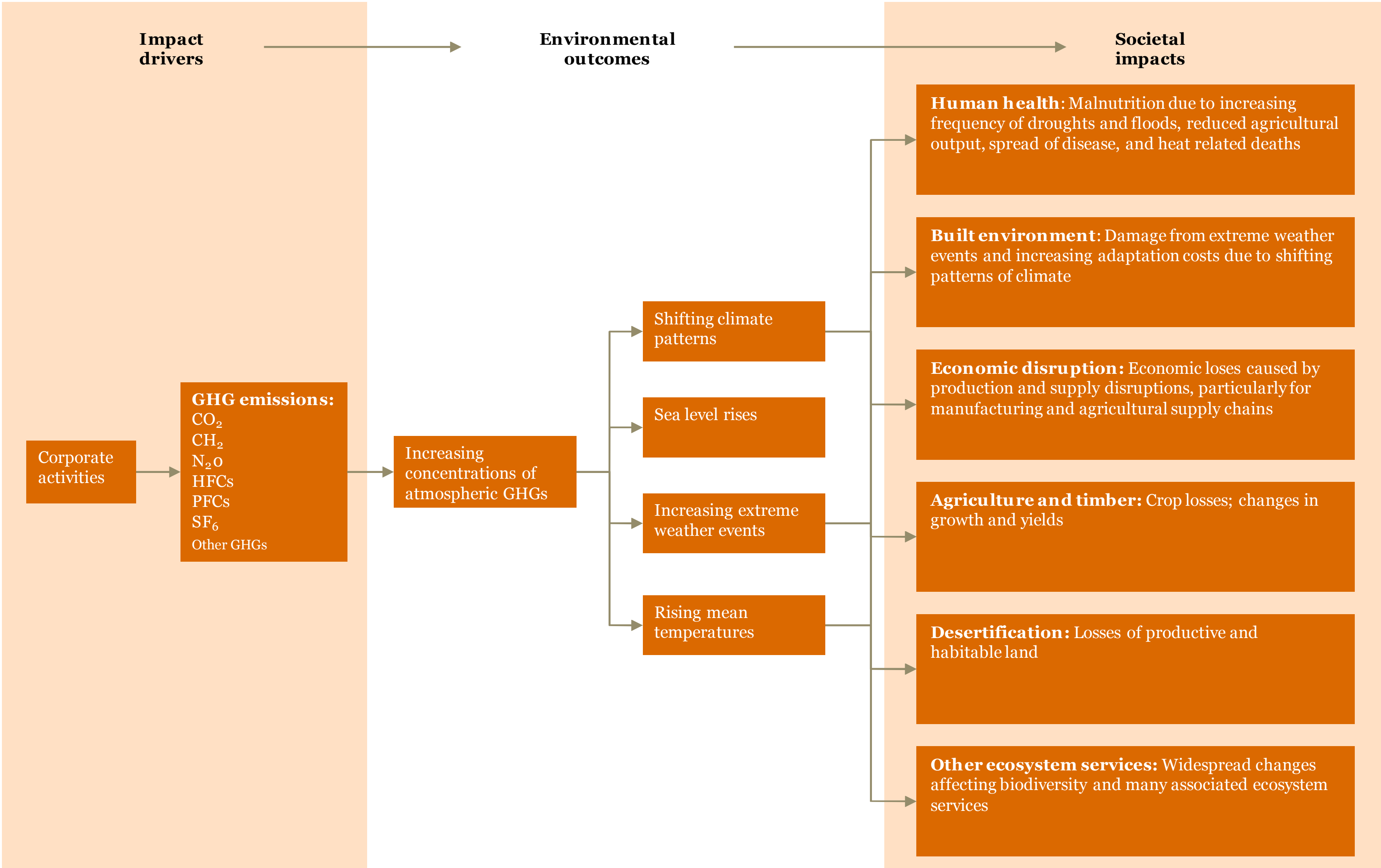
3. **Societal impacts:**

- Definition: These are the actual impacts on people as a result of changes in the environment (environmental outcomes).
- For GHGs: These may include disruption of business operations, changing patterns of disease, impacts on human health, agricultural production, or land and culture.

The three stages of the impact pathway are shown in Figure 2 overleaf. The label 'out of scope' identifies elements of the impact pathway which are not addressed in detail in our methodology. The reasons for any such limitations of scope are explained at the end of this chapter.

[1] A note on language: In this report, the measurement unit for any 'impact driver' is an 'environmental metric.' Therefore, GHG emissions are the impact driver, and tonnes of CO_2, CH_4, etc. are the environmental metrics.

Figure 2: Impact pathway for GHG emissions



2 Summary of methodology

2.1 Introduction

Our framework for estimating societal impacts of GHGs is structured around the impact pathway framework in Chapter 1. In aligning the two, we are able to demonstrate links between corporate activities, GHG emissions, and societal costs resulting from climate change. Our framework has three main steps:

1. **Obtain environmental metric data**: The starting point for each of our methodologies is data on emissions. These metric data are based on an understanding of the corporate activities which they result from. The data can come from a variety of sources, some of which (e.g., life cycle assessment (LCA) or environmentally extended input-output modelling (EEIO)) are subject to their own distinct methodologies[2].

Table 2: Environmental metric data for GHGs

Impact driver (emission or resource use)	**Environmental metric data**
Six primary GHGs: CO_2, CH_4, N_2O, SF_6, HCFs, PFCs (As well as any other GHGs deemed relevant)	Mass of emissions from corporate activities (tonnes[3])

2. **Quantify environmental outcomes**: We quantify physical changes in the environment resulting from corporate emissions or resource use (as measured by the metric data). This is discussed further in Table 3, column 2. For GHGs, we combine Steps 2 and 3 by using an estimate of the Societal Cost of Carbon[4] (SCC).

3. **Estimate societal impacts**: We estimate the societal cost (impact on people) resulting from environmental changes which in turn are the result of corporate activities. We use an estimate of the Societal Cost of Carbon (SCC) for this purpose.

2.2 Summary of methodology

2.2.1 Estimating GHG emissions

Specific methods for estimating GHG emissions are not the subject of this methodology which assumes adequate GHG emissions data as its starting point. Corporate GHG emissions can be estimated directly using information provided by companies, or indirectly through techniques such as LCA or EEIO analysis. Potential sources of emissions data are explored in somewhat greater detail in Chapter 3.

2.2.2 Estimating the societal cost of carbon

The core of the methodology revolves around identifying an appropriate estimate for the societal cost of carbon (i.e., the current and future economic damages from emission of a tonne of GHGs) to estimate the value of the current and future impacts of GHG emissions. Our approach is summarised in Table 3, overleaf.

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[2] The sources of metric data are outlined in Chapter 3. The assumed starting point for this methodology is the form specified in Table 2.

3 Care should be taken to ascertain whether these are metric tonnes (=1,000kg) or imperial tons (short ton = 2000 lb; long ton = 2240 lb)

[4] Often referred to in the literature as the 'Social' Cost of Carbon

Arriving at a primary estimate of the societal cost of carbon typically involves a number of complex steps (see appendix II for further detail): (1) selecting an emissions scenario (typically one of the IPCC scenarios) (2) constructing a climate model to project the likely future changes in climate (3) developing impact assessment models to quantify associated impacts on society (4) estimating the total economic costs associated with these impacts (5) discounting back the total cost estimate to the present-day using a social discount rate, and finally (6) apportioning the net present value of climate damages according to the volume of anthropogenic GHGs emitted. The result is an estimate of the societal cost of carbon (SCC) per tonne of CO_2 equivalent (tCO_2e).

To produce our estimate of the SCC, we chose to analyse the extensive academic literature which already exists on the subject. Alternative approaches would have involved either: a) undertaking a new climate modelling and valuation exercise from first principles or b) selecting an SCC estimate from a single study. We chose our meta-analytic approach in preference to the alternatives because, while the SCC has been subject to a significant amount of research by academics and government agencies - hence a novel study in the absence of new information would be of marginal benefit; there is not a single preferred approach – hence selecting a single study would be difficult to justify. Our approach is not a purely statistical meta-analysis (since it incorporates a number of non-statistical factors), but it shares some of the key benefits of a conventional statistical meta-analysis, particularly the ability to incorporate the results of multiple studies applying a range of different methods and scenarios. It also has the significant advantage that once a set of rules for selecting a sub-set of studies is defined, an automatic and un-biased mechanism to update the SCC estimate over time (as new research becomes available) is also established.

2.2.3 Summary tables

The following pages include two tables (Tables 3 and 4). Collectively, these provide an overview of our approach to valuing the societal impact of GHG emissions:

- Table 3 is a summary of the overall impact valuation methodology aligned with the impact pathway illustrated in Chapter 1
- Table 4 provides a summary of the workings behind our meta-analysis of existing SCC estimates (full details are included in the chapters that follow).

Table 3: Overview of our impact valuation methodology: estimating societal impacts from GHG emissions

Quantify environmental outcomes	Estimate societal impacts
Methods	• We quantify environmental outcomes and estimate societal costs of GHG emissions in one step by drawing on the population of existing estimates of the SCC from the extensive academic literature on the subject (see table 4 for details). - We select a sub-set of SCC estimates from the overall population based on criteria including the quality of the study, the age of the study and the discount rate used; - We then normalise our sub-set of estimates using a number of standard and transparent adjustments; - Finally we calculate estimates of the SCC from this normalised sub-set of estimates by identifying the arithmetic mean and median values. The choice of mean or median value is for the user but our default suggestion is to use the mean value. • A total societal cost estimate for corporate GHGs is reached by multiplying the tonnes of carbon dioxide equivalent (tCO_2e) associated with the corporate activity by the SCC.
Key variables	• The 12 key variables needed to derive our central estimate of the SCC are explained in Table 4.
Assumptions and justification	• A series of choices and assumptions underpin the methodology for estimating SCC via meta-analysis which are explained in Table 4. • We select the SCC as a better approximation of the impact on society from GHGs than the marginal abatement cost (MAC) or market prices. - MAC does not measure a company's impact on society, showing instead the cost to the company of reducing that impact at a point in time given prevailing technology. - Carbon market prices do not currently reflect the value of a company's impact on society as a result of GHG emissions. Instead, in the case of the European Union Emissions Trading Scheme (EU ETS) (for example), prices reflect the equilibrium in a relatively inflexible regulated market. As such, they give the current private cost of GHG emissions for regulated installations, but are widely accepted to be a poor proxy for the societal cost of those emissions.

Table 4: Summary of key methodological decisions and steps in our meta-analysis

Factor	Methodological choice in estimating SCC	Assumptions and justification
Selection of a restricted sub-set of SCC estimates		
Quality of study	Only estimates from peer reviewed studies will be used.	Peer review is the only widely accepted measure of quality applicable to studies of the societal cost of carbon. The significant and apparently systematic difference in values (peer reviewed values are typically lower) suggests that this is an important criterion.
Age of study	Only estimates from the ten most recently published peer-reviewed studies in our dataset are included.	Studies and estimates are generally perceived to have improved over time as both climate modelling and economic damage assessment methods have improved. We therefore deem it appropriate to focus on more recent estimates of the SCC, while maintaining a reasonable number of estimates to reflect the diversity of views about underlying assumptions. In order to do this, we use estimates from the ten most recently published peer-reviewed studies that conform to our methodology choices. While recognising that ten studies is a somewhat arbitrary figure, we note that choosing a study age criterion based on a number of studies has the additional benefit of providing a useful rule for future updates to the SCC based on newly published studies.
Discount rate	Only estimates that apply Pure Rate of Time Preference (PRTP) = 0% are included. We do not select SCC estimates according to the values they use for future economic growth rates and income elasticity of marginal utility.	A discount rate is used to convert future damage costs to their present value. In established economic theory (Ramsey F., 1928), the discount rate includes the Pure Rate of Time Preference (PRTP), a forecast of economic growth, and the marginal elasticity of utility with respect to income. We consider it ethically defensible and aligned with notions of inter-generational equity commonly found in the climate change literature to value the wellbeing of future generations equally to our own. It is not possible to select a subset of estimates that use specific values for income elasticity of marginal utility and economic growth rate because not all studies disclose this information. However, those that do disclose their assumptions show a sample average of 2.5%.
Treatment of outliers	Estimates more than three standard deviations from the mean are excluded.	Eliminating outliers helps to prevent extreme values from unduly distorting 'sample statistics'. However, the possibility of catastrophic climate outcomes (however remote) is generally accepted, and estimates of the SCC have been observed to follow a 'fat-tailed' distribution. We acknowledge the likelihood of this type of distribution by including estimates up to three standard deviations from the mean, but consider estimates outside this range to be true outliers and exclude them from our sample statistics on this basis.
Equity weighting	We do not select for SCC estimates according to the equity weighting used.	Equity weighting adjusts societal costs between different economic groups in underlying studies. No consensus exists on the appropriate method or degree of 'equity weighting' to use. We note that around 33% of our sub-set use some form of equity weighting and the overall effect on the results is modest.
Damage valuation approach	We do not select for SCC estimates according to the damage valuation approach used to derive the economic	Variation in underlying studies is relatively limited and there is no consensus on the preferred method.

cost of climate change.

	Methodological choice in estimating SCC	Assumptions and justification
Calculation of SCC from the restricted population of estimates		
Weighting of estimates	A multiple estimates weighting has been applied to values from studies which contain more than one estimate.	Studies with multiple estimates are weighted such that the sum of weightings for all estimates from a single study is 1. This is as applied by Tol (2011) and prevents individual studies containing large numbers of estimates crowding the sample and distorting the average SCC obtained towards the methods they employed. The technique also attempts to reflect the confidence placed by the author in each estimate.
Monetary inflation	Monetary inflation has been addressed by inflating each SCC estimate using World PPP-adjusted GDP deflators.	The value of a given monetary unit typically decreases over time as a result of monetary inflation. As the underlying studies relate to different years, the estimates need to be adjusted for monetary inflation to be comparable. Most studies explicitly or implicitly assume constant real exchange rates into the future. In practice real exchange rates have varied materially in the past twenty years; for this reason, World PPP adjusted GDP deflators are calculated for inflating older SCC estimates. Not all studies publish the year the SCC has been calculated for; therefore, the inflation rate from the 'rounded' year of publication of each study has been applied.
Growth rate of SCC over time	Growth rate of SCC assumed to be 3% per year.	Because the profile of anticipated climate damages is weighted into the future, and GHGs reside in the atmosphere for a limited period, the climate impact of an additional tonne of CO_2e rises over time. Three percent is the mid-point of the IPCC estimated range (2 – 4%) for this rate of increase.
Unit conversion	Conversion of \$/tC to \$/tCO_2e has been carried out by multiplying societal costs expressed in tC by the fraction 12/44.	Estimates of the SCC from the academic literature are typically expressed in: \$/tCe. We wish to present our results in the industry-standard units, \$tCO_2e. We therefore adjust for the difference in weight between a single atom of carbon (atomic mass = 12u) and a molecule of CO_2 (molecular mass = 44u).
Distribution of data	No fitted distributions are applied for the purpose of producing the SCC.	The sub-set of estimates selected (after applying the criteria set out above) does not clearly fit a specific distribution. We therefore consider it more transparent to use unfitted data to derive our averages.
Sample statistics	We present both the arithmetic mean and the median results of our meta-analysis and leave the choice of mean or median to the user. Our default suggestion is to use the mean.	There are valid statistical and ethical reasons for choosing either a mean or median value in this context. The mean takes more account of very high estimates derived from potentially catastrophic climate scenarios and therefore reflects a more precautionary approach to potential climate change impacts. The median, by contrast, is less affected by a few very high values and should therefore better reflect the consensus view, but takes limited account of catastrophic scenarios. Whichever value is chosen, the implications of using the other can be tested using sensitivity analysis.

3 Data requirements

3.1 Introduction

Gathering appropriate data is the starting point for valuing environmental impacts. The availability of high quality input data is a key determinant of the accuracy of impact quantification and valuation. Three types of data are required for the quantification and valuation of most environmental impacts. However, because the impact of GHGs does not vary based on where they are emitted, only two types of data are needed in the case of GHGs.

- **Environmental metric data**: Quantities of GHG released into the atmosphere as a result of corporate activities.
- **Contextual data**: Provides additional relevant information about the basic metric data. For example, describing the context in which pollutants are released (e.g. location, surrounding population density, local weather patterns). Due to the global nature of climate change, these data are not required for calculating the societal impact of GHGs.
- **Other data**: These are typically factors derived from the academic literature which are used to convert metric and contextual data into value estimates. For calculating the SCC using our method the relevant information is that which describes the treatment of the key variables in our meta-analysis. This information can be obtained from the original studies from which SCC estimates are selected.

While methods for the collection or estimation of basic metric data are not the subject of this paper, the data generation methods used are nonetheless relevant. This chapter describes the most likely sources of metric data across a typical corporate value chain and other key data needed.

3.2 Environmental metric data

In the case of GHG emissions, the primary units of measurement are metric tonnes of carbon dioxide equivalent (tCO_2e). These are theoretical units which express the global warming potential (GWP) of GHGs relative to the most prevalent greenhouse gas: CO_2 (see Chapter 1 for list of principal GHGs). This conversion allows for their impacts to be compared consistently.

For some companies, CO_2 will be the only noteworthy GHG that is emitted from their direct operations. However, it is likely that the emission of other GHGs will be associated with other stages along the corporate value chain. Where companies do not already collect or report these emissions, they can be estimated with reference to Life Cycle Inventories (LCI) or Environmentally-Extended Input-Output (EEIO) modelling as well as other methods and sources. Once the quantities of these gases have been obtained or estimated, they should be converted into tCO_2e using the conversion factors provided in the IPCC's fifth assessment report (IPCC, 2013, p714 & p731-737).

Table 5: Global warming potentials (GWPs) for selected GHGs (source: IPCC, 2013)

Greenhouse gas	Global warming potential (100 year, including climate-carbon feedbacks)
CO_2	1
CH_4	34
N_2O	298
HFC-134a	1,550

Likely data availability

Table 6 summarises the type of information that is likely to be available at each stage of a company's value chain.

Table 6: Likely data availability across a corporate value chain for GHG emissions

Value chain stage	Environmental metric data
Raw materials	Tonnages may be available directly from suppliers. Alternatively LCI or EEIO modelling may be used to derive estimated volumes.
Raw material processing	
Manufacture and assembly	GHG emissions may well be available from the company's own management information system; either as tonnes of gas emitted, or with the use of conversion factors based on fuel use or other industrial activities.
Distribution and sales	
Use	GHG estimates may be estimated with the aid of surveys of customer use, collected directly from products for certain electronic goods, or else can be informed by LCI databases or other existing market studies.
Disposal/re-use	

4 Detailed methodology: Selecting our approach to valuing GHG emissions

In this chapter, we briefly explain our decision to use the societal cost of carbon as a measure of the impact associated with GHG emissions (versus other approaches like market price or marginal abatement cost). Appendix I explains the alternative measures and the rationale for our choice in more detail. The mechanics of our calculations are covered in Chapter 5.

4.1 Selecting the appropriate measure of the cost of carbon

There are three main ways in which values are commonly placed on greenhouse gas emissions:

- The societal cost of carbon (SCC) approach: This approach values carbon by taking into account the costs of the impacts of climate change which are associated with it.
- The market price approach: Values can be derived from market prices induced by governments through carbon taxes or cap-and-trade systems.
- The marginal abatement cost (MAC) approach: The costs of greenhouse gas emissions are based on the cost of achieving a certain level of GHG emissions reductions.

Applying a societal cost of carbon is the only approach consistent with the wider conceptual framework for the E P&L which is based on the theory of welfare economics and seeks to measure changes in human well-being associated with corporate environmental impacts. However, the alternative approaches are used by other organisations and in various contexts to place values on GHG emissions and/or their abatement. Table 7 summarises these approaches.

Table 7: Summary of different approaches to valuing carbon

Method	What does it represent?	How does this differ from the societal cost?
Societal Cost of Carbon	The total cost to society resulting from the environmental impacts of GHG emissions.	N/A
Market price approach	**Carbon market price**: The equilibrium point where supply of permits (set by the regulator) meets demand from eligible companies. In theory, the quantity of emissions permits could be capped at a level which leads to full internalisation of GHG externalities. **Carbon tax price**: The amount payable as a tax set by government, either directly attributed to a tonne of CO_2e or indirectly via taxes on GHG intensive products (e.g. fuel or energy) or activities. In theory, this could be set at the level that fully represents the societal	**Carbon market price**: The amount payable for GHG emissions under a cap and trade scheme is a function of the supply and demand for credits within that scheme and bears little practical relation to the impacts associated with emissions. There are a range of different market prices for carbon because the global market is highly fragmented, the geographical coverage of the various national and regional schemes is incomplete, and the schemes themselves follow markedly different rules. The equilibrium prices observed in the various major carbon markets tend to be significantly below consensus estimates of the SCC. **Carbon tax price**: This is a measure of the amount payable for GHGs rather than the impacts associated with GHGs. For political and practical reasons, the level set by governments tends to be below consensus

Method	What does it represent?	How does this differ from the societal cost?
	cost of carbon (a so-called 'Pigouvian' tax).	estimates of the SCC.
Marginal Abatement Cost	The financial cost to reduce emissions by a given amount, at a given point in time, typically by investing in abatement technology. In theory, to maximise welfare we should seek to abate emissions from all activities up to the point at which the MAC is equal to the SCC.	The MAC indicates the financial cost to reduce emissions by a given amount, rather than the benefit to society of that emissions reduction (as a corollary to the cost of increasing emissions). The MAC faced by each industrial sector and indeed each company within those sectors is likely to vary. Companies that have made little effort to reduce emissions may initially face negative abatement costs (net savings from implementing abatement options), whereas companies who have exhausted all common abatement options may face abatement costs which exceed the SCC.

4.2 Primary estimation versus meta-analysis

There are three broad means to obtain an estimate of the societal cost of carbon:

- Primary estimation: Deriving estimates of the total cost of climate change using a multi-step modelling process which assesses the biophysical impacts of GHG emissions, quantifies the likely consequences in economic and welfare terms and then discounts these into present value terms and apportions them over relevant emissions (see appendix II for further details).
- Adopting a single SCC estimate from a single existing study.
- Meta-analysis: deriving an estimate of the SCC by analysing many existing primary estimates.

We chose to follow a broadly meta-analytic approach in preference to the alternatives for two primary reasons:

1) The SCC has been and continues to be subject to a significant amount of research by academics and government agencies - hence undertaking a new primary estimation exercise in the absence of new information would likely be of marginal benefit.
2) In spite of the significant work done in this area there is not a high degree of consensus on the preferred approach to several of the key steps and underlying assumptions required to calculate the SCC – hence selecting an SCC based on a single study would be difficult to justify.

As noted in chapter 2, our approach is not a pure statistical meta-analysis (since it incorporates a number of non-statistical factors), but it shares some of the key benefits of a conventional meta-analysis, particularly the ability to incorporate the results of multiple studies which apply a range of different methods, assumptions and scenarios. It also has the significant advantage that once a set of rules for selecting a sub-set of studies is defined, an automatic mechanism to update the SCC estimate over time (as new research becomes available) is also established.

Considering 'sample' size

As already noted, our analysis relies on a large overall population of SCC estimates identified in the climate change literature. However, we use a number of objective and subjective criteria to select a sub-set of these estimates from which to derive our own estimates. The application of these criteria reduces the number of estimates from which we actually calculate our 'sample' statistics to a sub-set of 33.

It is worth noting therefore that the sub-set of estimates that we identify is not intended to infer some properties of the overall population of estimates from which it is drawn (e.g. the mean or the median of the population, which we can calculate from the full population if desired), and hence it is not a 'sample' in the traditional statistical sense. Rather, the criteria we apply are primarily designed to identify the highest quality or most reliable estimates from the overall population (e.g. favouring peer reviewed studies and more recent studies and eliminating outliers). For this reason, the relatively small number of estimates remaining is not a particular cause for concern.

However, a key benefit of our approach (relative to producing a new primary estimate, or selecting a single estimate from a single study), is that it can reflect a divergence of views on underlying methods, assumptions and scenarios, where there is still much debate and limited consensus. For this reason, retaining a reasonable number of estimates remains desirable. We acknowledge this potential trade off at various points in this paper, and limit our selection criteria to those we consider most fundamental, to ensure that we retain a diversity of estimates (and therefore academic opinions).

With a relatively small number of estimates, the robustness of 'sample' statistics - particularly the median - to small changes in the number of estimates included (e.g. when updating our estimates based on new studies) may be a separate cause for concern. This and the related issue of the distribution of SCC estimates is discussed further in appendix III.

5 Detailed methodology: Executing a meta-analysis of SCC estimates

In this chapter we describe our methodology for calculating the SCC through a meta-analysis of existing estimates. Table 8 gives a simple overview of the methodology.

Table 8: Summary of methodology (from chapter 2)

Quantify environmental outcomes	Estimate societal impacts
Methods	• We quantify environmental outcomes and estimate societal costs of GHG emissions in one step by drawing on the population of existing estimates of the SCC from the extensive academic literature on the subject. - We select a sub-set of SCC estimates from the overall population based on criteria including the quality of the study, the age of the study and the discount rate used; - We then normalise our sub-set of estimates using a number of standard and transparent adjustments; - Finally we calculate estimates of the SCC from this normalised sub-set of estimates by identifying the arithmetic mean and median values. The choice of mean or median value is for the user but our default suggestion is to use the mean value. • A total societal cost estimate for corporate GHGs is reached by multiplying the tonnes of carbon dioxide equivalent (tCO_2e) associated with the corporate activity by the SCC.

5.1 Selecting a sub-set of SCC estimates from the overall population

This section discusses the basis on which SCC estimates are included in the calculation of our central estimate. Figure 3 shows the criteria by which certain estimates are excluded and the order in which these are applied.

There has been a significant amount of discussion (often unresolved) about many of these criteria in the literature so, along with our chosen approach, each sub-section includes context about the basic theory and alternative approaches if relevant.

Figure 3: Steps for selecting a sub-set of SCC estimates

1. **Quality of study:** Exclude non-peer reviewed studies
2. **Discount rate:** Restrict to estimates with PRTP of 0%
3. **Age of study:** Restrict to estimates from the 10 most recent studies
4. **Treatment of outliers:** Remove SCC estimates more than three standard deviations from the mean

5.1.1 Step 1: Quality of study

Context

Studies that have been peer-reviewed produce, on average, lower estimates of the SCC than those which have not been, with less variation between them (Tol, 2005, 2008, 2011).

Approach taken

Only estimates from studies that have been peer-reviewed in academic journals are selected to calculate the average SCC on the basis that peer review is the only widely accepted measure of quality applicable to studies of this nature. The significant and apparently systematic difference in values suggests that this is an important criterion.

Alternative approaches

While we believe that the decision to use only peer reviewed estimates is well justified on the grounds of quality, we could have analysed all published estimates. One potential problem could be that selecting only peer reviewed studies may emphasise any bias inherent to the academic community, but we consider that this risk is relatively small due to high academic standards and the apparent diversity of views amongst academics in the field, and that this risk is outweighed by the level of quality assurance provided by peer review.

5.1.2 Step 2: Discount rate

Context

"Global climate change unfolds over a time scale of centuries and, through the power of compound interest, what to do now is hugely sensitive to the discount rate that is postulated" Martin Weitzman, 2007

In order to aggregate costs and benefits accruing over time and across generations, as is the case with climate change, we must scale our results using a factor which describes relative societal preferences for consumption at different points in time. Economists and governments tend to use a societal discount rate (SDR) rather than a market one, in order to reflect society as a whole and to account for the existence of market imperfections. The broad academic consensus is to use a rate defined by the Ramsey model (Ramsey, 1928) which is:

Equation 1: The Ramsey model for deriving a social discount rate

$$s = \rho + \mu g$$

Where s is the societal discount rate (SDR)[5], ρ is the pure rate of time preference (PRTP), μ is the elasticity of marginal utility with respect to income, and g is the future economic growth rate.

Pure Rate of Time Preference (PRTP)

The societal discount rate discounts the future for two reasons. The first, embodied in the PRTP, says that we prefer a given amount of money now rather than later. This may be due to some level of impatience inherent in human nature; or due to other more fundamental considerations like the risk of the human race not existing (and therefore we may as well enjoy ourselves now rather than save for the future).

Regarding intergenerational issues, assigning any number greater than zero carries the implication that benefits accruing to future generations are inherently less valuable than those accruing to ourselves. For this reason Ramsey (1928) argued that discounting '*later enjoyments in comparison to earlier ones [...] is ethically indefensible*'. Such a view is representative of the *perceptive* approach, which argues that the value of the PRTP is an ethical choice which should be chosen on a societal basis, for example through some democratic process or on the basis of equality between people in different generations, all else being equal (Bicket, 2011).

In contrast, the SDR advocated by UK Government for policy and project appraisal includes a PRTP of 1.5%, based on studies of past long-term returns received by savers in the UK (HM Treasury, 2003). Such an approach to PRTP is consistent with the *descriptive* approach, which argues that PRTP should be determined empirically, often from the observed behaviour of savers. The decision to save money rather than spend it can be expected wherever the real interest earned is sufficient to offset savers' impatience to spend it[6]. Another alternative, used by the EU (2008), is to base the PRTP on the risk of dying, on the basis that death of oneself or of future generations is a compelling reason to prefer present over future consumption.

The academic and philosophical debate over which rate to use continues and, as such, is unresolved (see, for example, Weitzman, 2007).

The second reason the SDR discounts the future is to take account of how much better off future generations are expected to be, relative to present generations. This has two components which are explained below.

Future economic growth

The first component is expected long run economic growth. This factor is simply the expected change in GDP over time and so lacks a moral dimension, although it does differ across countries (and between forecasters).

The elasticity of marginal utility with respect to income

The second component is the elasticity of marginal utility with respect to income. The marginal utility of income is the amount of utility obtained from gaining one extra unit of income; the elasticity of marginal utility with respect to income is the percentage change in marginal utility associated with a one percent change in income (hereafter 'income elasticity'). This describes the rate at which the utility gained from an additional unit of income changes with the level of income.

Marginal utility tends to fall as income increases, so this factor gives more weight to benefits and costs accruing to people with lower incomes. Academic consensus puts the value of this parameter between 1 and 4 (that is, as income rises, marginal utility falls at a faster rate) (Weitzman, 2007). This also accords with common

[5] Also known as the consumption discount rate.

[6] This is holding consumption constant; consumption depends on income, as does the utility it brings.

perceptions of fairness, with the same amount of money being 'worth' more to a poorer person than to a richer person.

In principle, because we are applying a social discount rate to the valuation of a global externality, each country's growth rate and income elasticity should be considered individually. However, for practical reasons a simplifying assumption to apply a single SDR is generally made.

Considering all three parameters

The PRTP says how much we inherently discount income received in the future. The marginal elasticity of utility with respect to income says how much we value additional income depending on the level of income we have. Economic growth forecasts indicate how much income we expect to have in total. Combining these three parameters in the SDR, as per Equation 1, tells us how society values present consumption relative to future consumption.

Approach taken

We have restricted our population of SCC estimates to those where the Pure Rate of Time Preference (PRTP) = 0%. We consider 0% to be the least arbitrary choice because it represents the ethical viewpoint that future generations should be treated the same as our own, given the same level of wealth. The choice of PRTP is a question of ethics rather than one of uncertainty over physical outcomes and it is difficult to objectively critique the choice of any discount rate.

Comparing assumptions

The methodology presented in this paper involves averaging SCC estimates across a number of academic studies, some of which do not explicitly disclose the social discount rate that they apply.

Of the studies in the sub-set that we use that do disclose their societal discount rates, the average is 2.0%. This is slightly lower than conventional (usually intra-generational rather than inter-generational) discount rates used in policy analyses. For example, the UK Government uses an SDR of 3.5% (declining to 1% in the very long-term) while the EU (2008) suggests an SDR of between 3.5 and 5.5%.

Given that we have restricted our sub-set of SCC estimates to those where PRTP = 0%, this suggests that the average value for the product of economic growth and income elasticity, is also 2.0%. We can also compare this with equivalent values used in public policy: the UK Government uses 2% (HM Treasury, 2003) while the EU uses 2.5-4.5% (EU, 2008)[7].

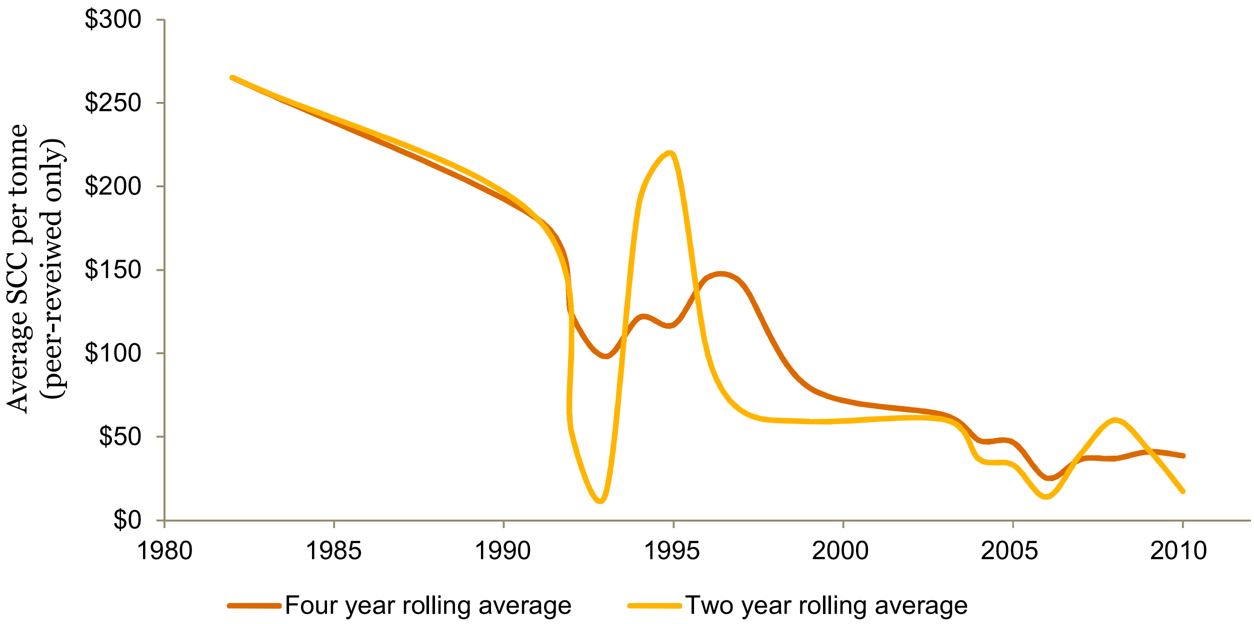
5.1.3 Step 3: Age of study

Context

Estimates of the SCC have, on average, decreased over time (Figure 4). The earliest estimate of the SCC in our total population of 311 estimates dates from 1982. Between 1991 and 2010, there was at least one study estimating the SCC published every year. Estimates varied significantly in earlier years. However, in the last 15 years, average estimates of the SCC have stabilised somewhat, although variation still exists. This trend was identified by Tol (2011), who suggested that the downward trend in the SCC reflects progress in our understanding of the impacts of climate change.

[7] The European Commission advises that a SDR of 5.5% should be used for EU countries eligible for the Cohesion Fund (fast-growing eastern European) and 3.5% for all other 'mature' EU countries (EU, 2008). The same guidance uses observed mortality-based statistics (equating to a 'death rate') as a proxy for PRTP, giving an estimate of 1%. Subtracting this 1% from the SDR leaves a range of 2.5-4.5% for the product of CDR and income elasticity.

Figure 4: Four year and two year rolling average estimates of the SCC (peer-reviewed studies only) in US dollars



Approach taken

To calculate the average SCC, we use only estimates from the 10 most recent studies that meet our other criteria (e.g., peer reviewed and using a PRTP = 0%). This approach is considered appropriate due to the observed stabilisation of estimates over time, which is speculated to be the result of improvements in estimation techniques. Using ten studies should ensure that our average SCC is calculated from a sufficient number of estimates (each study typically includes a number of estimates reflecting differing parameterisation of the underlying models) to reflect the diversity of views about underlying assumptions. The estimate can be updated as newer studies are published in future which ensures that our SCC reflects prevailing thinking, but is not unduly skewed by a single study.

Alternative approaches

Alternatives considered included:

- Defining a cut-off date (e.g. 1990 or 2000) and excluding studies published before this date. However, no specific and widely recognised breakthrough in estimating the SCC has occurred over time, and there are no time ranges that are statistically significantly different from any others due to the large degree of variation within temporal sub-samples. Furthermore, as the pool of studies included increased over time, the relative impact of new (and presumably improved) studies would tend to reduce, while the importance of inflation related adjustments (themselves generalised and approximate) to older estimates would continue to increase.
- Defining a period of time prior to the date of the new estimate (e.g. the past 10 years) and including only studies published within this period. However, this approach relies on the continued regular publishing of new peer reviewed studies on the SCC to maintain a sufficient sample size for future estimates. If the science and economics continues to converge towards a consensus estimate, then the publishing of novel studies may well become less frequent.

In conclusion, while the choice to use the ten most recent studies is itself arbitrary, it is an effective way to avoid both of the pitfalls described above, whilst still recognising the apparently increasing in quality of newer studies.

5.1.4 Step 4: Treatment of outliers

Context

The total population of 311 SCC estimates ranges in US dollar terms from positive four-figure values to negative values. Even when our sub-set is restricted to 34 estimates by applying the selection criteria described above (i.e. limiting the population to those from the ten most recent peer-reviewed studies with a PRTP = 0%), the estimates range between $6 and $622/t$CO_2$ (in 2012 US dollars). Given this large range, the fact that only three of the 34 estimates are more than one standard deviation from the mean implies the existence of outliers. However, the possibility of catastrophic climate outcomes (however remote) is generally accepted, and estimates of the SCC have been observed to follow a 'fat-tailed' distribution.

Including genuine outliers would tend to overstate the average SCC calculated. Whereas, excluding extreme but nonetheless valid estimates would tend to understate the risks of catastrophic climate outcomes.

Approach taken

We acknowledge the likelihood of a fat-tailed distribution by including estimates up to three standard deviations from the mean (whereas a common statistical 'rule of thumb' is two standard deviations), but consider estimates outside this range to be genuine outliers and exclude them from our sample statistics on this basis.

In this instance a single value falls more than three standard deviations from the mean. Grubbs' test[8] also identifies this value as an outlier at the 99% significance level.

5.1.5 Equity weighting

Context

Another variable in SCC estimation methodologies is the treatment of costs and benefits accruing across countries. Just as we discount future generations' income because they are likely to be wealthier than us, we may also discount developed country incomes relative to those of developing countries. In the context of climate change this is sometimes called 'equity weighting' and would attach greater weight to, say, $10m worth of flood damage in Bangladesh than $10m worth of flood damage in Germany.

Approach taken

We do not restrict our sub-set of estimates based on their approach to 'equity weighting' for the largely pragmatic reason that to do so would reduce the number of studies in our sub-set to a handful. We therefore include in our sub-set both SCC estimates that apply equity weighting and those that do not.

We note that our final estimate therefore implicitly includes some level of equity weighting; and that the impact on our sample statistics of including or excluding equity weighting is modest (see sensitivity analysis in chapter 6).

Alternative approaches

We could have opted to exclude estimates that apply some form of equity weighting on the basis that it is often applied in a somewhat arbitrary fashion, and that there appears little consensus on how it should be applied. Or we could have opted only to consider estimates which do apply equity weighting, on the basis that perceived inequities in the distribution of climate impacts are a significant issue in the literature.

We note that equity weighting typically increases derived SCC estimates. For example, Tol (2005) reported a mean SCC of $90/tC for estimates with no equity weights, and a mean SCC of $101/tC with equity weighting.

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[8] Grubbs' test (Grubbs 1969 and Stefansky 1972) is used to detect a single outlier in a univariate data set that follows an approximately normal distribution.

5.1.6 Damage valuation approach

Context

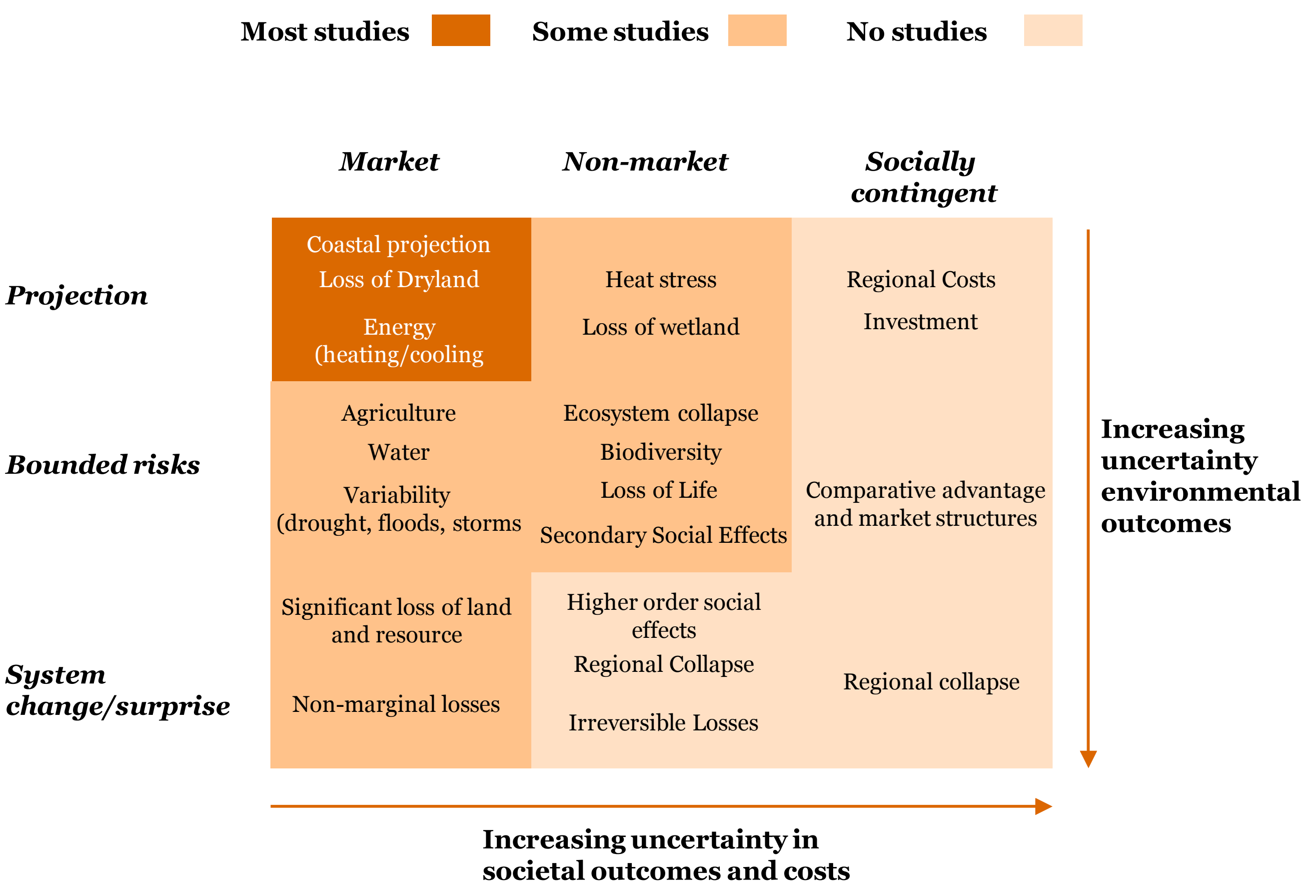
"[Estimates of the total cost of climate change]…tend to ignore interactions between different impacts, and neglect higher order effects on the economy and population; they rely on extrapolation from a few detailed case studies; they often impose a changing climate on a static society; they use simplistic models of adaptation to climate change; they often ignore uncertainties; and they use controversial valuation methods and benefit transfers." Tol (2008)

The physical and economic damages resulting from climate change (the environmental outcomes resulting from GHG emissions) are difficult to predict and value. Estimates vary for a range of reasons driven by inherent uncertainty and different theoretical approaches.

However, by focusing on more tangible areas such as market impacts and marginal changes, rather than attempting to predict widespread social change (such as migration or war) and irreversible impacts (such as the melting of permafrost or changes in oceanic circulation), studies are able to give a comparable range of estimates. Indeed, the uncertainty over such damages is actually less than that which surrounds the discounting of these damages into present value terms or the appropriate weights to attach to poorer countries. As Hope and Newbury (2006) argue: *"once economists and climate scientists agree on how to treat the future and aggregate regional impacts, the disagreement almost entirely disappears'"*.

Figure 5 outlines the types of damage estimates that are included and excluded in the existing SCC literature. As we go from left-to-right on the horizontal axis, the uncertainty in valuation increases and, as we go from top-to-bottom on the vertical axis, the uncertainty in forecasting the physical effects of climate change increases.

Figure 5: Societal Cost of Carbon Risk Matrix



Source: Adapted from Watkiss & Downing (2008)

Some academics have argued that the tendency of studies to exclude systemic and higher order impacts as well as socially contingent outcomes will tend to underestimate the true impacts of climate change. However, others argue that studies tend to underestimate the ability of societies and economies to adapt to climate change, particularly at a macro level, by overlaying the physical consequences of climate change on future settlements that are (in most cases) linearly extrapolated from today's.

Approach taken

We do not select for SCC estimates according to the damage valuation approach used to derive the economic cost of climate change.

As discussed in appendix II, the 300+ SCC estimates used in the meta-analysis originate from nine underlying estimates of the total damages caused by climate change, each of which uses slightly different methods to derive the total damage (or societal cost). Effectively therefore our SCC estimate reflects the average of a number of these underlying estimates, and is driven by each to a greater or lesser extent depending on its popularity with the authors of subsequent studies.

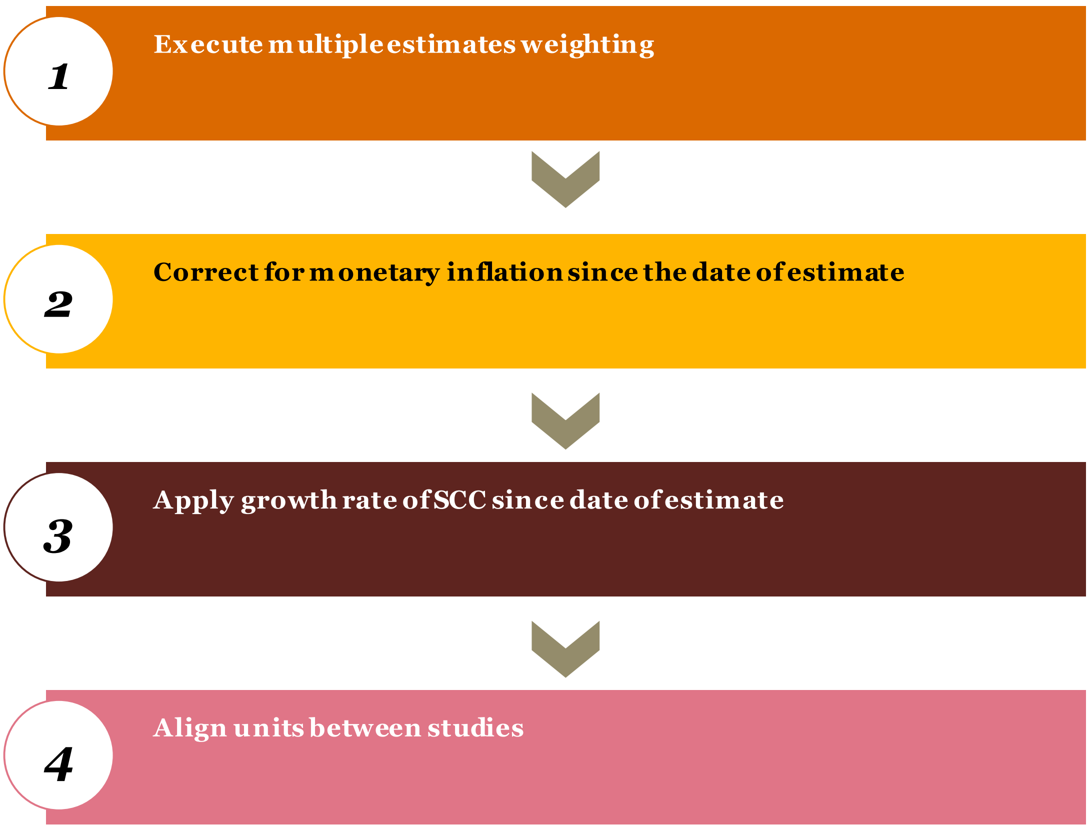
Alternative approaches

An alternative approach to using an average of the different methods for estimating damage cost would be to choose one approach only. We believe that selecting a single method would be arbitrary and difficult to justify, based on the current understanding of climate risks and their impacts on the global economy.

5.2 Normalizing the sub-set of SCC estimates

Once we have selected our sub-set of estimates according to the principles above, we apply a series of normalisations with the objective that the remaining estimates are as comparable as possible. These adjustments ensure that a single source is not over-represented, account for monetary inflation and the growth of the SCC over time and align units of measurement. Figure 6 shows the steps for normalization.

Figure 6: Steps for normalising our sub-set of SCC estimates



5.2.1 Step 1: Weighting of estimates

Context

A potential problem with a population of estimates which includes numerous estimates from some individual studies is that those studies' methodologies would exert a disproportionate influence on sample statistics. A simple approach would be to weight multiple estimates from a single study equally. However, estimates included in the population include both 'central estimates', in which the author has the most confidence, and estimates made as part of a sensitivity analysis to demonstrate the effect of changes in the underlying assumptions.

Approach taken

A 'multiple estimates weighting' consistent with Tol (2011) has been applied to values from studies which contain more than one estimate. This is applied such that the sum of the weightings of all estimates in a single paper is equal to one.

Individual weightings reflect whether an estimate is one of several others from the same study and, if so, the confidence the author has placed in the particular estimate. This introduces some subjectivity since interpreting the importance placed by an author in a particular estimate and representing this in a numerical weighting requires judgment. However, there is no clear alternative which would effectively account for multiple estimates from individual studies.

Alternative approaches

Ostensibly, less subjectivity would be introduced if all estimates from a single study were weighted equally. However, if study authors have proposed one or a few values as their central estimates, and have calculated others only to demonstrate the effect of changes in the underlying assumptions, then it seems inappropriate to consider all to have equal weight.

5.2.2 Step 2: Monetary inflation

Context

SCC estimates are calculated to apply to emissions at a point in time. However, the nominal value of money decreases over time as a result of monetary inflation. So for example, a 1995 US dollar would purchase more goods or services than a 2012 US dollar. As the underlying studies from which the SCC estimates are taken were published in different years, the SCC estimates must be converted into present-day dollars before estimating sample statistics.

While adjusting for monetary inflation is typically straightforward, complications arise when adjusting SCC estimates. Climate change damage estimates are aggregated estimates of impacts occurring all around the world which are calculated for a certain point in the future and in a specific currency (typically US dollars), using the prevailing exchange rate at the date of the study. The damage values therefore (implicitly or explicitly) assume constant real exchange rates (i.e. constant Purchasing Power Parity (PPP)) when costing future impacts around the world.

Studies that then go on to estimate the SCC today, discount these total future climate change damage values to approximately the date of the study using a single social discount rate. As a result, these studies also incorporate the assumption of constant PPP held across all countries from the date of the study into the future.

If this assumption were correct, the inflation rate of any country could be used to inflate the SCC estimates from the date of the study to the present-day (converting from US dollars to the chosen currency in the study year and back to US dollars in the target year) with consistent results. However, as shown in

Figure 7 and Figure 8, over the period in question this assumption did not hold. While world monetary inflation averaged almost three times US inflation through the period, the nominal exchange rate between the US dollar and a trade weighted basket of world currencies was almost identical at the beginning and end of the period (1995 – 2011), meaning that the real exchange rate US dollars and other world currencies fell significantly during the period. As a result, the choice of which country's inflation rate to use is significant.

Figure 7: US versus global monetary inflation

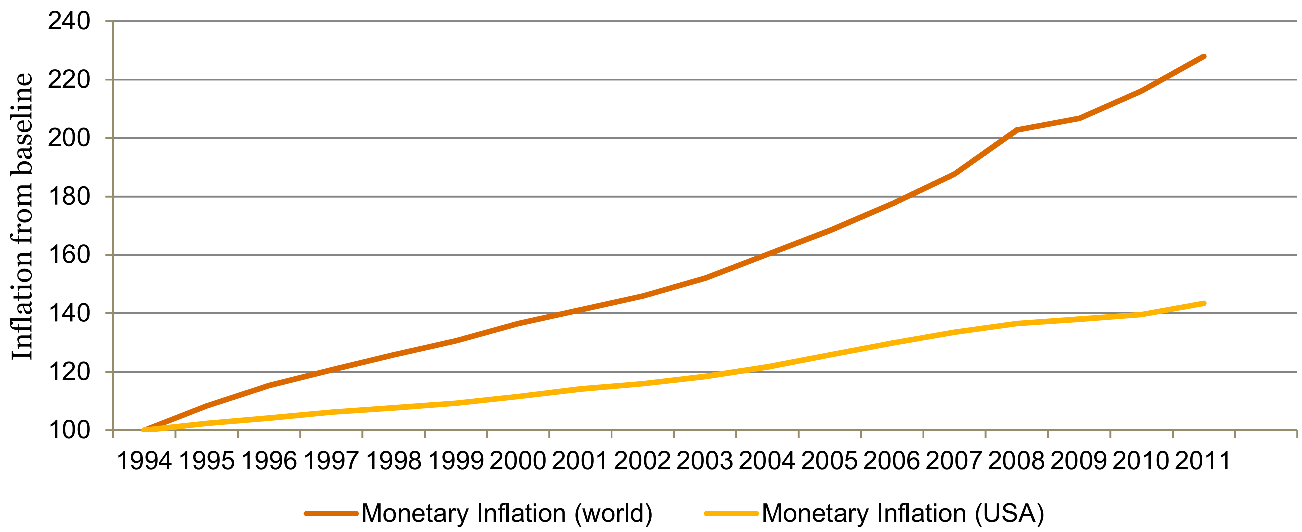
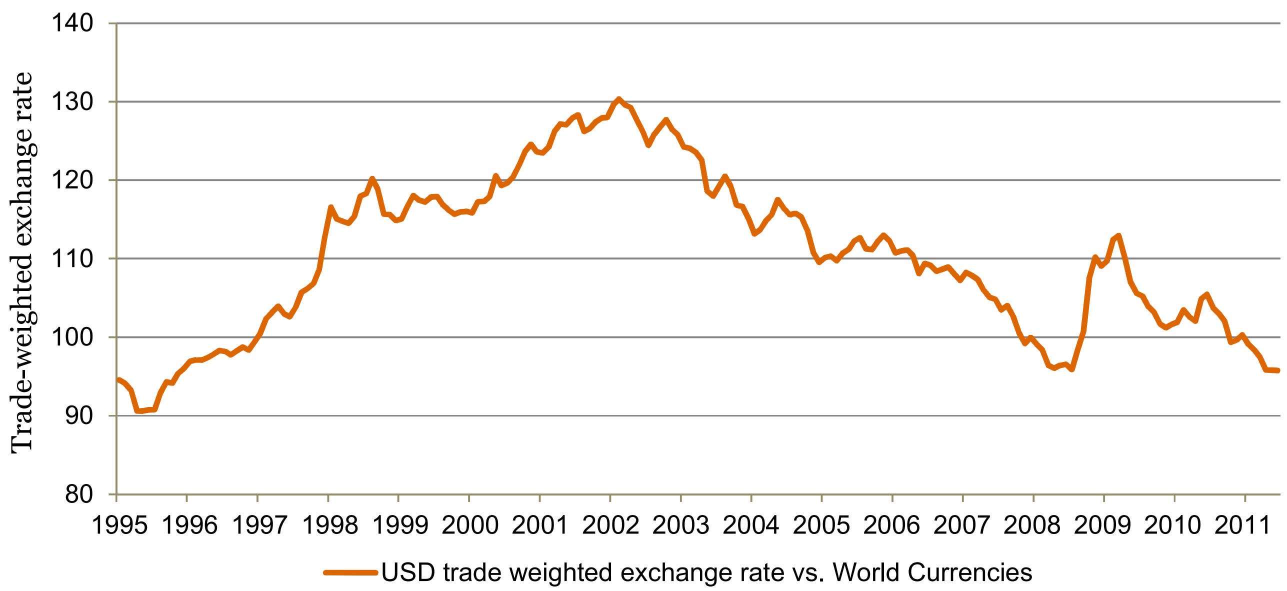


Figure 8: USD trade weighted exchange rate versus world currencies



Approach taken

In order to inflate nominal SCC estimates from a study from a certain date in the past into present-day US dollars, we adjust global GDP deflators (updated annually by the World Bank and OECD) for changes in PPP over the period in question. We calculate this series by multiplying the PPP ratio[9] for each country by their nominal GDP (expressed in US dollars and converted from local currency at market exchange rates) and summing the results. The rate of growth in this series from one year to the next provides the World PPP-adjusted GDP deflator in US dollars.

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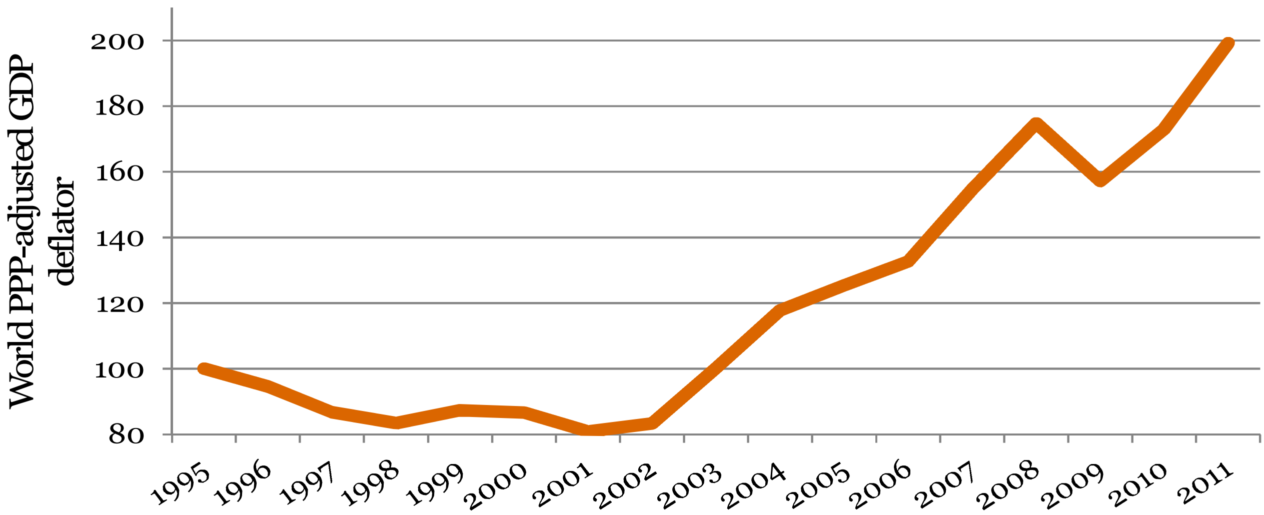
[9] The PPP ratio for a country accounts for changes in prices, changes in real GDP, and exchange rate movements, and is calculated by dividing the PPP conversion factor by the market exchange rate.

Figure 9 shows the World PPP-adjusted GDP deflator for the period 1995 (the date of the earliest study in our sub-set) to the present day.

Figure 9: World PPP-adjusted GDP deflator over time



A final consideration is that not all studies disclose the year for which the SCC estimate(s) are calculated. For those that do, the nominal year is typically within the five years previous to the ultimate publication date. We consider that the best approximation is therefore to round the study down to the nearest five year interval and inflate the estimates using the World PPP-adjusted GDP deflators from this point[10]. This is the same approach used to inflate for growth in the SCC.

5.2.3 Step 3: Growth rate of the SCC over time

Context

SCC estimates are provided for a given year. Because the profile of anticipated climate damages is weighted into the future, and GHGs reside in the atmosphere for a limited period, the expected climate impact of an additional tonne of CO_2e rises over time. Thus, the real SCC rises over time[11]. The IPCC suggests a range of 2 – 4% per year for this growth rate (IPCC, 2007; Chapter 20).

Approach taken

Selected estimates of the SCC are increased by 3% per year (the midpoint of the IPCC estimate) to reflect the greater impact of later emissions (converting them to present day estimates). Consistent with our approach to monetary inflation, estimates are increased from the nearest five year interval before the publication date.

Alternative approaches

The choice of the growth rate of damages applied affects SCC estimates. Choosing a different growth rate (from within the IPCC's range of 2-4%) would result in a change for the SCC central estimates. The change would be around plus $9 or minus $8 from the central estimate mean of $78, or of about plus or minus $4 around the central estimate median.

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[10] For example, if a study was published in 2002, we assume that the SCC estimates relate to the year 2000 and inflate for SCC growth to 2012 units from there.

[11] The use of the phrase 'real' SCC denotes that this growth is additional to price inflation.

5.2.4 Step 4: Unit conversion

Context

The SCC estimates compiled in Tol's (2011) database are expressed in units of $/tCe. We wish to present our results in the industry-standard units: $\$tCO_2e$, which also allows for easy comparison between greenhouse gases.

Approach taken

We adjust for the difference in weight between a single atom of carbon (12u) and a molecule of carbon dioxide (44u) by multiplying SCC estimates expressed in units of $/tC by the fraction $^{12}/_{44}$.

5.3 Calculating the SCC from the normalized sub-set

Once the sub-set of SCC estimates has been selected and normalised (as outlined above) we calculate both the median and the arithmetic mean.

There are valid statistical and ethical reasons for choosing either a mean or median value in this context. The mean takes more account of very high estimates derived from potentially catastrophic climate scenarios and therefore reflects a more precautionary approach to potential climate change impacts. The median, by contrast, is less affected by a few very high values and should therefore better reflect the consensus view, but takes little account of catastrophic scenarios. For this reason we leave the choice of mean or median value to the user. Our default suggestion is to use the mean.

Whichever value is chosen, the implications of using the other can be tested using sensitivity analysis.

Central Estimates of the SCC in 2012 US dollars:

Mean: US$78/$tCO_2e$

Median: US$62/$tCO_2e$

6 Sensitivity analysis

6.1 Sensitivity analysis

While the basic science of climate change is considered by most scientists to be 'settled', the scale of disruption that climate change will cause to human societies in the future remains highly uncertain. The appropriate use of economics to quantify the problem, and to express this in values that mean something to present day decision makers, is also the subject of much conceptual and practical debate. As a result, a wide range of estimates of the social cost of carbon have been produced.

Our approach relies on selecting a sub-set from this large population of estimates and normalising them so that they are expressed in common units and a common year. As shown in table 9 below, our final SCC estimates are most sensitive to the methodological decisions that determine which estimates are included or excluded from our meta-analysis. As discussed in the body of this paper, we apply a series of restrictions to the total population of estimates with the overall aim of leaving only those considered to be most reliable (e.g., excluding non-peer reviewed estimates, favouring more recent studies). Varying any of these restrictions (e.g., including non-peer reviewed estimates), has a significant impact on the results of our analysis.

The normalization of estimates (e.g., SCC growth rate or multiple estimates weighting) is generally less impactful, although it is somewhat surprising to note that the choice of approach to applying monetary inflation is the most significant of these.

Table 9: Assessing sensitivity to our methodological decisions by calculating the change in the SCC when each decision is changed to the most likely alternative

	Variable	Alternative tested	Impact rating[12]	Mean (% change)	Median (% change)
Restrictions	Age of study	Include studies regardless of their age	High	73%	5%
	Quality of study	Include non-peer reviewed papers	High	78%	1%
	PRTP	Include all estimates with PRTPs from 0-2	High	-53%	-65%
	Treatment of outliers	Include estimates more than three SDs from the mean	High	35%	5%
	Equity weighting	Include only estimates with equity weighting applied	Med	18%	8%
Normalisations	Monetary inflation	Apply US inflation rates rather than world PPP adjusted inflation	Med	-14%	-29%
	Growth rate of SCC over time	Use the upper bound of the IPCC range (4% p/a rather than 3% p/a)	Med	12%	6%
	Multiple estimates weighting	Do not apply any multiple estimates weighting	Low	-4%	-16%

6.2 Conclusions

Our central estimates are highly sensitive to several of the methodological decisions explained in this paper. As shown in table 9, alternative decisions could increase our mean SCC estimate by more than three quarters, or reduce it by almost two thirds. Given this high degree of sensitivity we have taken the time to carefully research and evaluate each methodological decision, to arrive at a set of decisions that we believe are both individually defensible and collectively coherent.

It is important that any decision maker intending to use an estimate of the social cost of carbon is aware of the uncertainty around it and able to make an informed judgement about the implications of that uncertainty for their particular decision context.

[12] Low = impact on mean is less than 5%

Med = impact on mean is between 5% and 20%

High = impact on mean is greater than 20%

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Appendices

Appendix I: Alternative measures of the cost of carbon

Market price approach

Definition of market price approach

Under the market price approach, the cost of carbon equals the equilibrium market price for permits which confer the right to emit carbon, or the price implied by a government-imposed tax. Market prices for carbon exist in certain countries due to government intervention.

Following economic theory, a government which seeks to maximise societal welfare should set a carbon price equal to the marginal social cost of carbon (the SCC). However, for many reasons, actual existing carbon taxes and market prices tend to be below reasonable estimates for the SCC[13] and rates vary significantly between countries.

Table 10 presents a selection of carbon taxes from around the world.

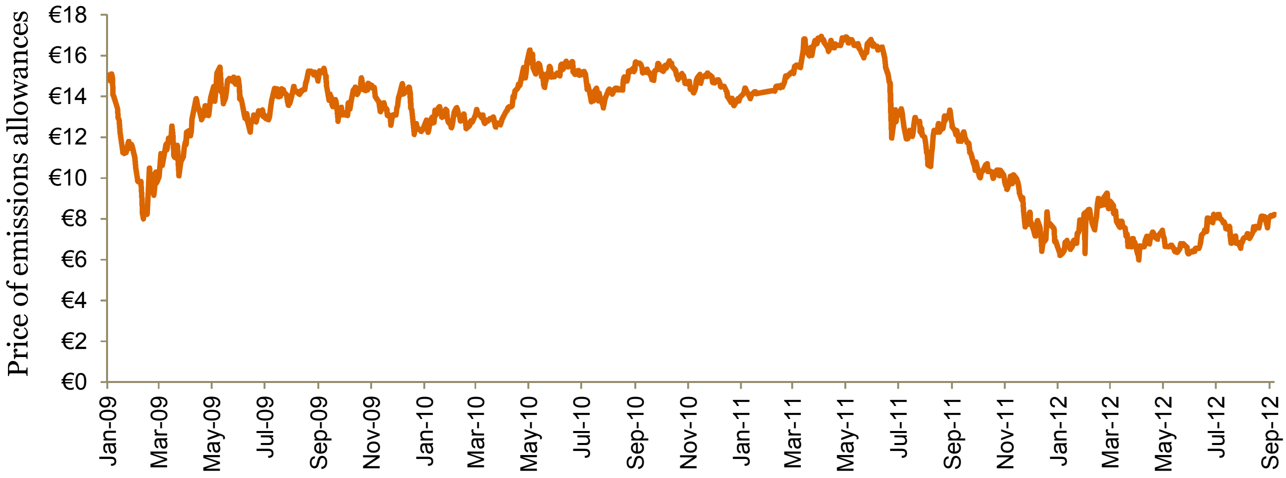
Table 10: Selected international carbon taxes

Country	**2009/10 rate (per tonne of CO_2)**
Carbon taxes that apply to coal used in electricity generation	
Canada (British Columbia)	CAD 15 (USD 13)
Canada (Quebec)	Vary by fuel (CAD 8/tonne of coal) (USD 7)
Carbon taxes with an exemption for coal used in electricity generation (EU ETS countries)	
Norway	Vary by fuel (Coal not taxed; up to NOK 354/tCO_2)(USD 57)
Sweden	EUR 108 (USD 144)
Finland	EUR 20 (USD 27)
Ireland	EUR 15 (USD 20)

Similarly there is no global 'market price' for carbon but a company could choose to take an average traded price from an established carbon market such as the EU Emissions Trading Scheme (ETS) (see Figure 10).

[13] Examples include regulatory capture and concerns about national competitiveness. More generally, GHG emissions abatement is a global public good, such that there are incentives for countries to free-ride on the mitigation efforts of others.

Figure 10: EU Allowance Prices January 2009 - September 2012 (EUR)



Source: Thomson Reuters Point Carbon (2012)

Evaluating the market price approach

Table 11 shows the potential uses of the market price approach based on its strengths, and shows how it is generally inappropriate for E P&L based on its weaknesses.

The main reason for rejecting the market price approach as a basis for placing a value on GHG emissions as part of an E P&L is that prices do not reflect the value of a company's impact on society. Instead, they reflect governmental policy, showing the private cost of a regulated firm's emissions rather than the total societal costs of those emissions. Furthermore, for a government to design carbon markets or taxes with prices equal to the SCC, that government would first need to arrive independently at an estimate of the SCC.

Table 11: Strengths and weaknesses of the market price approach

Potential uses (strengths)	Reasons for rejection of approach (weaknesses)
Avoids the need for calculating MAC curves or agreeing on an emissions reduction target. Is directly observable and does not rely on assumptions about the future.	Market prices do not directly measure the value of a company's environmental impact on society, instead measuring the private financial cost of this impact under specific policy regimes. It is unclear which value to use, as the boundaries of climate policy regimes are not aligned with the boundaries of many firms, particularly where supply chains are concerned. Observed carbon prices tend to be considerably lower than the majority of SCC estimates. This suggests that they do not reflect the true extent of externalities imposed on society by GHG emissions.

Marginal abatement cost (MAC) approach

Definition of the MAC

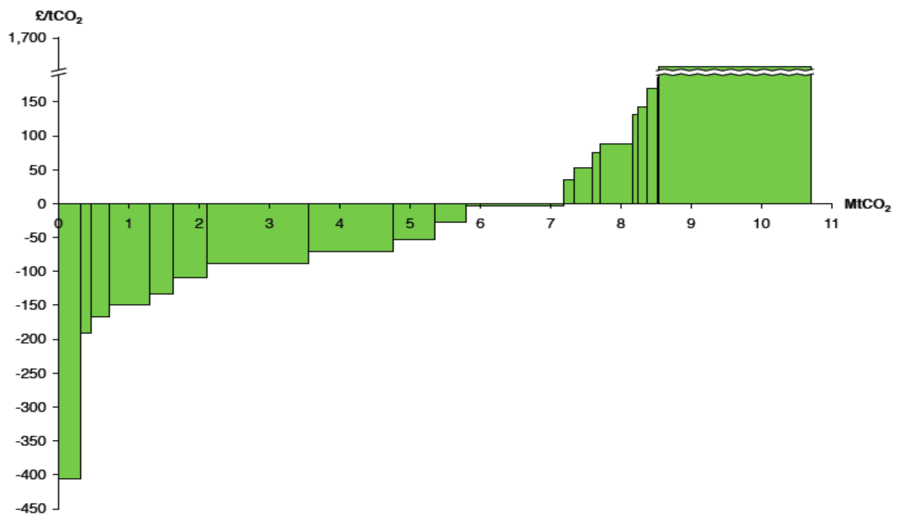
Under the MAC approach, the cost of carbon equals the cost of avoiding emitting that same unit of carbon, measured as emissions abatement relative to some business-as-usual scenario. Marginal abatement cost analysis shows the cost associated with a certain quantity of emissions abatement. The results of this type of analysis are typically displayed in a MAC curve. These illustrate the cost of reducing GHG emissions by one unit (typically a tonne) for the object of analysis. In the context of an E P&L, this object would be the company but, for other purposes it could also be an industry or entire economy. MACs are often created through analysis of the available technology options, but can describe other types of mitigation actions such as policy activities.

MAC analysis comes in a variety of forms. Kensicki (2011) distinguishes between expert-based and model-derived MAC curves. The former involve expert analysis of individual abatement measures, while the latter are the output of industry and energy sector models. The former involve bottom-up calculations by technology, while the latter may involve either bottom-up sector modelling or top-down whole-economy modelling.

For particular companies looking to abate their GHG emissions, expert-based MAC curves are likely to be the most appropriate as they are readily tailored to individual circumstances and are well-suited to a situation where only private, financial costs must be considered. The calculation process starts by looking at the abatement technologies available and the cost of each at a given point in time. Technologies could range from specific energy efficiency measures, to planting trees or implementing carbon capture and storage on fossil fuel-burning power plants. Once each technology's potential has been exhausted, the next technology is examined and its cost estimated. When placed in ascending cost order, these technologies form a MAC curve. MACs are time-specific and their form is driven by a range of assumptions, including those about technological innovation, efficiency of implementation and the state of the future economy (Ekins et al., 2011).

An example MAC curve is illustrated graphically in Figure 11. Each of the bars in the histogram represents a different technology which could be adopted to reduce emissions. The width of each bar shows the GHG abatement potential of the technology, while the height shows its cost per unit of emissions abatement when installed to maximum mitigation potential. The total cost of each technology is shown by the area of each histogram bar. The carbon price required to achieve a given emissions reduction can be identified from the MAC curve by locating the desired quantity of emissions reductions on the x (horizontal) axis and reading the price associated with the relevant bar from the y (vertical) axis. In the example shown in Figure 11, reducing emissions by 8MtCO_2 would require a carbon price of around £90/tCO_2 ($142).

Figure 11: Example MAC curve



Source: UK Climate Change Committee (2008)

Evaluation of the MAC approach

The MAC approach is widely used in private and public sector organisations, including European governments and major firms. However, it is important to consider the pros and cons of the approach in relation to E P&L in particular. Table 12 shows the potential uses of the MAC approach based on its strengths and shows how it is generally inappropriate for E P&L due to its weaknesses.

The core reason for rejecting the MAC approach as a basis for placing a value on GHG emissions as part of an E P&L is that it does not directly measure the value of a company's impact on society. Instead, it shows the cost of reducing that impact.

However, it is anticipated that an understanding of the MAC approach may still be useful in implementing an E P&L in other ways:

- Governmental and regulatory engagement: where a company wants their E P&L to support their engagement with regulatory authorities, they may wish to consider using the costs of carbon recommended by that authority for project and policy appraisal and evaluation. These are sometimes based on the MAC approach. One prominent example is the UK Government's 'shadow price'[14] for carbon in the non-traded sector (which is not covered by the EU-ETS). It was derived from MACs in an attempt to ensure that the UK hits its target to reduce GHG emissions by 80% by 2050 target. In 2010, the price was £52/tCO_2e ($82), rising to £200/t$CO_2$e ($316) in 2050.
- Analysis which builds on E P&L: some companies may wish to understand how to reduce the societal impacts of their activities once an E P&L has been produced. Overlaying the results of MAC analysis on E P&L results could help show how to achieve this most efficiently.

Table 12 summarizes the strengths and weaknesses of the MAC approach.

Table 12: Strengths and weaknesses of the MAC approach

Potential uses (strengths)	Reasons for rejection of approach (weaknesses)
• It is useful for setting a price for carbon which is consistent with an organisation or state achieving its emissions reductions goals. • It is useful for prioritising and designing more specific interventions to reduce GHG emissions, including the decision of whether to abate GHG emissions, or to pay a carbon tax or buy carbon credits. • The price derived is based on known costs and may have a narrower range of uncertainty than the SCC.	• To realise the decision-making benefits of a MAC curve within a company's value chain would require a thorough technological assessment of its operations and supply-chain to produce a firm-specific MAC curve. • MACs do not directly measure the value of a company's environmental impact on society. Instead, they measure the cost of reducing this impact.

[14] In economic theory, shadow prices are conceptually the same as the social cost of carbon, but the UK Government uses this terminology to distinguish the MAC approach from their former SCC approach.

Appendix II: Calculating a primary estimate of the societal cost of carbon

Although we chose to conduct a form of meta-analysis based on existing studies, it is nonetheless instructive to consider how primary estimates of the societal cost of carbon are derived to help understand the challenges and uncertainties which ultimately underpin our results.

Deriving estimates of the total cost of climate change typically involves a multi-step modelling process which assesses the biophysical impacts of GHG emissions and values the likely consequences in economic terms.

Step 1: Using IPCC scenarios to project GHG emissions

The first step is to project future atmospheric GHG concentrations. This relies on projecting future GHG emissions which, in turn, relies on assumptions about future development trends including population growth, economic growth and technological change.

Given the significant uncertainty associated with these assumptions, climate scientists consider a range of different scenarios. The IPCC uses a set of scenarios derived from four different 'storylines', as described in the IPCC's Special Report on Emissions Scenarios (SRES) (IPCC, 2000). Each storyline describes and applies a quantitative interpretation to one particular direction for future demographic, social, economic, technological, and environmental changes. For each storyline, several scenarios were developed using different modelling approaches, resulting in a total set of 40 SRES scenarios. Out of these, the IPCC chose six 'marker', or illustrative, scenarios (referred to as A1F1, A1T, A1B, A2, B1, and B2). The output published by the IPCC using the six marker SRES scenarios is a range of projected GHG emissions in the years 2020, 2050 and 2100.

Step 2: Using global climate models to project future climate changes

The second step is to input projected GHG emissions, along with many other factors, into general circulation models (GCMs). GCMs are mathematical models used to simulate the Earth's climate system. Their outputs are projections of climate-related variables (e.g. atmospheric and oceanic temperatures, precipitation, currents, sea ice cover, and wind) at a given location at a given point in time.

While GCMs are extremely complex and rely on a variety of assumptions, the IPCC has 'considerable confidence' that they provide 'credible quantitative estimates' of future climate change for the following reasons:

- Model fundamentals are based on established physical laws such as conservation of mass, energy and momentum.
- Climate models have shown increasing ability to simulate observed features of current climate systems.
- Models have been able to reproduce key features of past climates and climate changes.

At the same time, the IPCC recognises that the models still show 'significant errors', generally at regional rather than global scales. For example, there is still considerable uncertainty associated with the representation of clouds and how clouds respond to climate change. There are also difficulties in projecting conditions at a local-scale, particularly for regions that are relatively poorly studied by GCMs.

Step 3: Using impact assessment models to project environmental impacts

Outputs from GCMs (e.g. a given temperature rise at a given time in the future) are used in impact assessment models to project an impact of climate change on the environment. A wide range of such environmental impact assessment models have been developed, including storm and tropical cyclone models to model weather events, inundation models to model sea-level rises and use of geographic information systems to model asset exposure.

Step 4: Using economic methods to value environmental impacts

The next step is to assign a monetary value to a given future environmental impact. There are two main approaches to this: enumerative and statistical. The enumerative approach takes various economic techniques and models - including market and surrogate market approaches, and aggregates the monetary values across all environmental impacts. This gives a total damage valuation of climate change for a particular point in the future.

The statistical approach directly estimates climate welfare impacts using observed spatial variations in prices and expenditures. This aims to isolate the differences in incomes and costs that can be attributed to differences in climate between one region and another. Both enumerative and statistical approaches are used to derive the 14 estimates of total welfare loss due to climate change cited by Tol (2011).

Step 5: Discounting back to present-day dollars

The total damage valuation derived in step 4 is typically given as a percentage impact on GDP at a certain point in future. In order to derive an SCC, this total damage value must be discounted to present-day US dollars. The 300+ estimates of SCC derive from only nine estimates of total damage valuation, which illustrates the diversity of techniques used for discounting. Selecting a social discount rate is contentious, as is the question of how to account for the fact that disproportionate damage will be caused to people in developing rather than developed countries (see Chapter 5).

Integrated assessment models

Integrated assessment models consider scientific, social and economic factors together in order to apply a more holistic approach to analysing climate change. Essentially, a combination of steps 1 to 4 described above is integrated into a single model. Integrated assessment models tend to be used to provide information for decision/policy-makers. For example, the UN Food and Agriculture Organisation (FAO) has developed MOSAICC (Modelling System for Agricultural Impacts of Climate Change), which is an integrated assessment model used to model each step from climate scenarios down to economic impact analysis of climate change on agriculture at a national level. MOSAICC is divided into four components: climate data-processing models, crop models, hydrological models, and economic models.

Appendix III: Statistical methods – the distribution of SCC estimates in our sub-set

"The possibility of catastrophic climate change needs to be taken seriously ... the bad tails of the relevant probability distributions should not be ignored" Weitzman (2009)

One of the more recent debates over the SCC is the treatment of 'fat-tailed' probability distributions of climate damages. This refers to the finding that - when the estimates from many studies are analysed side by side - extremely large climate damage estimates are far more frequent than net positive estimates of comparable magnitude, relative to a normal distribution around the mean value. This issue is now prominent in the climate change debate with some arguing for a precautionary approach which puts significant weight on extreme scenarios (e.g. by choosing a mean estimate), and others arguing that such an approach overstates the risks and that there is greater convergence around lower estimates (e.g. for which the median would provide a better proxy).

The choice between mean and median allows for this divergence of views to be accommodated; but raises a further issue for consideration. By applying restrictions to the overall population of estimates we arrive at a relatively small sub-set of estimates (33), from which to generate our 'sample statistics'. The reliability and stability of all sample statistics, but particularly of the median, can be affected by a small sample size. As discussed in chapter 4, our sub-set of estimates is not a 'sample' in the traditional statistical sense and the relatively small number of observations is therefore not a primary concern. But a lack of robustness of our median estimate to small changes in the number of observations (e.g. when updating our estimates based on new studies) would nonetheless be undesirable.

One option to try and counter this would be to 'fit' the observations to a statistical distribution – either one that they appear to follow, or one which they might be expected to follow (e.g. a fat-tailed distribution such as the Fisher-Tippet[15] distribution). This approach has been followed in several studies (e.g. Tol, 2008; 2009; 2011).

However, after evaluation of a number of potential candidate distributions (Fisher-Tippet, Log-Logistic, Log-Normal and Pearson) we opted not to fit the data to a pre-defined statistical distribution. Our primary reason was that none of the tested distributions demonstrated a clear alignment with our sub-set of estimates. But in addition, those exhibiting the closest fit had minimal impact on our sample statistics and did not materially alter the robustness of the median to marginal changes in the number of observations[16].

Furthermore, there has been some criticism of fitting estimates of the SCC into a single distribution which points out that the studies are not drawn from a single underlying distribution (whereas, for example, measurements of different trees of the same species are), that the various estimates are not independently generated (many come from the same original damage estimates) and that it is artificial to apply a sample size to a single estimate (a necessity when assuming it represents a distribution) (Nordhaus, 2011).

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[15] The Fisher-Tippet distribution was evaluated as the best fit for the overall population of SCC estimates by Tol (2005; 2011).

[16] I.e. the impact on the median of removing a value from the top or bottom half of the sub-set was not significantly different whether calculating from fitted or un-fitted data. The relative stability of our median estimate produced using the un-fitted data provides some comfort about its robustness.

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Valuing corporate environmental impacts: Land use and biodiversity

PwC methodology paper

Version 2.5

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Contents

Table of Tables

Table of Figures

Table of Equations

Abbreviations and acronyms

Abbreviation	Full name
COPI	Cost of policy inaction
E P&L	Environmental Profit and Loss
EEIO	Extended input-output modelling
FAO	Food and Agriculture Organisation
GDP	Gross domestic product
GNI	Gross national income
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
OECD	Organisation for Economic Co-operation and Development
NPV	Net Present Value
PPP	Purchasing power parity
SCC	Social Cost of Carbon
TEEB	The Economics of Ecosystems and Biodiversity
UK NEA	United Kingdom National Ecosystem Assessment
UN	United Nations
USD	United States dollar
WWF	World Wildlife Fund

1. The environmental impacts of land use

1.1. Introduction

Natural land areas – often rich with biodiversity, provide essential services to society which 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 (MA, 2005). A subsequent analysis requested by the G8+5 environment ministers, The Economics of Ecosystems and Biodiversity (TEEB), estimated that the economic cost imposed by degradation and loss of biodiversity and ecosystem services each year is between USD 2 and 4.5 trillion[1] (TEEB COPI, 2008).

The flow of ecosystem services from natural land areas are provided to society every year and, as the extent of natural land areas decreases, so the annual flow of ecosystem services is reduced. The impact of the conversion of a natural area is therefore felt every year, until that area is restored such that it resumes its production of ecosystem services.

The principal cause of on-going losses of biodiversity and declines in associated ecosystem services is the conversion of natural land areas for agriculture (OECD, 2008). Cultivated systems now cover one quarter of the Earth's terrestrial surface and it is predicted that a further 10% to 20% of natural grassland and forestland may be converted by 2050 (MA, 2005).

1.2. Overview of impact area

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 society gains from ecosystems, such as climate regulation, bioprospecting, food and fuel. They also include non-use values from cultural experiences or education, for example, and option values that reflect that we recognise that we might have future use values.

In this methodology, and in the E P&L in general, we only seek to value the benefits that people receive from biodiversity and natural areas; we do not attempt to measure the intrinsic 'value' of nature outside of the realm of human preferences.

An important consideration for this methodology is the temporal dimension because many natural areas were converted long ago, and have changed uses 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 its 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 conversion of the land.

An alternative approach would be to consider the net present value (NPV) of lost ecosystem services at the time when the natural ecosystem was converted and assign this total impact to those directly responsible for the conversion. Another approach, employed by some regulations and certification bodies, is to only consider negative impacts of land converted after a predefined date.

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[1] The Economics of Ecosystems and Biodiversity (TEEB), Cost of Policy Inaction Report, USD2-USD4.5 trillion is the present value of net ecosystem service losses from land based ecosystems caused in 2008 and continuing for 50 years, based on discount rates ranging from 1-4%.

The first approach (valuing and attributing in year losses to the current occupant) is most appropriate for the E P&L for three important reasons:

I) It reflects the flow of impacts which are created as a result of occupation, and are dependent on the management practices which the current occupier chooses to employ (even if others are responsible for the pre-conditions). For example, UK floods in 2013 were exacerbated by poor land management leading to reduced farmland water interception and retention on land converted hundreds of years ago. This is in accordance with the broader principles of the E P&L approaches.

II) It incentivises current land occupiers to minimise the loss of ecosystem services, for example through sustainable land management practices.

III) It avoids making highly uncertain assumptions as to the future extent of lost ecosystem services or the date of past conversions.

1.2.1. Types of land use conversion

Some natural areas are converted each year. However, a lot of natural areas have been converted many years ago when natural areas were less scarce than they are today. In this methodology, we distinguish between use (occupation) of already converted land and new conversion of natural ecosystems, introduced here and discussed in more detail later.

- Use of previously converted land: This methodology values the loss of ecosystem services in the current year and attributes these impacts to the current occupier of the land.

  All previously converted land is valued the same (all else being equal) irrespective of when it was converted. This is because within a given area, all converted land contributes equally to the prevailing deficit in ecosystem services (all else being equal). For example, consider two one-hectare plots that were previously natural forests. One plot was converted two years ago and the other 10 years ago. Each plot represents one hectare of forest ecosystem services that are not supplied in the current year; they therefore contribute equally to the value of lost ecosystem services, even though they were converted at different times. The average of marginal values of lost ecosystem services as a result of increasing ecosystem scarcity (particularly evident over time) is therefore the appropriate value for land use on previously converted land. This is discussed in more detail in Chapter 4.

- New conversion of ecosystems: Natural ecosystems that are converted in the current year should be valued at the current marginal value. The current marginal value is higher than the average value over time, which will reflect lower historical levels of scarity. Areas of new conversion are therefore treated differently from use of previously converted land. New conversions result in increased scarcity, so the average marginal cost per hectare of lost ecosystem services, which is applied to converted land, will increase in subsequent years if scarcity increases. A year after conversion, the average marginal value is applied (which increases as a result of the previous year's conversions and increasing scarcity, assuming no restoration).

Results of the E P&L for these two types of land use can be shown separately or together.

1.2.2. Environmental and societal outcomes

Table 1 presents a classification of different ecosystem services which can be affected by the conversion and occupation of land. It is these services which deliver use, non-use and option values to society.

Table 1. Classification of final ecosystem services

Service Class	Specific eco-system 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 & medicinal plants	Global
	Ornamental products	Regional
	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

We only consider final ecosystem services here as the inclusion of intermediary services would lead to double counting (UK NEA, 2011). This is also in-line with the recommendations of CICES (Common International Classification of Ecosystem Services, 2013). For example, supporting services are excluded. Supporting services include those that are necessary for the production of 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 the supporting services.

In the event of natural land conversion, and its subsequent occupation, the extent of impacts can be determined by considering how each of these services is affected by the change in land use. This depends on what the land was converted from and what the new land use is. For example, conversion of tropical forest to grassland pasture will result in an almost complete loss of climate regulating services. However, conversion to rubber plantations will only result in a partial loss. Different services will be affected differently depending on the conversion and the type of land management practices which are employed during occupation. The change in ecosystem service provision is termed the ‘environmental outcome’.

The potential extent to which people around the world are affected by the loss in specific services will depend on the geographical level at which these services operate. For example, harvesting of food and fibre from natural areas tends to be local, while climate regulation is global (Table 1). This geographical scope defines the population that could be impacted as a result of the loss of these ecosystem services from an area. The actual extent to which people are affected depends on how vulnerable they are to losses in a specific service (this is discussed further in Chapter 4).

Similarly, the ways in which people are affected is highly context specific. 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. Loss of soil fertility and associated provisioning services could lead to malnutrition and displacement for subsistence farmers, but in developed countries the impacts are more likely to be a loss in revenue or profitability, or loss of recreational opportunities, for example.

The causality between the conversion of land, the environmental outcome and the different types of impact on people is discussed in the next section, while the methodology to quantify them is presented in Chapter 4.

1.3. Impact pathway

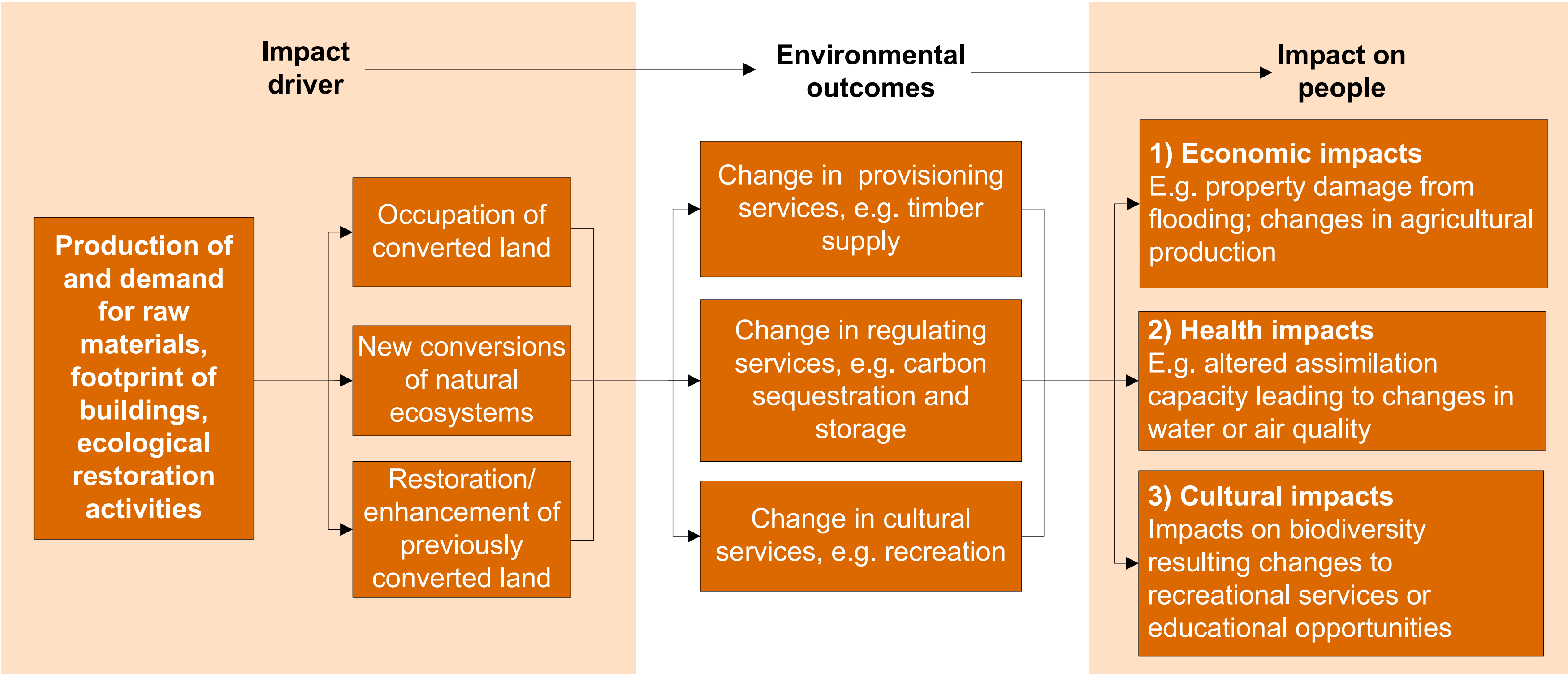
In order to value the impacts of corporate land occupation and conversion, we need to understand how their activities lead to changes in ecosystem service provision and how these changes affect people. Therefore, we define impact pathways that describe the links between corporate activities, the environmental outcomes from those activities and the resultant societal impacts. Our impact pathway framework consists of three elements:

- Impact drivers:
  - *Definition:* These drivers are expressed in units which can be measured at the corporate level, representing either an emission to air, land or water, or the use of land or water resources.[2]
  - *For land use:* Demand for land for agriculture, other raw materials and living/working space by businesses drives the occupation of land and conversion of natural ecosystems.
- Environmental outcomes:
  - *Definition:* These describe actual changes in the environment which result from the impact driver (emission or resource use).
  - *For land use:* The loss of ecosystem services. This can include near complete loss of services from a natural area if, for example, a forest is converted to an office block, or partial loss if the forest is converted to a timber plantation.
- Impacts on people:
  - *Definition:* These are the actual impacts on people as a result of changes in the environment (environmental outcomes).
  - *For land use:* The range of impacts is quite broad depending on the types of ecosystem services lost. For example, reduced resilience to floods may result in property damage, a variety of economic losses and impacts on health and life.

The three stages of the impact pathway are shown in Figure 1 overleaf.

[2] A note on language: in this report, the measurement unit for any 'impact driver' is an 'environmental metric.'

Figure 1: Impact pathways for land use



1.4. Prioritising which impacts to quantify and value

This methodology attempts to include all material impacts of land use and conversion in the valuation of corporate impacts. We value each ecosystem service individually (as a value per hectare per year), aggregating services to obtain a total economic value (TEV) for the ecosystem services of a particular area of land.

Different types of ecosystem service are more or less valuable in different contexts. Firstly, service provision is variable depending on the eco-region. For example, carbon sequestration is relatively immaterial in most deserts compared to tropical forests. Secondly, the extent to which people are dependent on specific services is contingent on their circumstances and the broader context. The relative dependence of people is addressed in the application of the values to specific contexts in the valuation methodology presented in Chapter 4. Here we consider the relevance of ecosystem services to define our scope for different eco-regions.

In Table 2 we present the ecosystem services in scope for different eco-regions. This scope is based on the wealth of research into the value of ecosystem services in different eco-regions over the last thirty years. In particular, our methodology builds on the approach and dataset of The Economics of Ecosystem and Biodiversity[3] (TEEB, 2010, Van der Ploeg, 2010). At the time of publishing in 2010, the TEEB dataset was the most comprehensive dataset of ecosystem service valuations. Over the last few years, we have updated it with the latest literature, such that it now contains over 1,500 estimates of individual ecosystem service values.

While our scope is limited to an extent by the availability of data, coverage of estimates across different ecosystem services and eco-regions is generally good with estimates available for the major ecosystem services for each eco-region. The academic literature has not yet considered all services from all eco-regions[4] and while there are gaps, we do not consider these to be significant as they are for services of lower significance. For example, air purification is not generally considered to be an important service provided by deserts or wetlands. Deserts/arid grasslands have significantly less coverage than the other eco-regions: this is partly because they do not (typically) supply all these ecosystem services. Their relatively lower value compounds this: they tend to be of less interest from a societal value perspective and have not been the focus of much work by environmental economists.

Table 2: Coverage of valuation estimates by ecosystem service and eco-region

Ecosystem service	**Tropical Forests**	**Temperate and boreal forest**	**Grasslands**	**Desert/Arid grassland**	**Inland Wetlands**	**Coastal Wetlands**
Food from natural/semi-natural ecosystems	Y	Y	Y	Y	Y	Y
Fibre, other raw materials	Y	Y	Y	Y	Y	Y
Domestic and industrial water	Y	Y	Y		Y	Y
Bio-prospecting & medicinal plants	Y	Y	Y		Y	Y
Ornamental products						Y
Air purification	Y	Y	Y			
Recreation	Y	Y	Y		Y	Y
Spiritual and aesthetic		Y			Y	Y
Cognitive and learning opportunities		Y			Y	Y
Stable climate	Y	Y	Y		Y	Y
Pollution control and waste assimilation	Y	Y	Y		Y	Y
Erosion control	Y	Y	Y		Y	Y
Disease and pest control		Y				Y
Flood control and protection from extreme events	Y				Y	Y

[3] TEEB is a global initiative focused on drawing attention to the economic benefits of biodiversity. Its objective is to highlight the growing cost of biodiversity loss and ecosystem degradation. It was set up following the G8+5 Potsdam conference, publishing a series of papers drawing on expertise from over 2,000 scientists, economists and business people and policy makers.

[4] Ecosystem services with fewer than 2 valuation estimates are excluded from our analysis and are not presented in Table 2.

1.4.1. Limitations of scope

As discussed above, while there are fewer ecosystem services covered for deserts, we do not believe there to be significant omissions in scope, with each of the major ecosystem services for each eco-region covered.

Our methodology follows the ecosystem approach by valuing the services provided by ecosystems, rather than the individual constituents of a specific ecosystem. This is generally accepted as the most robust approach to the measurement of societal values relating to land use changes and degradation of ecosystems by academics and policy makers.

However, it is an evolving approach and this on-going development is relevant in a number of important respects:

1. The ecosystem services typology set out in Table 1 and 2 is a significant simplification of the many, and varied, benefits that people receive from the environment and it follows that any valuation based on this typology will itself be a simplification of reality.
2. Methods for the valuation of ecosystem services are themselves evolving rapidly and the choice of method can have a significant impact on the resulting valuation. At present, the basic alignment between economic concepts of direct use, indirect use and non-use value, and ecosystem service classifications is also imperfect.
3. Even if the alignment were perfect, the difficulties that ecologists face in linking changes in biodiversity with changes in the provision of ecosystem services, coupled with the simplifications required in economic analysis, mean that ascribing precise values to marginal changes in biodiversity (in all but a few unusual cases) remains some way off.

A key implication is that, in situations where an individual species is affected (e.g. due to wild hunting) without a discernible impact on the supply of ecosystem services (either due lack of data or an incomplete understanding of ecosystem functioning), it may not be possible to estimate the changes in human welfare – i.e. to ascribe a societal cost. This is particularly likely where the affected species is not 'charismatic'[5] and does not provide directly measurable benefits (via ecosystem services) to society, such as through tourism, bioprospecting or pest control.

[5] A charismatic species is usually large and noticeable organism which acts as icon or symbol for a defined habitat. Different cultures will have different charismatic species of particular meaning to them. *www.wwf.panda.org* accessed Feb 2014.

2. Summary of methodology

2.1. Introduction

The impact pathway presented in Chapter 1 identifies how emissions and resource use can lead to different types of impacts. Our valuation framework is structured to follow this pathway, at each stage demonstrating the causal links between corporate activities (which result in changes in ecosystem service provision) and societal costs or benefits.

To understand the value of societal impacts associated with each of the drivers, it is necessary to:

1. **Obtain environmental metric data:** The starting point for each of our methodologies is data on the amount of land use. These metric data are based on an understanding of the corporate activities which they result from. The data can come from a variety of sources, some of which (e.g., life cycle assessment (LCA) or environmentally extended input-output modelling (EEIO)) are subject to their own distinct methodologies[6].

Table 3: Environmental metric data

Impact driver (emission or resource use)	Environmental metric data
Land conversion and occupation of newly converted land	Total land for corporate activities (m^2), identifying land cover type, extent of ecosystem service loss (or suitable proxy, see Chapter 4)

2. **Quantify environmental outcomes:** We quantify the biophysical changes in the environment resulting from anthropogenic pressures (as measured by the metric data). This is discussed further in Table 4.
3. **Estimate impacts on people:** We estimate the societal cost of how people are affected by environmental outcomes stemming from corporate activities. This is discussed further in Table 4.

It is not always necessary or appropriate for environmental economic valuation to go through each of these steps explicitly. A single methodological step may often cover several steps at once. However, considering each E P&L methodology through the lens of this valuation framework helps ensure rigor, transparency, and consistency.

2.2. Summary of methodology

The methodology presented here is intended for use with global supply chains. It presents a method to calculate the quantity of land use where actual data are not available and value the lost ecosystem services across many countries simultaneously. Where highly localised valuations are required, a more locally-focused approach should be applied (based on similar principles to those described here but tailored on a case by case basis).

The first step of this methodology is to measure or estimate the quantity of land in use and converted in the current year (metric data). Where available, this can be obtained directly from company data. However, companies that do not directly manage the production of their raw material inputs are unlikely to have data on the most material areas of land use. In such cases, land use can be estimated through productivity modelling (this is generally the preferred approach and is described in Appendix II), or by using techniques such as environmentally extended input output modelling (EEIO), or life cycle assessment (LCA).

This paper focuses on quantifying changes in the environment and valuing these changes in terms of societal costs and benefits. The steps to do this are summarised in Table 4. This table relates back to the second two columns of the impact pathway: identifying the environmental outcomes (change in ecosystem services) and valuing the impacts on people.

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[6] The sources of metric data are outlined in Chapter 3. The assumed starting point for this methodology is the form specified in Table 3.

Table 4: Summary of land use societal impacts calculation methodology, key variables and assumptions

Quantify environmental outcomes		Estimate impacts on people
Land use pathway impact pathway		
Method	• Where the eco-region affected is not known, this can be identified using WWF's Wildfinder GIS database. • The extent of ecosystem service loss associated with the land use management practices, relative to the natural eco-region, are estimated based on changes in biomass and species richness. • The pattern of land occupation and conversion also affects service provision and can be used to estimate losses in ecosystem services in different plots of land. • Where multiple outputs are produced in the same land area (e.g. leather and meat from cattle farms), the impacts are attributed based on the relative economic value of each output.	• Per hectare valuation estimates are calculated for different ecosystem services by eco-region and in different countries or sub-national regions. • 1,500 estimates of ecosystem services are classified into eco-regions and medians taken across each ecosystem service to estimate the current marginal value of ecosystem services. • An econometric approach was tested however the number of relevant variables is too large, each with limited explanatory power, and not enough data of sufficient consistency to identify a systematic relationship. Van der Ploeg et al. (2010) came to the same conclusion during their analysis of the TEEB database. • Country and eco-region specific values of individual ecosystem services are adjusted for the socio-economic context. In particular, the proportion and concentration of rural populations are used as a proxy for dependence and vulnerability of people on local and regional services. The values for these services are also PPP adjusted to account for differing willingness to pay. Global services are not adjusted for country-specific parameters. • A portion of the service value is applied to each hectare of land use based on the proportion of ecosystem service loss. • For occupation of previously converted land, an average of the different marginal values due to differing scarcity of land in the past is applied. • The current marginal value is only applied to new conversions.
Key variables	• WWF Wildfinder eco-regions. • Biomass and species richness of natural eco-region and new land use by country.	• Existing primary estimates of ecosystem services. • GDP, inflation, GNI, population density.

Assumptions and justification	• The 6 broad eco-regions (corresponding to our valuation database) are considered appropriate because the principle driver of value is the nature of the ecosystem service itself within an eco-region, together with characteristics of the benefiting population. • Changes in biomass and species richness pre/post conversion are acceptable indicators for changes in ecosystem service provision.	• Underlying estimates provide a representative sample of the ecosystem services provided by each of the 6 eco-regions. The database used is the most comprehensive repository of primary estimates available. However, distributed across 14 ecosystem services and 6 eco-regions, the number of values for each ranges from 2 to 90 (after all exclusions) which are subject to a level of natural variation. Despite this, we believe they are sufficient to give a strong indication of the likely scale of value that can be delivered by different ecosystem services. • The distribution of estimates in the underlying dataset shows a long tail, with many estimates towards the low end and a few very high estimates. We select the median value as the most representative estimate of the likely impacts of sourcing from a given eco-region because it is more robust (outliers have less influence on the result).

3. Data requirements

3.1. Introduction

Gathering appropriate data is a precursor to valuing the environmental impacts from land use. The availability of high quality input data is a key determinant of the appropriateness of impact quantification and valuation.

Three broad categories of data are required for quantification and valuation:

- **Metric data:** Quantity and location of land use and conversion. Where this is not available, the quantity and source location of raw materials.
- **Contextual data:** These relate to the context of land use. Our methodology requires two types of contextual data:
  - Eco-region type of occupied and converted land; and
  - Socio-economic characteristics around land use and conversion areas.
- **Other coefficients:** Numerical estimates of ecosystem value derived from the academic literature or other credible sources which are required to convert metric and contextual data into value estimates.

While methods for the collection or estimation of basic metric data are not the subject of this paper, the data generation methods used are nonetheless relevant to the likely availability of contextual data and therefore the viability of different potential valuation approaches. This chapter therefore has two purposes: firstly, it describes the most likely sources of metric data across a typical corporate value chain and the implications for contextual data availability; secondly, it sets out key contextual and other coefficient data requirements and the preferred sources for these.

3.2. Environmental metric data

The ideal dataset would specify the land area used and the location of these areas (at least at the country level) across the supply chain. In practice, it is the production of raw materials, particularly agricultural raw materials, which are likely to represent the vast majority of land use for most companies outside of the services sector.

However, in many cases, only companies that are directly associated with the production of raw materials will know the area used. Data sources which are likely to be most readily accessible for an average company are outlined in Table 5.

Table 5: Likely metric data sources

Value chain stage	Metric data
Own operations	Land use footprint of buildings should be available from company management information
Immediate suppliers	Land use footprint of buildings may be available from suppliers Where this is unavailable, gaps in metric data can be filled using modelling techniques such as EEIO.
Upstream/ supply chain	Footprint of buildings can be estimated using EEIO and LCA (or inferred from other suppliers). Land use footprints of 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

Value chain stage	Metric data
	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. 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 on 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/ re-use impacts	Land use area can be modelled using EEIO or LCA techniques. This may be further informed by customer surveys or industry information.

3.3. Contextual and other data

The contextual data and other data requirements are set out in Table 6. These include other key inputs (beyond volume of land use) needed to execute our models and coefficients from literature needed to execute valuation.

Table 6: Contextual data requirements for quantifying land use and valuing the impacts of lost ecosystem services

Information	Purpose	Default metrics
Quantifying environmental outcomes		
Location of land use	Eco-regions and ecosystem service supply varies geographically.	Company data, supplier questionnaires. Other options include trade data from UN Trade, government statistics and multi-region input-output models.
Co-productions: Relative value of raw material versus other products from land	For raw materials that are produced in conjunction with other materials in the same area, we use an economic allocation (e.g. meat and hide from cows)	Company data, supplier questionnaires, FAO Stat and other commodity price data sources.
Eco-region distribution	Specifies the type of natural ecosystem for the location	WWF Wildfinder, and supporting academic literature
Biomass	Used to approximate the change in ecosystem service function as a result of conversion	IPCC provides generic biomass (tonne/ha) estimates for different land uses and approved methods for estimating a change following conversion. Other sources are available in the ecological academic literature.
Species richness	Used to approximate the change in ecosystem service function as a result of conversion	Ellis *et al.* (2012) estimate changes in species' richness (count of functional groups) associated with different land uses in different eco-regions. Other location specific data on species richness are available in the ecological academic literature.

Information	Purpose	Default metrics
Estimating societal impacts		
Unit and currency conversions	Estimates of ecosystem services should be inflated to current prices and converted to the same unit and, where appropriate, PPP adjusted.	Exchange rates, inflation and GNI for PPP adjustments are sourced from the World Bank.
Population distribution and density	The more people who benefit from an ecosystem service, and the more dependent those people are on it, the higher its value. Population density and the rural – urban distribution of population is used to adjust value estimates to account for regional differences.	The World Bank provides data on average of country population density and the distribution of population between urban and rural areas. More location specific values can be used where appropriate and available.
Ecosystem services value	Estimate the impact of ecosystem service loss in a given context	Peer-reviewed environmental economic literature

4. Valuation module: Loss of ecosystem services from land use and conversion

This chapter presents a method to estimate the societal impacts of corporate land use, including occupation and conversion. This estimate incorporates the societal impacts associated with all land use that can be attributed to a company's operations and therefore may need to cover multiple geographies across expansive global supply chains. The valuation approach first traces the share of land use for which the company is responsible before going on to estimate the extent of service loss that occurs as a result of the change in use. An estimate of the societal impacts of these lost services is then determined in the final step.

A summary of the valuation approach is provided in Table 7 (from chapter 2). The following sections build on the data requirements (and likely availability) outlined in Chapter 3 and discuss our methodology in detail.

Table 7: Summary of land use societal impacts calculation methodology

	4.1 Quantify environmental outcomes	4.2 Estimate societal impacts
Valuation module: Loss of ecosystem services from land use and conversion		
Methods	• Where the eco-region affected is not known, this can be identified using WWF's Wildfinder GIS database. • The extent of ecosystem service loss associated with the land use management practices, relative to the natural eco-region, are estimated based on changes in biomass and species richness. • The pattern of land occupation and conversion also affects service provision and can be used to estimate losses in ecosystem services in different plots of land. • Where multiple outputs are produced in the same land area (e.g. leather and meat from cattle farms), the impacts are attributed based on the relative economic value of each output.	• Per hectare valuation estimates are calculated for different ecosystem services by eco-region and in different countries or sub-national regions. • 1,500 estimates of ecosystem services are classified into eco-regions and medians taken across each ecosystem service to estimate the current marginal value of ecosystem services. • An econometric approach was tested however the number of relevant variables is too large, each with limited explanatory power, and not enough data of sufficient consistency to identify a systematic relationship. Van der Ploeg et al. (2010) came to the same conclusion during their analysis of the TEEB database. • Country and eco-region specific values of individual ecosystem services are adjusted for the socio-economic context. In particular, the proportion and concentration of rural populations are used as a proxy for dependence and vulnerability of people on local and regional services. The values for these services are also PPP adjusted to account for differing willingness to pay. Global services are not adjusted for country-specific parameters. • A portion of the service value is applied to each hectare of land use based on the proportion of ecosystem service loss.

- For occupation of previously converted land, an average of the different marginal values due to differing scarcity of land in the past is applied.
- The current marginal value is only applied to new conversions.

4.1. Environmental outcomes

To estimate the environmental outcomes of land use, we must assign the correct portion of land use and then estimate the extent of ecosystem services lost on that land.

Figure 2: Steps for estimating environmental impacts of land use & conversion

1. **Apprortion land use to the company**
2. **Assign land use to eco-regions**
3. **Estimate the extent of ecosystem service loss in each eco-region in each location**

4.1.1. Step 1: Apportion land use to the company

The first step is to determine the quantity of land in use or converted, in hectares, by location. See Chapter 3 for potential metric data sources, and Appendix II for an example of a recommended method to estimate land use requirements for raw materials in agriculture.

While the company may not be the owner or operator of the land in use or under conversion, the objective of the E P&L is to calculate all the impacts associated with the company's operations and supply chain. As discussed in Chapter 1, we therefore assign the impacts of all land required to produce all goods and support all services associated with a company, to the company.

In some cases, land may have several uses, only some of which are associated with the company – for example, where several economic goods are produced in the same area. This is the case for intercropping and agroforestry and for co-products from the same production process. For example, leather is a co-product of cattle rearing along with meat, blood, bone and offal.

We apportion land use based on the average economic share which aligns with the economic incentives for land use and land conversion. An alternative approach frequently used in Life Cycle Assessment (LCA) literature is to apportion responsibility according to mass balance (each co-product is assigned impact proportionally to its weight relative to the total of all co-products). We believe, however, that an economic allocation better reflects the motivations behind land conversion and occupation, which are primarily economic decisions.

To this end, the land use from area, l, should be attributed to the company(ies) demanding each raw material output, o, of the land based on their relative value (Equation 1).

Equation 1: Calculate economic share

$$\boldsymbol{Economic\ share}\ (\%)_{\boldsymbol{ol}} = \frac{\boldsymbol{output\ value}\ (\$)_{\boldsymbol{ol}}}{\boldsymbol{total\ value\ of\ outputs\ from\ area_{l}}}$$

To apportion the land use, the total land area required is then multiplied by the percentage attributable to the raw material in question (Equation 2).

Equation 2: Calculate attributable area

$$\boldsymbol{Attributable\ area}\ (\boldsymbol{ha})_{\boldsymbol{ol}} = \boldsymbol{Area}\ (\boldsymbol{ha})_{\boldsymbol{l}} \times \boldsymbol{Economic\ share}\ (\%)_{\boldsymbol{ol}}$$

Box 1 presents an example for leather. Note that, where an output has no economic value, it would have no impacts allocated to it.

Box 1: Attribution of impacts to leather

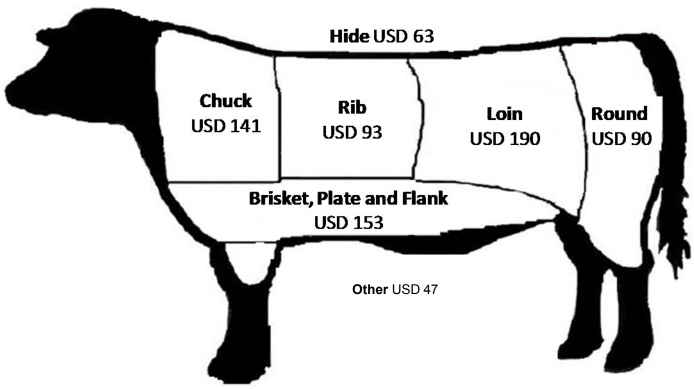
Since demand for leather only forms part of the economic case for cattle-rearing as a land use, the cost of impacts should be attributed between leather, meat and the other outputs proportionally to their relative value.

For a calf reared and sold in France, on average this gives 8 %, made up of a hide value of $105.17 and a total carcass value of $1,314.29. Another example for a US beef steer is presented below.

Some commentators argue that cattle hides are a pure by-product of meat production and therefore should not hold any responsibility for the environmental impacts of cattle rearing. However, it is more accurate to consider the hide as an economic co-product because for some farmers it can be an important part of the economic case for raising cattle. Indeed, the hide represents between 5 and 15% of the total value of cattle products, depending on quality and location. Beef Issues Quarterly, a major commentary on the industry supported by the National Cattlemen's Beef Association, notes that the value of a hide and other non-edible beef products 'are an important factor in beef packer returns'[7]. To support US ranchers, USDA Agricultural Marketing Service reports hide price data on a weekly basis[8].

Figure 3: The average value of different cattle meat and hide[9]

Cattle product	Value of total
Loin	25%
Plate, Brisket, Flank	20%
Chuck	18%
Rib	13%
Round	11%
Hide	7%
Other	6%



4.1.2. Step 2: Assign land use to eco-regions

Land in the desert provides different services from land on the coast. It is therefore necessary to designate distinct eco-region types to each area of land use in order to accurately assess the extent of ecosystem service loss.

At a global scale, arguably the most complete dataset for defining eco-regions is the WWF Wildfinder which presents the distribution of 16 biomes (based on a more detailed set of 867 ecosystem types), Figure 4. WWF's ecoregions have been classified using biogeographical data and are the result of an extensive collaboration with over 1,000 biogeographers, taxonomists, conservation biologists and ecologists from around the world. Existing estimates of ecosystem valuation are not available in sufficient quantity at this level of detail so we map the 16 biomes to six eco-regions, presented in Table 8. While these six groups are broad, increased differentiation is

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[7] Beef Issues Quarterly, http://beefissuesquarterly.com/Beefbyproductvaluesimportanttocattleprices.aspx Accessed July 2011.

[8] United States Agricultural Marketing Service. www.ams.usda.gov Accessed July 2011.

[9] Meat price and proportional weight data based on USDA and CattleFax, June 2010 report. Hide price data are an average of global prices across a bundle of hide types. Carcass and hide weights are assumed to be 277 kg and 27 kg, respectively.

introduced when they are valued using other location-specific information. In addition, as more original valuation estimates are conducted and our valuation dataset is updated, we anticipate being able to apply a more granular breakdown of eco-region type.

Where the exact locations of land use and land conversion activities are known, the WWF Wildfinder map can be used to identify the appropriate eco-region classification. However, where the exact locations are not known (for example if LCA is used to estimate the extent of supply chain activities), the relative coverage of each eco-region in a given location (e.g., at a country level) can be used to estimate a proportional split of eco-regions for each location. For example, areas of land used for cattle ranching in Australia could be assigned to an eco-region based on the distribution of each eco-region type within key cattle ranching areas of Australia.

At the end of this step, the land area (ha) required for the production of a given raw material should be available by eco-region within a given country and sub-national location.

Figure 4: WWF Wildfinder biomes

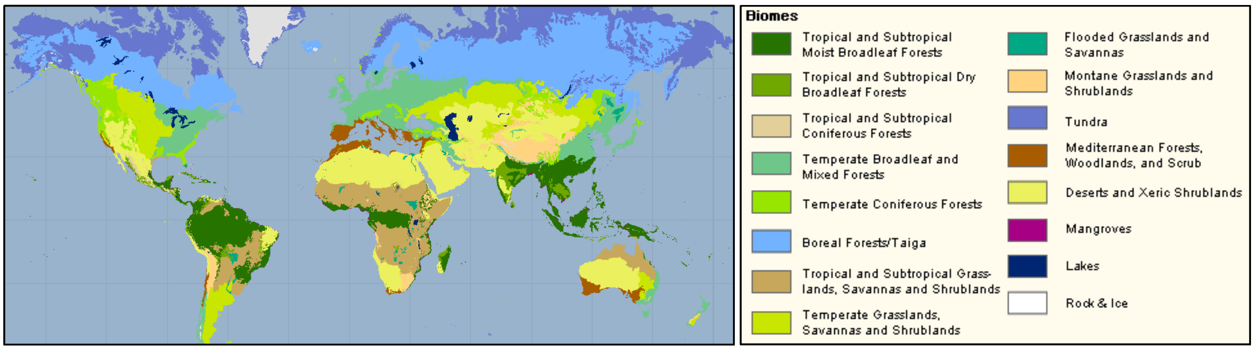


Table 8: Mapping biomes to eco-regions

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

4.1.3. Step 3: Estimate the extent of ecosystem service loss in each eco-region in each location

The objective of this step is to calculate the change in ecosystem services and therefore define the environmental outcomes of the land use in each location. The change in ecosystem service provision depends on the type of natural ecosystem displaced and the current land use activity. The extent of ecosystem service loss, expressed as a percentage, can vary significantly according to the type of land use change.

The extent of service loss can be determined directly when the exact location is known and data are available on specific estimates of ecosystem service provision. In such cases the pattern of conversion can also be taken into account – this can have an important impact on the extent of service losses, particularly in neighbouring natural areas (see discussion of edge effects in, for example, Chaplin-Kramer *et al.,* forthcoming, and Skole and Tucker, 1994).

However, for corporates with extensive global supply chains, it is not possible to measure specific changes in service provision for a range of different eco-regions. In such cases, the relative biomass (tonnes/ha) and species richness (count of functional groups) 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 ecosystem functioning at a general level (Hooper *et al.*, 2005). However, we recognise that this method is a crude approximation of the complexity of different ecological systems globally, with many other important interactions present in nature. We test for the potential significance of this assumption in our sensitivity analysis.

Table 9 identifies the proxy variables used for each ecosystem service which are intended for use when minimal details of the 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 for the gathering of food or fibre.

Where sustainable land management practices are employed in the production of the raw material in question, specific analysis is required to understand how the sustainable management practices in use affect the provision of each ecosystem service and percentage changes estimated for each.

Table 9: Proxies to estimate the relative change in ecosystem services for use where 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
	Bio-prospecting & medicinal plants	Total loss - N/A
	Ornamental products	Total loss - N/A
	Air purification	Partial loss - Biomass
Cultural services	Recreation	Partial loss - Biomass & species richness
	Spiritual and aesthetic	Partial loss - Biomass & species richness
	Cognitive and learning opportunities	Partial loss - Biomass & species richness
Regulating services	Stable climate	Partial loss - Biomass
	Pollution control and waste assimilation	Partial loss - Biomass & species richness
	Erosion control	Partial loss - Biomass
	Disease and pest control	Partial loss - Biomass & species richness
	Flood control and protection from extreme events	Partial loss - Biomass

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 3).

Equation 3: Calculate the extent of ecosystem service loss using a proxy

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

Where both biomass and species richness are used together, the average of the two percentages is used to infer the extent of ecosystem service loss. Table 10 presents an example of estimated ecosystem services loss for an area of pasture land use in an Australian Grassland.

Table 11 presents the data requirements and sources for estimating environmental outcomes and Table 12 presents the key assumptions.

Table 10: 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 Pasture	12,300 ha	Food from natural/semi-natural ecosystems	100%
			Fibre, other raw materials	100%
			Domestic and industrial water	100%
			Bio-prospecting & medicinal plants	100%
			Ornamental products	100%
			Air purification	78%
			Recreation	64%
			Spiritual and aesthetic	64%
			Cognitive and learning opportunities	64%
			Stable climate	78%
			Pollution control and waste assimilation	64%
			Erosion control	78%
			Disease and pest control	64%
			Flood control/protection from extreme events	78%

Table 11: Data required to estimate ecosystem service loss

Variable	Suggested data source(s)
Relative value of raw material vs other products from land	Company data, supplier questionnaires, FAO Stat and other commodity price data sources
Eco-region distribution	WWF Wildfinder
Biomass	IPCC (2007) provides generic biomass estimates for different land uses and approved methods for estimating a change following conversion. There is also a huge academic literature in carbon storage rates found in specific sites around the world.
Species richness	Ellis *et al*. (2012) estimate changes in species' richness (count of functional groups) associated with different land uses in different eco-regions. Other location specific data on species richness are available in the ecological academic literature.

Table 12: Key assumptions to estimate ecosystem service loss

Assumption	Explanation
Economic allocation of impacts between multiple raw materials from same area	Typically, allocation of impacts is either made on a per mass basis or using an economic allocation. We opt for an economic allocation because we believe that it better reflects the decision making behind land use choices.
Mapping of WWF biomes to 6 eco-regions is appropriate	The eco-regions are used as the starting point for the valuation, assuming that the underlying valuation studies are representative of the ecosystems classified within the eco-region. These broad groups are considered appropriate because the principle driver of value is the nature of the ecosystem service itself, together with characteristics of the benefiting population rather than type of ecosystem the service was derived from. For example, both mangroves and coastal marshes provide coastal protection which is valued in a similar way despite the significant differences in ecology.
Changes in biomass and species richness pre/post conversion is a suitable indicator for changes in ecosystem service provision	This adjustment is considered an acceptable approximation for applications at a global scale in the absence of other commonly applicable indicators with available data. Although we recognise that there are a great many other factors which affect ecosystem functioning, and that changes in species richness or biomass will not necessarily lead to proportional changes in functioning, for example due to keystone species or functional duplication across multiple species.

4.2. Societal impacts

In this section, we estimate the change in economic value associated with the loss of services to society estimated in the previous section (4.1).

The value of an ecosystem is driven by the different ways in which its' services provide benefits to society. The total value of a hectare of a given ecosystem is the sum of the value derived from each of the ecosystem services it provides.

Going into the valuation, the impacts are presented in hectares of land use with an associated percentage loss of each ecosystem service split by the eco-regions present in each location. The valuation method discussed here considers, in economic terms, how these losses in ecosystem services affect people.

Our methodology draws on the wealth of research into ecosystem valuation. Our database contains 1,500 individual estimates of ecosystem service values. These are used to estimate values for ecosystem services from different eco-regions in different contexts by averaging across the available studies.

Figure 5: Methodological steps for estimating societal impacts of land use conversion

1. **Standardise valuation estimates in database to be expressed in 2011 USD per hectare per year**
2. **Group ecosystem services by eco-region type and calculate the median value of each ecosystem service. This represents the current marginal value.**
3. **Calculate the average of marginal values through increasing scarcity (and time), assuming a relationship between scarcity and value**
4. **Apply the ecosystem service values to changes in ecosystem services associated with land occupation or conversion to estimate the impacts**

4.2.1. 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. We therefore make a number of adjustments to the individual published estimates contained within the database to express them in 2011 USD per hectare per year:

- Converting values to per hectare per year: Some values in the original publication are expressed as per household, 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 applied. This ensures inflation is applied consistently and avoids some potentially large fluctuations based on variations in inflation.
- Applying income adjustment to correct for Purchase Power parity (PPP) differences across estimates: we apply current PPP adjustments (based on the ratio of local Gross National Income (GNI) to U.S. GNI) to local and national ecosystem services. We do not apply income adjustments to global services as these are typically already based on a global (or at least international) willingness to pay (includes a classification of ecosystem services by scale of the service delivery – local, national or global.)

At this stage, 284 values have to be excluded from the database, reducing our sample size, either because it is not possible to express them as per hectare per year values, or because they do not have a specific eco-region associated to them. Every effort is made to include values where possible. For example, estimates of grouped ecosystem services (e.g. labelled as TEV or total of provisioning services) are retained and estimates covering several countries are also retained (with exchange rates calculated based on the relevant basket of economies).

4.2.2. Step 2: Estimating 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 ecosystems today, given the prevailing level of ecosystem services (and scarcity of natural ecosystems). The objective of the methodology is to estimate the current marginal value of the different ecosystem services provided by each of the 6 eco-regions in different contexts.

Remove outliers by ecosystem service and eco-region

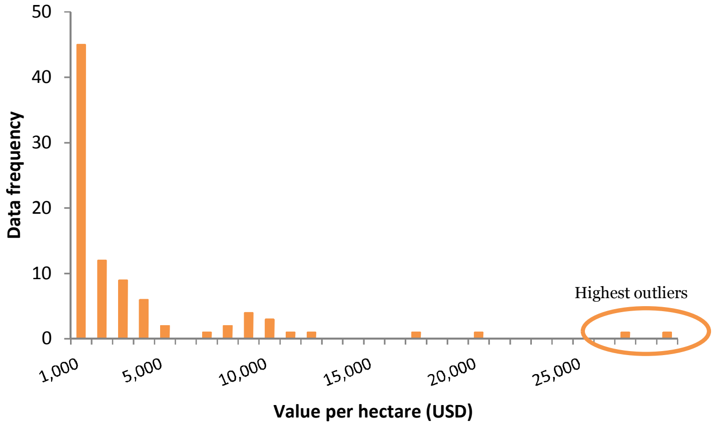
To estimate the current marginal value of ecosystem services we take the average across estimates in the database. The estimates are split by ecosystem services within each of the 6 eco-regions. We do not segregate the estimates by country or by geographical region. We consider averaging across eco-regions to be a better approach than averaging estimates by country or region because (i) there is more similarity in terms of ecosystem services across eco-regions in different countries than between different eco-regions in the same country and (ii) there is insufficient data coverage to provide reliable and comparable estimates by country or region. TEEB takes a similar approach, emphasising 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 quite high values. As a result, the mean values tend to be higher than the medians, with quite large standard deviations (see Figure 6 for an example).

This analysis calculates an average value in order to give an indication of the central tendency within our distribution of values of ecosystem services in a given eco-region. In nature there is considerable variation within eco-regions, furthermore there is even greater variation in the way human society interacts with (and therefore gleans value from) ecosystems. It is therefore 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 skew the results disproportionately (even if the median is used). We therefore opt to exclude estimates which are 2 standard deviations higher (or lower) than the mean.

Figure 6 presents an example with results before and after the exclusion of two outliers. 146 estimates are excluded across all ecosystem services and eco-regions, leaving 1,061 estimates. An alternative approach would be to retain all values in our pool for calculating marginal values, but we feel this exposes the mean and median values to bias and reduces reliability and comparability across eco-regions.

Figure 6: Example distribution of all estimates for food services from coastal wetlands (USD/ha/yr)



Calculating average values for ecosystem services by eco-region

Following exclusion of 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 give the average total economic value of each of our 6 eco-regions. In doing so, we exclude ecosystem services for which we only have one value estimate. 13 estimates are excluded across all ecosystem services and eco-regions. Table 2 presents the final coverage of ecosystem services included in the valuation for each eco-region.

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 can be included at the point of aggregation to a TEV for each eco-region. They are weighted by the number of estimates, to give all data points equal weight in the average TEV (for example, 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 this aggregated TEV). Table 13 shows the results of this analysis for Tropical Forests. These results illustrate the difference between the mean and median estimates. We recommend using the median because it is a more robust estimate of the central tendency, particularly given the long tailed distribution of the estimates.

Table 13: Ecosystem service values for Tropical Forests

		Mean USD/ha/yr	Median USD/ha/yr	Count (n)	Min USD/ha/yr	Max USD/ha/yr	Standard deviation USD/ha/yr
Provisioning services	Food from natural/semi-natural ecosystems	454	236	38	<1	3,984	862
	Fibre, other raw materials	906	749	36	6	3,896	914
	Domestic and industrial water	338	62	8	<1	2,648	918
	Bio-prospecting & medicinal plants	204	16	47	<1	2,049	479
	Ornamental products	excluded[10]	excluded	1	67	67	
	Air purification	256	256	2	11	501	346
Cultural services	Recreation	1,853	510	24	4	8,991	2,794
	Spiritual and aesthetic	-	-	-	-	-	
	Cognitive and learning opportunities	-	-	-	-	-	
Regulating services	Stable climate	193	190	13	1	8474	243
	Pollution control and waste assimilation	1,323	1,084	7	1	4,384	1,620
	Erosion control	2,385	1,376	12	16	13,117	3,800
	Disease and pest control	excluded	excluded	1	14	14	
	Flood control/protection from extreme events	107	39	8	12	376.77	139
	Cultural service [general]	excluded	excluded	1	12	12	
	TEV	2,144	2,504	6	162	4,069	1,625
	TOTAL	**7,846**	**4,458**	**197**			

[10] Services for which there are only one estimate are excluded.

Calculating values of ecosystem services by country

The objective of the calculations in 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 are dependent on different services in different contexts. For example, rural communities tend to be more reliant on ecosystem services (directly or indirectly), and 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.[11] Similarly, if those 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 for country level data, however where more locally specific information on land use is available the adjustments should be made to reflect the local conditions. As before, services accruing at the international level are not adjusted for local conditions because the values in the underlying estimates already reflect international preferences. There are two key adjustments applied to local and national services to transfer the median eco-region estimates to different countries:

1. **Income adjustment**

   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.

2. **Population dependancy and distribution**

   The proportion of the population living in rural areas, together with the concentration of the urban population is used to adjust country-specific values, such that countries with a higher proportion of rural population have higher valuation estimates. A population adjustment factor between 0 and 1 is calculated based on 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 14 presents the value of ecosystem services from tropical forests in a number of countries.

Table 14: Ecosystem service values of tropical forests in different countries

USD/ha/yr		Brazil	Colombia	Congo, Rep.	Indonesia
Total		**1,377**	**1,159**	**649**	**845**
Food from natural/semi-natural ecosystems	Local	54	41	11	23
Fibre, other raw materials	Local	170	130	36	72
Domestic and industrial water	Regional	14	11	3	6
Bio-prospecting & medicinal plants	Global	16	16	16	16
Air purification	Global	253	253	253	253
Recreation	Regional	116	88	24	49
Stable climate	Global	188	188	188	188
Pollution control and waste assimilation	Regional	246	188	52	104
Erosion control	Regional	312	238	66	132
Flood control and protection from extreme events	Regional	9	7	2	4

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

The principle data requirements and key assumptions required for the calculations are presented in Table 15 and Table 16.

Table 15: 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 to this from journals, including: Journal of Environmental Economics and Management, Ecological Economics, American Journal of Agricultural Economics, Land Economics, Environmental and Resource Economics, Environment and Development Economics, 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 population between urban and rural areas. OECD calculates a population concentration index.

Table 16: Key assumptions to estimate the current marginal value of ecosystem services

Assumption	Explanation
Underlying estimates provide a representative sample of the ecosystem services provided by each of the 6 eco-regions	Although, the our database of values is arguably the most comprehensive repository of primary estimates currently available, these are distributed across 14 ecosystem services and 6 eco-regions, the number of values for each ranges from 2 to 90 (after all exclusions). Although there is less confidence at the lower end of this range, we believe these data are sufficient to give a good indication of the likely scale of value that can be delivered by different ecosystem services.
Estimates more than 2 standard deviations from the mean are excluded	The objective of this valuation is to provide an estimate of the average value of different ecosystem services in a generalised eco-region. It is clear from studying the largest outliers that these are very special cases and do not give a fair representation of the average values (e.g. strawberry growing in deserts) and are therefore 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 one another. Given the distribution of the data and given the objective of the study is to identify the most likely impact, we consider the median to be 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 can also proxy for appetite for trading off environmental goods (some of which could be considered a luxury good in the short term) for other economic gains. Population adjustments reflect the number of people who are likely to benefit from the services and their reliance on ecosystem services.

4.2.3. 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 represent the impacts of additional land conversions today. This value is applied to new in-year conversions.

However, 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 through time. There are two factors which contribute to this: the increasing scarcity value and the increasing marginal damage costs associated with cumulative environmental degradation. This is particularly the case as ecosystems display threshold effects, whereby the damages increase exponentially after a particular point of loss of functioning.

Rather than applying the current marginal value, the appropriate measure for land converted in the past is the average of the marginal values through 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 being equal). Box 2 illustrates this point with an example.

Box 2: Whythe 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), depicted in Figure 7.

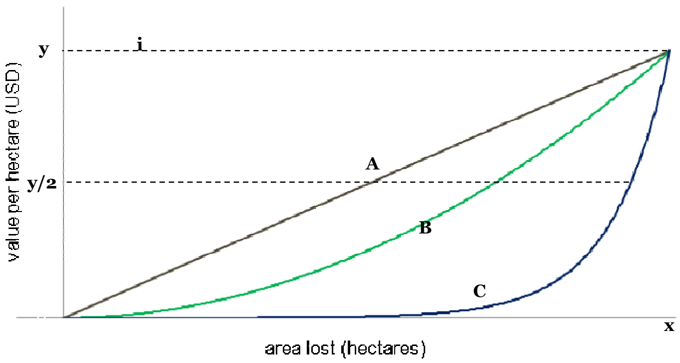
Year 1	**$1**							
Year 2		**$2**						
Year 3			**$3**					
Year 4				**$4**				
Year 5								

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 value. In year 5, the total cost of lost ecosystem services is $10 and the average marginal value through 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 of marginal values we need to assume a relationship between the extent of natural land areas lost (through time) and the corresponding value loss associated with converting an additional hectare. Figure 7 illustrates a number of the possible relationships. The graph demonstrates that, if the current marginal value (y) was applied to all areas of land use, the impact given by the area under line i would be a gross over-estimate. Three different curves are shown to illustrate the possible relationship: in curve A, costs increase linearly while, in B and C, the incremental costs increase slowly at first and then more rapidly as a greater total area is lost. Whilst one of these relationships may hold true, the actual relationship will differ across ecosystem services in different contexts.

Figure 7: Ecosystem services have increasing marginal value as more natural areas are lost



Given this, we assume a linear relationship (curve A) in our calculations. This is a conservative approach and leads to higher estimates of potential impacts (since any other convex relationship would suggest impacts of past conversions are lower). In this instance, it is straightforward to calculate the average marginal cost, as it is half the current marginal cost. Table 17 presents the average marginal ecosystem service values for a number of countries.

Table 17: Average marginal ecosystem service values for tropical forests in different countries

		Brazil	Colombia	Congo, Rep.	Indonesia
Total		**689**	**579**	**325**	**422**
Food from natural/semi-natural ecosystems	Local	27	20	6	11
Fibre, other raw materials	Local	85	65	18	36
Domestic and industrial water	Regional	7	5	1	3
Bio-prospecting & medicinal plants	Global	8	8	8	8
Air purification	Global	126	126	126	126
Recreation	Regional	58	44	12	24
Stable climate	Global	94	94	94	94
Pollution control and waste assimilation	Regional	123	94	26	52
Erosion control	Regional	156	119	33	66
Flood control and protection from extreme events	Regional	4	3	1	2

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

Once 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 E P&L estimate of the lost value as a result of ecosystem services reductions associated with land use.

For newly converted land within the year of analysis, we use the current marginal value of ecosystem services. For previously converted land, we use the average of marginal values.

In cases where services are only reduced, we calculate the appropriate portion of lost value based on the percentage change in service provision (Equation 4).

Equation 4: Calculate the lost ecosystem service value, per hectare per eco-region

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

We then calculate the total losses in ecosystem service. This value is then multiplied by the area over which it has been lost or reduced (Equation 5).

Equation 5: Calculate the total lost ecosystem service value, per eco-region

$$\begin{aligned}&\boldsymbol{Total\ lost\ ecosystem\ serice\ value}\ (\$)_{\boldsymbol{el}}\\&\quad= \boldsymbol{Attributable\ area}\ (\boldsymbol{ha})_{\boldsymbol{ml}} \times \boldsymbol{Lost\ ecosystem\ service\ value}\ (\$/\boldsymbol{ha})_{\boldsymbol{el}}\end{aligned}$$

Calculating the global societal cost requires simply summing the ecosystem services value from each region and impact. The key assumptions are presented in Table 18.

Table 18: Key assumptions to value ecosystem service losses

Assumption	Explanation
Ecosystem service value is directly proportional to the scarcity of ecosystem services (as per relationship *A* in Figure 7)	The actual relationship will be different for different ecosystems in different contexts. We take the conservative approach and assume a linear relationship.

Sensitivity analysis

Module-specific sensitivity analysis

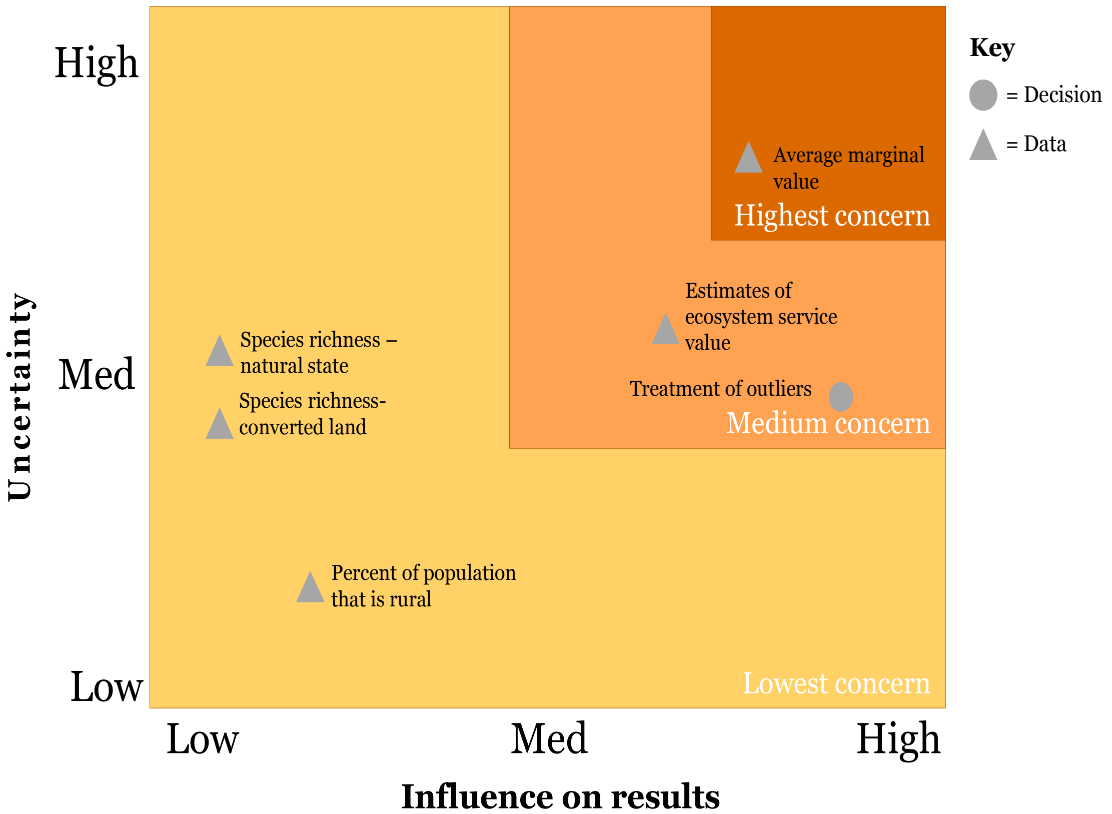
Overall summary and considerations for model use

This section presents a summary of the findings of our sensitivity analysis, more detailed discussion on the parameter influence on results and uncertainty follows.

In order to assess how sensitive our esimates are to different input parameters and decisions we flex these and examine how our results change. This allows us to categorise the parameters as Low to High Concern.

The key parameters tested in our sensitivity analysis are mapped in Figure 8 on an influence/uncertainty matrix. The underlying ecosystem service values, and the treatment of outliers within these data, are the most important parameters. The underlying valuation estimates and the treatment of outliers are classified as Medium Concern, while the calculation of the average marginal value across changing scarcity has been classified as of High Concern. This is because there are no data on which to base the average marginal calculation, so we have opted for the most conservative estimate.

Figure 8: Impact/uncertainty matrix summarising the sensitivity assessment summary for key parameters, split into data and decisions



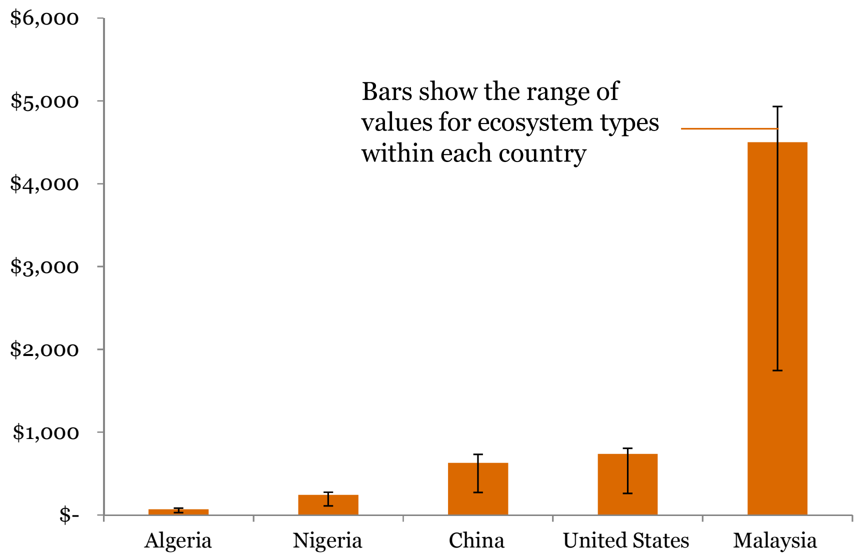
Selection of parameters to test

To recap, there are three steps to estimating land use impacts, with this methodology considering only steps two and three:

Obtain environmental metric data (quantity and ecosystem type)
Quantify environmental outcomes
Estimate societal impacts

While the quantity of land use has an important impact on the results (it has a directly proportional relationship) it is not considered in this methodology or sensitivity analysis because it does not relate to the valuation methodology. Related to this is the correct identification of the type of ecosystem which are being occupied or converted. Figure 9 presents the range of values of different ecosystems in selected countries. As this also relates to the quality of the metric data, rather than the valuation methodology, it is not considered this sensitivity either.

***Figure 9*: The average across all end use types for the value of a converted hectare of land in illustrative countries**



The second step estimates the proportion of ecosystem service losses following conversion to the specified land use. In this methodology we recommend the use of site specific data, but present an alternative proxy based on change in species richness and biomass where this is unavailable, which we test here.

The third step estimates the change in welfare associated with the change in ecosystem services. We test both the data underlying the valuations and our method of applying the valuations to different countries.

Parameter influence on results

Table 19 presents the influence of changes in the key parameters for Steps 2 and 3 of the calculation.

The parameters and decisions which have the greatest influence on the results are the underlying data, and its treatment in the calculation.

***Table 19:* Assessing parameter impact by assessing the change to the overall societal cost per unit of land converted to cattle grazing**

Variable	Flex	Impact summary [12]	Algeria (% change)	China (% change)	Malaysia (% change)	Nigeria (% change)	US (% change)
Average marginal value	Line B in Figure 7	High	-56%	-56%	-56%	-56%	-56%
Estimates of ecosystem service value	+10%	Med/High	10%	10%	10%	10%	10%
Exclude outlier valuation estimates (central estimate 2 St. dev.)	3 St. dev.	High	14%	16%	14%	22%	11%
	1 St. dev.	High	-25%	-16%	-21%	-21%	-20%
Species richness in natural state	+10%	Low	0.28%	0.12%	0.08%	0.05%	0.02%
Species richness in converted land uses	+10%	Low	-0.54%	-0.33%	-0.37%	-0.16%	-0.06%
Percent of population - rural	+10%	Low	0.35%	0.68%	1.08%	0.69%	0.91%

[12] Low = average response for overall impact for five countries is less than 1%

Med = average response for overall impact for five countries is 10% or less

High = average response for overall impact for five countries is greater than 10%

Parameter uncertainty

Table 20 presents a qualitative ranking of parameter uncertainty.

The most uncertain parameters and decisions are those regarding the underlying ecosystem values in the database. The estimates of ecosystem service values show significant variation (see Table 13). While this is to be expected due to natural and socio-economic variation (as discussed in Section 4.2.2) it does reflect the potential variation in values of the ecosystems occupied or converted by the corporate.

***Table 20:* Assessing the uncertainty of key parameters based on the reliability of the measurement and the variance in attempts to measure the parameter**

Variable	Uncertainty rating	Reliability/quality of measurement	Indicative variance of the number measured
Average marginal value	High	There are no measures of change in marginal value with increasing scarcity for ecosystem services; the most conservative option was used.	An order of magnitude
Estimates of ecosystem service value	Med	Well documented uncertainty in non-market valuation. Database represents most comprehensive available, and precedent for use in meta-analysis by TEEB, and subsequent studies	An order of magnitude
Exclude outlier valuation estimates	Med	Decision based on distribution of data and analysis to remove outlying data points which are disproportionately driving results	N/A
Species richness in natural state	Med	Base data are measured for currently intact ecoregions (by Elis et al.,) in equal-area hexagonal grid cells of 7,800 km^2 and averaged at a regional level. Best available if global coverage is required. Additional uncertainty given natural state no longer in existence for converted areas.	An order of magnitude
Species richness in converted land uses	Med	Base data are measured for currently converted land (by Elis et al.,) in equal-area hexagonal grid cells of 7,800 km2 and averaged at a regional level. Best available if global coverage is required.	An order of magnitude
Percent of population - rural	Low	Census data, assembled by World Bank into global database.	~10%

Appendices

Appendix I – Bibliography

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Appendix II – Productivity modelling to estimate land use area

For most companies that do not manage the production of their raw material inputs, it will be difficult to identify the areas of land which are used to produce the materials. However, companies should know the quantity of raw materials they use and may also know from which countries these materials are sourced. We can use this information to estimate the land use by assigning the company a share of the total impact of the raw material production industry in each location. For example, if, in a given country, there are some producers which are high impact and others which are low impact, this approach estimates the impact of both and assumes the company draws on each producer relative to the volume of their production.

To identify the location and estimate the quantity of land used or converted, the following steps are necessary for each type of raw material, details on the data sources and assumptions follows.

Step i: Collecting company data on the quantity of raw materials used to produce their products in the year of interest

Raw material data should be collected from the company and its suppliers. This should cover the most land-intensive materials (typically agricultural materials) used in any meaningful quantity. For each raw material, the data should represent the total quantity required for production, including any wastage from production processes.

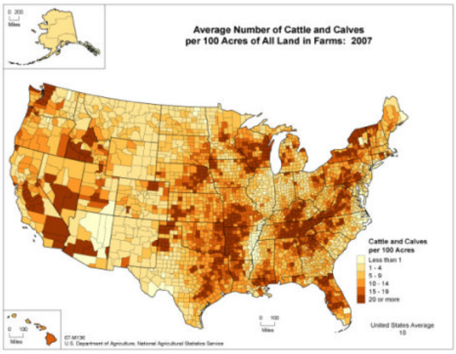
Step ii: Identifying the location(s) of production for the raw material

It is important to know the source location because this defines what type of ecosystem services are being lost and to what extent different populations are affected. In some cases, the source location of the material will be easy to identify, particularly where it is specified by the company as part of its procurement policies for quality assurance. In other cases, it may be necessary to ask suppliers. Where suppliers are unable to provide the locations, trade flow data can be used to identify the most likely source country of raw materials based on imports.

Where only the country of origin is known, additional publically available information can allow greater precision on the likely sub-national locations of production. This is particularly important for countries with many different eco-regions because it will allow more precision in the identification of environmental outcomes and the subsequent valuation of impacts.

Figure 10 presents an example: the range in cattle density across the United States which can be used to identify the main locations contributing to the countries meat and leather production

Figure 10: Cattle density across the United States



Step iii: For each location calculating the production yield

Production yields describe the amount of land required to produce a unit of the raw material. Production yields can be highly variable: for example, the yield in hectares of production area per head of slaughtered cattle in Australia is 42.9, while it is 7.1 in the US and 0.3 in Japan . Intra-country variation can be equally large, as illustrated in Figure 10. It is recommended to calculate yields at least at a sub-national level: this will also allow more accurate identification of eco-region type (discussed in Step 2).

Equation 6 describes the yield in hectares per unit production of material m in location l.

Equation 6: Calculate the yield

$$\boldsymbol{Yield\ (ha/unit)_{ml} = \frac{total\ area\ (ha)_{ml}}{total\ production\ (unit)_{ml}}}$$

Step iv: Calculating the quantity of land required from each location

To calculate the quantity of land used for material m in location l, the yield is multiplied by the quantity of raw material used by the company, as per Equation 7.

Equation 7: Calculate the land use area

$$\boldsymbol{Area\ (ha)_{ml} = yield\ (ha\ per\ unit)_{ml} \times quantity\ (unit)_{ml}}$$

Where locations are only known at a country level and data on the distribution of production in the country is being used to gain precision, we assume that the company demands from all in-country producers proportionally to their output. In such cases, the raw material demand from the country should be distributed across sub-national locations according to the relative production of each sub-national location. Equation 7 is used to calculate the land area in use for the raw material demand within each sub-national location. The sum of these areas represents the total area for the raw material demand for the country. However, at this stage, it is better to keep the land use areas within each sub-national location separate for examining the eco-regions in Step 2.

Table 21: Variables to estimate land use from productivity modelling

Variable	Suggested data source(s)
Quantity of raw material used	Company data, supplier questionnaires
Source location of raw material	Company data, supplier questionnaires. Other options include trade data from UN Trade, government statistics or multi-region input-output models
Raw material production yield by location	FAO Stat, government statistics, industry reports

Table 22: Key assumptions to estimate land use from productivity modelling

Assumption	Explanation
Where precise locations of land use are not known, the raw materials are assumed to be drawn from all in-country producers relative to their production	Intra-country variance of impacts per unit of raw material can be high: this approach gives the best approximation of the impacts of sourcing from a given country.

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Valuing corporate environmental impacts: Solid waste disposal

PwC methodology paper

Version 2.7

pwc

Contents

Table of Tables

Table of Figures

Table of Equations

Abbreviations and acronyms

Abbreviation	Full name
CH_4	Methane
CO	Carbon monoxide
CO_2	Carbon dioxide
CO2e	Carbon dioxide equivalent
CVM	Contingent value modelling
DOC	Degradable organic carbon
E P&L	Environmental Profit and Loss
EEIO	Environmentally extended input-output modelling
ERQ	Environmental Regulatory Quality
EU	European Union
GHG	Greenhouse gases (CO_2, CH4, N2O and O3)
GNI	Gross national income
HARAS	Hazard Rating System model
HDM	Hedonic demand modelling
HPF	Hedonic pricing factor
HM Treasury	Her Majesty's Treasury (United Kingdom)
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
IQ	Intelligence quotient
kWh	Kilowatt hour
LCA	Life cycle assessment
LFG	Landfill gas
LFGTE	Landfill-gas-to-energy
MCF	Methane correction factor
MSW	Municipal Solid Waste
NH_3	Ammonia
N_2O	Nitrous Oxide
NO_x	Nitrogen monoxide (NO) and Nitrogen Dioxide (NO2)
OECD	Organisation for Economic Co-operation and Development
PM_{10}	Coarse particulate matter (diameter under 10μm)

$PM_{2.5}$	Fine particulate matter (diameter under 2.5µm)
PPP	Purchasing power parity
SCC	Societal cost of carbon
SO_x	Sulphurous oxides
VSL	Value of statistical life
WTE	Waste-to-energy

1. The environmental impacts of solid waste disposal

1.1. Introduction

Corporate activities in all sectors result in some level of solid waste generation. 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 these costs in monetary terms.

Fluid waste is considered in the PwC methodology paper *Valuing corporate environmental impacts: water pollution;* gaseous waste is considered in the PwC methodology paper *Valuing corporate environmental impacts: air emissions*. The majority of material impacts associated with solid waste are covered in this paper, but two classes of related impacts are partially addressed in other papers. For greenhouse gas (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 but valued according to their respective PwC methodology papers: *Valuing corporate environmental impacts: greenhouse gases* and *Valuing corporate environmental impacts: air pollution*. 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 the design or production inefficiencies which may be indicated by the presence of waste.

1.2. Overview of impact area

For solid waste disposal, the type of waste and the method of its disposal are key factors that dictate the profile of the resultant environmental outcomes. The common types of waste, disposal approaches and environmental outcomes are listed below. The impact pathway (Figure 1) describes how these factors influence environmental outcomes and subsequently impact people.

1.2.1. Types of waste

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

- **Hazardous waste:** Waste that is defined as particularly dangerous or damaging to the environment or human health, usually through inclusion on an official listing by the relevant regulator.

  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 has an important influence on the type and extent of impacts associated with different disposal techniques. More specific classifications may be required to ensure the right impacts are allocated to the waste produced by a given activity. For example, inert waste is a subclass of non-hazardous waste which is chemically unreactive and does not decompose, and therefore does not release GHGs.

1.2.2. 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 below:

- **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 the landfill also has a disamenity impact for surrounding residents and visitors to the vicinity. Landfill quality varies dramatically. Here we use the term to cover everything from unmanaged dumpsites where leachate and GHGs can escape unabated into the environment at one end of the spectrum, to carefully managed, impermeably lined, sanitary landfills where these emissions are collected and processed, and in some cases combusted 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. 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 these emissions (which is done in the same way as 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 on an appropriate basis. Recycling does not therefore have a dedicated section in this valuation paper.
- **Specialist processing:** Local regulations may mandate or recommend specialist treatment of some solid waste products, particularly hazardous waste (such as hydrocarbons and radioactive waste). The nature of the treatment and the resulting impacts will be highly specific to each situation and we do not therefore present a generalised methodology here.

The transport of waste to the treatment site also creates impacts, such as GHGs and air pollution from the burning of fuel. The valuation of these impacts is covered by the relevant methodology paper. Given that these impacts are driven by the creation of waste they should be allocated to waste in the presentation of E P&L results.

1.2.3. Environmental outcomes and societal impacts

Waste disposal can lead to a number of environmental outcomes which bring adverse societal impacts. These include the following impact areas:

- **Disamenity:** The loss of environmental quality resulting from the presence of a waste management site. The presence of waste sites can lead to a range of aesthetic changes in the environment that cause displeasure to people in the immediate vicinity, including visual intrusion, odour, noise, and pests.
- **Leachate release:** The release of liquid produced in landfill sites, principally due to the infiltration of rainfall. As waste breaks down, the liquids produced can percolate through the landfill and contaminate the soil and local ground and surface water. This has the potential to affect agricultural output, as well as the health of ecosystems and the local population.
- **Climate change:** Waste disposal in both landfill and incineration contribute to climate change by releasing GHGs into the atmosphere (see PwC methodology paper: *Valuing corporate environmental impacts: greenhouse gas emissions*); the majority of the GHGs from incinerators are in the form of carbon dioxide (CO_2) while those from landfill sites are methane (CH_4).
- **Air pollution:** The emission into the air of substances that reduce air quality (see PwC methodology paper: *Valuing corporate environmental impacts; air pollution*). In the context of waste disposal, reduced air quality is a by-product of incineration. Societal costs tend to be dominated by health impacts, but visibility, agriculture, forests, the built environment, and amenity value are also affected. The most relevant pollutants to waste disposal include particulate matter ($PM_{2.5}$ and PM_{10}), nitrogen oxides (NO_x), sulphurous oxides (SO_x), carbon monoxide (CO), ammonia (NH_3), volatile organic compounds (VOCs), heavy metals and dioxins.
- **Land use:** Individual waste management sites can occupy large areas and, if poorly managed, may contaminate the land they occupy and surrounding areas (see PwC methodology paper: *Valuing corporate environmental impacts: land use*). Land contamination caused by landfills is considered under leachate.

1.3. Impact pathway

In order to value corporate environmental impacts, we need to understand how the treatment and disposal of solid waste affects humans. Therefore, we define impact pathways that describe the links between corporate activities, the environmental outcomes from those activities, and the resultant societal impacts. Our impact pathway framework consists of three elements:

- **Impact drivers:**
  - *Definition:* These drivers are expressed in units which can be measured at the corporate level, representing either an emission to air, land, or water; or the use of land or water resources.[1]
  - *For solid waste disposal:* The type and quantity of waste produced and its treatment.
- **Environmental outcomes:**
  - *Definition:* These describe actual changes in the environment, which result from the impact driver (emission or resource use).
  - *For solid waste disposal:* These include reduced air, water or landscape quality.
- **Societal impacts:**
  - *Definition:* These are the actual impacts on people as a result of changes in the environment (environmental outcomes).
  - *For solid waste disposal:* These may include negative impacts on human health, agricultural outputs and reduced enjoyment of the environment.

The three stages of the impact pathway are shown in Figure 1 overleaf. Solid waste disposal is a complex pathway, with multiple impacts each playing a role in multiple environmental and societal outcomes. The label 'out of scope' identifies elements of the impact pathway, which are not addressed in detail in our methodology. The reasons for any such limitations of scope are explained at the end of this chapter.

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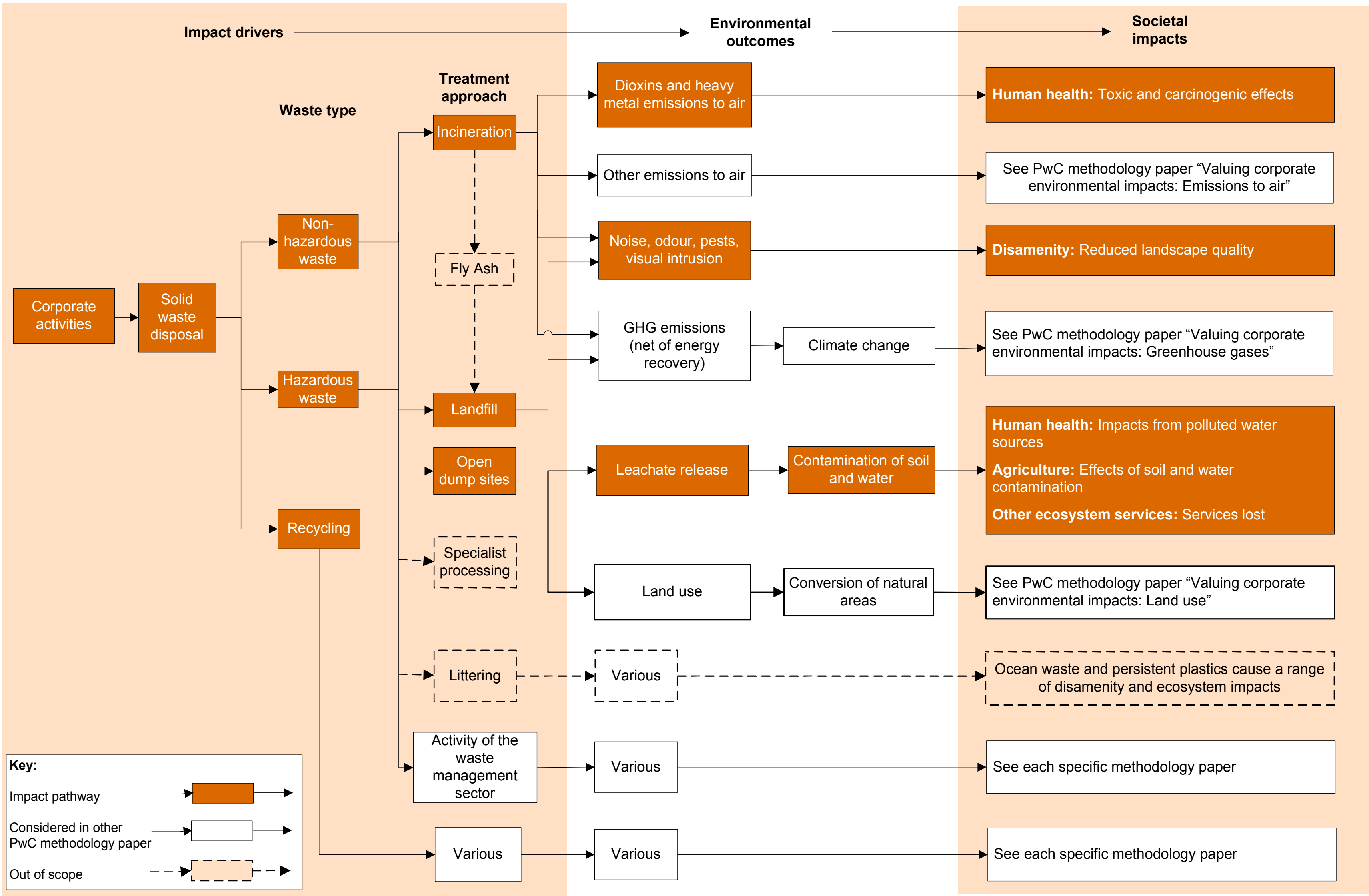
[1] A note on language: In this report, the measurement unit for any 'impact driver' is an 'environmental metric.' Therefore, solid waste disposal is the impact driver, and tonnes of waste are the environmental metrics.

Figure 1: Impact pathway for solid waste



1.4. Prioritising which impacts to quantify and value

This section outlines the key solid waste disposal impact areas and pathways that will be quantified (in biophysical units) and valued (in monetary terms). It also defines those impact areas and pathways that are beyond the scope of this methodology.

Consistent with the literature on disposal methods, we focus on the impacts associated with treating solid waste through incineration and landfill, including unmanaged dumpsites (COWI, 2000b; Eshet et al., 2005b). For most businesses, these two disposal methods will capture the vast majority of the associated environmental outcomes. In cases where a business has a significant volume of waste treated or disposed of in other ways, an additional methodology to quantify environmental outcomes may be required. The impacts of recycling should be quantified as per other industrial processes and valued based on the appropriate methodology paper. It does not therefore have a dedicated section in this report.

We seek to include as many impact areas as possible and only exclude areas where there is particularly strong evidence of low relative materiality in the literature. The impacts that this PwC methodology paper covers and excludes are summarised in Table 1. More detail can be found in later sections.

Table 1: Summary of valuation priorities

Impact pathway	**Quantified in this PwC methodology paper**		**Valued in this PwC methodology paper**	
	Incineration	**Landfill / dumpsite**	**Incineration**	**Landfill / dumpsite**
Disamenity	✓	✓	✓	✓
Leachate	× Immaterial	✓	× Immaterial	✓
Greenhouse gas emissions	✓	✓	Other PwC methodology paper	Other PwC methodology paper
Air pollution	✓	× Immaterial	Other PwC methodology paper	× Immaterial
Land use	Other PwC methodology paper	Other PwC methodology paper	Other PwC methodology paper	Other PwC methodology paper
Recycling	Treated like any industrial process	Treated like any industrial process	Treated like any industrial process	Treated like any industrial process
Specialist processing	Not covered	Not covered	Not covered	Not covered
Littering and ocean waste	Not covered	Not covered	Not covered	Not covered

1.4.1. Impacts covered by this methodology paper

1.4.1.1. Disamenity

Disamenity includes unpleasant odours, visual intrusion, noise and pests. The academic literature consistently shows that the societal costs from disamenity can be significant in the areas surrounding waste disposal sites (e.g. as indicated by a decrease in house prices).[2]

1.4.1.2. Leachate

Leachate from landfill can result in material impacts if a site is not properly managed. Impacts will vary depending on the characteristics of the landfill site (e.g. whether it has a liner system that can adequately contain leachate), and the quantity and composition of leachate, which varies over time and is particularly dependent on waste composition and weather conditions (COWI, 2000a).

Leachate from incineration facilities is not considered in this paper; this view is consistent with the literature. At incineration sites, waste is generally only stored in modest quantities for short periods of time prior to burning. Waste ash from incinerators is typically disposed of through stabilisation (e.g. in concrete) prior to landfill.

1.4.2. Impacts partially covered by this methodology paper

1.4.2.1. Greenhouse gas emissions

The societal impacts of GHGs arise from their contribution to climate change. This is already having or is expected to have a range of negative consequences, including impacts on health, damage to crops and infrastructure, and disruption to ecosystems (see PwC methodology paper: *Valuing corporate environmental impacts: greenhouse gases*). GHGs are produced by the decomposition of waste materials at landfill sites and from burning waste in incinerators.

1.4.2.2. Air pollution

At landfills, emissions to air (i.e., 'landfill gas' (LFG)) are generally 50-55% CH_4 and 45-50% CO_2 (which are both included as GHGs, and are in scope), with small amounts of other gases including nitrogen dioxide (IPCC, 2006c; Rierdevall et al., 1997, COWI, 2000a). The small quantities of other gases from landfill are deemed immaterial and are not considered in this paper (consistent with IPCC, 2000a; COWI, 2000a; Eunomia, 2002).

One of these other gases may be hydrogen sulphide; however the concentrations are not typically sufficient to result in impacts to human health. At low concentrations the principle impacts are associated with the odour, which is included in the disamenity impact pathway.

Waste incineration produces a wider variety of air pollutants. PM_{10}, NO_x, and SO_x are particularly important (EXIOPOL, 2009) and are quantified in this methodology, but valued in the Air Pollution methodology paper. Small amounts of other pollutants, such as dioxins and heavy metals are considered, as they can have significant societal consequences (e.g. causing cancer or loss of intelligence via developmental harm).

1.4.3. Impacts covered by other PwC methodology papers

1.4.3.1. Land use

Landfills receive waste from a number of companies and households over many years, and the modelled impact of an individual firm in a given year is generally small. We do not present a specialist methodology here, but where appropriate the impacts can be measured and valued in line with the methodology presented in the PwC paper: *Valuing corporate environmental impacts: land use.*

[2] In some locations open landfill sites may provide an important source of livelihoods for local communities (e.g. waste pickers). This should be taken into consideration if it is likely to be a significant factor given the balance of locations being considered.

1.4.3.2. Recycling

Emissions and resource use associated with recycling should be quantified in the same way as for other industrial processes (e.g. using direct measurement or Life Cycle Assessment) and valued according to the relevant impact methodology. Any benefits of using recycled materials relative to virgin raw materials can either be assigned to the purchaser of the recycled materials by default, or allocated between the purchaser and the supplier of the recycled material on an appropriate basis (for example, this may be done in proportion to the financial costs of the recycling activity borne by each party).

Because recycling is treated like any other industrial activity it does not warrant a dedicated section in this or any other methodology paper (the starting point for which is units of emissions or resource use as a result of industrial activities).

1.4.4. Limitations of scope

1.4.4.1. Specialist processing

The materiality of specialist waste processing is highly dependent on the type of business in question. For most value chains specialist processing is not relevant. Given the potential range of processes and contexts, and general low materiality we have not attempted to present a generalised methodology here. Where appropriate, specific impacts should be estimated and valued on a case by case basis as required.

The potential for land contamination due to leachate is an important consideration; these outcomes are covered by the methodology for quantifying and valuing leachate release.

1.4.4.2. Littering, ocean waste and persistent plastics

This paper does not cover the impacts caused by littering, ocean waste or persistent plastics. Depending on the context these could include disamenity, ecosystem degradation, human and eco toxicity. While there is some limited research into the extent of these impacts (see for example, UNEP, 2014) more work is required to understand the causal links in these impact pathways.

2. Summary of methodology

2.1. Introduction

The impact pathway presented in Chapter 1 identifies how emissions can lead to different types of impacts. Our valuation framework is structured to follow this pathway, at each stage demonstrating the causal links between corporate activities resulting in waste and societal costs. To understand the value of environmental outcomes associated with each of the impact types, it is necessary to:

1. **Obtain environmental metric data:** The starting point for each of our methodologies is data on emissions. These metric data are based on an understanding of the corporate activities from which they result. The data can come from a variety of sources, some of which (e.g., life cycle assessment (LCA) or environmentally extended input-output modelling (EEIO)) are subject to their own distinct methodologies[3].

Table 2: Environmental metric data

Impact driver (emission or resource use)	Environmental metric data
Solid waste disposal	Metric tonnes of hazardous waste Metric tonnes of non-hazardous waste Further data on waste characteristics (e.g. fossil carbon percentage) or composition (e.g. principle materials) if known

2. **Quantify environmental outcomes**: We quantify physical changes in the environment resulting from corporate emissions or resource use (as measured by the metric data). This is discussed further in Table 3, column 2.
3. **Estimate societal impacts**: We estimate the societal cost (impact on people) resulting from environmental changes which in turn are the result of corporate activities.

2.2. Detailed methodology

The first step is to estimate the environmental metric which is the waste flows by composition (particularly hazardous and non-hazardous) and disposal method (landfill and incineration). These can be estimated directly, using information provided by companies, or indirectly through techniques such 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.

Our methodology for estimating the societal cost of environmental outcomes from solid waste disposal is summarised in Table 3, below. The table includes one page on each of the relevant pathways (1) GHGs from landfill and incineration; (2) Disamenity from landfills and incineration sites; (3) Leachate from landfills; (4) Air pollution from incineration. The left hand column covers the quantification of environmental outcomes and the right hand column summarises how these outcomes are subsequently valued.

[3] Potential sources of metric data are outlined in Chapter 3. The assumed starting point for this methodology is data in the form specified in Table 2.

Table 3: Overview of our impact valuation methodology: estimating societal impacts from solid waste

Quantify environmental outcomes		Estimate societal impacts
Greenhouse gas emissions from landfill and incineration impact pathways		
Methods	• Environmental outcomes (contribution to climate change) and the societal impacts associated with these are evaluated in one step by applying the societal cost of carbon (SCC) to net GHG emissions (see PwC methodology paper *Valuing corporate environmental impacts: Greenhouse gases* for more information on the methodology). • Net GHG emissions from waste are estimated as follows: – GHG emissions (principally CH_4) from waste sent to landfill are estimated over 90 years using the Intergovernmental Panel on Climate Change (IPCC, 200a) Waste Model based on the mass and type of waste, and the conditions of the landfill. – The present value of the associated impacts is then calculated by applying a social discount rate of 3.5%. – GHG emissions from incineration of waste are quantified using waste emission factors based on the fossil carbon content of the type of waste in question. – Where energy recovery is present, avoided GHG emissions are estimated by multiplying waste tonnage (sent to either landfill-gas-to-energy (LFGTE) sites or incinerators with energy recovery) by the energy potential of the waste and the average carbon intensity of the national grid.	• Estimate of the SCC (see PwC methodology paper *Valuing corporate environmental impacts: greenhouse gases*)
Key variables	• Mass and type of waste. • IPCC Waste Model parameters, including climate, landfill characteristics, and organic carbon content for relevant waste types. • Presence and efficacy of energy recovery at waste management sites and carbon intensity of grid electricity.	• See *Valuing corporate environmental impacts: greenhouse gases for key variables*
Assumptions and justification	• Assumptions underpinning the SCC can be found in PwC methodology paper *Valuing corporate environmental impacts: greenhouse gases.* • Assumptions related to net GHG emissions: – National grid energy mixes should be used for estimating avoided GHGs, unless a specific energy source is known to be substituted. • Energy recovery rates are assumed to be zero unless otherwise indicated.	• See *Valuing corporate environmental impacts: greenhouse gases* for key assumptions

Quantify environmental outcomes	Estimate societal impacts

Disamenity (landfill and incineration) impact pathway

Methods	• Environmental outcomes (increases in odour, noise and changes to visual amenity) and societal impact are evaluated in one step using a hedonic pricing model which uses price information from a surrogate market (in this case the housing market) to measure the implicit value of a non-market good or bad (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 from the academic literature. • This function is used to estimate WTP (to avoid disamenity) based on local average house prices, household density and the housing market discount rate. • Societal cost of disamenity is then expressed in terms of the estimated per tonne of waste based on site lifetime and waste flow data.
Key variables	• The estimated coefficients from underlying hedonic pricing studies which describe the degree to which waste disposal sites affect house prices around the point of waste disposal. • House prices, housing (and, therefore) household density around waste disposal facilities (using national average data if unavailable), flow of waste to sites, remaining lifetime of disposal site (default if unavailable), housing market discount rate.
Assumptions and justification	• House price differentials (at given distances from the site) relative to house prices not in close proximity to waste management sites are assumed to reflect the societal costs of disamenity of waste facilities, controlling for other factors which affect house prices. This is currently the standard approach used by academics and governments. • A hedonic transfer factor is derived from six previous primary studies from five countries 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.

	Quantify environmental outcomes	Estimate societal impacts
Leachate release (from landfill) impact pathway		
Methods	• The likelihood and severity of potential environmental outcomes associated with leachate from landfill are estimated on a scale of 1 to 1000 using the Hazard Rating System (HARAS) leachate risk model (Singh *et al.*, 2012), based on source-pathway-receptor relationships.	• Societal impacts are assessed by first identifying a worst-case estimate of leachate clean-up costs as a proxy for worst case societal impact, and subsequently adjusting this estimate by multiplying it together with the HARAS risk score (expressed as a fraction between 0 and 1).
Key variables	• 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 the five variables below: – Proportion of hazardous waste. – Climatic conditions and precipitation. – Presence of a liner at landfill sites. – Geology and soil permeability. – Population density in proximity to sites.	• Estimate of the worst-case leachate clean-up cost per tonne of waste to landfill. • Local purchasing power parity (PPP) relative to the US (where the leachate clean-up cost estimates are sourced from).
Assumptions and justification	• 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.	• Clean-up costs are widely used as a proxy to estimate the value of non-market impacts where damage costs are unavailable. In practice, they are likely to be a lower bound proxy for societal cost of leachate impacts where 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 = 1000) and that scaling this worst-case damage costs according to a risk factor is appropriate because the impacts of leachate are uncertain for any individual case. 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.

	Quantify environmental outcomes	Estimate societal impacts
Air pollution (from incineration) impact pathway		
Methods	• Dioxin and heavy metal emissions: Emissions are calculated using incineration emission factors. Estimate change in the incidence of cancer and lost intelligence quotient (IQ) points by multiplying emissions by linear dose-response functions.	• Multiply increased incidence of cancer and lost IQ points by the weighted societal cost of cancer (value of statistical life (VSL) and of non-fatal cancer) and the WTP to avoid loss of IQ points.
	• Traditional air pollutants (NOx, SOx, NH3, PM2.5, PM10, VOCs): Emissions are calculated using incineration emission factors. Environmental outcomes (increased ambient concentration of pollution) of traditional air pollutions are considered in the PwC methodology paper *Valuing corporate environmental impacts: Emissions to air*. • Avoided emissions are estimated as per avoided GHG emissions from incineration, with air emissions intensity of electricity and heat generation replacing carbon intensity.	• The welfare values associated with health, agriculture and visibility impacts of air pollutions are considered in the PwC methodology paper *Valuing corporate environmental impacts: Emissions to air*.
Key variables	• Dose response functions for the impact of dioxins and heavy metals on IQ and cancer rates.	• Value of Statistical Life (VSL). • Cost of non-fatal cancer.
Assumptions and justification	• Dose-response functions are based on epidemiological studies at a given ambient concentration and emission level.	• VSL estimates are representative of the welfare loss associated with health endpoints.

3. Data requirements

3.1. Introduction

Gathering appropriate data is a precursor to valuing the environmental impacts from waste and the first step in our valuation methodology for each impact area. The availability of high quality input data is a key determinant of the accuracy of impact quantification and valuation. There are three types of data required for quantification and valuation:

- **Environmental metric data:** These relate to companies' waste, and have three general characteristics:
  - Waste quantity;
  - Waste type or composition;
  - Waste treatment approach.
- **Contextual data:** They provide additional relevant information about the basic metric data. For example, describing the context in which waste is disposed (e.g. location, surrounding population density, local weather patterns). The availability of useful contextual data will depend to an extent on the source of the metric data. For example, in the case of directly collected data, location and location characteristics should be known. Whereas in the case of data sourced from an EEIO model, it is likely that only the country and perhaps the industrial sector will be known. While there are some overlaps with metric data, these additional types provide greater contextual detail e.g.:
  - Landfill site and incinerator characteristics;
  - Socio-economic characteristics around waste sites.
- **Other coefficients:** Typically numerical values derived from the academic literature or other credible sources which are required in calculations to convert metric and contextual data into value estimates. Some of these are applied directly, such as the hedonic pricing coefficient used in the disamenity methodology or the SCC used to value GHG emissions, while others are built into more complex models into which data are inputted, such as the IPCC model used to calculate CH_4 emissions from landfill.

While methods for the collection or estimation of basic metric data is not the subject of this paper, the data generation methods used are nonetheless relevant to the likely availability of contextual data and therefore the viability of different potential valuation approaches. This chapter therefore has two purposes: firstly, it describes the most likely sources of metric data across a typical corporate value chain and the implications for contextual data availability; secondly, it sets out key contextual and other coefficient data requirements and the preferred sources for these.

3.2. Environmental metric data

This section discusses the likely availability of metric data for solid waste.

The amount of waste produced by business activities is typically calculated in tonnes, either by on-site direct measurement or estimation. If direct information is not available, techniques such as LCA and/or EEIO can also be used.

Different types of waste, particularly hazardous and non-hazardous waste, will have different environmental outcomes in certain circumstances, and so they are often recorded separately. This distinction is particularly relevant to the impact on GHGs and leachate from landfill, as well as GHGs and air pollution from incineration. Despite inconsistencies in the definition 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. It is therefore important to understand how much waste is disposed of through landfill or incineration. If mode of treatment is not known, it can be approximated using country or state level information, recognising that this will only provide a picture of the average impacts in a given region. This can be obtained from publicly accessible sources, primarily through government environment ministries or international databases such as those maintained by the OECD or World Bank[4].

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 this information. This is likely to vary across a company's value chain as described below in Table 4.

Table 4: Likely metric data sources

Value chain stage	Metric data
Own operations	Waste tonnage, broken down by waste type and composition, should be available from company management information. The other estimation techniques detailed for the supply chain can also be used if direct data are unavailable.
Immediate suppliers	Waste tonnage, broken down by waste type and composition, may be available from some suppliers. Where this 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 may be further informed 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 may be further informed from customer surveys or industry information.
End of life/ re-use 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 may be further informed from customer surveys or industry information.

3.2.1. Limitations and uses of EEIO modelling for waste related GHGs

If EEIO modelling is used in the environmental metric quantification phase of a project, it will estimate the GHG emissions associated with waste from each sector in the economy. Typically these emissions are included within the emission intensity of each 'parent' sector, rather than being aggregated and assigned to the waste management sector.

[4] For example:

China, Brazil and a large number of other countries: http://stats.oecd.org/

USA: http://www.epa.gov/epawaste/nonhaz/municipal/msw99.htm '2010 Date Tables PDF'

England: http://www.defra.gov.uk/statistics/environment/waste/wrfg23-wrmsannual/ 'England and the regions data downloads - 2000-01 to 2010-11'

The emissions calculated by the EEIO are, however, likely to be incomplete because the environmental extensions in EEIO models generally rely on in-year estimates of GHGs from sectors; i.e. they do not include the projected emissions over the lifetime of waste decomposition. It would therefore likely be a significant underestimate to only include the waste GHG emissions from an EEIO model.

It is, therefore, necessary to calculate the emissions profile of the waste separately, as described here. As it is generally not possible to disaggregate the EEIO in-year waste emissions from the rest of a sector's emissions there will be some double counting in the results. We prefer to slightly over- than significantly under-estimate impacts so we recommend including both the EEIO results and the CH_4 emissions calculated separately (as presented above) thereby including the total GHG profile of the waste.

3.3. *Contextual and other data*

Table 5 (overleaf) summarises the data required to estimate and value the environmental impacts of solid waste, grouped by impact area (e.g. GHGs or leachate), the purpose of the data, and default metrics for gathering it.

In general, where waste characteristics and the characteristics of treatment approaches and locations are known, this specific contextual information should be used as it improves accuracy. However, in the event that not all data are available, regional or country-specific averages can be used. Data required by other E P&L methodologies, as summarised by the other reports in the PwC methodology paper series, are not included here, readers should refer to the relevant papers *(Valuing corporate environmental impacts: Greenhouse gases* and *Valuing corporate environmental impacts: Air pollution).*

Table 5: Contextual data requirements

Information	**Purpose**	**Default metrics**
Various impact areas		
Quantity of waste sent to different treatment method (e.g., landfill and incineration, including waste-to-energy (WTE) and land fill gas to energy sites (LFGTE)).	The quantity of waste disposed of through each method is the basic prerequisite for calculating total impacts and impacts per tonne. Classification of this waste into hazardous versus non-hazardous categories, and additional, finer distinctions where possible, are needed for methodologies in various impact areas.	Tonnes of waste, with data on type (hazardous, non-hazardous, and specific types where available; see below for details). Type of disposal facility used. Where specific facilities cannot be identified apportion based on country-level trends. If no data on WTE or LRFTE assume no energy recovery.
Gross national income (GNI) and purchasing power parity (PPP) statistics.	These are required to transfer values between countries.	GNI and PPP data from the World Bank.
Monetary inflation rate.	Inflation describes the rate at which money's value changes for a given country. This is required to update some reference values to their present day equivalents.	World Bank national consumer price inflation data.
Greenhouse gases		
Degradable organic carbon (DOC) content.	DOC is the organic carbon that is accessible to biochemical decomposition (IPCC, 2000c), expressed as a percentage of the quantity of	Categorise waste and apply DOC factors (IPCC default values available

Information	Purpose	Default metrics
	waste.	for classified waste).
Methane correction factor (MCF).	Landfill sites managed in different ways and using different technologies will release different amounts of CH_4 for the same quantity of waste. This can be represented by an MCF.	Categorise landfill management quality and apply relevant MCFs.
Country-specific waste composition.	Where specific types of waste are not known average DOC figures can be used for different countries.	Use national factors for municipal waste, or constant value for industrial waste.
Climatic conditions.	Climatic conditions at landfill sites affect the decomposition of waste and the quantity of CH_4this process emits into the atmosphere.	Use specific data, or regional or country data where this is unavailable.
Methane capture presence and effectiveness.	Some landfill sites capture and burn CH_4, reducing GHG emissions in carbon dioxide equivalent (CO2e) terms.	Use technology specific values where available. Where no information, assume no capture. Where presence but not effectiveness of capture, use IPCC default 20% effectiveness.
Carbon content of wet waste.	The quantity of waste which is carbon, in percentage terms. This determines CO_2 emitted by incineration.	Use specific data if known. Otherwise use IPCC default values.
Fossil carbon fraction.	The percentage of waste's carbon content which is fossil carbon. This determines CO_2 emitted by incineration.	Use specific data if known. Otherwise use IPCC default values.
Combustion efficiency.	The efficiency of an incinerator in burning waste and producing GHGs from waste. This depends on the technology used, and determines CO_2 emitted by incineration.	Use technology-specific factors, if known. Otherwise use IPCC default values.
Energy potential of waste.	Energy produced by LFGTE and WTE plants, measured in kilowatt-hours per tonne of waste sent.	Use specific data where available. Otherwise, use IPCC default values.
Grid carbon intensity.	The amount of CO_2 released by energy generation, expressed as CO2e/kilowatt hour (kWh).	Use International Energy Agency data on national and regional grid intensity around the world.
Disamenity		
Hedonic function transfer factor.	The hedonic function transfer factor is a coefficient which describes how house prices change as a function of distance from waste sites (incinerators and landfills). It is used in the calculation of disamenity value.	The factor is calculated from existing studies.
Average house price.	This is required to calculate the effect of	Use national statistics (providing regional and local house price

Information	Purpose	Default metrics
	waste sites on house prices.	statistics) and property websites.
Household density.	Household density is calculated from population density and average number of people per household.	Actual population density should be used where location is known. Otherwise, average national population density can be used. Data are available from the World Bank. Household size data are available from various sources, such as the OECD.
Housing market discount rate.	The discount rate implicit in the relevant housing market is required to discount waste flows, and can be approximated using standard medium-term discount rates.	Discount rates are available from Her Majesty's Treasury (United Kingdom) (HM Treasury) Green Book.
Remaining site lifetime.	The length of time over which a waste site operates. This affects the time period over which disamenity effects from waste should be calculated.	Data specific to the disposal site where available, otherwise use national averages.
Leachate release		
Presence of liner.	Lined and unlined landfills have significantly different risks of leachate release.	Use specific data where available. Otherwise, use national Environmental Regulatory Quality (ERQ) score and waste collection rates as proxies.
Waste hazard classification.	Hazardous and non-hazardous waste have significantly different risks of leachate release.	Use specific data where available. Otherwise apply national or regional averages.
Soil permeability.	Soil permeability is an indicator of how readily leachate will infiltrate the water and soil systems.	Use specific data where available. Otherwise, use regional information to classify permeability as best, medium, or worst case. If landfill location is unknown, assume medium case, or use country average data where there is limited variation within a country.
Population density.	Population density is an indicator of how many people are likely to be affected by leachate.	Use local population density around waste site; where unavailable use regional or national population density.
Clean-up cost of worst-case leachate remediation.	Clean-up costs are commonly used as a lower-bound proxy for societal costs, and can be scaled according to the risk of leachate release.	Use most relevant national or regional case study.
Air pollution		

Information	Purpose	Default metrics
Emission factors.	Emissions factors show the air emissions intensity of waste incineration (pollution mass released from incineration of a given mass of waste).	Use locality- and technology-specific factors if available. Otherwise, use default values from IPCC or European Union (EU).
Grid air emissions intensity.	Describes the emissions to air of each substance arising from the production of electricity, expressed as a national or regional average (pollutant mass released per kWh energy produced).	Use national or regional data.
Dose-response rates.	Dose-response factors describe a simplified empirical relationship between emissions of certain substances and cases of related health endpoints amongst a given number of people who are exposed. Relevant dose response factors here are for the number of cases of cancer and neurotoxicity effects, the latter measured by reduced IQ points.	Use published, peer reviewed dose-response rates.
Value of IQ points.	This is used as a proxy for the societal costs of neurotoxicity, describing one of its potential consequences.	Use published, peer reviewed value estimates based on WTP studies.
Value of statistical life.	Widely used by policy makers, the VSL places a monetary value on statistically probable mortality.	We use values recommended by the OECD based on their meta-analysis.
Value of non-fatal cancer.	Widely used by policy makers, these data provide a monetary value for statistically probable morbidity.	We use values recommended by the OECD based on their meta-analysis.
Cancer survival rate.	This is used to apportion cancer between fatal and non-fatal cases.	Data from medical research institutes, such as Cancer Research UK.

4. Detailed methodology: Greenhouse gases

This chapter describes the detailed methodology for calculating GHG emissions relating to waste disposal, and translating them into societal impacts. The methodology looks at the net total GHG emissions from landfills and from incineration. In this chapter, landfill and incineration pathways are described separately for GHG emissions, and considered together for avoided GHG emissions. These methodologies take the data discussed in Chapter 3 as inputs.

Table 6: Summary of greenhouse gases methodology

	4.1 Quantify environmental outcomes	4.2 Estimate societal impacts
Greenhouse gas emissions from landfill and incineration impact pathways		
Methods	• Environmental outcomes (contribution to climate change) and the societal impacts associated with these are evaluated in one step by applying the societal cost of carbon (SCC) to net GHG emissions (see PwC methodology paper *Valuing corporate environmental impacts: Greenhouse gases* for more information on the methodology). • Net GHG emissions from waste are estimated as follows: – GHG emissions (principally CH_4) from waste sent to landfill are estimated over 90 years using the Intergovernmental Panel on Climate Change (IPCC, 200a) Waste Model based on the mass and type of waste, and the conditions of the landfill. – The present value of the associated impacts is then calculated by applying a social discount rate of 3.5%. – GHG emissions from incineration of waste are quantified using waste emission factors based on the fossil carbon content of the type of waste in question. – Where energy recovery is present, avoided GHG emissions are estimated by multiplying waste tonnage (sent to either landfill-gas-to-energy (LFGTE) sites or incinerators with energy recovery) by the energy potential of the waste and the average carbon intensity of the national grid.	• Estimate of the SCC (see PwC methodology paper *Valuing corporate environmental impacts: greenhouse gases*)

4.1. Quantify environmental outcomes

The methodology for estimating GHG emissions from waste needs to quantify two key quantities: emissions and avoided emissions. Firstly we describe the methodology to estimate the GHG emissions from landfill and incineration (separately) and then we describe the methodology to estimate the avoided GHG emissions from energy recovery. These emissions and avoided emissions can then be summed to quantify the net emissions.

We have developed a model to estimate the GHG emissions from landfill. The model uses a modified version of the IPCC model to estimate CH_4 emissions from landfill sites, in accordance with its 2006 'Guidelines for National Greenhouse Gas Inventories' (IPCC, 2006c).[5] The methodology estimates landfill GHG emissions from both hazardous and non-hazardous waste, split by specific types of waste. The landfill methodology focuses on CH_4 emissions, because biogas from waste decomposition dominates the four main sources of GHG emissions associated with landfill sites (Mendes et al., 2004) (the other sources being: Site construction, Site management and Transport (moving and compacting the waste)).

We also use the IPCC methodology as the basis for our approach to estimating the emissions of fossil carbon from incineration. We take into account the specific combustion efficiency of incinerators, and fossil carbon content of the type of waste.

At some waste disposal sites, technology is present to capture energy and generate electricity (in the form of landfill gas to energy combustion or incineration). When landfill gas or incinerated waste is used to generate electricity it replaces the need for that electricity to be generated by other means, therefore the associated potential emissions from that generation have been avoided. These avoided emissions offset a portion of the GHG emissions from landfills or incineration. A summary of our approach is shown in Figure 2.

Figure 2: Process steps required to estimate waste-related GHG emissions

1. **Calculate GHG emissions from landfill**
2. **Calculate GHG emissions from incineration**
3. **Calculate avoided GHG emissions from energy recovery (from landfill and incineration)**
4. **Calculate the net emissions**

4.1.1. Step 1: Calculate GHG emissions from landfill

Biogas represents more than 99% of the GHGs from landfills (Rierdevall et al., 1997), and is therefore the focus of this approach. Construction, site management and transport-related emissions are insignificant relative to

[5] The US EPA have an equivalent model (LandGEM, 2005), however this is designed specifically for the US and is not as broadly applicable.

emissions from biogas[6] (Hong et al., 2006; Mendes et al., 2004; Rieradevall et al., 1997) and are therefore excluded from the scope of this methodology[7].

The constituent gases emitted from a landfill depend on the type of waste and the conditions of decomposition, however the general consensus is that landfill gas is around 50-55% CH_4 and 45-50% CO_2, with a small amount of other gases including nitrogen dioxide (IPCC, 2006c; Rierdevall et al., 1997). Only the CH_4 emissions from landfill are considered anthropogenic (Liamsanguan & Gheewala, 2008; Rieradevall et al., 1997; US EPA, 2006). Waste degradation produces CH_4 only under the anaerobic conditions that can prevail in landfill sites. The carbon content of waste would naturally be released as CO_2 as part of the carbon cycle and is therefore non-anthropogenic (IPCC, 2000a). CH_4 has a global warming effect 34 times greater than CO_2; non-anthropogenic[8] CO_2 emissions need to be netted off, so 33 should be used.

CH_4 emissions over the lifetime of the landfill site are the principal factor determining anthropogenic GHG emissions from landfill sites. The IPPC model used in this methodology allows the user to adjust for the different conditions present in landfills (e.g. different climatic conditions, shallow dumpsite versus deep landfill), as well as the characteristics of waste, which determine the rate of decomposition and formation of CH_4 relative to CO_2.

The methodology estimates landfill GHG emissions from both hazardous and non-hazardous waste. Hazardous waste is sometimes pre-treated prior to disposal in a landfill, which reduces emissions considerably (IPCC, 2006b). However, there is limited data to quantify this at a country level, so we treat hazardous and non-hazardous waste in the same way in relation to pre-treatment. This is consistent with much of the literature (for example: COWI, 2000b; Eshet et al., 2005).

In some cases, particularly in developed countries, CH_4 is captured from landfill sites. The captured CH_4 is either flared or used to generate energy (Spokas et al., 2006). Beyond the avoided emissions (described in 4.1.3), this action converts a portion of the CH_4 into biogenic CO_2 which, as discussed above, is not considered anthropogenic and lowers the GHG emissions from the landfill.

As stated earlier, direct data on waste volume and type should be used when available as they will improve accuracy. The model would ideally calculate the CH_4 emissions associated with each tonne of waste over that tonne of waste's entire lifetime at a landfill site.

Table 7 describes the principal factors which affect GHG emissions from landfill, and how each is addressed in the model. The model uses these values to estimate the tonnes of CH_4 emissions produced each year, over a given number of years - 90 is recommended as beyond this emissions are insignificant (IPCC, 2006).

Table 7: Key variables in the custom IPCC landfill methane model

Variable	How the variable is addressed
Amount of waste.	Weight in tonnes is the primary input.
Type of waste.	DOC is the organic carbon that is accessible to biochemical composition (IPCC, 2000c). The IPCC have derived DOC values for different categories of both municipal solid waste (MSW) and industrial waste (see Appendix I, Table 25, Table 26). Where waste composition is known and can be characterised as one of these

[6] Rierdevall et al (1997) calculates that GHG emissions from transport, management and biodegradation of waste in landfill sites contribute 0.15% of total emissions.

[7] They will, however, be quantified as part of indirect GHG emissions if EEIO modelling or broad boundary Life Cycle Inventory estimates are used to estimate environmental metric data.

[8] It is the anaerobic conditions of the landfill which result in methane emissions, rather than the 'natural' emissions of carbon dioxide.

	categories, the corresponding DOC should be entered into the model. Where it is not possible to characterise waste type – then the IPCC default value for industrial waste should be used.
Management of landfill vs. open dump sites.	The customised IPCC tool includes a methane correction factor (MCF) that adjusts for the way in which a site is managed (e.g. depth affects the extent of anaerobic digestion). The default setting for the MCF is 0.6, which corresponds to 'uncategorised solid waste disposal sites' (IPCC, 2006). Where it is possible to characterise sites into one of the four categories used by the IPCC, the corresponding MCF should be used (Appendix I, Table 27).
Country.	Methane capture rate varies across countries. Local figures can be used if available or alternatively national average capture rates.
Climatic conditions.	Weather affects the decomposition rate. Countries can be classified into one of the following four categories, depending on the Mean Annual Temperature (MAT), Mean Annual Precipitation (MAP) or Potential Evapotranspiration (PET) (IPCC, 2006a): • Dry temperate • Wet temperate • Dry tropical • Moist and wet tropical

Table 8: Assumptions required for chosen approach to estimate GHG emissions from waste sent to landfill

Assumption	Explanation
Methane emissions are the only GHG emissions considered significant from landfill.	CO_2 emissions from landfill sites – either emitted directly within landfill gas, or indirectly through the flaring of CH_4– are considered to be non-anthropogenic and therefore not contributing to human caused climate change (IPCC, 2000a). Other GHGs are produced in trivial quantities (IPCC, 2006c; Rierdevall et al., 1997).
Hazardous and non-hazardous waste can be considered together.	DOC of waste varies based on its composition, which changes primarily as a result of the industry type; not whether it is hazardous or non-hazardous. The IPCC model provides DOC's for a default industrial waste category, or more specific options, where this is known. The IPCC provides no DOC value for hazardous waste (IPCC, 2006b); the values of industrial waste and clinical waste – both have the same DOC value of 0.15.
Methane capture rate = 0% unless specific capture rates are reported. Where only the proportion of sites with capture is known, but their capture efficiency is unknown, 20% efficiency is assumed for those sites.	The rate of CH_4 capture significantly influences the final GHGs estimated using the IPCC model. It is therefore deemed prudent to assume this is 0% unless rates are known, and this is consistent with the guidance in IPCC (2006c). The IPCC advises that, where the prevalence of CH_4 capture is known, in the absence of better information, that a 20% capture rate efficiency should be assumed (IPCC, 2000c; Chapter 5).

Assumption	Explanation
Methane emissions considered over 90 years.	90 years is the maximum duration for the landfill gas generation phase recommended by the IPCC (Tabasaran, 1981; as cited by IPCC, 2000a). This is considerably longer than the 25 – 30 years after which COWI (2000a) suggest landfill gas reaches insignificant amounts. Since the CH_4 emissions estimated by the IPCC model follow a first order decay function, and hence tend towards insignificant amounts in later years, we believe taking the uppermost limit is prudent to capture all the impacts.
Rate of methane production from waste decomposition can be characterised by climate conditions of country.	The IPCC model uses a first order decay function to model CH_4 produced by deposited waste. This function is explained in full by the IPCC (2006c) which presents a number of reaction constants based on the climatic conditions of the country of interest. The use of this function – set out in IPCC (2006c) and subsequently used in the IPCC model (IPCC, 2006a) – is taken as sufficient evidence this assumption is valid.
All sites classed as 'uncategorised solid waste disposal sites' (and related MCF used) unless management details about the specific site are known (e.g. depth and moisture content).	The IPCC advise that the MCF for 'uncategorised solid waste disposal sites' should be used 'if countries cannot categorise their solid waste disposal site' (IPCC, 2006c). This is deemed to be consistent with the situation where a company cannot provide specific information on the relevant waste disposal site(s).

4.1.2. Step 2: Calculate GHG emissions from incineration

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.

When waste is incinerated, CO_2 is released into the atmosphere. Some of this is biogenic (or non-anthropogenic), such as wood or plant matter which forms part of the carbon cycle. However, incineration also results in anthropogenic emissions because carbon stored in an otherwise stable form is released (e.g., fossilised biological carbon that would otherwise remain out of the carbon cycle; for example from plastics and other materials derived from petrochemicals[9]). Such carbon is referred to as 'fossil' carbon (IPCC, 2000b).

Alongside the large quantities of CO_2 that are released into the atmosphere, much smaller quantities of nitrous oxide (N2O) and CH_4 are also released. According to the IPCC (2000c) CO_2 is the most significant GHG from waste incineration by at least two orders of magnitude and for this reason only CO_2 emissions are considered further in this methodology.

Unlike GHG emissions from landfill sites, which are emitted for many years after the waste is disposed of, GHG emissions from incineration of waste are instantaneous (Ozge Kaplan et al., 2009). When country-specific data on industrial waste management are not available from other sources, the management is assumed to follow the same pattern as management of MSW (IPCC, 2000c). Fossil CO_2 per tonne of incinerated waste can be estimated as the product of the carbon content of wet waste, the % of carbon that is fossil carbon and the efficiency of combustion (ibid). The IPCC has derived default values for each of these variables, depending on whether waste is hazardous or non-hazardous, as shown in Table 9.

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[9] Note that some authors consider all CO2 as relevant to climate change (for example, EXIOPOL, 2009). However, this is inconsistent with IPCC guidance.

Table 9: Variables influencing CO_2 emission per tonne of incinerated waste (IPCC, 2000c)

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 (depending on incinerator type)	95%	99.5%
Tonnes fossil CO_2 per tonne incinerated waste[10]	**0.557**	**1.642**

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.

4.1.3. Step 3: Calculate avoided GHG emissions from energy recovery (from landfill and incineration)

The principal environmental benefit of energy recovery is in the form of avoided emissions. When landfill gas or incinerated waste is used to generate electricity it replaces the need for that electricity to be generated by other means, therefore the associated potential emissions from that generation have been avoided.

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

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

Landfill-gas-to-energy

In some countries, a proportion of waste sent to landfill each year will be used to generate electricity (LFGTE) and will therefore displace 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 found to be present, then the avoided GHG emissions from energy recovery per tonne of waste should be estimated using Equation 1.

Equation 1: Avoided emissions from LFGTE

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

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

The variables required are further explained in Table 10.

Table 10: Variables required to estimate avoided GHG emissions per tonne waste from LFGTE

Variable	Explanation
Tonnes of waste sent to LFGTE site	The World Bank estimated the number of LFGTE plants worldwide (Willumsen, 2002) 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 will depend not only on the type of waste but also the technology used to collect and convert it. For example, Mendes et al. (2004) provide a value of 166 kWh/tonne, assuming 50% of the CH_4 is captured, and that it is burnt in a gas engine with 30% energy recovery efficiency[11]. Site specific values should be sought wherever possible.
Carbon intensity of national or local electricity grid, CO2e/kWh	The International Energy Agency (IEA, 2011a: p.111) 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 1. In particular, if the tonnage of waste sent to LFTGE sites is not known, the calculation can be modified based on national prevalence of LFGTE (as a percentage).

Incineration waste-to-energy (WTE)

The same approach detailed above should be used to estimate the avoided GHGs when energy is recovered from waste incineration. The variables set out in Table 10, can be used with the exception of the energy potential of waste, which should be replaced with a variable specific for 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 fitted, this technology is common between countries and suggest one of the two energy potentials shown in Table 11, derived for use in European policy making (COWI, 2000b).

Table 11: Energy potentials per tonne of incinerated waste

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

The lower energy potential (625 kWh/t) should be used as a default for countries where it is known that energy recovery occurs, but not what type of recovery technology is prevalent. For countries where it is known that energy recovery is commonly used to produce heat as well as electricity, 2,075 kWh/t should be used.

[11] 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.

4.1.4. Step 4: Calculate net emissions

Once we have calculated the emissions from landfill and incineration, as well as the avoided emissions from those two sites, we can calculate the net emissions arithmetically. The tonnages of avoided GHG emissions (for both landfill and incineration) should be subtracted from the total additional emissions to calculate the net emissions related to waste disposal.

4.2. Estimate societal impacts

GHG emissions from landfill and incineration are first converted to units of CO_2e using Global Warming Potential factors estimated by the IPCC (see: *Valuing corporate environmental impacts: greenhouse gases, chapter 3)*. Then to value the associated societal impacts we apply an estimate of the Social Cost of Carbon calculated according to the PwC methodology paper: *Valuing corporate environmental impacts: greenhouse gases.*

5. Detailed methodology: Disamenity

This chapter describes the detailed methodology for estimating the value of disamenities associated with waste disposal, including visual intrusion, noise, odour and pests frequently caused by waste disposal facilities. We estimate and value the impacts using the established proxy of changes in house prices in proximity to waste management sites ('hedonic pricing'). While the approach to valuing disamenity through hedonic pricing (discussed below) is well established there are few studies comparing disamenity impacts of landfills and incinerators, and they are typically considered together. We follow this convention and use the same methodology for both.

Table 12: Summary of disamenity methodology

5.1 Quantify environmental outcomes	5.2 Estimate societal impacts
Disamenity (landfill and incineration) valuation module	
Methods	• Environmental outcomes (increases in odour, noise and changes to visual amenity) and societal impact are evaluated in one step using a hedonic pricing model which uses price information from a surrogate market (in this case the housing market) to measure the implicit value of a non-market good or bad (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 from the academic literature. • This function is used to estimate WTP (to avoid disamenity) based on local average house prices, household density and the housing market discount rate. • Societal cost of disamenity is then expressed in terms of the estimated per tonne of waste based on site lifetime and waste flow data.

5.1. Quantify environmental outcomes

Adverse localised environmental outcomes of waste management sites including noise, odour, pests and visual intrusion are considered together under the heading of 'disamenity'. In theory it would be possible to consider these outcomes separately through willingness to pay surveys, however it would be difficult to avoid double counting as people struggle to segregate their preferences for these closely related attributes of waste disposal sites. It is therefore established practice in the literature to use hedonic pricing methods, which evaluate all disamenity impacts together. Disamenity is therefore quantified and valued in a single step, as a function of the presence of waste sites (along with other factors) rather than via enumeration of specific outcomes.

5.2. Estimate societal impacts

Our approach to valuing the disamenity impacts leverages the existing body of published research.

The calculations described here estimate the societal cost using reduced house prices as a proxy for the disamenity associated with living close to a waste management facility, holding other attributes of the housing market and local context constant. They assume that the reduction in the price people are prepared to pay for a house on the basis of its proximity to a waste management site reflects the net present value of the disamenity they will incur over the lifetime of the landfill.

This approach is called hedonic pricing, a type of 'revealed preference' method for valuing environmental goods. It is based on observations of a surrogate market (for housing) that 'reveal' people's preferences towards the environmental good of interest. Hedonic pricing is considered best practice for quantifying disamenity impacts associated with waste disposal facilities. However, it is likely to underestimate the total welfare effects because it only considers the welfare of local residents (excluding visitors and people passing through for example).

This paper uses a linear hedonic price function (HPF) describing the change in house price as a function of how far the property is from a waste management facility. The function was derived from an average of different functions estimated around the world and is applied to all countries consistently.

Function transfer is generally accepted to produce more accurate results than simply transferring a unit value for an environmental good, mainly because it allows for more specific information on the target location to be taken into account (see for example Eshet, 2007a). Our approach uses the function to estimate WTP to avoid disamenity per tonne of waste in different countries and locations.

If the required location-specific data are not available (e.g., average housing price and average household density). An alternative approach would be to use unit transfer of published disamenity values from the literature (adjusting them at a country level for income and PPP).

Appendix II provides further background detail to the selected approach, including: a review of the methods and values obtained through primary valuation of disamenity from waste treatment sites, an evaluation of using the benefit transfer approach in the context of disamenity, and a discussion of 'stock' and 'flow' externalities and their relevance to waste.

Figure 3: Steps required to estimate waste-related disamenity

1. **Derive a hedonic pricing function**
2. **Estimate total disamenity from site through transfer of linear hedonic pricing function**
3. **Estimate disamenity per tonne of waste**
4. **Calculate total societal cost**

5.2.1. Step 1: Derive a hedonic pricing function

Our methodology derives a hedonic pricing function (HPF) based on the approach described by Eunomia (2002).

A number of linear hedonic price functions have been developed over the years for landfills and incineration sites. 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. 2007b; Du Preez & Lottering, 2009; Nahman, 2011; Isoto & Bashaasha, 2011; Akinjare et al., 2011) and one meta-analysis of ten landfills and one incinerator in the US (Brisson & Pearce, 1995). From these seven studies, six were selected based on whether it was possible to derive the maximum impact on house prices (the 'y intercept' of the function) and the distance from the site at which there was no longer a noticeable effect on house prices (the 'x intercept' of the function) because these two data points are necessary to calculate a composite function. No systematic trend in these values was evident (for example, a correlation between GDP or HDI and maximum proportional effect on house prices) and so a mean value of all x and y intercept values was taken in order to obtain a composite linear function that could be applied more generally. The studies included in this analysis are summarised in Appendix III.

In order to extract the relevant values from the literature for the purpose of deriving a function, it was necessary to assume that all functions were linear and, in some instances (Du Preez & Lottering, 2009; Akinjaare, 2011), that the effect on disamenity of the landfills became negligible at any distance greater than those reported by the study.

The standard deviation for the population of estimates is fairly large, reflecting substantial variability between studies. This is principally due to the inclusion of one site in the study by Nahman (2011), which exhibited a particularly large maximum effect of 38.3%. Removing the study by Nahamn (2011) from the sample reduced variance but also reduced the mean maximum effect on house prices (the mean maximum effect falls from 11.05% with a standard deviation of 6.40, to 8.61%, with standard deviation of 2.59). We opted to retain this study in the sample to reflect the existence of sites with particularly high disamenity impacts. Ideally we would directly account for this type of variability by deriving separate relationships for different contexts. However, this is generally unrealistic given (a) the number of sites which may be in scope in relation to a single corporate value chain, (b) the paucity of hedonic pricing studies to draw from, and (c) the practical fact that the specific waste management sites in question are frequently not known.

5.2.2. Step 2: Estimate total disamenity from site through transfer of linear hedonic pricing function

Disamenity is principally associated with the physical presence of a site rather than the gradual influx of waste over time (a stock rather than a flow disamenity – see Appendix II for further discussion) so obtaining a total disamenity value per site is a necessary step. The attributable change in house prices around the waste site is taken to represent the capitalised net present value of the loss of amenity caused by proximity to the site over its remaining life (Cambridge Econometrics et al., 2003); i.e. it is assumed that house prices affected by proximity to the site reflect the net present value of all future loss of amenity caused by the site from 'now' until the site closes.

The hedonic pricing function derived in Step 1 can be used to estimate total disamenity at specific sites. The percentage change in the price of an individual house as a function of distance from a landfill or incineration sites is shown in Equation 2, based on the meta-analysis described above.

Equation 2: Linear HPF derived from meta-analysis of six studies

$$\textit{Percentage change in house price with distance from landfill} = 4.06r - 11.05$$

r is the hedonic pricing radius, which is the distance from the waste site, measured in km beyond which the effect on house prices falls to zero.

The total change in house values attributable to the presence of a waste site, can be approximated by integrating this function over what are actually discrete rings around the waste management site, relating to the distance at which each house is from the site boundary (Eunomia, 2002) [12], which gives the hedonic function transfer factor, F. This is shown in Equation 3.

Equation 3: Total disamenity cost per site from hedonic pricing function

$$\textbf{Total disamenity cost per site} = -\int_{o}^{r} P(11.05 - 4.06r)\rho\, 2\,\pi\, r.dr$$

P is the average house price in locality/country and ρ is the household density around the waste site (household/km[2]). The latter is calculated as shown in Equation 4.

Equation 4: Calculating household density

$$\rho = \frac{\textbf{Population density}}{\textbf{Household size}}$$

Equation 3 can be simplified as follows, assuming P and ρ do not vary with distance r:

$$\textbf{Total disamenity cost per site} = -P\,\rho\,2\,\pi \int_{0}^{2.72} (11.05 - 4.06r)\,r.\delta r$$

$$= -P\,\rho\,2\,\pi\,[5.52r^2 - 1.35r^3]_0^{2.72}$$

Between r=0 (waste site boundary) and r=2.72 (where effect on house price falls to zero, see appendix III).

A hedonic function transfer factor, F, can be derived if we define F as shown in Equation 5.

Equation 5: Deriving a hedonic function transfer factor

$$\textbf{Total disamenity cost per site} = FP\rho$$
$$FP\rho = 0.86P\rho$$
$$F = 0.86$$

5.2.3. Step 3: Estimate disamenity per tonne of waste

In order to obtain a value that can be used to attribute impacts to different sources of waste, it is necessary to apportion the disamenity impacts to each tonne of waste entering the landfill over its lifetime (RDC Environment & Pira International, 2003; COWI, 2000a&b; Eunomia, 2002; Cambridge Econometrics et al., 2003).

COWI (2000b) estimate daily tonnages for landfill and incinerator sites (200 tonnes and 400 tonnes per day respectively) and scale up over 5 days a week and 50 weeks a year (equal to 100,000 tonnes and 200,000 tonnes per year, respectively) and assume a life of 50 years[13].

Cambridge Econometrics et al. (2003) use a similar approach, having estimated the loss of amenity from landfill across the whole of the UK using a nation-wide hedonic pricing study. This is then apportioned, per tonne of waste, by estimating the total annual waste for the UK (using UK Government figures from 1998/99) over an estimated average remaining lifetime for all UK landfills of 28 years. A significant difference in this

[12] The integration provides an approximation since in fact the distribution of houses in the area affected by the landfill is more properly represented by a discrete variable as opposed to a continuous one. This means, in practice, that the lower is the population density, the greater will be any errors associated with reducing the discrete distribution of households by a continuous function (Eunomia, 2002).

[13] In the example calculations provided in COWI 2000b, it is not clear whether the 50 years relates to total lifetime or remaining lifetime.

approach to that used by COWI (2000b) is that Cambridge Econometrics et al. (2003) discount these annual waste flows over the 28 years, in order for the tonnes of waste to be comparable to the change in house prices[14].

The discounting of waste flows (Cambridge Econometrics et al., 2003) is also preferred by others (Pearce, 2005) and is the approach adopted here. Theoretically the discount rate used should be the same as the underlying rate implicit in the housing market (Pearce, 2005). Since this is rarely possible to identify, we follow Pearce (2005) and propose a 3.5% discount rate to approximate the implicit discount in the housing market since this is used widely as discount rate appropriate for the medium-term (HM Treasury, 2011).

In summary, in order to express the total disamenity associated with a landfill or incineration site per tonne of waste going to that site, we divide total disamenity by the discounted waste that flows to the site over its remaining lifetime. When considering a specific waste site, annual forecast waste flows can be obtained and discounted for the site's remaining lifetime.

When a specific location in a country is not known or local data are unavailable, Equation 5 can be multiplied by the number of landfill or incineration sites in the country of interest, and the resulting value (an estimate of total disamenity associated with all landfill or incineration sites in that country) divided by the discounted national annual waste flows to landfill or incineration for that country. This gives the average WTP to avoid disamenity per tonne of waste sent to landfill and to incineration for that particular country, as shown in Equation 6.

Equation 6: WTP per tonne waste

$$\boldsymbol{WTP\ per\ tonne\ waste\ (landfill, incinerator)} = \frac{\boldsymbol{P\ \rho\ F\ n}}{\sum_1^T \boldsymbol{W/(1-DR)^t}}$$

where F = hedonic function transfer factor, n = number of waste sites in the country/location (landfill and incineration), W = annual national waste to landfill/incinerator (tonnes per year), DR = discount rate, and t = remaining site lifetime (years).

5.2.4. Step 4: Calculate societal impact

The figure for disamenity per tonne of waste (calculated in step 3) for each in scope location can then be multiplied by the volume of waste related to that location.

Potential data sources for the variables and parameters used in this approach are discussed in Table 13.

Table 13: Parameters and variables required to estimate WTP per tonne waste

Parameter or variable	Metric	Source	Approach
Average house price in area/country 'P'	USD (constant)	National statistics and property websites	• Use average house price without any effect of proximity to the site • When using tonnages per country (i.e. without any site-specific information), use the national average house price
Estimated population density	No. of people per square km	World Bank (2012b)	• Actual population density for known location • Average national population density for country (Appendix IV)

[14] The effect of proximity to landfill is implicitly discounted over time within house price (which includes the capitalised net present value of future flows of disamenity). In order to express this as a ratio of tonnes of waste sent to landfill over a site's remaining life, these waste flows should be discounted at the same rate. Otherwise the effect will be to understate the ratio of cost to tonnes in future years, which will decrease the estimated cost per tonne. For more detailed explanation, please see Cambridge Econometrics et al (2003: Paragraph 5.51).

Parameter or variable	Metric	Source	Approach
around site			
Estimated household density around the site ‘ρ’	No. of households per square km	Average number of people per household: Various[15]	• Calculated as per Equation 4.
Hedonic function factor ‘F’	0.86	Calculated	• See discussion above.
Number of waste sites ‘n’	Number	Various[16]	• Aggregate disamenity depends on the number of waste sites in a given area.
Annual waste flow to landfill and incineration ‘W’	Tonnes per year	Various[16]	• Use most recent waste flow data for the waste site of interest or, when applying the methodology at the national level, for the country (using a single annual figure in this way assumes that flows to each landfill site or incinerator are uniform over their remaining lifetimes. This follows the precedent of other studies, for example: COWI, 2000b; Cambridge Econometrics et al., 2003). • Disamenity per tonne of waste depends on the tonnes of waste processed by each site, which often vary systematically between landfill and incineration sites. Therefore, it is necessary to calculate WTP per tonne waste separately for landfill and incineration sites.
Discount rate implicit in housing market ‘DR’	% rate	HM Treasury (2010)	• Our approach uses a discount rate of 3.5%, the medium-term social discount used widely in policy making (HM Treasury, 2010) and suggested for estimating WTP per tonne waste by Pearce (2005).
Remaining life of landfill and incinerator ‘t’	Years	Cambridge Econometrics et al. (2003)	• The duration for which a site operates varies between and within countries. If available, use the national average remaining site life. • Use the UK average figure for landfill sites of 28 years if country-specific data are not available. This is broadly consistent with estimates in other studies and countries (Cambridge Econometrics et al., 2003; COWI, 2000b). We found no usable alternative value for incinerators.

[15] For example, a list relating to OECD countries exists here http://www.oecd.org/dataoecd/62/22/41919509.pdf for China, here http://ye2.mofcom.gov.cn/aarticle/aboutchina/population/200603/20060301780750.html and for India, here http://en.wikipedia.org/wiki/Indian_states_ranking_by_household_size

[16] For example USA: http://www.epa.gov/epawaste/nonhaz/municipal/msw99.htm ‘2010 Date Tables PDF’

England: http://www.defra.gov.uk/statistics/environment/waste/wrfg23-wrmsannual/ ‘England and the regions data downloads - 2000-01 to 2010-11’

South Africa: http://www.sawic.org.za/index.php?menu=15

This methodology relies on a number of assumptions which are further evaluated in Table 14.

Table 14: Assumptions made in estimating WTP per tonne waste

Assumption	**Explanation**
Meta-analysis function comprising mean values of six studies from five countries is applicable internationally.	Use of function transfer is more sophisticated and is generally considered to produce more accurate results that unit value transfer mainly because it transfers more information (Eshet, 2007a). The studies included in the meta-analysis show general alignment in the shape of HPF's for waste disamenity internationally, suggesting that preferences are somewhat consistent with regard to waste disamenity (Appendix III). This in turn suggests that, with appropriate adjustment for income differentials (represented here by house price differentials), and household density, reasonable estimates can be obtained using a single transfer function, providing methodological consistency across our estimates.
A site receiving hazardous waste has a similar disamenity effect to sites receiving non-hazardous waste.	Empirical evidence from the UK suggests that disamenity associated with hazardous waste processing sites is not significantly different from that associated with average sites (Cambridge Econometrics et al., 2003).
Disamenity arising from landfill sites is similar to that arising from incineration sites.	There are repeated references in the literature to disamenity values from incinerator sites not being materially different from those arising from landfill sites, and these values are often deemed to be interchangeable (Brisson & Pearce, 1998; Eshet et al., 2005a). Brisson & Pearce (1995) observe that the HPF from a study of house prices surrounding an incinerator in the US is nested within the HPFs derived from the US studies of house prices surrounding landfills. This makes intuitive sense given that incineration sites generally store some quantity of waste ready for burning and hence many of the same issues arise. Note that the value produced in this methodology does differentiate between the two when calculating a value per tonne due to varying volumes of waste flow going to incinerators and landfills.
There is uniform household density surrounding waste management sites equal to the national average household density.	Where local population density or the specific waste management site is not known approximations are required. National average is an imperfect measure, because it doesn't necessarily reflect the actual population density around landfills. However, it is used here in the absence of better data (see Appendix IV for further discussion).
National average house price reflects the value of houses around the site.	Where waste management site location is not known an approximation of house prices is required. The national averages approach has a range of precedents in the literature (Brisson & Pearce, 1995; Eunomia, 2003; RDC Environment & Pira International, 2003).
Waste flows to landfill and incineration sites are uniform over remaining site lifetime.	This assumption is routinely made in disamenity calculations of this type (Cambridge Econometrics et al., 2003; COWI, 2000b).

Assumption	Explanation
Average remaining lifetime of landfill and incinerator = 28 years.	This figure was derived empirically from UK data for the remaining lifetime of UK landfills and is consistent with landfill lifetimes assumed in a range of studies (Cambridge Econometrics et al., 2003; COWI, 2000b). No contrary evidence was found to suggest an alternative assumption for incinerators.

Consideration of waste site quality

Intuitively, it would seem reasonable for the level of disamenity to increase with decreasing quality of site management. For example a poorly managed landfill site, where waste is not covered might lead to more nuisances such as odour or the presence of vermin. Poorly managed sites in more tropical climates may be associated with even greater disamenity due to the increased rates of decomposition. However, at present empirical evidence is not available to support this intuition and therefore it is not possible to make a meaningful adjustment to disamenity value estimates to account for the quality of site management.

6. Detailed methodology: Leachate release

This chapter lays out the detailed methodology for estimating environmental outcomes from leachate release and translating these into societal impacts. Since leachate is only related to landfill, and not incineration, only landfill sites are considered here. A summary of the methodology is shown in Table 15.

Table 15: Summary of leachate release methodology (from chapter 2)

	6.1 Quantify environmental outcomes	6.2 Estimate societal impacts
Leachate release (from landfill) impact pathway		
Methods	• The likelihood and severity of potential environmental outcomes associated with leachate from landfill are estimated on a scale of 1 to 1000 using the Hazard Rating System (HARAS) leachate risk model (Singh *et al.*, 2012), based on source-pathway-receptor relationships.	• Societal impacts are assessed by first identifying a worst-case estimate of leachate clean-up costs as a proxy for worst case societal impact, and subsequently adjusting this estimate by multiplying it together with the HARAS risk score (expressed as a fraction between 0 and 1).

6.1. Quantify environmental outcomes

One of the environmental concerns about landfills is escape of leachate (i.e., contaminated fluid) into the surrounding environment. The associated environmental outcomes of leachate are reductions in water quality and soil contamination. Both can affect people's health, impact on economic activities such as agriculture and can damage ecosystems resulting in the loss of other ecosystem services.

There are a number of variables which influence the likelihood of occurrence and consequent severity of leachate. These can be split by source, pathway, and receptor:

- **Source:** This refers to the amount and composition of the waste. Although the classification and composition of hazardous waste varies, as a general rule it is more likely to result in leachate that is directly harmful to human health, than non-hazardous waste (Singh *et al.* (2012), CSERGE (1993)). Non-hazardous waste can also cause impacts, particularly associated with elevated concentrations of nitrates and other organics which can result in eutrophication of waterways. However, the severity of leachate impacts from non-hazardous waste is generally significantly lower than those associated with hazardous wastes.
- **Pathway:** This refers to how the leachate escapes the landfill and enters into surrounding systems. The presence of an impermeable liner has the biggest single impact on whether leachate impacts occur at landfill sites, but the permeability of the soil, depth of aquifers and distance to waterways are also relevant.
- **Receptor:** This relates to the way in which leachate is likely to result in specific societal impacts. For example, the presence of groundwater used by human or livestock populations, or proximity to sensitive ecosystems are relevant factors.

The key variables in each category are outlined in Table 16.

Table 16: Key variables which influence the likelihood and severity of leachate

Classification	Variable
Source	Quantity of waste
	Composition of leachate – determined by composition of waste
	Generation of leachate from waste – largely determined by annual precipitation that infiltrates the landfill. This in turn is most heavily influenced by local precipitation rates and the presence and type of cover
Pathway	Escape of leachate – determined by leachate collection system, quality of liner and geology of site
	Aquifer characteristics – including vadose zone (the zone between soil surface and top of the water table) and aquifer zone
Receptor	Presence of groundwater near to site
	Use of groundwater near to site (i.e. drinking water, irrigation of crops, livestock, other sensitive ecosystems)

Adapted from Singh *et al.* (2012)

6.1.1. Step 1: Use the HARAS model to generate a leachate risk factor

Ideally, as in other areas this methodology would apply a specific impact pathway approach, identifying the causal link between disposal of waste and the different impacts of leachate – including to health via drinking water and agriculture via groundwater. However, there is no credible generalizable approach to do this because the occurrence of leachate is highly site specific, and typically occurs over a prolonged period with a range of impacts over this time.

Given the practical difficulties in identifying causal links between the specific end point impacts of leachate and the disposal of waste, a risk-based approach is typically used in the literature, for example, in the UK Government assessment done by CSERGE (1993); and, Miranda & Hale (1997).

This methodology does likewise, calculating a risk adjusted estimate of the social cost of leachate, with the aid of a hazard risk model which assesses the likelihood of leachate impacts resulting from a given landfill site and the likely severity of impacts should leachate occur. The risk factor generated by the model is applied to a cost estimate which reflects the impacts associated with a worst case scenario. This section describes how the model is used to generate the risk factor.

Even with precise information on each of the variables in Table 16 for a given site, the true extent and severity of leachate impacts through time is hard to predict. Nevertheless a number of models have been developed for use in policy and planning to determine the risk and probable severity of leachate impacts arising.

We recommend using the groundwater contamination HARAS model (Singh *et al.*, 2012). This calculates a leachate risk factor, which represents the likelihood and likely severity of leachate impacts, based on source, pathway, and receptor characteristics. The leachate risk factor is measured on a scale ranging from zero (no risk) to one thousand (highest risk). This model has two key strengths which make it the most suitable for the E P&L:

- **Risk differentiation:** The HARAS model has a high level of risk differentiation, which allows it to better distinguish between different landfill sites in different locations than other models. This is particularly important for creating an E P&L, as companies' value chains frequently extend to diverse geographical areas around the world and have the potential to include both very poorly- and very well-managed landfill sites, from open dumpsites to advanced sanitary landfill sites.

- **Data flexibility:** The HARAS model has both complex and simplified variants, allowing flexibility regarding input data. This is important for creating an E P&L, because for a company with a global value chain, it is likely that data will vary geographically in its quality and completeness. The complex model is more data intensive than the simplified version, but allows more accurate assessment. The simplified model uses a smaller set of widely available proxy variables to classify source, pathway, and receptor characteristics as best, medium, and worst case.

The complex HARAS model estimates the likelihood and probable severity of leachate impact using 26 variables which describe source, pathway and receptor characteristics. Appendix VI includes a detailed list of the variables.

If the data for the complex HARAS model are not available, the simplified version of the HARAS model can be used. We employ a set of proxy variables to categorise the source, pathway, and receptor characteristics of landfills in different locations and thus calculate leachate risk ratings. The variables used and the underlying assumptions are presented in Table 17.

Table 17: Variables in the simplified HARAS model used to identify best, medium and worst case scenarios for leachate risk

Variable	Recommendation	Explanation
Presence of a liner	• Use actual data on presence of liner if available • Alternatively, estimate percentage likelihood of a liner being present in a given country or region using the approach specified (right)	A number of sources can be used to ascertain if landfills in a given country are likely to be lined or not, including: • Reported country specific data • Percent of formal waste collection (with the remainder likely to be waste disposed of in open dumpsites which have no liner) • Measures of environmental regulatory quality (ERQ)[17] • Where specific data on the presence of liners are available this is used, in other cases: – If collected waste accounts for over 99% of total waste and the ERQ is greater than 1 then 100% of landfills are assumed to have liners – If the ERQ score is below zero, then 0% are assumed to be lined – If the percent of waste collected is unknown and the ERQ score is between 0 and 2.5 (the maximum) then the percent lined is estimated in five equal increments of 20% (0>0.5 = 20%, 0.5>1 = 40% etc.) In appendix VI provides some example results based on following the above approach at a country level for a selection of countries. By testing the approach on some countries where the % of landfills currently in use that have liners is known (e.g. the UK), we are able to validate that the results for those countries are reasonable. However, for many countries and regions it is not possible to validate the probabilities generated in this way, hence the need for an estimation

[17] The ERQ score is derived from: Esty. 'Environmental Regulatory Regime Index'. The index grades countries based on their environmental regulations in place and environmental performance.

Variable	Recommendation	Explanation
		approach.
Source		
Hazardous vs. non-hazardous	• Hazardous waste will be treated as the worst case • Non-hazardous waste will be treated as the best case	• Whether waste is hazardous or non-hazardous is the most important of the source variables and has the second largest influence on the risk rating, after liner presence • Our approach allows differentiation between hazardous and non-hazardous, which has a significant influence on the extent and severity of leachate • This simplified approach does not allow for any greater level of differentiation based on specific toxicity
Pathway		
Soil permeability (logk)[18]	Regional information based on Gleeson et al. (2011) where location is known: • 10^{-15} to 10^{-17} logk m^2 treated as best case • 10^{-12} to 10^{-15} logk m^2 treated as medium • 10^{-10} to 10^{-12} logk m^2 treated as worst case • Medium assumed where no data are available	• Soil permeability is used as an indicator of how readily leachate will infiltrate the water and soil systems. • Where specific landfill locations are not known medium can be used. • The pathway has a relatively small impact on the risk rating relative to the presence of a liner and the nature of the source.
Receptor		
Population density	National average population density: • 0 to 100 people per km^2 treated as best case • 100 to 250 people per km^2 treated as medium case • Over 250 people per km2 treated as worst case	• Population density is used as an indicator of how many people are likely to be affected by leachate. • Human health impacts typically dominate estimates of the social cost of leachate (UK Health Protection Agency, 2011). • National average population density is clearly an imperfect measure but could reasonably be expected to correlate with population density around landfill sites. Furthermore, national population density may influence the availability of space for siting landfills away from population centres. See Appendix IV for further discussion.

Source: PwC analysis

[18] Describes the flow of a fluid through a porous medium. A larger value indicates greater permeability and increased risk of leachate penetration to an aquifer

The simplified HARAS model translates these into leachate risk ratings, the detail of which is shown in Appendix VI.

6.2. Estimate societal impacts

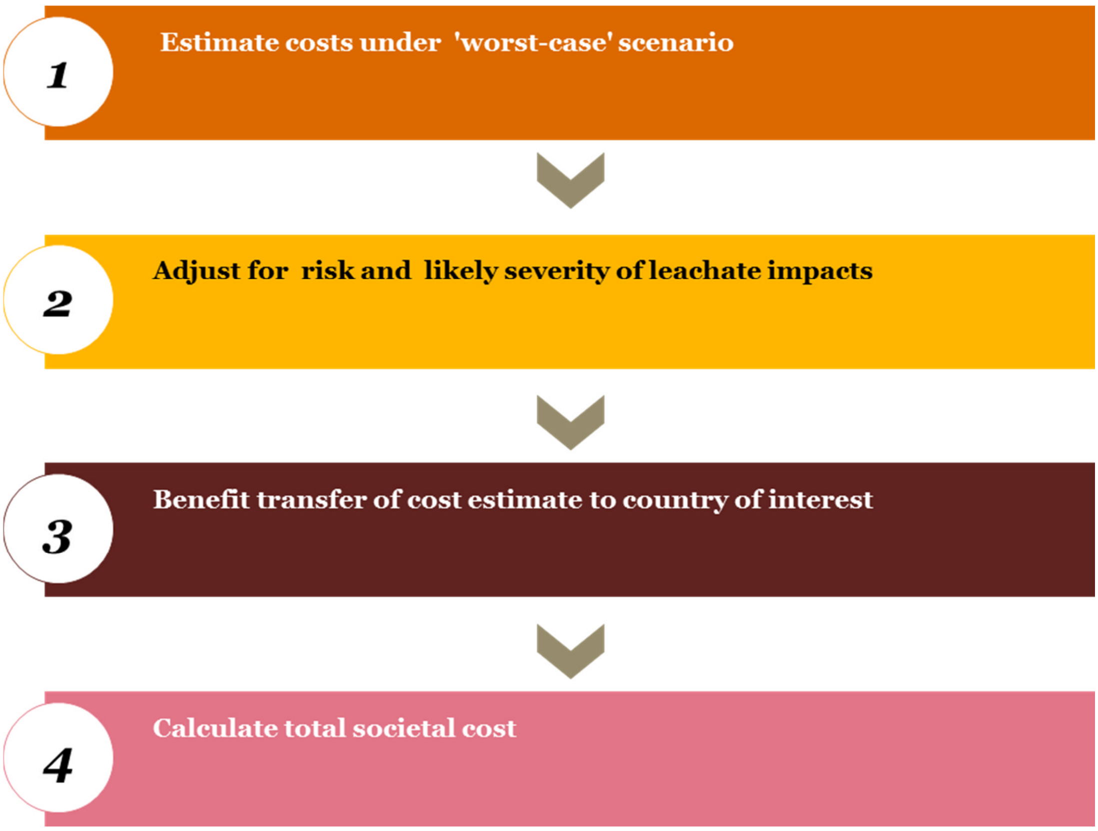
This section describes how societal costs of leachate are estimated and how risk factors can be applied to adjust these cost estimates. Societal impacts are assessed by first identifying a relevant worst-case estimate of leachate clean-up costs as a proxy for worst case societal impact, and subsequently adjusting this estimate by multiplying it together with the HARAS risk score expressed as a fraction between 0 and 1 (where 1 is equal to the worst-case estimate).

The principal environmental outcomes resulting from leachate are associated with water pollution and soil contamination. The welfare impacts of these are heavily dependent on characteristics of the local population and the local environment, but include impacts to health, economic output (e.g. agriculture), and loss of other ecosystem services potentially including fishing, hunting and recreation.

We follow convention in the literature and use clean-up costs as a proxy for societal costs, because estimating the impacts of leachate in a given location is difficult and subject to high uncertainty. This is consistent with the approach taken in a number of academic and government studies (CSERGE, 1993; Miranda & Hale, 1997).

We then adjust the clean-up cost estimates using the leachate risk factor from the HARAS model which reflects both the probability of leachate occurring and the potential scale of the damages. CSERGE (1993) also use the probability of leachate occurring to scale clean-up costs in the UK according to the likelihood of severe, moderate and minor incidents and US EPA (1976) applied a similar method in Illinois.

Figure 4: Steps required to estimate societal impacts of leachate



6.2.1. Step 1: Estimate costs under 'worst case' scenario

Clean-up costs are defined as the cost to remediate the effects of leachate to a pre-landfill level. The chosen cost represents a 'worst case' scenario, as defined in the HARAS model with a score of 1000. Using the simplified model (described in Appendix VI), this worst case comprises:

- No landfill liner
- Hazardous waste
- High soil permeability
- High population density.

It is appropriate to use a worst case estimate of leachate impacts as a starting point because the risk adjustment approach described here will always scale down the estimated value of the impact based on the probability of leachate occurrence and the expected severity of leachate impacts. The worst case estimate will only apply in full in rare cases where a landfill liner is absent and the source, pathway and receptor conditions suggest that severe leachate impacts are certain to occur (i.e. where the risk score fraction = 1).

To calculate the clean-up costs associated with the worst case scenario an appropriate worst case example is needed. The appropriate 'worst case' will vary depending on the location. In the US, for example, the 'Superfund' can be used as a useful source of significant leachate events.

The example we present below draws on the Onalaska Municipal Landfill, Wisconsin. The landfill was operational from 1969 to 1980, with a capacity of 130,000 m^3.[19] Over that period a mixture of municipal and hazardous wastes were disposed of, including a relatively high proportion of chemical wastes, such as industrial solvents naphtha, toluene, and trichloroethene. The site was unlined, in an area of high soil permeability, and affected a relatively large population. Full site remediation[20], including reducing levels of organic and inorganic contaminants in the local aquifer, cost the US Federal Government and the Wisconsin State Government about USD 8,950,000[21].

Equation 7: Example of worst-case societal cost of leachate per tonne of waste

$$\boldsymbol{Societal\ cost\ of\ leachate} = \frac{\boldsymbol{Total\ cost\ of\ remediation}}{\boldsymbol{Landfill\ capacity}} = \frac{\boldsymbol{USD\ 8,950,000}}{\boldsymbol{130,000\ tonnes}} = \boldsymbol{69\ \$/tonne}$$

6.2.2.Step 2: Adjust for risk and likely severity of leachate impacts

Based on the methodology described in 6.1, the characteristics of the waste and the conditions of disposal within the given country can be used to assess the source, pathway and receptor using the worst, medium and best case classifications. Each risk rating combination has a corresponding risk score (Table 32 in Appendix VI). This scoreis used to estimate the risk-adjusted societal cost as shown in Equation 8.

Equation 8: Risk-adjusted societal cost of leachate per tonne of waste

$$\boldsymbol{Societal\ cost\ of\ leachate} = \frac{\boldsymbol{risk\ score}}{\boldsymbol{1,000}} \times \boldsymbol{\$/tonne}$$

Example results of this calculation using the output of Equation 7 are shown in Table 33 in appendix VI.

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[19] Onalaska Landfill physical characterisation: 7 acres, with an average depth of waste disposal of 15 feet. Conversion factor used: 1 m^3/tonne of waste.

[20] Remediation activities included: the installation of a landfill cap; the installation of an air injection system within the area of soil contamination to enhance the bioremediation of organic contaminants; the installation of a groundwater extraction and treatment system to capture and treat VOC contaminants in the groundwater immediately down gradient of the landfill; and, the implementation of a groundwater, surface water, and sediment monitoring program to ensure the adequacy of the clean-up.

[21] Expenditures 2010: Wisconsin State – USD 4,200,000; US Federal Government: USD 4,620,000. Committed but not yet expended: Wisconsin State USD 130,000. All values are 2010 USD.

6.2.3.Step 3: Benefit transfer of cost estimate to country of interest

Where it is necessary to transfer societal cost estimates from one country to another (e.g. where local worst case examples are not effectively documented), it is necessary to adjust for PPP and inflate values using the country inflation rate from the year of the cost estimate to the year of waste disposal.

6.2.4.Step 4: Calculate total societal cost

Once we have established the location or country-specific societal costs of leachate per tonne of waste disposed, we can calculate the overall cost arithmetically by multiplying these figures by the volume of waste in each location.

The principal assumptions of this approach are outlined in Table 18.

Table 18: Assumptions required for estimating the social cost of leachate

Assumption	Comment on purpose and reasonableness
Clean-up costs are an acceptable proxy for the societal costs of leachate	Clean-up costs are often used as a lower bound proxy for welfare costs where more specific data are unavailable. There has been limited study of the societal costs of leachate, particularly in the developing world. Accordingly, several academic and government studies have used clean-up costs (CSERGE, 1993; Miranda & Hale, 1997). CSERGE (1993) note that 'there is little more to go on'. More than two decades on the usable evidence base remains limited.
Clean-up costs can be acceptably transferred between countries	Where country or location focused analysis is required we recommend using a worst case example which is appropriate for the specific country. For multi-country assessments a single worst case (or average of several) may be preferred to provide consistency and comparability. In certain countries, clean-up costs may not actually be incurred as a result of leachate incidents. For example, the effort expended by the US Environmental Protection Agency to clean-up the Onalaska case study used here may be greater than that which would be expended in other countries. However, in developing countries, limited clean-up expenditure is more likely to reflect weaker governance or capacity to respond than in the USA, rather than the absence of societal costs, and preferences to avoid these. Therefore, transferring the cost of a thorough clean-up is likely to be more reasonable as a proxy for welfare impacts than using actual clean-up costs in developing countries. The international transfer of values adjust only for PPP and assume an income elasticity of 1 and constant preferences for health, economic prosperity and environmental quality. This treatment is common in the literature on the basis that there is insufficient evidence on which to base systematic adjustment for these factors.

7. Detailed methodology: Air pollution

This chapter lays out the detailed methodology for calculating the environmental outcomes related to air pollution from incinerating waste, and translating them into societal impacts. Landfills are not addressed in this chapter, as they produce trivial volumes of non-GHG emissions (EXIOPOL, 2008). The air pollutants from incineration fall into two categories; dioxins and heavy metals, and traditional air pollutants (e.g. PM, NO_x, SO_x). Dioxins and heavy metals are discussed first, followed by traditional air pollutants.

Table 19: Summary of air pollution methodology (from chapter 2)

	7.1 Quantify environmental outcomes	7.2 Estimate societal impacts
Air pollution (from incineration) impact pathway		
Methods	• Dioxin and heavy metal emissions: Emissions are calculated using incineration emission factors. Estimate change in the incidence of cancer and lost intelligence quotient (IQ) points by multiplying emissions by linear dose-response functions.	• Multiply increased incidence of cancer and lost IQ points by the weighted societal cost of cancer (value of statistical life (VSL) and of non-fatal cancer) and the WTP to avoid loss of IQ points.
	• Traditional air pollutants (NOx, SOx, NH3, PM2.5, PM10, VOCs): Emissions are calculated using incineration emission factors. Environmental outcomes (increased ambient concentration of pollution) of traditional air pollutions are considered in the PwC methodology paper *Valuing corporate environmental impacts: Emissions to air.* • Avoided emissions are estimated as per avoided GHG emissions from incineration, with air emissions intensity of electricity and heat generation replacing carbon intensity.	• The welfare values associated with health, agriculture and visibility impacts of air pollutions are considered in the PwC methodology paper *Valuing corporate environmental impacts: Emissions to air.*

7.1. Quantify environmental outcomes (dioxins and heavy metals)

To quantify the environmental outcomes associated with dioxin and heavy metal emissions, we follow the methodology set out in EXIOPOL (2009); estimating the change in incidence of cancer and lost IQ points by multiplying emissions by linear dose-response functions.

The materiality of air pollution relative to the overall societal impact of waste is highly variable in our results. For example, it represents just 1% of the waste incineration impacts in the US, but 28% of the impacts China.

The difference is driven by dioxin and heavy metal emissions which are much higher in the absence of flue gas treatment. Regulation is a major factor in determining the presence of such safeguards.

Given the low materiality of air pollution relative to the total environmental impacts from incineration in much of the developed world, it may be acceptable to consider it out-of-scope in an E P&L. Particularly for companies with operations (and supply chain) focused in countries with well-enforced incineration regulations.

Figure 5: Steps required to estimate dioxin and heavy metal emissions driven by corporate waste

1. **Calculate dioxins and heavy metal emissions**
2. **Calculate the number of health endpoints associated with the emissions**

7.1.1. Step 1: Calculate dioxins and heavy metal emissions

The majority of heavy metals and dioxins are highly damaging to health and, if inhaled, can cause cancer, and may lead to neurotoxicity, reducing IQ (in the case of mercury).

This methodology follows the standard academic and policy approach, estimating the quantity of emission released using emission factors, applying dose response functions to calculate the likely number of attributable health outcomes, and then valuing these health outcomes (see for example: EXIOPOL, 2009). Appendix VII includes some examples of previous value estimates from the literature.

Reliable measured data on heavy metal and dioxin emissions from incinerators is extremely rare, but if available should be used. More realistically, heavy metal and dioxin emissions per tonne of waste can be approximated using national or regional emissions limits for waste incineration (EXIOPOL, 2009). The IPCC recommend using the factors provided in Table 35 in Appendix VIII. However, in some countries more stringent regulations are enforced and the IPCC factors would therefore lead to an over estimation of the impacts (Table 36 in Appendix VIII). For example, the EU emissions limits in Directive (2000/76/EC) are lower than the IPCC factors in some cases (chromium, dioxins and lead). Where lower emission limits are known to be enforced these should be used following the precedent of EXIOPOL (2009) and Rabl et al. (2008).

Given the potentially large difference in results when using the IPCC versus the EU figures, it is recommended that a consistent rule be applied to guide allocation of emission factors to countries where more specific data are unavailable. The Environmental Regulation Quality index (Esty, 2002), discussed in the context of leachate, can be used to guide this decision: EU factors can be used for countries with 'good' scores (>0.0), while the IPCC factors can be used for countries with 'poor' scores (<0.0).

7.1.2. Step 2: Calculate the number of health endpoints associated with the emissions

Dose response functions describe how many health endpoints (response) are likely to be associated with a given level of emissions (dose). ExternE (2004) publishes dose response functions for cancer.

Table 20: Dose response rate (cancer) per kg heavy metal

Pollutant	Impact
Arsenic (As)	4.0E-05 cancers/kg
Cadmium (Cd)	2.0E-05 cancers/kg
Chromium (Cr)	1.0 E-04 cancers/kg
Nickel (Ni)	1.9E-06 cancers/kg
Dioxins	9.3E+01 cancers/kg

Source: ExternE (2004)

Lead (Pb) and Mercury (Hg) are not considered to be carcinogenic in the concentrations that result from waste incineration (ExternE, 2004) but can have severe neurotoxic impacts. Neurotoxic impacts are typically measured in terms of how many IQ points are reduced by a given exposure to the pollutant. Table 21 presents does response functions for Lead and Mercury.

Table 21: Dose response rate (neurotoxicity) per kg heavy metal

Pollutant	Impact
Lead (Pb)	6.0E-02 IQ points/kg
Mercury (Hg)	8.0E-01 IQ points/kg

Source: ExternE (2004)

7.2. Estimate societal impacts (Dioxins and heavy metals)

In this section, we set out our methodology for valuing the societal cost of the impacts on human health expected to result from emissions of dioxins and heavy metals from waste incineration. These impacts are cancers (associated with arsenic, cadmium, chromium, nickel and dioxins), and lost IQ points (associated with lead and mercury).

To estimate the societal impacts of dioxins and heavy metals, we multiply the increased incidence of cancer and lost IQ points by the weighted societal cost of fatal (value of statistical life (VSL)) and of non-fatal cancer, and the value of lost IQ points.

Figure 6: Steps to value the societal cost of dioxins and heavy metal emissions

1. **Calculate weighted societal cost of fatal and non-fatal cancer or lost IQ points**
2. **Adjust for inflation and for income at PPP if desired**
3. **Calculate total societal cost**

7.2.1. Step 1: Calculate societal cost of fatal and non-fatal cancer and lost IQ points

The output of the dose-response calculation is number of cases of cancer. In order to value this we first need to distinguish between fatal and non-fatal cases. Cases which are fatal are valued using the value of statistical life (VSL). Please see the PwC methodology paper *Valuing corporate environmental impacts: Air pollution* for detailed discussion of our approach to the VSL.

A portion of cancer cases will not be fatal, the OECD (2006) note considerable variation in the WTP to avoid cases of non-fatal cancer based on the type of cancer as well as the method and sample of the study. We apply a figure which is 10.5% of the VSL (the mid-point of the studies quoted by the OECD), resulting in a value of USD 360,000 in 2012 USD.

To calculate the portion of non-fatal cases national statistics can be used; for example in 2009 Cancer Research estimated that 43% of cases of cancer in the UK were non-fatal (Cancer Research UK, 2009).

A range of values exist in the literature for the societal cost of lost IQ points, between USD 4,000 and USD 19,000, mostly based on lost earnings or remedial education (Spadaro & Rabl, 2004). We follow the precedent of both Spadaro & Rabl (2004) and ExternE (2004) in taking an intermediate value of USD 17,500 per IQ point (in 2011 prices).

7.2.2. Step 2: Adjust for inflation and for income at PPP if desired

Before transferring the societal cost of fatal and non-fatal cancer or the cost of lost IQ points, it is necessary to inflate using the country inflation rate to the year of waste disposal. If income adjustments are to be made, these also need to be applied using a GNI ration, described in the air pollution methodology paper. If income adjustments are to be made equity considerations should be kept in mind.

7.2.3. Equity considerations

Most countries operate a principally market-based economy, where the allocation of resources is determined largely by the forces of supply and demand, which also establish prices in the economy. In this context, an individual's income determines the quantity of marketed goods that they can obtain. When estimating the monetary value of goods (or 'bads') which are not currently traded in markets, the income constraint must therefore be considered.

As peoples income changes, their level of demand for a good usually changes, and the amount they would pay for each unit of the good also changes. Empirical evidence for environmental goods (or avoidance of 'bads') suggests that this 'income effect' is positive – people are prepared to pay more as their income increases (Pearce, 2003). For this reason, if values estimated in one location are to be used in a different location, they need to be adjusted to take account of differences in the income constraints of people in each location.

This is best illustrated using an example. Suppose a survey of people living beside a lake in the USA finds that they value the leisure time they spend around the lake at $1,000 per year. This represents about 2% of their average annual income. Combining this with the number of people who live in close proximity to the lake allows for an estimate of the value of the lake for leisure purposes to be produced. This non-market value estimate can be taken into account when decisions which might affect the future of the lake (e.g. new developments) are considered.

Now suppose we wish to estimate the value of a similar lake in Uganda. Resources to conduct a new survey aren't available but the number of people living near to the lake can be estimated, and it is known to be a popular recreation area. However, the average per capita income in Uganda is 1/100th of the average per capita income in the USA[22]. So assigning the same value of $1,000 per person in the Ugandan context would clearly be inappropriate; suggesting that local people would pay twice their average annual income for a year's worth of leisure at the lake. In order to estimate the value that local people place on the lake, relative to their other priorities, it is necessary to adjust for the differences in income constraints.

This central concept of income effects in non-market valuation of environmental goods is relatively uncontroversial, as is the practice of adjusting for differences in income and purchasing power when transferring value estimates between countries. However, when valuing goods (and bads) relating to human health, equity considerations become more apparent.

As with environmental goods, empirical evidence demonstrates that the amount individuals' would pay to maintain good health and to reduce risks to life increases with income (Viscusi and Aldy, 2003; Scotton and Taylor, 2010; OECD, 2010). This is reflected in estimates of the Value of a Statistical Life (VSL)[23]. When applying a VSL estimate calculated in one location to health outcomes in another location, it is common practice in the health literature (see for example: OECD, 2012; Hammitt and Robinson, 2011) to adjust the VSL to reflect the income differential between those locations, as described above.

These differences in preferences for life and health between locations may reflect a genuine acceptance of greater health risks, particularly in the context of other priorities such as economic development or employment. However, because preferences of this nature are often considered to be constrained by the limited choices available in low income contexts, the use of differing VSLs is contentious where decisions may relate to inter-regional resource allocations. In recognition of these concerns, the OECD (amongst others) recommend that where decisions may relate to allocations between regions a single VSL estimate should be used in policy analysis across those regions.

Given the range of possible decision-making contexts where E P&L results may be considered[24] it is important that the decision maker is aware of this potential issue and is in a position to make an informed decision. Whether the primary presentation includes or excludes income adjustments to health related values is therefore a decision for the ultimate user.

Either way we suggest that the effect of differing income levels on the results of an EP&L is assessed through sensitivity analysis.

[22] Even after accounting for differences in purchasing power the ratio is 1/40th.

[23] *"Value of a Statistical Life (VSL), … represents the value a given population places ex ante on avoiding the death of an unidentified individual. VSL is based on the sum of money each individual is prepared to pay for a given reduction in the risk of premature death, for example from diseases linked to air pollution."* OECD, 2012

[24] For example, some decision contexts will be confined to a single country and could involve comparing environmental values to other factors (outside the E P&L) determined by prices or incomes within that country; while others could require prioritisation of impacts across many countries.

Where the decision context has implications for inter-regional allocations, two sets of results should be presented: one which reflects equity concerns without any income adjustment to health related values, and a second which does take into account income differentials.

The decision maker will still need to consider a range of factors beyond pure environmental or health impacts. For example, a study which does incorporate income adjustments across a range of countries could provide incentives to shift polluting activities to lower income countries where the implied cost of impacts would be lower – this may be undesirable. However, a similar study which does not adjust for differences in income may deter foreign investment in lower income countries; investment which could have created improvements in well-being in excess of any health related losses.

For this reason decision makers may also wish to consider a more holistic decision making framework such as PwC's Total Impact Measurement and Management (TIMM) which values environmental impacts alongside economic, fiscal and social impacts[25].

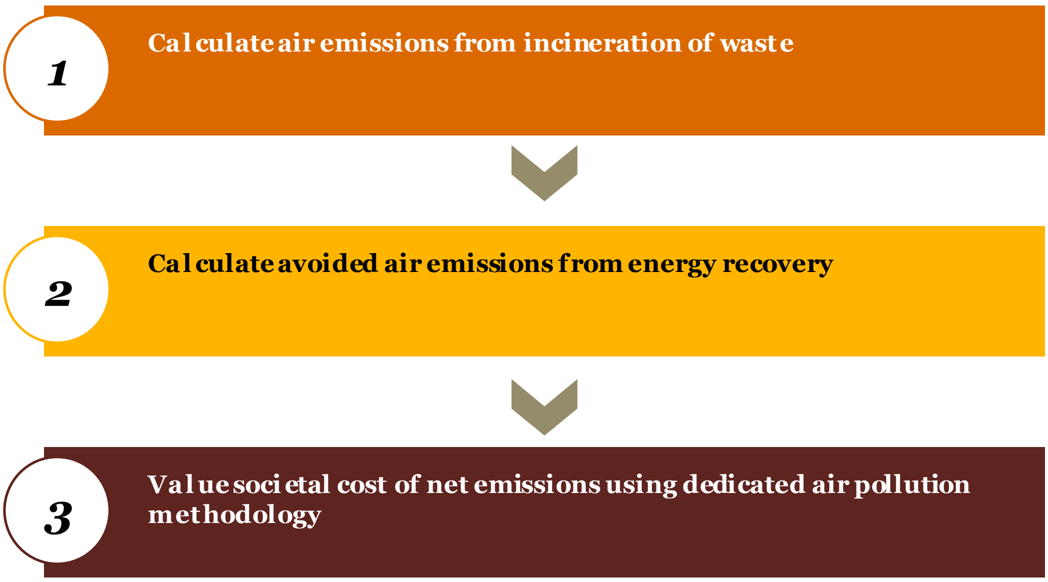
7.2.4. Step 3: Calculate total societal cost

Once the number of health endpoints (e.g. cancer or lost IQ) associated with each tonne of waste, and the societal cost per health endpoint, have been calculated for each relevant location, the results can be multiplied by the tonnage of waste going to each location to calculate the total societal cost of dioxins and heavy metals attributable to the company in question.

7.3. Quantify and value traditional air pollutants

Incinerating waste also produces traditional air pollutants (NO_x, SO_x, PM_{10} and $PM_{2.5}$). We estimate those pollutants using specific data wherever available otherwise defaulting to average values from respected sources. As with the GHG module in Chapter 4, we subtract any avoided air pollutants from heat and power generation. And finally we value the net air emissions using our methodology from the Air Pollution paper. The process for estimating environmental outcomes and societal impacts is shown in Figure 7.

Figure 7: Steps required to estimate traditional air pollutant emissions driven by corporate waste



[25] See "Measuring and managing total impact: A new language for business decisions", PwC 2013: http://www.pwc.com/gx/en/sustainability/publications/total-impact-measurement-management/assets/pwc-timm-report.pdf and: http://www.pwc.com/totalimpact for more information.

7.3.1. Step 1: Calculate air emissions

Location-specific waste data and emissions factors should be used to estimate air pollutants if the data are known. Similarly, if waste facility specification is known, then the technology-specific emissions factors, such as those provided by the European Environment Agency, should be used[26].

If this information is not available, then it will be necessary to apply a general estimate of the air emissions resulting from waste incineration. In reality, the amounts of air pollutants from waste incineration will vary depending on the composition of waste flows (such as their nitrogen and sulphur content). However, representative data on air emissions from waste incineration are difficult to obtain (EXIOPOL, 2009; Rabl et al., 2008). In the absence of better information, the general air emissions factors for industrial waste, provided by EMEP & EEA (2009a), can be used. These are shown in Table 22.

Table 22: Air emissions factors for use with tonnes of incinerated waste

Pollutant	Value[27]	Unit	Reference
NO_x	0.870	kg/Mg waste	European Commission (2006)
SO_x	0.047	kg/Mg waste	European Commission (2006)
PM_{10}	0.007	kg/Mg waste	US EPA (1996) applied on TSP
$PM_{2.5}$	0.004	kg/Mg waste	US EPA (1996) applied on TSP

Source: EMEP & EEA (2009a)

7.3.2. Step 2: Calculate avoided air emissions from energy recovery

In the same way that energy recovery from incineration results in avoided GHG emissions, so too will it result in avoided air emissions.

The avoided air emissions can be estimated by using Equation 1, as described in chapter 4, using the air emissions intensity of the national or regional electricity grid (NO_x, SO_x, PM_{10} and $PM_{2.5}$ emissions per kWh) in place of the CO_2 intensity.

Emissions intensity per kWh for these four pollutants in the country of interest can be derived by using the average air emissions (for NO_x, SO_x, PM_{10} and $PM_{2.5}$) per kWh for the principal fuel types used to generate electricity and heat (EMEP & EEA, 2009b) and scaling each of these emissions intensities by the proportion of total electricity and heat generation that these fuel types make up respectively in the country of interest (IEA, 2011b).

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[26] The IPCC Emissions Factors Database (EFDB) recommends use of the European Environment Agency's 'EMEP/CORINAIR' Emission Inventory Guidebook 2009 which contains emissions factors for incineration of both industrial waste and MSW, for a variety of incinerator specifications (EMEP & EEA, 2009a).

[27] These are the default values provided by the European Environment Agency in the EMEP CORINAIR emission inventory guidebook (EMEP & EEA, 2009a) and assume an averaged or typical technology and abatement implementation in the country (using only particle emission abatement equipment for controlling emissions).

7.3.3. Step 3: Estimate societal impacts (traditional air pollutants)

Once the volumes and locations of traditional air pollutants (NO_x, SO_x, and PM) are calculated, the environmental outcomes and societal costs are estimated as described in the PwC methodology paper: *Valuing corporate environmental impacts: Air Pollution.*

8. Sensitivity analysis

8.1. Module-specific sensitivity analysis

8.1.1. Summary and considerations for model use

This section presents a summary of the findings of our sensitivity analysis, more detailed discussion on the parameter influence on results and uncertainty follows.

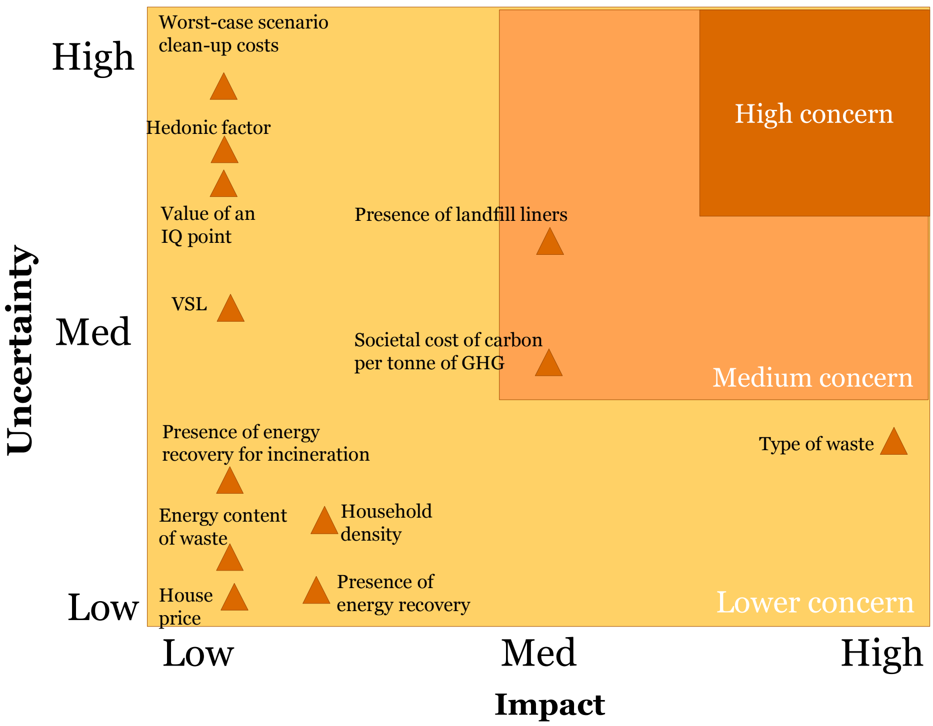
The key parameters tested in our sensitivity analysis are mapped in Figure 8 on an impact/uncertainty matrix.

In general, the estimates around GHG emissions are of most interest, in particular the type of waste disposed of and the societal cost of carbon emissions. When using the model, it is important to get the best data possible on the type of waste, due to its high impact. With adequate data, the type of waste and its emissions profile can have low uncertainty, but granularity of data will vary.

Greenhouse gases dominate the societal cost of waste disposal, and the SCC is the main driver of GHG waste impact. However, determination of the SCC has been the subject of extensive work by PwC and many others as demonstrated in the methodology paper: *Valuing corporate environmental impacts: Greenhouse gases*.

Some of the other parameters for the other modules have high uncertainty, largely due to poor availability of sufficiently detailed data globally. For example, our estimate of the hedonic price factor is based on a meta-analysis of six studies with quite high variation. As indicated by Figure 8, despite the higher uncertainties in these modules the overall influence on the total impacts of waste is low.

Figure 8: Impact/uncertainty matrix summarising the sensitivity assessment for key parameters and decisions

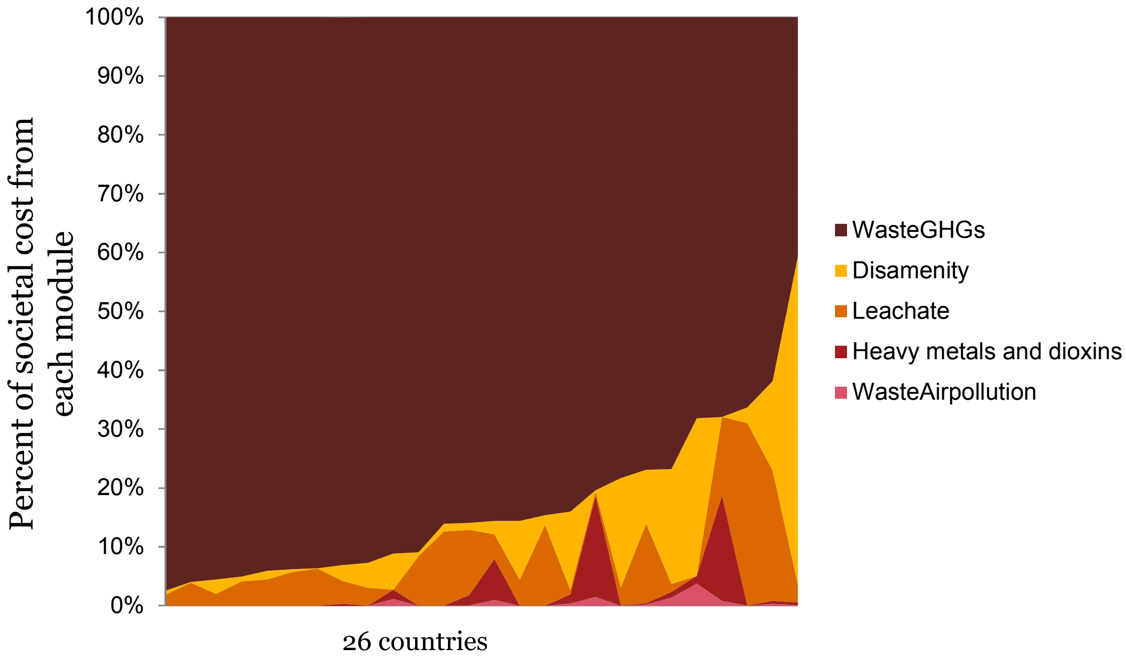


8.1.2. Materiality

Figure 9 presents the relative contribution of each of the different impact pathways to the overall impact of waste disposal (averaged across hazardous and non-hazardous, E P&L value per kg) in different countries. GHGs dominate the impacts, but in some cases disamenity represents a significant portion (up to 45%), while leachate represents up to 30% in some countries but is predominantly below 10%. Dioxins and heavy metals are also less than 10% in most instances with values up to 20% in a few cases, while other air emissions are never more than 5%.

Based on this materiality assessment, we pay particular attention to the inputs to the GHG module for the sensitivity analysis.

Figure 9: Percent of societal cost from each module for a selection of 26 countries



8.1.3. Parameter impact

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Table 23 presents the parameters considered in this sensitivity analysis, split by the impact module which they influence. Most parameters have a 10% change applied to show how strongly they influence the results. Parameters which influence the GHG module tend to also have a significant influence on the overall societal cost of a tonne of waste. Type of waste is an important driver of the quantity of GHGs released. Changes in the SCC have a directly proportional impact on the results of the GHG module.

For leachate, disamenity and air pollution, even though some parameters have an important influence on the module results they tend to have a low impact on the overall societal cost of a tonne of waste.

Table 23: Quantitative assessment of module and overall sensitivity to changes in key parameters (average of hazardous and non-hazardous)

Variable	Flex	Impact rating [28]	US (% change to module)	US (% change to overall cost)	China (% change to module)	China (% change to overall cost)	Nigeria (% change to module)	Nigeria (% change to overall cost)
Module: GHG								
SCC per tonne of GHG	10%	Med	10%	**9.5%**	10%	**8.6%**	10%	**9.8%**
Type of waste	Textile[29]	High	51%	**32%**	46%	**28%**	60%	**36%**
Presence of energy recovery[30]	10%	Low	-0.6%	**-0.6%**	-1%	**-1%**	0%	**0%**
Module: Leachate								
Worst case scenario clean-up costs	10%	Low	10%	**0.8%**	9.2%	**1.1%**	10%	**0.2%**
Presence of landfill liners	-10%[31]	Med	105%	**4.5%**	14%	**1.2%**	9.1%	**0.2%**
Module: Disamenity								
Home price	10%	Low	10%	**0.2%**	10%	**0.1%**	10%	**0.03%**
Household density	10%	Low	10%	**0.3%**	10%	**0.2%**	10%	**0.01%**
Hedonic factor	10%	Low	11%	**0.3%**	11%	**0.2%**	11%	**0.1%**
Module: Air pollution								
Value of an IQ point	10%	Low	9.2%	**0.03%**	0.30%	**0.02%**	0.0%	**0.0%**
Presence of energy recovery[30]	10%	Low	-23%	**-0.6%**	-72%	**-1.0%**	0.0%	**0.0%**
VSL	10%	Low	0.8%	**0.0%**	9.2%	**0.6%**	0.0%	**0.0%**

8.1.4. Parameter uncertainty

Table 24 presents a qualitative assessment of the uncertainty surrounding the data for each of the parameters. The most uncertainty lies within the leachate, disamenity and air pollution modules. Uncertainty in GHGs, as the major impact arising from waste disposal, can have a greater influence on the results. The principle uncertainty associated with GHGs relates to the future damage costs of climate change.

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[28] Low = average response for **overall cost** for three countries is less than 1%

Med = average response for **overall cost** for three countries is 10% or less

High = average response for **overall cost** for three countries is greater than 10%

[29] The baseline for comparison was the IPCC default values for waste to IPCC values for textile waste.

[30] Energy recovery technology affects both the GHG and air pollution modules. We show both to demonstrate the relative impacts on the modules, but the impact on the overall societal value is the same.

[31] This parameter was varied -10% because the presence of liners is currently 100% in the US.

Table 24: Qualitative assessment of parameter uncertainty

Variable	Uncertainty rating	Reliability/quality of measurement	Estimated variance of the number measured
GHG			
Societal Cost of Carbon per tonne of GHG	Med	Approach is broadly accepted but a range of estimates exist	<50%
Type of waste	Low	Depends on company data	N/A
Presence of energy recovery for incineration	Low	Data availability varies, but typically reported by government agencies where recovery exists	<10%
Leachate			
Worst case scenario clean-up costs	High	Data used as proxy for societal welfare impact	<100%
Presence of landfill liners	Med	Data availability varies, ERQ is an imperfect proxy	<50%
Disamenity			
House price	Low	Data reported by government agencies and reliable market data available	<10%
Household density	Low	Data reported by government agencies	<10%
Hedonic factor	High	Meta-analysis of available literature is best available approach, but there is limited literature available	<100%
Air pollution			
Value of an IQ point	High	Value from a credible peer reviewed study, however high uncertainty in actual value of welfare impact	<100%
Presence of energy recovery	Low	Data availability varies, but typically reported by government agencies where recovery exists	<10%
VSL	Med	Estimated, method used is peer reviewed and broadly accepted although variation exists between estimates	<50%

8.2. Conclusions

Our overall averaged results for the societal cost of a tonne of waste in three example countries are shown to be relatively robust to changes in the majority of variables. They are sensitive to a few key variables, most notably the SCC, the composition of waste and the presence of a liner. The SCC is undoubtedly the most thoroughly explored amongst these and while its precise value remains uncertain, we can have confidence at least that the reasons for this uncertainty are well understood. Data quality in relation to the composition of waste is also highly variable but importantly it is within the control of the company in question to improve these data if they are thought to be material. Presence of a liner is perhaps the trickiest of the sensitive variables to reliably ascertain but a number of plausible approaches are shown to be possible to approximate the likelihood and the overall impact on the societal cost is modest.

It is important that any decision maker intending to use estimates of the societal cost of waste disposal in different locations is aware of the uncertainty around them and able to make an informed judgement about the implications of that uncertainty for their particular decision context.

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Appendices

Appendix I: Custom IPCC model default values

Table 25: Default DOC values by waste type

Waste category (MSW and industrial)	Default Value (%)
Food waste	15
Garden	2
Paper	4
Wood and straw	43
Textiles	24
Sewage sludge	5
Rubber	39
Bulk MSW waste	18
Industrial waste	15

Source: IPCC (2006b)

Table 26: Default DOC values by industry type for industrial waste

Industry type DOC	Industry type DOC (%)
Food beverages and tobacco (other than sludge)	15
Textile	24
Wood and wood products	43
Pulp and paper (other than sludge)	40
Petroleum products, Solvents, Plastics	0
Rubber	39
Construction and demolition	4
Other	1

Source: IPCC (2006b),

Table 27: Methane correction factors (MCF) for different types of waste management site

Type of Site	MCF Default Values
Managed – anaerobic[32]	1.0
Managed – semi-aerobic[33]	0.5
Unmanaged[34]– deep (>5 m waste) and/or high water table	0.8
Unmanaged[35] – shallow (<5 m waste)	0.4
Uncategorised SWDS[36]	0.6

Source: IPCC, 2006c

[32] Anaerobic managed solid waste disposal sites: These must have controlled placement of waste (i.e., waste directed to specific deposition areas, a degree of control of scavenging and a degree of control of fires) and will include at least one of the following: (i) cover material; (ii) mechanical compacting; or (iii) levelling of the waste.

[33] Semi-aerobic managed solid waste disposal sites: These must have controlled placement of waste and will include all of the following structures for introducing air to waste layer: (i) permeable cover material; (ii) leachate drainage system; (iii) regulating pondage; and (iv) gas ventilation system.

[34] Unmanaged solid waste disposal sites – deep and/or with high water table: All SWDS not meeting the criteria of managed SWDS and which have depths of greater than or equal to 5 metres and/or high water table at near ground level. Latter situation corresponds to filling inland water, such as pond, river or wetland, by waste.

[35] Unmanaged shallow solid waste disposal sites; All SWDS not meeting the criteria of managed SWDS and which have depths of less than 5 metres.

[36] Uncategorised solid waste disposal sites: Only if countries cannot categorise their SWDS into above four categories of managed and unmanaged SWDS, the MCF for this category can be used.

Appendix II: Valuing the societal cost of disamenity

A.2.1. Methods for primary valuation of disamenity

While not explicitly traded in markets, amenity can be valued through stated preference methods such as Contingent Valuation (CV), or by revealed preference methods such as Hedonic Demand Modelling (HDM) which examines transactions in the housing market. Whilst there are examples in the literature of both methods being used to value the disamenity resulting from waste sites; Eshet et al. (2005b) carry out a thorough review of the literature up to 2003 and suggest that HDM seems to be the favoured approach to value disamenity. This is consistent with Cambridge Econometrics et al. (2003) who cite an established criticism that CV may not adequately measure WTP for the environmental quality, assuming as it does that people understand the nature of the amenity in question. HDM is therefore the method used to value disamenity of waste sites in our methodology.

HDM uses the assumption that houses are a 'differentiated good' (differing in a variety of characteristics but so closely related in consumers' minds as to be considered one commodity (Cambridge Econometrics, EFTEC & WRc, 2003)) and so house price value represents a composite of an individual's willingness to pay for each characteristic of the house from the number of bedrooms to the quality of the local environment. If a landfill or incineration facility is located near the house, the price is likely to be affected by the individual's willingness to accept a price reduction to live near the facility. HDM examines bid curves for house prices to extract this information, taking other influential attributes of the house price into account. Data for HDM can be obtained based on actual sales within a specific time period, interviews with real estate agents or based on market valuations of a sample of properties within the area (Nahman, 2011).

A.2.2. Stock and flow externalities

Externalities can be conceptually divided into being either independent of emission quantity (and so fixed or 'stock' externalities) or proportional to emission quantity (and so variable or 'flow' externalities)[37]. Whilst most of the externalities arising from waste that have been considered so far can be considered as 'flow' – valuation of them being dependent on biophysical quantities of the emissions resulting from waste produced – disamenity could be thought of as having both stock and flow components. Some of the impact of a landfill on amenity may vary with volume of waste processed at the site, for example the impact of noise and disturbance from increased traffic, and so could be valued proportional to quantity of waste. Likewise, a significant component of the impact of a landfill on amenity will arise from its mere existence and so should be valued per site, and in relation to the distribution of houses around the site.

The relative significance of stock versus flow externalities from waste sites is debated in the literature. However, several studies identify stock externalities as most significant (see for example COWI, 2000) either implicitly or explicitly assuming that the disamenity associated with a marginal ton of waste is approximately zero (Eunomia, 2002). Whilst a per site measure is therefore the most readily justifiable for valuing the stock disamenity from a waste site – and there are a number of HDM studies that do this (for a summary see Eshet et al., 2005b) – there is still precedent to use HDM to estimate average disamenity cost per tonne of waste. This latter approach has more practical use in the context of the E P&L approach.

Estimating average disamenity cost per tonne of waste can be rationalised by considering the effect on house price as being equivalent to the capitalised net present value (NPV) of the loss of amenity caused by the site over the number of years it operates. This NPV can be apportioned to each ton of waste that has gone to the site by dividing the total house price variance at any point in time by the total discounted waste flows to the site over

[37] The following paragraph summarises the discussion in Cambridge Econometrics et at (2003), p.5-8

its lifetime (Pearce, 2005). This assumes that the disamenity effects of the site are constant (i.e. that the effect of a new landfill is the same as an older one) and that waste flows to the site are constant over time.

A.2.3. Primary valuations of disamenity in the literature

A number of primary studies have been carried out that use either CVM or HDM to value disamenity from both landfill and incineration sites. For this methodology, valuations were obtained from those studies using HDM both for the reasons described above and because values derived from CVM were not in an accessible format. Of around 19 HDM studies, 13 were carried out in the USA, 2 in Europe and one study each from Israel, South Africa, Nigeria and Uganda (see Table 28, adapted from Eshet, 2005a). These studies presented valuations in a variety of units: including percentage reduction in house price per unit distance from waste site, or cost per ton waste. In addition to these primary studies, a number of studies were identified that employ various benefit transfer techniques.

Table 28: Primary studies deriving social cost of disamenity

Study	Country	Disamenity valuation
Brisson & Pearce (1995)	USA	2.4% km^{-1} reduction in house prices up to 5.25km from site dHP% = 12.8 – 2.35r (where r is distance in km)
ExternE (1995)	Italy	2.8% km^{-1} reduction in house price EUR 13.2 ton^{-1} waste
Cambridge Econometrics et al. (2003)	Great Britain	Average UK: reduction of about USD 8,668 (7%) in house value in proximity of 0.4 km from landfill and USD 2,521 (2%) for 0.8 km; 0.8–1.6 km: 1.04% 1.6–3.2 km: 0.7% > 3.2 km: 0% GBP 1.52 – 2.18 $tonne^{-1}$ waste
Eshet (2007b)	Israel	Average of marginal price for each of first 4 km from site USD 5,031
Du Preez & Lottering (2009)	South Africa	dHP (ZAR) = 8246.03 + 0.36 r (where r is distance in m)
Nahman (2011)	South Africa	dHP (ZAR) = 318,469 + 6,940 r (where r is distance in km) for site 1; and dHP (ZAR) = 147,517 + 22,905 r for site 2 ZAR 75 ton^{-1} waste
Akinjare et al. (2011)	Nigeria	NGN 2,168 (6%) per 1.2 km away from landfill
Isoto & Bashaasha (2011)	Uganda	USD 534.34 km^{-1} up to 2 km

Source: adapted from Eshet, 2005a

Appendix III: Selected primary studies of hedonic pricing of disamenity

Table 29: Selected primary studies deriving hedonic price functions

Author	Location of study (number of sites)	Method used	Baseline HP used to derive maximum effect	Maximum effect (% baseline house price, average of sites)	Distance at which effect negligible (km, average of sites)
Akinjare et al., 2011	Nigeria (4)	Multiple regression assuming linear function, applied to four landfill sites in Lagos State.	Mean average HP for study area, quoted by the study	6	1.2
Nahman, 2011	South Africa (2)	Multiple regression assuming linear function, applied to two landfill sites in Cape Town.	Derived from function (i.e., estimated HP at y intercept)	23.22	4
Du Preez & Lottering, 2009	South Africa (1)	Multiple regression assuming linear function, applied to one landfill site in Cape Town. Function describes % increase in house price with distance from landfill, up to 2km, which is the easterly limit of housing	Derived from function (i.e., estimated HP at y intercept)	8.73	2
Eshet, 2007	Israel (4)	Multiple regression assuming semi log function with quadratic distance, applied to four waste transfer station sites in Israel.	Mean average HP for study area, quoted by the study	8.47	2.86
Cambridge Econometrics et al., 2003	UK (11,293)	Multiple regression assuming semi log function, applied national house price data and locations of landfills.	Mean average HP for study area, quoted by study	7.06	0.81

Author	Location of study (number of sites)	Method used	Baseline HP used to derive maximum effect	Maximum effect (% baseline house price, average of sites)	Distance at which effect negligible (km, average of sites)
Brisson & Pearce, 1995	US (11)	OLS meta-analysis of eleven HDM studies (ten landfills, one incinerator)	N/A	12.8	5.47
Mean				**11.05**	**2.72**
Standard deviation				**6.40**	**1.77**

Appendix IV: Population density around landfill sites

In the leachate and disamenity methodologies, we sometimes need to use national average population density to approximate the population density around landfill sites where local data are not available. This is an imperfect measure at best, however our research indicates that it is justifiable where no better data are available.

Reliable data on the actual average population density around landfill and incineration sites are available for the UK, based on a comprehensive study of 11,300 sites. This study finds the average population density around landfills to be 255.3 people per km². While this happens to be remarkably close to the UK national average population density of 259 people per km², it is acknowledged that this is by no means the case in all countries.

Use of national averages may appear particularly unsatisfactory in countries with large expanses of open space, and therefore a very low average population density, such as Australia. However, in such countries, population density tends to drop off very quickly outside of cities and towns where there is plenty of space for a landfill with only a small number of people suffering any disamenity (those within a 2.3km radius according to our analysis in Chapter 5 and Appendix III). Figure 10 shows the location of waste management facilities (landfills and transfer stations) across Australia, this shows that while the sites are clustered around urban areas, the majority are in fact distributed in the counties surrounding actual urban centres, where population densities are very much lower.

Figure 10: Location of waste management facilities (landfills and transfer stations) in Australia. Source: (Geoscience Australia)

Appendix V: Alternative approaches to valuing disamenity and leachate

A.V.1. Unit value transfer for disamenity

The method described above for estimating social cost of disamenity relies upon the availability of three types of location-specific data: average house price, average household density (derived from population density and average household size) and national or local data on waste quantities by treatment approach. If these data are not available, then a crude estimate of cost per tonne could be made using unit value transfer from existing primary studies of disamenity in the literature, adjusted for PPP and inflation and applied at a country level.

Table 30 includes estimates of WTP per tonne from primary hedonic pricing studies in three countries: Great Britain, South Africa and Italy.

Table 30: WTP per tonne waste from primary studies

Country	**Cost per tonne (2011 USD)**	**Study**
Great Britain[38]	3.09 – 4.43	Cambridge Econometrics et al. (2003)
South Africa	10.37	Nahman (2011)
Italy	25.12	ExternE (1995)

The specificity of the sites that the latter two estimates relate to, along with a shortage of credible primary values for other sites and countries, points to the use of a single consistent methodology as described in chapter 5. In addition, the recommended function transfer approach allows for the use of more specific local data where this is available, allowing for increasingly sophisticated value estimates as data improves.

A.V.2. Value specific outcomes based on leachate composition

Whilst estimating the specific composition of leachate across many sites is unrealistic, COWI (2000b) combines empirical observations of leachate composition by volume (Christensen et al., 1994) with assumed generation of leachate (White, 1995) to arrive at an assumed leachate composition in grams per tonne of MSW sent to landfill.

In theory, the environmental outcomes and corresponding societal impacts associated with this set of pollutants could be estimated by applying aspects of the PwC methodology paper: *Valuing corporate environmental impacts: water pollution*. However, a fundamental problem with using this approach in the E P&L context is that a universal leachate composition (derived from empirical observation of MSW sites) will not be applicable to all waste types in all locations. The COWI (2000b) method does not lend itself to adaptation to alternative waste types, which as shown in chapter 8 are a powerful determinant of the societal costs of waste disposal.

[38] Pearce (2005) points out that this particular value is calculated using a social discount rate of 6% which is at odds to that prescribed by UK Government recommended rate of 3.5% and proposed taking the upper limit the study proposes as a 'crude adjustment' to correct this.

We therefore conclude that following even this simplified approach to leachate composition would be unlikely to produce more accurate estimates of the social cost of leachate than the risk adjusted worst case scenario approach that we advocate.

Appendix VI: Estimating leachate risk using HARAS model

Table 31: Parameters in the complex version of the HARAS model

Category	Parameters
Source	Waste quantity (tonnes)
Source	Annual precipitation (mm)
Source	Cover system 1 Surface (top deck) grade (%) 2 Vegetative soil cover thickness (m) 3 Drainage layer thickness (m) 4 Clay layer thickness (m) 5 Geomembrane thickness (mm)
Source	Waste composition 1 Hazardous fraction (%) 2 Biodegradable fraction (%) 3 Construction and Demolition Waste fraction (%) 4 Other waste (dry inerts/recyclables etc.)
Source	Hazardous waste industries in vicinity of landfill with no hazardous waste landfill in region
Pathway	Leachate containment system a. Liner system Clay barrier thickness (m) Geomembrane thickness (mm) b. Leachate Collection and Removal System
Pathway	Vadose zone a. Vadose zone thickness (m) b. Vadose zone permeability (m/sec)
Pathway	Aquifer zone a. Thickness of groundwater aquifer (m) b. Aquifer permeability (m/sec) c. Groundwater gradient (%) d. Distance to nearest groundwater well (m)
Receptor	a. Groundwater aquifer b. Human population using groundwater from within 5,000 m of landfill

Category	Parameters
	c. Irrigation use of groundwater from within 5,000 m of landfill d. Livestock using groundwater from within 5,000 m of landfill *e.* Sensitive environment using groundwater from within 5,000 m of landfill

Source: Adapted from Singh et al. (2012)

Table 32: Leachate risk rating system in the simplified HARAS model

Inputs			**Outputs**	
Source rating	**Pathway rating**	**Receptor rating**	**Risk factor for unlined landfill**	**Risk factor for lined landfill**
Best	Best	Best	5	0.5
Best	Best	Medium	8	0.7
Best	Best	Worst	9	0.8
Best	Medium	Best	6	0.5
Best	Medium	Medium	10	0.9
Best	Medium	Worst	12	1
Best	Worst	Best	8	0.7
Best	Worst	Medium	14	1.2
Best	Worst	Worst	16	1.4
Medium	Best	Best	106	9
Medium	Best	Medium	180	16
Medium	Best	Worst	212	19
Medium	Medium	Best	135	12
Medium	Medium	Medium	229	20
Medium	Medium	Worst	270	24
Medium	Worst	Best	190	17
Medium	Worst	Medium	322	28
Medium	Worst	Worst	379	33
Worst	Best	Best	280	25
Worst	Best	Medium	476	42

Inputs			**Outputs**	
Source rating	**Pathway rating**	**Receptor rating**	**Risk factor for unlined landfill**	**Risk factor for lined landfill**
Worst	Best	Worst	560	49
Worst	Medium	Best	356	31
Worst	Medium	Medium	605	53
Worst	Medium	Worst	712	63
Worst	Worst	Best	500	44
Worst	Worst	Medium	850	75
Worst	Worst	Worst	1000	88

Source: Singh et al. (2012)

Table 33: Predicted societal costs of leachate per tonne of waste to landfill in Wisconsin (USA) in 2012 based on differing source, pathway and receptor characteristics

			Societal cost (USD 2012)	
Source	**Pathway**	**Receptor**	**Unlined**	**Lined**
B	B	B	0.35	0.03
B	B	M	0.55	0.05
B	B	W	0.62	0.06
B	M	B	0.41	0.03
B	M	M	0.69	0.06
B	M	W	0.83	0.07
B	W	B	0.55	0.05
B	W	M	0.97	0.08
B	W	W	1.10	0.10
M	B	B	7.31	0.62
M	B	M	12.42	1.10
M	B	W	14.63	1.31
M	M	B	9.32	0.83
M	M	M	15.80	1.38

			Societal cost (USD 2012)	
Source	**Pathway**	**Receptor**	**Unlined**	**Lined**
M	M	W	18.63	1.66
M	W	B	13.11	1.17
M	W	M	22.22	1.93
M	W	W	26.15	2.28
W	B	B	19.32	1.73
W	B	M	32.84	2.90
W	B	W	38.64	3.38
W	M	B	24.56	2.14
W	M	M	41.75	3.66
W	M	W	49.13	4.35
W	W	B	34.50	3.04
W	W	M	58.65	5.18
W	W	W	69.00	6.07

Appendix VII: Illustrative table of dioxin and heavy metal societal values

Table 34: Breakdown of dioxins and heavy metals from waste incineration

Pollutant Type	Pollutant	$/kg pollutant 2011 USD	$/tonne waste 2011 USD
Heavy Metals	Arsenic (As)	104	0.01
	Cadmium (Cd)	51	0.01
	Chromium (Cr)	261	0.00
	Mercury (Hg)	10,428	2.69
	Nickel (Ni)	5	0.00
	Lead (Pb)	782	0.44
Dioxins	Dioxins	240,000,000	0.13
	Total	**-**	**3.28**

Source: EXIOPOL (2009)

Table 34 shows that, in this impact pathway, the most significant societal costs per tonne of waste incinerated are likely to be from mercury, lead, and dioxins. However, in certain contexts, the proportions may vary, so this methodology considers the range of pollutants listed above.

Appendix VIII: Emissions factors and dose response rates for dioxin and heavy metal emissions

Table 35: IPCC emission factors for dioxins and heavy metals

Pollutant	Value[39]	Unit	Cited reference
Lead (Pb)	1.3	g/Mg waste	Theloke et al. (2008)
Cadmium (Cd)	0.1	g/Mg waste	Theloke et al. (2008)
Mercury (Hg)	0.056	g/Mg waste	European Commission (2006)
Arsenic (As)	0.016	g/Mg waste	Theloke et al. (2008)
Chromium (Cr)	0.3	g/Mg waste	EMEP & EEA (2009a)
Nickel (Ni)	0.14	g/Mg waste	Theloke et al. (2008)
Dioxins (PCDD/F)	350	µg I-TEQ/Mg waste[40]	UNEP (2005)

Source: EMEP & EEA (2009a)

[39] These are the default values provided by the European Environment Agency in the EMEP CORINAIR emission inventory guidebook (EMEP & EEA, 2009a) and assume an averaged or typical technology and abatement implementation in the country (using only particle emission abatement equipment for controlling emissions).

[40] I-TEQ, or international toxic equivalence, is a standardised scale by which all dioxins and furans (including PCDD and PCDF) are weighted relative to their toxicity to the most toxic dioxin, TCDD. This allows a single dose response rate to be applied a mass of any mixture of dioxins or furans, provided mass are stated in I-TEQ.

Table 36: EU emission limits for dioxins and heavy metals that are lower than emission factors from EMAP & EEA (2009a)

Pollutant	Emission factor	Unit
Lead (Pb)	0.57	g/Mg waste
Chromium (Cr)	0.0033	g/Mg waste
Dioxins	0. 515	µg I-TEQ/Mg waste

Source: EMEP & EEA (2009a); EC (2000)

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Valuing corporate environmental impacts: Water consumption

PwC methodology paper

Version 2.2

Contents

Table of Tables

Table of Figures

Table of Equations

Abbreviations and acronyms

Abbreviation	Full name
DALY	Disability-adjusted life year
DF	Damage Factor
EEIO	Environmentally extended input-output modelling
EF	Effect Factor
EP&L	Environmental Profit and Loss
FAO	Food and Agriculture Organisation
GHG	Greenhouse gas
GNI	Gross national income
GPS	Global Positioning System
HDI	Human Development Index
HHF	Human Health Factor
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
LCIA	Lifecycle impact assessment
NGO	Non-governmental organisations
OECD	Organisation for Economic Co-operation and Development
PPP	Purchasing power parity
UN	United Nations
UNDP	United Nations Development Programme
US	United States
VSL	Value of a statistical life
WDF	Water Deprivation Factor
WHO	World Health Organisation
WSI	Water stress index
WTP	Willingness To Pay

1. The environmental impacts of water use

1.1. Introduction

All corporate activity directly and indirectly relies on water availability. Water consumption is the volume of water that is evaporated, incorporated into a product or polluted to the point where the water is unusable (Mekonnem & Hoekstra, 2011). Consumption of water reduces the amount of water available for other uses, which, depending on the level of competition and the socio-economic context, can have consequences for the environment and people. It is valuing the impacts associated with corporate consumption which is the focus of this methodology.

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

As we will demonstrate in the discussion that follows, the availability of water is typically not the sole (and moreover not the most significant) driver of impacts of corporate water consumption. Areas where competitive water consumption impacts are highest are typified by poor sanitation, inadequate water supply infrastructure, basic public health care, poverty and high malnutrition. The responsibility for impacts driven by water consumption is shared not just with the corporate users but with other water consumers and most importantly with local and national governments. The methodology presented here estimates the impacts of corporate water consumption taking the local context as a given, and does not consider the level of responsibility for the prevailing socio-economic context[1].

The impacts of corporate water pollution are not considered in this report. The PwC methodology paper *Valuing corporate environmental impacts: Water pollution* provides a method to value the impacts of specific water pollutants which are emitted as a result of corporate activity.

1.1.1. Environmental and societal outcomes

Where corporate water consumption leads to a reduction in available clean water to other users reliant on the same source societal impacts could include:

- **Human health - Malnutrition:** In water scarce areas corporate water consumption may reduce the water available to agricultural users, reducing yields. In areas dependent on local food production this may lead to increases 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

[1] 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 assessments (see, for example, PwC's Total Impact Measurement and Management framework).

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 are dependent on groundwater and are extracting it at an unsustainable rate leading to groundwater depletion and an inflow of saline water. Over exploitation of non-renewable water supplies will lead to future impacts associated with the increased scarcity and cost of supply, unless other sources are secured.
- **Other ecosystem services:** Removal of fresh surface water can reduce the functioning of ecosystems, particularly in riparian areas. The associated loss in ecosystem services may lead to a reduction in ecosystem services and the associated impacts for the local population, including market and non-market losses from fishing and recreation, for example.
- **Subsidy cost of water:** Water pricing rarely reflects the financial cost of water supply, and is frequently subsidised. Corporate use therefore increases the financial burden for tax payers.
- **Economic opportunity cost of water:** Where there is direct competition for water, and the corporate using the water is not the most economically productive user (based on the marginal private and public benefits of production) there is an opportunity cost of water use.
- **Environmental impacts of water supply sector:** The supply of water prior to use by corporates requires energy and raw materials, which will have other environmental impacts associated, including greenhouse gases (GHGs), air emissions and waste, water pollution and land use.

As stated above, the extent and severity of the impacts from corporate water consumption is highly dependent on the local conditions. Table 1 provides a list of the some of the main factors that typically influence the extent of impacts.

Table 1: Main variables known to influence societal impacts from corporate water consumption

Impact pathway	Variable
All	Water scarcity and competition between users
Health - malnutrition	Prevailing malnutrition rates
	Resilience of food production systems to water shortages
	Ability to secure food from alternative sources
Health – infectious water-borne diseases	Water supply and treatment infrastructure
	Quality of alternative water sources
	Prevailing infectious water-borne disease rates
	Level and availability of health care
Resource depletion	Dependence on groundwater and rate of natural recharge
	Availability and cost of alternative water sources
Other ecosystem services	Resilience of local ecosystems to water withdrawals
Subsidy cost of water	Level of subsidy
Opportunity costs of water	Level of direct competition for water, relative value of other uses
Environmental impacts of water supply sector	Technology of water and electricity/energy sectors.

1.2. *Impact pathway*

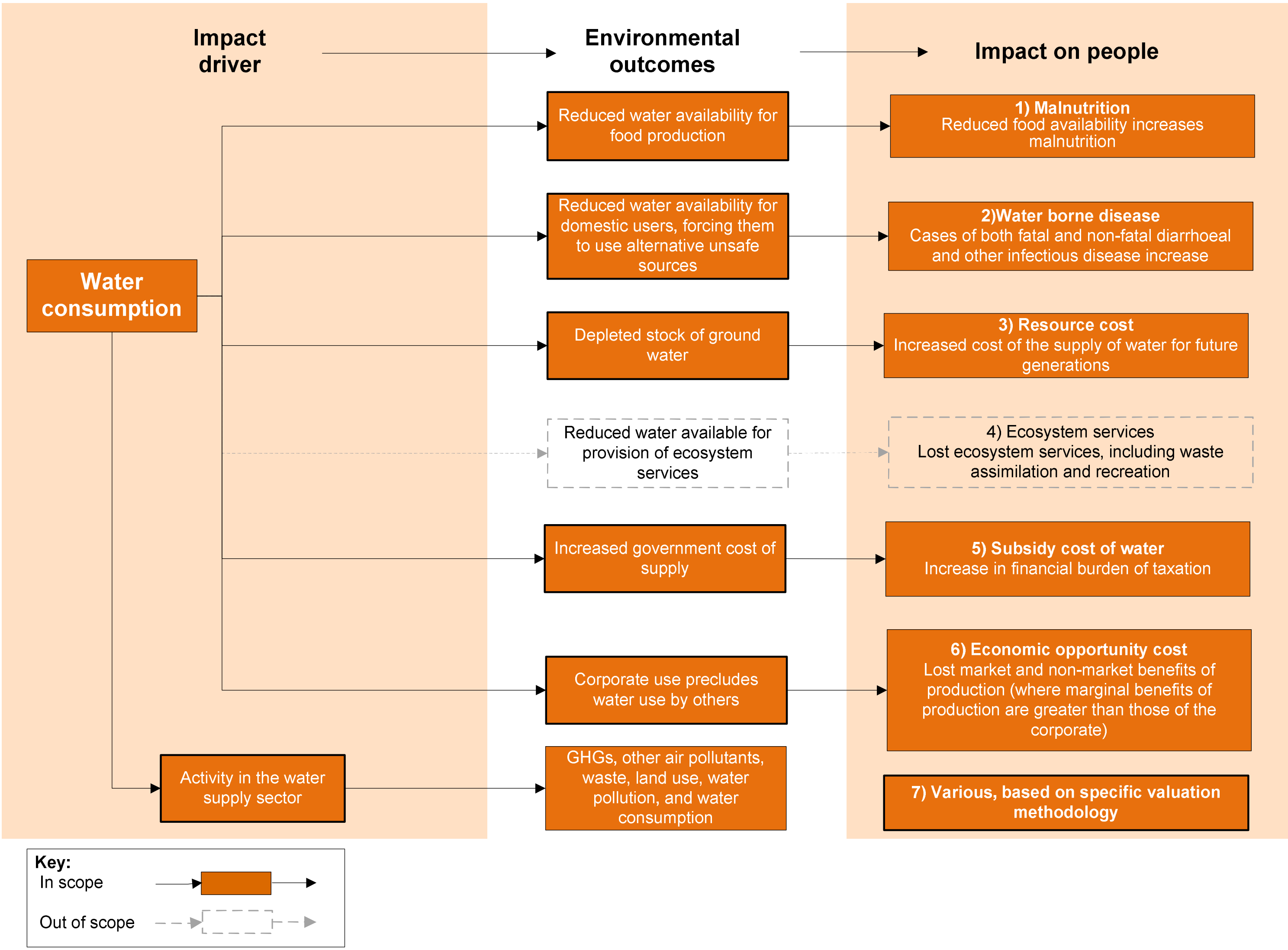
In order to value corporate environmental impacts, we need to understand how corporate water consumption affects societies now and in the future. We use impact pathways to depict the causal links between corporate activities, their environmental impacts, and the resulting societal outcomes. Our impact pathway framework consists of three elements:

- **Impact driver:**
  - *Definition:* These drivers are expressed in units which can be measured at the corporate level, representing either an emission to air, land, or water, or the use of land or water resources[2].
  - *For water consumption:* The volume and location of corporate water consumption.
- **Environmental outcomes:**
  - *Definition:* These describe actual changes in the environment which result from the impact driver (emission or resource use).
  - *For water consumption:* Reduced availability of water for other users and depletion of groundwater reserves at an unsustainable rate and the impact of the water supply sector.
- **Societal impacts:**
  - *Definition:* These are the actual impacts on people as a result of changes in the environment (environmental outcomes)
  - *For water consumption:* Human health impacts, future costs to society of alternative water sources and impacts via GHGs, air pollution and waste from the water supply sector.

The three stages of the impact pathway are shown in Figure 1 overleaf. The label 'out of scope' identifies elements of the impact pathway which are not addressed in detail in our methodology. The reasons for any such limitations of scope are explained at the end of this chapter.

[2] A note on language: In this report, the measurement unit for any "impact driver" is an "environmental metric." Therefore, water consumption is the impact driver, and m^3 of water used is the environmental metric.

Figure 1: Impact pathway for water consumption



1.3. Prioritising which impacts to quantify and value

This section outlines the impact pathways for corporate water consumption that will be quantified and valued. It also identifies those that are beyond the scope of this paper.

The United Nations Environment Programme and The CEO Water Mandate conducted an assessment of methods and tools for measuring water use and impacts (UNEP, 2013), which highlighted that most existing approaches focus purely on use measurement in m^3 and very few actually consider the impacts or consequences of water consumption. The most relevant studies come from the LCIA (lifecycle impact assessment) community, most notably Pfister *et al.'s* (2009) study into the environmental impacts of freshwater consumption and Motoshita *et al.'s* (2010) paper on the impact of infectious diseases caused by domestic water scarcity. Our approach to prioritisation builds on these studies and UNEP's review to evaluate the potential materiality of the impact pathways identified.

1.3.1. Valuation priorities

1.3.1.1. Valuation priorities covered by this PwC methodology paper

1.3.1.1.1. Health impacts

Our priority focus in this analysis is the indirect impact of corporate water consumption on health. The way in which corporate water consumption will affect human health is dependent on the context in any given location. The two health impacts of reduced clean water availability considered here are malnutrition and water-borne diseases which are also the focus of the two studies noted above.

1.3.1.1.2. Resource depletion

Some countries are exploiting groundwater reserves faster than they are naturally replenished. For example, it is predicted that some regions of the US will completely exhaust their supplies of groundwater within the next 30 years (Steward, 2013). Similar to health impacts, resource depletion is only an issue in certain locations with a specific set of conditions including a depleting groundwater supply and lack of alternative fresh water sources. The potential future impacts of groundwater depletion are hard to predict, but could be severe if alternative sources are not found. The projected costs of generating alternative fresh water are relatively high. For example despite predicted technological advances, desalinisation is still energy intensive and relatively expensive, particularly where transport is required.

1.3.1.1.3. Subsidy cost of water

This paper identifies a methodology to assess the subsidy costs imposed on others as a result of water use. The impacts may not always be material, particularly in areas with poor infrastructure, and can therefore be excluded in those instances. However, in some developed countries subsidy costs may be one of the principle impacts.

1.3.1.1.4. Opportunity cost of water

Like subsidy costs, opportunity costs will not always be relevant, however if they are, the impacts could be significant. Opportunity costs are only considered where there is direct competition for water, and the corporate using the water is not the most economically productive user in that locality.

1.3.1.1.5. Environmental impacts of water supply sector

The operations of the water supply sector can be fairly energy intensive, depending on the location in question. The quantification of impacts is considered in Chapter 3, but valuation methodologies are not presented here because the methodology should follow those discussed in each of the dedicated papers for greenhouse gases (GHGs), air emissions and waste, water pollution and land use. Water use by the water supply sector should be valued as per this methodology.

1.3.1.2. Limitations of scope

1.3.1.2.1. Other ecosystem services

Pfister *et al.* (2009) present a method to assess ecosystem impacts of water consumption measured using an index call potentially disappeared fraction of species. While a useful basis for comparison, we consider this indicator to be too removed from the actual impacts on people to be included here. Therefore, in this methodology, we do not present an approach for assigning causality between impacts on other ecosystem services and corporate water use. While in certain locations and at certain times, the impacts of water scarcity on ecosystems can be a very significant local issue, the principle consequence for people is via health – which is captured above. In more developed countries, other impacts may include reduced recreation and aesthetic enjoyment or reduced biodiversity. However, for most ecosystems these impacts are primarily triggered by reduced rainwater. However, where a corporate is known to be drawing water from a heavily depleted system resulting in the loss of ecosystem services, these can be valued using bespoke site specific methodologies.

1.3.1.2.2. Health impacts – social unrest and the displacement of people

Some of the literature, particularly from the NGO community, considers how water scarcity does or could contribute to social unrest and the displacement of people. We do not seek to include the potential for these impacts here because the drivers of impact are complex and corporate consumption is likely to be a minority driver and impossible to model in a generic and transferable approach.

2. Summary of our methodology

2.1. Introduction

To understand the value of environmental impacts associated with corporate activities, it is necessary to:

1. **Obtain environmental metric data:** The starting point for each of our methodologies is data on emissions or resource use. These metric data are based on an understanding of the corporate activities which they result from. The data can come from a variety of sources, some of which (e.g., life cycle assessment (LCA) or environmentally extended input-output modelling (EEIO)) are subject to their own distinct methodologies[3].

Table 2: Environmental metric data for water consumption

Impact driver (emission or resource use)	Environmental metric data
Water consumption	Volume of water consumed (m^3)

2. **Quantify environmental outcomes:** We quantify physical changes in the environment resulting from corporate emissions or resource use (as measured by the metric data). This is discussed further in Table 3, column 2.
3. **Estimate societal impacts:** We estimate the societal impact (impact on people) resulting from environmental changes which in turn are the result of corporate activities. This is discussed further in Table 3, column 3.

It is not always necessary or appropriate for economic valuation of the environment to go through each of these steps explicitly. A single methodological step may cover some or all steps at once. However, developing each EP&L valuation methodology by following a clearly defined impact pathway helps to retain a causal link and ensure rigor, transparency, and consistency.

2.2. Summary of methodology

Environmental metric data on corporate water consumption are the starting point for this methodology paper and hence the methods for collecting or estimating these data are not exhaustively covered. However, for the purposes of valuation it is important to understand any additional characteristics of the metric data that are likely to be available. For water consumption, the geographical resolution of these data (country, state, city, watershed, GPS location) is particularly important as the general availability of water in the area of consumption will drive how much water is being deprived from other users. Because these additional metric data characteristics depend to some extent on the source of the data, we outline the most likely data sources in Chapter 3, Table 4.

This methodology for taking the metric data on water consumption and quantifying and valuing the associated impacts on society is summarised in Table 3 below. The table describes the methodologies for each of the major impact pathways that we value in turn:

1. Malnutrition impacts
2. Infectious disease impacts
3. Ground water depletion impacts
4. Subsidy costs

[3] The sources of metric data are outlines in Chapter 3. The assumed starting point for this methodology is the form specified in Table 2.

5. Economic opportunity costs

Considering:

- The key methods and steps;
- The key variables for which data must be collected at each step;
- The key assumptions and justifications underlying each methodological choice.

Table 3: Overview of our impact valuation methodology: estimating societal impacts from water consumption

	Quantify environmental outcomes	Estimate societal impacts
Malnutrition impact pathway		
Methods	• Drawing on the analysis of Pfister *et al.* (2009), the reduction in the available fresh water for agriculture is calculated at the watershed level, considering the volume of corporate water consumption and the level of water stress in the specified watershed.	• A cause-effect chain is established by Pfister *et al.* to estimate the malnutrition impact on the local population of increased water scarcity. The extent to which water is a limiting factor in agricultural production is used to estimate a reduction in agricultural output. • This is translated into a number of cases of malnutrition using the minimum water requirement per capita to avoid malnutrition. • The results are adjusted according to the prevalence of malnutrition in that area to indirectly take into account the prevalence of food shortages and the ability to import food. • The number of cases of malnutrition is converted into Disability Adjusted Life Years (DALYs) using a regression of country level malnutrition cases and DALYs associated with malnutrition. • A monetary value of each DALY is calculated based on the value of a statistical life (VSL) and the lost DALYs associated with the VSL estimate.
Key variables	• Corporate water consumption (m^3) per capita. • Water stress index (WSI).	• Proportion of water used by agricultural sector. • Human Development Index (HDI). • Malnutrition rate (number of DALYs per capita). • Malnutrition rate (% of people affected). • VSL estimate. • Years of lost life associated with VSL estimate.

	Quantify environmental outcomes	Estimate societal impacts
Assumptions and justification	• The approach assumes that increased corporate water consumption would directly reduce the water available to agricultural users (based on the WSI), which is likely to be the case if the water infrastructure is linked to the same source for both users. • This assumption can be adjusted using more specific data if the analysis is being carried out at a local level, however at a regional or national level this is a necessary simplification.	• The methodology doesn't explicitly include the ability to import food from other regions or countries when the local agricultural sector doesn't provide enough. This is indirectly accounted for by including the prevailing malnutrition rate which will be lower in countries where it is easier to import food from an alternative source.
Infectious water-borne disease impact pathway		
Methods	• The volume of water that could be withdrawn by domestic users if it were not consumed by corporate users is estimated using the WSI for the location of consumption. The WSI is thus used as a measure of the competition for water between corporate and other users (Pfister *et al.*, 2009).	• 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 explanatory variables used are selected to explain the socio-economic drivers of water-borne disease, they are: domestic water use, health expenditure, prevalence of undernourishment, government effectiveness and the water stress level. • The derived relationship is used to predict the fall in prevalence of water-borne disease if the quantity of water which corporates deprive domestic users of (based on the WSI) was reallocated to domestic users. • The resultant change in DALYs per capita is valued and allocated across the total corporate water use to give a welfare impact per m^3.
Key variables	• Corporate water consumption (m^3) per capita • Water stress index (WSI)	• DALYs per capita associated with infectious water-borne diseases • Domestic water use per capita • Corporate water use per capita

	Quantify environmental outcomes	Estimate societal impacts
		• Health expenditure • Prevalence of undernourishment • Health expenditure per capita • Government effectiveness • WSI • VSL estimate • Years of lost life associated with VSL estimate
Assumptions and justification	• The WSI represents the proportion of corporate water consumption which would otherwise be available to domestic users. The volume of water not available to domestic users is assumed to be total corporate consumption, adjusted by the WSI: this assumption will hold where conditions (e.g. infrastructure) are such that, when a corporate user reduces its water consumption, a domestic user can access it. This may not be the case in reality where 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 the associated error.	• Results from selected quantiles can be applied to other locations using the level of water-borne disease to assign the most appropriate coefficients. • The data shows that if we applied an OLS we would over estimate impacts for countries with low levels of water-borne disease, and under estimate impacts for countries with high levels. We therefore 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 different quantiles, we suggest sensitivity analysis to explore the potential impacts using the coefficients in both quantiles.
Groundwater resource depletion impact pathway		
Methods	• The rate of groundwater depletion and the expected time to depletion are used to estimate the future annual shortfall in water supply.	• We calculate replacement cost as a lower-bound estimate of likely societal impacts of groundwater depletion. • The cost is calculated based on predicted desalinisation costs and the cost to

	Quantify environmental outcomes	Estimate societal impacts
		transport desalinated water to main populated areas. • The total costs of the annual shortfall is allocated to current use to give a per m^3 value.
Key variables	• Rate of groundwater depletion • Time to depletion	• Desalinisation costs • Transport costs
Assumptions and justification	• New ground water reserves will not be discovered at the same cost of current extraction. This holds for most locations where sufficient hydrological data is present to identify depletion.	• Projected cost of supply is an appropriate proxy for the societal costs. While this is a likely to be a lower bound of potential impacts, replacement costs are deemed an acceptable proxy where better information is not available. • Desalinisation is the technology of choice, or has similar costs to preferred alternative technologies. 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. • Assumes that profits from ground water extraction aren't ring fenced as funding for the future supply water. • As with any projection, there is significant uncertainty over the future societal impacts associated with groundwater depletion occurring today.

Subsidy cost of water use		
Methods	• The environmental outcome is not required for this calculation.	• The subsidy cost is the difference between the water sector revenues (from water prices) and financial costs of supply.
Key variables		• Price of water or water sector revenue • Cost of water supply • Or level of subsidy per m^3
Assumptions and justification		• Corporate consumers are not supporting water subsidies through a ring-fenced fund. If companies are supporting subsidies through a ring-fenced fund, and that their contribution is at least proportional to the quantity of water they withdrawal, they are not putting an additional burden on other tax payers. This is assumed not to be the case unless information to the contrary is demonstrated.
Economic opportunity costs of water use		
Methods	• Identify the quantity of water that users which could deliver higher societal benefits per m^3 are deprived of as a result of corporate consumption.	• Estimate the loss in societal benefits (including private revenues and public gains) as a result of inefficient allocation of water resources, based on the marginal productivity of consumption.
Key variables	• Various, depends on location, available data and chosen method.	• Various, depends on location, available data and chosen method.
Assumptions and justification	• Various, depends on location, available data and chosen method.	• Various, depends on location, available data and chosen method.

3. Data requirements

3.1. Introduction

The availability of high quality data on corporate water consumption across the value chain, as well as the accessibility of relevant contextual information, are key determinants of the viability of different impact quantification and valuation techniques and will affect the ultimate level of uncertainty surrounding any results.

Gathering appropriate data is a precursor to valuing the environmental impacts from corporate water consumption. Therefore, this chapter introduces the types and potential sources of data required to value corporate water consumption.

Three types of broad data are considered here:

- **Metric data:** Volume of water consumed across a company's value chain.
- **Contextual data:** Provides additional relevant information about the basic metric data. For example, describing the context in which the water consumption occurs (e.g. location, local water stress). The availability of useful contextual data will depend to an extent on the source of the metric data. For example, in the case of directly collected data, the location characteristics should be known. Whereas in the case of data sourced from Life Cycle Assessment (LCA) databases, location specific data will be more limited.
- **Other coefficients:** Typically numerical values derived from the academic literature or other credible sources which are required in calculations to convert metric and contextual data into value estimates.

While methods for the collection or estimation of basic metric data are not the subject of this paper, the data generation methods used are nonetheless relevant to the likely availability of contextual data and therefore the viability of different potential valuation approaches. This chapter therefore has two purposes: firstly, it describes the most likely sources of metric data across a typical corporate value chain and the implications for contextual data availability; secondly, it sets out key contextual and other coefficient data requirements and the preferred sources for these.

The mythology that follows discusses how to use these data to estimate the impacts of corporate water consumption. The methodology is designed to be flexible to the level of available data, so the more detail that can be provided on metric and contextual data, the more specific the final results can be. The required level of detail based on the objectives of the analysis should always be kept in mind when assessing data availability and quality.

3.2. Environmental metric data

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 give different levels of data specificity depending on the application. For example, LCA databases are typically rich in data on plastics but statistical agencies of industry bodies are likely to have more up to date information on industrial water use intensities. Similarly with EEIO, the data are only as specific as the country and sector resolution provided in the model.

Likely metric data availability across the corporate value chain and implications for contextual data are shown in Table 4.

Table 4: Likely metric and locational data availability across a corporate value chain

	Metric data	Implications for contextual data
Own operations	Direct measurement water consumption. The other estimation techniques detailed for the supply chain can also be used if direct data is unavailable.	Based on knowledge about the location of the company and supplier, it should be possible to source contextual information from public sources, if not from the company and their suppliers themselves.
Immediate suppliers	Supplier questionnaires can be directed to areas of high materiality or those with limited quality data from other sources. Most companies would be expected to quantify their water consumption based on their water bills. Any water consumption that is released unpolluted back into the same fresh water source should be excluded from this measure of water consumption.	
Upstream/ supply chain	EEIO can be used to give an approximation of corporate water consumption based on a company's purchase ledger. LCA databases can be used for more process specific data where this is deemed appropriate. Other data sources include government and industry reports: those from the IPCC may be particularly relevant.	Depending on the visibility of the supply chain location, information may or may not be available for some suppliers. Tracing raw material flows can be a good method of determining the location of different activities and processes in the supply chain. Multi-region EEIO models and trade-flow data bases can be used to approximate this, or supplier questionnaires where feasible.
Downstream/ use phase	It is necessary to estimate the probable water consumption associated with the product or service over its life. For an item of apparel, this is the expected amount of water consumption used to wash the item over its lifetime.	Depending on the product, the location of sale could be used as an estimation of the use and disposal phase location. In some cases, it may be necessary to consider trade flows using data bases or multi-region EEIO.

3.3. Environmental metric data for the water supply sector

The impacts associated with supplying clean water for corporate water consumption may be a significant portion of the total water impacts, particularly if desalinisation or long distance water transportation is required. Metric data is therefore also required for these impacts, including emissions of GHGs, other air emissions, water pollution, waste and use of land and water consumption.

Where details of the specific location and water sector infrastructure is known the likely impacts can be modelled based on the supply technologies in place using industry and government data (see Box 1 for an example).

Other more generic estimation techniques include LCA databases and environmentally extended input output modelling (putting the price of water into the water supply sector will give an approximate estimate of triggered activity) can be used where less specific data is available, or materiality is considered to be lower.

Box 1: GHG and air pollution impacts of desalinisation in Cyprus

Cyprus has significantly invested significantly in water supply and distribution infrastructure over the last two decades. While this helps avoid significant health impacts, increased use of desalinisation leads to higher impacts sector associated with energy use in the water supply.

Desalinisation impacts can be calculated based on an average electricity requirement of 4.5kwh/m³ (Cyprus Water Development Department, 2010), combined with Cypriot electricity emission factors (Eurostat, 2010), as per the table below. These metric data are valued based on the methodology presented in the dedicated papers to estimate the societal impacts.

Emissions	**Emission factor** kg/KWh	**Emission per unit of water produced** kg/m3
GHG	0.768915	3.460117
NO_X	0.002365	0.010643
SO_X	0.003189	0.014349
NH_3	0.000001	0.000005
PM_{25}	0.000064	0.000288
PM_{10}	0.000032	0.000144
NMVOC	0.000275	0.001239

3.4. Contextual and other data

The impact of corporate water consumption is highly dependent on the location. If the exact location of a company's direct or indirect water consumption is known, location specific data should be applied in the valuation wherever possible. The contextual and other coefficient data used to quantify outcomes and value societal impacts of water consumption is set out in Table 5.

Table 5: Contextual data used to value the impact of corporate water consumption

Data	**Explanation**
Malnutrition impact pathway	
Water stress index	Water consumption as a proportion of total renewable supply, a measure of how much water other users are deprived of when one user consumes water.
Proportion of water used by agricultural sector	WSI is combined with the proportion of water used by the agricultural sector to estimate the impact of corporate water consumption on the ability of the agricultural sector to consume water.

Data	Explanation
Water requirement to avoid malnutrition	The amount of water required by the agricultural sector per capita to provide enough food to avoid malnutrition.
Human development index	An index of the general development of a country or a region. This is combined with the malnutrition rate (measured in DALYs) to create a malnutrition human development factor which is an indicator of the level of development of the country driven by malnutrition.
Malnutrition rate (DALYs per 100,00 people)	The number of annual DALYs per capita caused by malnutrition. This is used as above and combined with the malnutrition rate (%) to estimate the number of DALYs caused by each case of malnutrition.
Malnutrition rate (%)	Proportion of the population suffering from malnutrition. Explanation of use as above.
Value of a DALY	Monetary value of the damage function calculated in DALYs/m^3. DALY value is calculated based on OECD estimate of the value of a statistical life (VSL).
Infectious water-borne disease impact pathway	
Water-borne disease rate (DALYs per capita)	The number of DALYs attributable to infectious water-borne diseases per capita, broken down into two sub datasets. DALYs per capita attributable to diarrhoeal diseases and DALYs per capita attributable to other infectious water-borne diseases.
Domestic water withdrawal per capita	The volume of water used by domestic users. Used as an explanatory variable in the regression of DALYs attributable to water-borne infectious diseases.
Corporate water consumption per capita	The volume of water consumed by corporate water users. Adjusted using the WSI and applied to the function developed with regression analysis to predict the amount change in the predicted DALYs rate if all corporate water consumption were transferred to the domestic users.
Health expenditure per capita	Used as an explanatory variable in the regression of DALYs attributable to water-borne infectious diseases.
Household connection to water supply	Used as an explanatory variable in the regression of DALYs attributable to water-borne infectious diseases.
Undernourished population	Used as an explanatory variable in the regression of DALYs attributable to water-borne infectious diseases.
Water Stress Index	Used as an explanatory variable in the regression of DALYs attributable to water-borne infectious diseases. WSI is also used to estimate the amount of water deprived from domestic users when corporate users consume water.
Government effectiveness	Used as an explanatory variable in the regression of DALYs attributable to water-borne infectious diseases.

Data	Explanation
Value of a DALY	Used to put a monetary value on the damage function calculated in DALYs/m^3. DALY value is calculated based on OECD estimate of the value of a statistical life (VSL).
Ground water resource depletion	
Time to ground water depletion	The number of years until a country or area is expected to have until it completely depletes its ground water reserves.
Rate of ground water depletion	The amount that ground water reserves have been depleted by in the year of activity.
Cost of desalinisation	The cost of producing clean, usable water in the future when ground water reserves have been fully depleted.
Subsidy cost of water	
Financial data on water price or revenue, and cost of supply	Used to estimate cost recovery from water pricing and therefore level of subsidy.
Economic opportunity cost of water	
Volume of water which other, more productive users, are deprived of	Used to identify inefficient allocation of water resources.
Marginal private and public benefit of corporate water use and alternative use	Used to identify inefficient allocation of water resources.

4. Detailed methodology: malnutrition impacts

This chapter presents a methodology to estimate and value malnutrition impacts associated with corporate water use. We use the standard metric of Disability Adjusted Life Years (DALYs) to estimate the extent of impacts, drawing on analysis of Pfister *et al.* (2009). We value DALYs to estimate the welfare impacts per m^3 of water consumption. Impacts tend to be focused in areas with a high competition for water and where local populations are dependent on local agricultural production. Where these conditions are both not the case, impacts tend to be close to zero. For a summary of the analysis see Table 6.

Table 6: Summary of the methodology for the calculation of the societal impacts of malnutrition

4.1 Quantify environmental outcomes		4.2 Estimate societal impacts
Malnutrition impact pathway		
Methods	• Drawing on the analysis of Pfister *et al.* (2009), the reduction in the available fresh water for agriculture is calculated at the watershed level, considering the volume of corporate water consumption and the level of water stress in the specified watershed.	• A cause-effect chain is established by Pfister *et al.* to estimate the malnutrition impact on the local population of increased water scarcity. The extent to which water is a limiting factor in agricultural production is used to estimate a reduction agricultural output. • This is translated into a number of cases of malnutrition using the minimum water requirement per capita to avoid malnutrition. • The results are adjusted according to the prevalence of malnutrition in that area to indirectly take into account the prevalence of food shortages and the ability to import food. • The number of cases of malnutrition is converted into Disability Adjusted Life Years (DALYs) using a regression of country level malnutrition cases and DALYs associated with malnutrition. • A monetary value of each DALY is calculated based on the value of a statistical life (VSL) and the lost DALYs associated with the VSL estimate.

Box 2: What is a Disability Adjusted Life Years (DALY)?

DALYs measure the overall burden of disease, combining years lost due to premature death (YLL) and 'healthy' years lost to ill health or disability (YLD). The number of healthy years lost are calculated by multiplying the length of time the disease occurs and a disability weighting based on the severity of the disease as described in Prüss-Üstün *et al.*'s (2003) report for the WHO on assessing the environmental burden of disease.

Equation 1: DALY equation

$$DALYs = YLL + YLD$$

4.1. Environmental outcomes

The environmental outcome of water consumption is reduced water availability for other users. For the malnutrition impact pathway we focus on the amount of reduced water available to the agricultural sector (Pfister *et al.,* 2009). The next section considers whether this reduction in water affects agricultural output and if that impacts on local people's access to nourishment.

4.1.1. Calculate Water Stress Index (WSI)

Water stress indices describe the proportion of available water that is consumed by all users. They are used to indicate the level of pressure on water resources and provide a measure of the potential for competition between users.

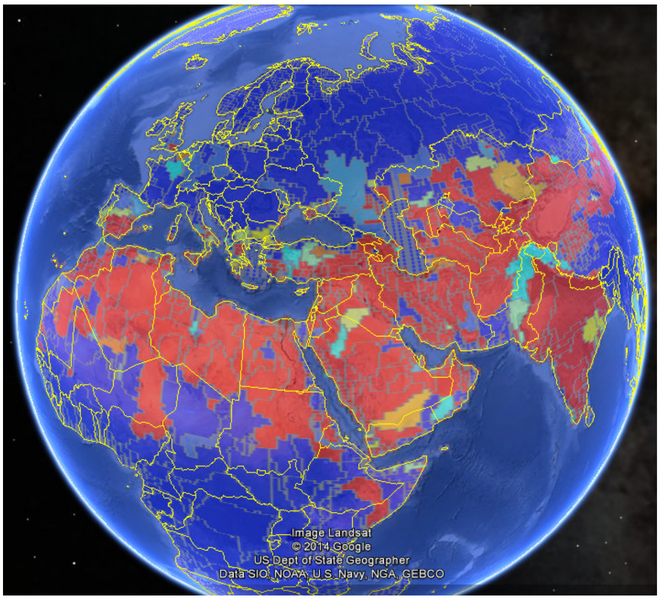
As such the WSI is an important variable in understanding the potential for malnutrition impacts because in locations with a high WSI corporate water use is more likely to reduce water availability for agriculture. Conversely in areas with plentiful water supply and no completion between users, corporate consumption will have no direct impact on agriculture or malnutrition.

There are various measures of WSI available. They are all a measure of the ratio of water withdrawals to the total water availability within a defined area, usually a watershed. Pfister et al. (2009) uses WaterGAP2 - a global model of 10,000 individual watersheds (Figure 2).

To calculate the WSI, the ratio of water withdrawal to available water is adjusted using a variation factor which takes into account how significantly water availability is affected by variations in monthly and annual precipitation. The ratio is transformed to a scale of 0.01 to 1 which indicates the average proportion of water consumption by one user that deprives another user of water in the given watershed (see Alcamo *et al.*, 2003 for more details on this calculation).

Figure 2: Physical water stress index by watershed visualised in Google Earth.

0-no water stress (blue) to 1 – extreme water stress (red)



4.2. Societal outcomes

To estimate the societal outcomes we draw on the work of Pfister *et al.* (2009) to consider if a reduction in water available to agriculture will affect agricultural production, and consequently local food supplies. The output of Pfister et al.'s (2009) work is Disability Adjusted Life Years (DALYs, see Box2) per m^3 of consumed water. We combine this with a value per DALY to produce a welfare estimate of the impacts.

Pfister *et al.* (2009) do not consider the potential economic impacts of reduced production or exports, or the potential for malnutrition impacts in would-be importing countries. We consider the economic impacts of reduced production and exports through our opportunity cost approach. Motoshita (2010) suggested an approach to consider malnutrition impacts in would-be importing countries, using trade data to estimate reduced international food availability. However, we believe that this approach is not sufficiently robust to be implemented. Particularly given Pfister *et al.'s* (2009) model uses a control variable to account for the ambient level of malnutrition, which can be seen as a proxy for the ability to import food.

The steps to the analysis are summarised in Figure 3 and described below.

Figure 3: Process steps for estimating the societal cost of malnutrition

1. **Calculate water deprivation factor (WDF)**
   Provides a measure of local competition for water.
2. **Calculate Effect Factor (EF)**
   The number of malnutrition cases caused by agricultural water deprivation.
3. **Calculate Damage Factor (DF)**
   Describes the severity of malnutrition cases on disability adjusted life years.
4. **Calculate Human Health Factor using WDF, EF, and DF**
   Change in disability adjusted life years per unit of water consumption.
5. **Estimate the value of a disability adjusted life years**
   Monetary valuation of welfare associated with change in disability adjusted life years.
6. **Calculate the cost of societal impacts**
   Combines value of a disability adjusted life year with the Human Health Factor to give monetary impact per unit of water consumption.

4.2.1. 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, as a proportion of total water consumption by agriculture ($m^3_{deprived}/m^3_{consumed}$).

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.

Figure illustrates this calculation. 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, $WU_{\%, agriculture, i}$ (see Equation 2).

Equation 2: Water depravation factor

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

4.2.2. Step 2: Calculate the Effect Factor

The Effect Factor (EF) is the annual number of malnourishment cases caused by 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 malnutrition rate expressed in DALYs (described in Box 2.) to provide an indication of malnutrition vulnerability and is a factor between 0 and 1.

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

Equation 3: Effect factor

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

4.2.3. Step 3: Calculate the Damage Factor

The Damage Factor (DF) estimates the amount of harm per case of malnutrition. The damage factor is derived from a linear regression of the malnutrition rate ($MN_{\%}$) and the DALY malnutrition rate ($DALY_{malnutrition, rate}$) at a country level to give a conversion from cases of malnutrition to DALYs. The regression gives a damage factor of 0.0184 DALYs/capita.year. Equation 4 shows the calculation of DF.

Equation 4: Damage factor

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

4.2.4. 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 5, units in parentheses).

Equation 5: Calculating the human health damage factor of malnutrition with units in parentheses.

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

4.2.5. Step 5: Estimate the monetary value to a DALY

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

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

In Lvovsky *et al.'s* (2000) paper, they derive the value of the DALY from the value of statistical life (VSL) based on the number of DALYs lost associated with that lost life (Equation 6). 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). The discussion below presents our application of this approach. The values used are consistent with the values used for the VSL in the other environmental impact methodologies.

Equation 6: Value of a DALY

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

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 early teens to mid-fifties (Figure 3: Age weighting for DALYsFigure 3); the DALYs are therefore age weighted (Prüss-Üstün *et al.*, 2003).

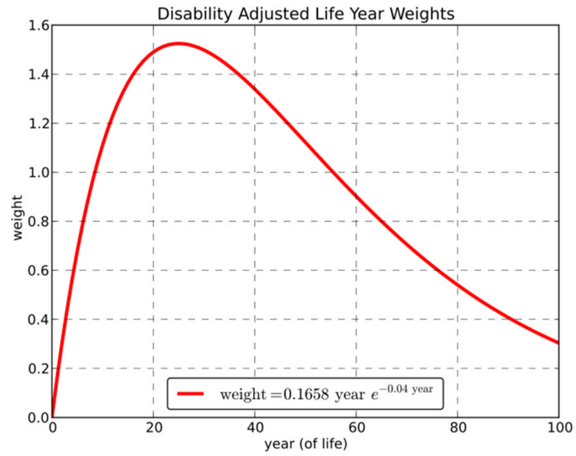
Prüss-Üstün et *al.* (2003) provide a formula and suggested coefficients to calculate the relative weighting of each year of life (X_w), which is set out in Equation 7

Equation 7: Age weighting formula for calculating DALYs

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

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 year of the 78 years of life expectancy associated with the OECD VSL estimate.

Figure 3: 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 Equation 8 below.

Equation 8: 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 Equation 9. This represents the proportion of life lost for a person who expected to live to 78 but died prematurely at 47.

Equation 9: 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. Table 7 contains the steps of the calculation that result in the value of DALY of $185,990 (in 2011USD).

Table 7: Value of a DALY

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

The value of a DALY for OECD nations is transferred to other countries. If an income adjustment is to be included (see section 4.2.5.1) differences between income per capita adjusted for PPP can be accounted for in accordance with Equasion 10. An income elasticity of 0.6 is recommended as a central estimate of the values presented in OECD (2010).

Equation 10: Income adjustment transfer function

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

Where:

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

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

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

4.2.5.1. Equity considerations

Most countries operate a principally market-based economy, where the allocation of resources is determined largely by the forces of supply and demand, which also establish prices in the economy. In this context, an individual's income determines the quantity of marketed goods that they can obtain. When estimating the monetary value of goods (or 'bads') which are not currently traded in markets, the income constraint must therefore be considered.

As peoples income changes, their level of demand for a good usually changes, and the amount they would pay for each unit of the good also changes. Empirical evidence for environmental goods (or avoidance of 'bads') suggests that this 'income effect' is positive – people are prepared to pay more as their income increases (Pearce, 2003). For this reason, if values estimated in one location are to be used in a different location, they need to be adjusted to take account of differences in the income constraints of people in each location.

This is best illustrated using an example. Suppose a survey of people living beside a lake in the USA finds that they value the leisure time they spend around the lake at $1,000 per year. This represents about 2% of their average annual income. Combining this with the number of people who live in close proximity to the lake allows for an estimate of the value of the lake for leisure purposes to be produced. This non-market value estimate can be taken into account when decisions which might affect the future of the lake (e.g. new developments) are considered.

Now suppose we wish to estimate the value of a similar lake in Uganda. Resources to conduct a new survey aren't available but the number of people living near to the lake can be estimated, and it is known to be a popular recreation area. However, the average per capita income in Uganda is 1/100th of the average per capita income in the USA[4]. So assigning the same value of $1,000 per person in the Ugandan context would clearly be inappropriate; suggesting that local people would pay twice their average annual income for a year's worth of leisure at the lake. In order to estimate the value that local people place on the lake, relative to their other priorities, it is necessary to adjust for the differences in income constraints.

This central concept of income effects in non-market valuation of environmental goods is relatively uncontroversial, as is the practice of adjusting for differences in income and purchasing power when transferring value estimates between countries. However, when valuing goods (and bads) relating to human health, equity considerations become more apparent.

As with environmental goods, empirical evidence demonstrates that the amount individuals' would pay to maintain good health and to reduce risks to life increases with income (Viscusi and Aldy, 2003; Scotton and Taylor, 2010; OECD, 2010). This is reflected in estimates of the Value of a Statistical Life (VSL)[5]. When

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[4] Even after accounting for differences in purchasing power the ratio is 1/40th.

[5] *"Value of a Statistical Life (VSL), … represents the value a given population places ex ante on avoiding the death of an unidentified individual. VSL is based on the sum of money each individual is prepared to pay for a given reduction in the risk of premature death, for example from diseases linked to air pollution."* OECD, 2012

applying a VSL estimate calculated in one location to health outcomes in another location, it is common practice in the health literature (see for example: OECD, 2012; Hammitt and Robinson, 2011) to adjust the VSL to reflect the income differential between those locations, as described above.

These differences in preferences for life and health between locations may reflect a genuine acceptance of greater health risks, particularly in the context of other priorities such as economic development or employment. However, because preferences of this nature are often considered to be constrained by the limited choices available in low income contexts, the use of differing VSLs is contentious where decisions may relate to inter-regional resource allocations. In recognition of these concerns, the OECD (amongst others) recommend that where decisions may relate to allocations between regions a single VSL estimate should be used in policy analysis across those regions.

Given the range of possible decision-making contexts where E P&L results may be considered[6] it is important that the decision maker is aware of this potential issue and is in a position to make an informed decision. Whether the primary presentation includes or excludes income adjustments to health related values is therefore a decision for the ultimate user.

Either way we suggest that the effect of differing income levels on the results of an EP&L is assessed through sensitivity analysis.

Where the decision context has implications for inter-regional allocations, two sets of results should be presented: one which reflects equity concerns without any income adjustment to health related values, and a second which does take into account income differentials.

The decision maker will still need to consider a range of factors beyond pure environmental or health impacts. For example, a study which does incorporate income adjustments across a range of countries could provide incentives to shift polluting activities to lower income countries where the implied cost of impacts would be lower – this may be undesirable. However, a similar study which does not adjust for differences in income may deter foreign investment in lower income countries; investment which could have created improvements in well-being in excess of any health related losses.

For this reason decision makers may also wish to consider a more holistic decision making framework such as PwC's Total Impact Measurement and Management (TIMM) which values environmental impacts alongside economic, fiscal and social impacts[7].

4.2.6. Estimate the societal cost

Once we have established the HHF and the value of a DALY, the societal cost can be estimated by multiplying the number of DALYs per m^3 of water consumption with the welfare value per DALY.

Assumptions and data required for this method are shown in Table 8 and 9.

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[6] For example, some decision contexts will be confined to a single country and could involve comparing environmental values to other factors (outside the E P&L) determined by prices or incomes within that country; while others could require prioritisation of impacts across many countries.

[7] See "Measuring and managing total impact: A new language for business decisions", PwC 2013: http://www.pwc.com/gx/en/sustainability/publications/total-impact-measurement-management/assets/pwc-timm-report.pdf and: http://www.pwc.com/totalimpact for more information.

Table 8: Data sources for malnutrition impact pathway

Data	Explanation
Water stress index (WSI)	Water consumption as a proportion of total renewable supply: a measure of how much water other users are deprived of when one user consumes water.
Proportion of water used by agricultural sector $WU_{\%,agriculture,i}$	WSI is combined with the proportion of water used by the agricultural sector to estimate the impact of corporate water consumption on the ability of the agricultural sector to consume water.
Water requirement to avoid malnutrition $WR_{malnutrition}$	The amount of water required by the agricultural sector per capita to provide enough food to avoid malnutrition.
Human development index HDI	An index of the general development of a country or a region. This is combined with the malnutrition rate (measured in DALYs) to create a malnutrition human development factor which is an indicator of the level of development of the country driven by malnutrition.
Malnutrition rate (DALYs per 100,00 people) $DALYs_{malnutrition,rate}$	The number of annual DALYs per capita caused by malnutrition. This is used as above and combined with the malnutrition rate (%) to estimate the number of DALYs caused by each case of malnutrition.
Malnutrition rate (%) $DALYs_{malnutrition,rate}$	Proportion of the population suffering from malnutrition. Explanation of use as above.
Value of a DALY	Used to put a monetary value on the damage function calculated in DALYs/m^3. DALY value is calculated based on OECD estimate of the value of a statistical life (VSL).

Table 9: Key assumptions for the malnutrition impact pathway

Assumptions	Comment on purpose and reasonableness of the assumption
$WR_{malnutrition}$ is assumed to be independent of location, when more granular data is absent.	The methodology can be adapted to more specific data when available. The amount of water per capita to avoid malnutrition is likely to be dependent on local climate and soil conditions (e.g. in different locations. differing quantities of foods could be produced with the same amounts of water). For global assessments we follow Pfister *et al.'s* (2009) approach and use a global average because there is insufficient data availability to estimate more specific values for all areas.
We do not specifically model nations importing food to address shortages.	The Effect Factor (EF) takes into account the local vulnerability to malnutrition which implies that high damage functions will only occur when where malnutrition is a real problem.

5. Detailed methodology: infectious disease impacts

This chapter discusses the valuation of human health impacts from water borne infectious diseases associated with corporate water use. In countries with poor water infrastructure and where corporate water use reduces the clean water available for others, people may be driven to consume dirty water resulting in health impacts including diarrhoea and other water-borne infectious diseases.

In this methodology we first estimate the level of scarcity and implied competition for water using the WSI in the Environmental outcomes section. The Societal outcomes section describes an econometric model which is used to identify the drivers behind the observed level of water-borne disease (measured in DALYs) in different countries. This allows us to hypothesise how the level of water-borne disease in a given location would decrease if corporate water use did not deprive domestic users. The econometric model is used to predict the reduction in impacts, which are then allocated to the total corporate water use to estimate an impact per m3. For a summary of steps, see Table 10.

Table 10: Summary of infectious disease societal impacts calculation methodology

	5.1 Quantify environmental outcomes	5.2 Estimate societal impacts
Infectious water-borne disease impact pathway		
Methods	• The volume of water that could be withdrawn by domestic users if it were not consumed by corporate users is estimated using the WSI for the location of consumption. The WSI is thus used as a measure of the competition for water between corporate and other users (Pfister *et al.*, 2009).	• 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 explanatory variables used are selected to explain the socio-economic drivers of water-borne disease, they are: domestic water use, health expenditure, prevalence of undernourishment, government effectiveness and the water stress level. • The derived relationship is used to predict the fall in prevalence of water-borne disease if the quantity of water which corporates deprive domestic users of (based on the WSI) was reallocated to domestic users. • The resultant change in DALYs per capita is valued and allocated across the total corporate water use to give a welfare impact per m3.

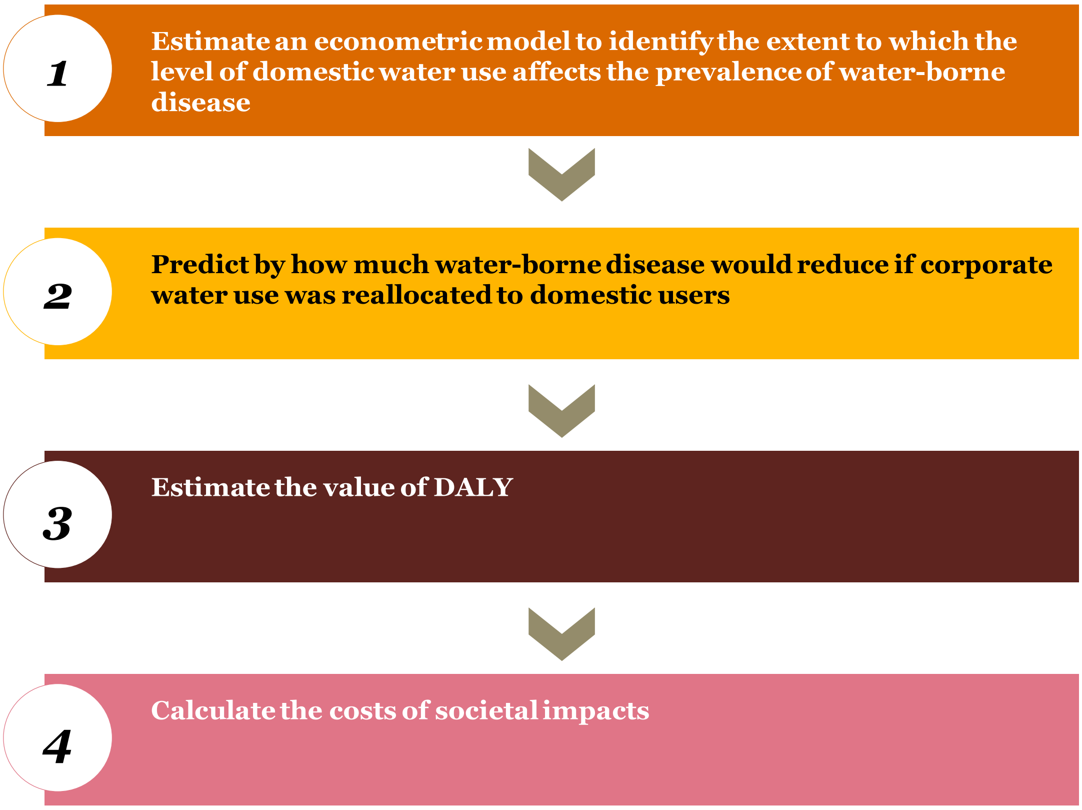
5.1. Environmental outcomes

The environmental outcome of corporate water consumption is the reduced water available to domestic users for drinking and sanitation. This is calculated by multiplying corporate water consumption by the WSI at a specified location. We take the WSI from published values in WaterGAP2 as described in Chapter 4. This value is used in Step 2 of the calculations that follow under Societal outcomes.

5.2. Societal outcomes

The analysis in this section follows four steps (Figure 4). We first use an econometric model to explain the variation in the observed prevalence of water-borne disease, and in particular identify the extent to which the level of domestic water use affects it. The second step considers the level of corporate and industrial water use, and uses the results of the econometric model to predict how the prevalence of water-borne disease would reduce if the volume of water that corporate users are depriving domestic users of (calculated in the Environmental outcomes above) was reallocated to domestic users. This gives us an estimate of DALYs per m^3 of corporate water use. The final two steps assign a value to DALYs to calculate the societal costs per m^3.

Figure 4: Steps in the estimation of societal outcomes for water-borne diseases



5.2.1. Step 1: Construct an econometric model for water-borne disease

Our analysis here builds on work by Motoshita *et al.* (2010) who demonstrate how an econometric model can be used to identify the extent to which corporate water use contributes to the prevalence of water-borne disease. Motoshita *et al.*'s (2010) analysis shows that water-borne disease decreases as household connection water (which is positively related to domestic water use) increases. The authors then assume corporate water use directly reduces domestic water use and they allocate impacts accordingly.

We draw on similar principles in our approach, but rather than assume corporate water use directly reduces domestic water use, we incorporate the WSI and use our econometric results to predict how water-borne disease would reduce if the portion of corporate water use that deprives other users of water was reallocated to domestic users. Furthermore, we set up our model using a quantile regression to take into account the different patterns in different contexts.

Our model is set up using publically available country-level data. However, the relationships that we estimate can be applied at more geographically specific level if data is available. Our approach is described in detail below, Appendix I provides a summary of Motoshita *et al.'s* (2010) approach.

Step 1.1: Identify relevant water-borne diseases

The WHO identifies two groups of water-borne disease in their Water Sanitation and Hygiene database:

1. Diarrhoeal diseases
2. Other non-diarrhoeal infectious diseases
    - Intestinal nematode infections
    - Protein-energy malnutrition
    - Consequences of malnutrition
    - Trachoma
    - Schistosomiasis
    - Lymphatic filariasis

The two malnutrition related diseases are included here rather than in the malnutrition impact pathway because they are not associated with a shortage of food (as per Chapter 4), but rather the inability to assimilate nutrients due to disease.

We retain these two groups in our analysis because it allows us to independently model the drivers of each.

Step 1.2: Set out hypothesis to be tested and key variables

The objective of the econometric analysis is to identify the extent to which a change in clean water availability for domestic use (drinking, cooking, washing, sanitation etc.) would influence the prevalence of water-borne disease. Our null hypothesis is that an increase in clean water availability for domestic users would reduce water-borne disease, if all else was constant.

In order to test this hypothesis we build two econometric regression models, one for each group of diseases. In both cases our dependent variable is the prevalence of water-borne disease, measured in DALYs per capita. Our key independent variable is domestic water withdrawal[8] per capita. It is the coefficient on this variable which we are most interested in, because it is this portion of the variation in prevalence of disease which is driven by availability of water to domestic users, which a change in corporate consumption might influence.

We test a range of socio-economic control variables to account for variation in vulnerability to disease and shortages of clean water. Table 11 summarises the variables considered for inclusion. Step 1.3 discusses the chosen model specification.

[8] Both domestic water consumption and withdrawal were tested as explanatory variables during model development. The results showed that withdrawal per capita has a stronger correlation with DALYs per capita. This intuitively makes sense as the amount of clean water people use for drinking and sanitation is more likely to drive prevalence of disease than the amount used for drinking only.

Table 11: Variables considered for inclusion in the regression model

Variable	Abbreviation	Reason for consideration
Dependent variables		
DALYs per capita per year associated with diarrhoeal water-borne diseases and non-diarrhoeal water-borne diseases	*DALYs_diar* *DALYs_nondiar*	Response variable
Key independent variable		
Domestic water withdrawal per capita per year	*dww*	Amount of water withdrawn by domestic users
Control variables considered		
Government effectiveness index	*ln_govt_eff*	Proxy for the availability and quality of public services and infrastructure which will reduce the likelihood of becoming ill and reduce severity (and therefore DALYs) should an individual become ill
Health expenditure per capita per year	*healthexp*	Higher health care expenditures will cure water-borne diseases more quickly and reduce the impact of disease
Actual Individual Consumption USD per capita per year	*aic*	Indicates the level of wealth and therefore vulnerability to disease
GDP, USD per capita per year	*gdp*	Indicates the level of wealth and therefore vulnerability to disease
Household connection to water supply (%)	*hcws*	Indicates quality of infrastructure and availability of clean water
Undernourished population (%)	*undernour*	Susceptibility to infectious diseases increases with lack of nutrition (Katona & Katona-Apte, 2008)
Water Stress Index	*wsi*	In areas of high water stress, it is more likely that corporate and domestic users will be competing for the same source and more water-borne disease

Step 1.3: Construct econometric models

Ideally we would have time series data for the variables listed above. Using a time series dataset would allow the regression to account for unobserved time-invariant location specific effects; failing to account for these effects could exacerbate problems caused by omitted variables. However, such rich data are not available and we are limited to a snapshot cross country dataset for our analysis. How we deal with this is discussed in more detail below after we present the model specification.

Variable selection and functional form

Data for the variables in Table 11 is available for 123 countries for diarrhoea disease and 112 for non-diarrhoeal diseases. We model the relationships at the country level, but the results could be applied at a more geographically specific level if data is available.

Table 12 below displays the correlation coefficients between the variables listed above. The first key conclusion is that neither the potential dependent variables, nor the control variables, are strongly correlated with domestic water use. This is an important starting point as it limits the risk of multicollinearity through correlated independent variables. Multicollinearity would reduce the robustness of the estimated coefficient on domestic water use, as well as raising the standard errors and, as a consequence, reduce the power of the model.

As expected, however, several of the other potential independent variables are correlated and are explaining similar effects (e.g. GDP and AIC). While we are less concerned with the coefficients on the other explanatory variables, as their coefficients are not used in our calculation, we use this to inform our model selection and to avoid over-specification.

Table 12: Correlation coefficients between key variables

	ln_dalys_d~r	ln_dalys_n~r	ln_dwu6	ln_hcws	ln_aic	ln_gdp	ln_undernour	ln_healthexp	ln_wsi	ln_govt_eff
ln_dalys_d~r	1.00									
ln_dalys_n~r	0.95	1.00								
ln_dwu6	- 0.70	- 0.70	1.00							
ln_hcws	- 0.79	- 0.76	0.68	1.00						
ln_aic	- 0.91	- 0.90	0.73	0.77	1.00					
ln_gdp	- 0.84	- 0.83	0.65	0.72	0.95	1.00				
ln_undernour	0.74	0.73	- 0.64	- 0.71	- 0.75	- 0.72	1.00			
ln_healthexp	- 0.84	- 0.84	0.64	0.72	0.95	0.98	- 0.72	1.00		
ln_wsi	- 0.34	- 0.36	0.52	0.38	0.40	0.28	- 0.33	0.28	1.00	
ln_govt_eff	- 0.70	- 0.70	0.48	0.59	0.77	0.78	- 0.58	0.79	0.35	1.00

Correlation > |0.8| Correlation |0.7| to |0.8|

Equation 11 presents our chosen specification for both the diarrhoea and non-diarrhoea models. We use the same variables in both regressions as the high level drivers are expected to be the same. However, the relative importance of each is not the same for diarrhoeal and non-diarrhoeal diseases.

Alongside domestic water use our explanatory variables are under nourishment, health expenditure, government effectiveness and the water stress index. These provide proxies for susceptibility to disease, likely severity of incidents of disease, quality of infrastructure and the level of competition for water, respectively. Further discussion is provided on why these variables where chosen is presented alongside the results below. The variables showed a non-linear relationship, so we selected a log-log functional form.

Equation 11: Water-borne disease regression model

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

Quantile regression

Our regression analysis uses the variables identified above to explain the observed variation in water-borne disease. We use a Quantile regression rather than a standard Ordinary Least Squared (OLS).

If we used an OLS regression our results would identify a single relationship across all countries in the dataset, minimising the sum of squared deviation from the mean. Quantile regressions allow for unequal (asymmetrical) variation in the data due to complex interactions between the factors in the system Koenker *et al.* (2000). They minimise the asymmetrically weighted sum of squared deviations from the mean. Quantile regressions order data in the response variable (in our case prevalence of water-borne disease) and weight the deviations for data (countries) around the chosen quantile more than deviations in other quantiles. The weighting allows the relationship which better fits a subset of the data to be identified, without splitting the data into small groups which would reduce the power of the estimation. The result of this is that for normally distributed data the 50^{th} percentile of the Quantile regression is the same estimator as an OLS (it is the central point so all data points are equally weighted), but the 75^{th} percentile results present the relationship which fits more closely the data around the 75^{th} percentile.

Our Quantile regressions provide us with a different relationship for different country groups. This is particularly attractive because:

- We would expect (and find that) the strength of different factors influencing the prevalence of water-borne disease varies across different countries. Using the results of our Quantile regression we can group countries with similar rates of water-borne diseases and apply the most appropriate relationship giving us a more specific estimate of impacts in any given location (see further discussion below).

- Quantile regressions are particularly useful for modelling complex systems, such as the prevalence of water-borne disease. For this reason they are often used in ecology where interactions between different factors lead to data with unequal variation (Cade *et al.,* 2003).

- Quantile regressions are more robust to outliers and heteroskedasticity (Koenker *et al.,* 2000).

Regression analysis

Figure 5 depicts the data on prevalence of water-borne disease (measured in DALYs per capita) across countries. It is the variation in this data which our regression is trying to explain. Through our analysis with the Quantile regression we see a different relationship between the variables for countries with different levels of disease.

Figure 5 presents the regression results for the 15th, 50th, and 85th percentiles. Each of the quantile regressions have high explanatory power, explaining over 70% of the variation in the level of water-borne diseases. However, for countries with very low prevalence of water-borne disease (see 15th quantile) we see that domestic water use does not significantly explain any variation in prevalence of disease. Undernourishment and health expenditure are significant, however. This is not surprising because these tend to be developed or middle-income countries where the population (on average – we are using country level data to derive the overall relationship) have sufficient clean water, but that if undernourishment is high or health expenditure low prevalence of water-borne disease increases slightly.

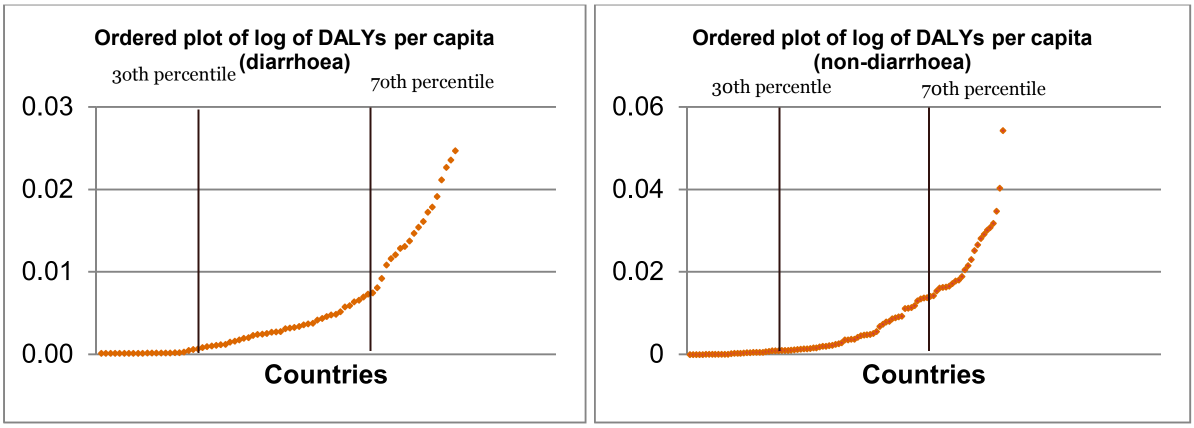
As the prevalence of water-borne disease increases over the 30^{th} percentile (corresponding to 0.0016 DALYs/capita/yr for diarrhoea and 0.0009 DALYs/capita/yr for non- diarrhoea) the significance and strength of the effect of domestic water use increases. We estimate the results into the top and bottom 30 percentiles and the middle 60 percentile based on the distribution of data in Figure 5. The central point of each therefore represents the median trend.

At the 50^{th} and 75^{th} percentiles (midpoints of the middle 60 and upper 30 percentiles), the coefficient for non-diarrhoea diseases increase from -0.30 to -0.48, both significant at the 99% level. We also see government effectiveness and health expenditure become more significant, while under nourishment becomes less significant. This corresponds to what we might expect; in countries where water-borne disease is most prevalent, it is the volume of clean water and quality of the health care system which are most strongly associated with it. It may be that undernourishment is also an important factor but is explaining similar effects as the other explanatory variables. We retain it in all models because it is important at the lower quartiles, and increases the significance of the coefficients on the other variables.

The magnitude of the coefficients on the variables indicates that government effectiveness and health expenditure are explaining most of the variation. The regression only indicates correlation, not causation, but this suggests that functioning institutions and particularly a healthcare system is potentially more important in tackling water-borne diseases than the quantity of water available to domestic users. This reinforces the point made in Chapter 1, that corporates are only partially responsible for addressing domestic water issues.

In most of our regression results the WSI is not significant. This is not particularly surprising because there are many developed countries which have high WSI and low prevalence of disease (e.g. Australia), with the converse also true (e.g. Congo). We decided to leave WSI in the regression however, because the other coefficients show a stronger significance with it included.

Figure 5: Ordered plots of country data on DALYs per capita per year for water-borne disease



5.2.1. 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 corporate water use that is depriving other users of water was reallocated to domestic users. While the relationship was derived at a country level, it could be applied to estimate impacts at a more location specific level if data is available.

Step 2.1 Predict baseline prevalence of disease

Our regression analysis has shown that for locations where 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 locations with disease levels below this we consider there to be no impact of corporate water use on the prevalence of water-borne disease. While these values are empirically derived, the 'cut off' is partially arbitrary based on observed changes in significance. For locations close to these cut off values, we recommend conducting sensitivity analysis to understand the impacts using the 50th percentile relationship.

For locations with disease levels above these values we predict what the DALYs per capita per year for each group of diseases are based on 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 Equation 11 above, inputting the values for domestic water use, health expenditure, undernourishment, government effectiveness and the WSI, and multiplying these by the appropriate coefficient, depending on whether the actual prevalence of disease falls between the 30th to 60th percentiles or 61st and 100th.

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

The second part of this calculation is re-predicting the prevalence of disease, this time including the corporate water use. We multiply the total corporate and industrial water use for the region[9] by the WSI to give the portion that deprives other users of water. It is this quantity of water which we reallocate to domestic users to hypothesise how much lower DALYs per capita per year would be if this water was available.

The estimated reduction in DALYs per capita per year is multiplied by the population for the region and allocated between the total corporate water use (not just the portion which is depriving others) giving a DALY per m^3 of corporate water consumption in a given year.

5.2.2. Step 4: Assign the value of a DALY

To value the impacts of disease, we assign locally-specific DALY values to our DALY/m^3 estimates generated in the previous step. The calculation of these DALY values is the same as in 4.2 (above) and utilises an age-weighting adjustment and parameter estimates from the OECD. The detailed methodology is outlined in Chapter 4.

5.2.3. Step 5: Calculate the societal impacts of disease

Once we have established the damage factor of corporate water use in DALYs lost to disease per m^3 of water withdrawal and the location-specific value of a DALY, we can estimate the overall societal cost per m^3.

[9] The calculation is non-linear, so in order to get an average impact per unit of corporate water consumption this data must represent the total industrial and corporate water for the same geographical region as the other data inputs. If 1m3 is used, this would give the marginal impact.

Table 13: key assumptions for water-borne infectious diseases valuation

Assumptions	Comment on purpose and reasonableness
Group of quantiles to apply quantile regression coefficients	The data shows that if we applied an OLS we would over estimate impacts for countries with low levels of water-borne disease, and under estimate impacts for countries with high levels. We therefore use a quantile regression to better reflect the impacts in different locations.
The WSI represents the proportion of corporate water consumption which would otherwise be available to domestic users	The volume of water not available to domestic users is assumed to be total corporate consumption, adjusted by the WSI: this assumption will hold where conditions (e.g. infrastructure) are such that, when a corporate user reduces its water consumption, a domestic user can access it. This may not be the case in reality where 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 the associated error.
Data from a single year on the correlation between disease and explanatory factors is representative of that relationship in other years	We use best available data. Due to data constraints, the regression models to estimate disease impacts were run using data from a single year.
Data from different years are representative of other years and driven by underlying commonality	The datasets used to generate the regression coefficients are based on different base years. Where possible, data from 2004 was used to match the base year of DALYs datasets. However, in the case of water consumption and water stress index the dataset based on annual averages were used because they are the best available data at this point in time. If better data becomes available then the model can be updated.

6. Detailed methodology: depletion of ground water resources

This chapter presents an approach for estimating the contribution of current corporate water use to future costs associated with the depletion of ground water resources. In many areas of the world groundwater resources are being used at an unsustainable rate. The extent of future impacts will depend on whether infrastructure is put in place to access alternative supplies. The approach presented here draws on the available data on depletion timescales to estimate the future shortfall in supply. Given the uncertainty over future impacts, replacement costs, in the form of desalinisation and transportation costs, are used as a proxy for the societal impacts. Table 14 presents a summary of the approach.

Table 14: Summary of ground water depletion societal impacts calculation methodology

	6.1 Quantify environmental outcomes	6.2 Estimate societal impacts
Groundwater resource depletion impact pathway		
Methods	• The rate of groundwater depletion and the expected time to depletion are used to estimate the future annual shortfall in water supply.	• We calculate replacement cost as a lower-bound estimate of likely societal impacts of groundwater depletion. • The cost is calculated based on predicted desalinisation costs and the cost to transport desalinated water to main populated areas. • The total costs of the annual shortfall is allocated to current use to give a per m^3 value.

6.1. Environmental outcomes

In those areas of the world where ground water reserves are being extracted and consumed at an unsustainable rate, it is likely that the reserves will become completely depleted, or degraded to a point where it is unusable due to saltwater intrusion. Assuming that demand for water does not fall an alternative source of water will be required in the future.

The environmental outcome of corporate groundwater consumption is the reduced stock and ultimate depletion of groundwater reserves.

The first step is to identify locations (ideally specific water basins) that are likely to fully deplete their ground water reserves given the current rate of extraction and estimate the number of years until groundwater reserves are completely depleted. The second step calculates the future annual shortfall of freshwater supply (Figure 6).

Figure 6: Process steps for estimating the environmental impacts of groundwater depletion

1 **Estimate the years to depletion of ground water**

2 **Estimate the shortfall in the future water supply**

6.1.1. Step 1: Estimate years to depletion of groundwater in a location

The locations that are likely to deplete their groundwater reserves are identified by examining the ratio of annual groundwater withdrawal to the total renewable water reserves. Further research on the specific reservoir should yield data on the years to depletion. If this information is not available it can be calculated for a given location using information on the total ground water reserves, renewable groundwater available each year and ground water extraction rates.

6.1.2. Step 2: Estimate the shortfall in the future water supply

To calculate the current years contribution to future depletion we subtract the current year's renewable groundwater from the total volume withdrawn from groundwater reserves. This assumes that demand does not decrease. If demand increases this will be an overestimate of the actual shortfall, however given we are only interested in the contribution of current consumption we do not need to predict the level of demand increases.

Although this calculation is a simplification of reality it is sufficient to provide an indication of likely future infrastructure requirements. Alternative supplies are likely to come on line gradually, such that shortfall is not suddenly experienced in the year of reserve depletion.

Table 15: Data requirements for groundwater environmental outcomes

Data	Explanation
Location-specific data on groundwater availability and consumption	Location specific data is available from a variety of sources including government statistics, international organisations, international research bodies and academic literature depending on the exact location
Country level groundwater availability and consumption	If more specific information is not available about the location of groundwater removal, we use FAO AQUASTAT

Table 16: key assumptions for groundwater depletion environmental outcomes

Assumptions	Comment on purpose and reasonableness
Demand for water does not reduce	Although there is likely to be some elasticity in demand due to price or conservation regulation, the likely continued increase in population size will likely see at least an increase in demand.

6.2. Societal outcomes

Depletion of groundwater may have important societal impacts in areas where groundwater reserves are being depleted at an unsustainable rate. The extent of impacts will largely depend on the ability of governments to ensure alternative supplies are put in place before a shortfall is reached.

If available, location specific estimates can be developed to estimate the societal costs based on the predicted socio-economic impacts in the given context. In general however, these impacts are likely to be hard to predict. Here we suggest using an increased cost of supply as a lower bound proxy for potential societal impacts. Once the future costs of the annual shortfall are calculated, they are allocated to current water use. Figure 7 presents the steps in the analysis.

Figure 7: Process steps for estimating the societal impacts of groundwater depletion

1 **Estimate the cost of future water supply**

2 **Estimate the cost per unit of water withdrawn in current year**

6.2.1. Step 1: Estimate the cost of future water supply

We use desalinisation and transportation costs as the proxy for societal costs, income adjusted to the location of interest. Numerous sources for desalinisation costs are available, for example, Zhou & Tol (2005) provide a useful review of the technology costs and average transportation costs in different locations.

6.2.2. Step 2: Estimate the cost per unit of water withdrawn in current year

To estimate a value of the average impact of current corporate water consumption, the future cost of the ground water depletion is averaged over the total water withdrawal. This is done by dividing the discounted of future water supply associated with current year depletion by the total water withdrawal within that location[10].

The data and assumptions for the societal impacts of ground water depletion are shown in Table 17 and Table 18.

[10] If the source of corporate water consumption is known to be ground water then the average impact per unit of ground water withdrawn could be alternatively calculated by dividing the total discounted cost by total ground water withdrawal.

Table 17: Data requirements for groundwater societal impacts

Data	Explanation
Cost of desalination and transport of water	The cost of desalinisation and transport of the water is based on research published in the Water Resources Research journal

Table 18: key assumptions for groundwater depletion societal impacts

Assumptions	Comment on purpose and reasonableness
Desalinisation is the most appropriate technology to estimate lower bound societal costs	This assumption is based on the adoption rate of desalinisation by fresh water-stressed locations compared to the adoption rates of other strategies.
Water depleted in the current year will create a shortfall in water supply that will need to be replaced in the first year after depletion	It is assumed that the water depleted in the current year will create a shortfall in water supply that will need to be replaced in the first year after depletion. In reality, the shortfall of water supply caused by the current year's depletion should be apportioned equally between all future years when desalinisation will be used and the cost discounted back from the relevant years. Information is not available on how many years desalinisation will be required for after total depletion. By apportioning the total water shortfall to the first year after depletion, we are slightly overestimating the cost to society, which is consistent with the precautionary principal applied in these methodologies.
The cost of desalinisation technology will vary between countries by the ratio of PPP adjusted GNI	The cost of desalinisation is PPP-adjusted between different countries. This assumes that the cost of desalinisation technology will vary between countries by the ratio of PPP adjusted GNI. A possible alternative would be to adjust the cost of the water supply using the relative cost of different technologies in each country. However, this would introduce a bias which is unrelated to the societal costs and it was therefore deemed more appropriate to use a fixed cost which is PPP-adjusted.
Current levees or charges for ground water extraction applied by governments aren't ring-fenced funding for the future supply water	It is assumed that the levees or charges for ground water extraction applied by governments aren't ring-fenced funding for the future supply water. If this were the case, part of the societal cost would already be captured in the company's balance sheet.

7. Detailed methodology: subsidy cost of water

This chapter presents an approach for estimating the financial burden imposed on tax payers as a result of subsidies on corporate water use. In many locations the price of water is a poor reflection of the financial cost of extracting or producing clean water and distributing it to users. In some developed countries where investment in advanced infrastructure has overcome shortfalls in water supply (and therefore the other impacts associated), subsidy costs may be the most material portion of the impact of corporate water consumption.

As well as the financial burden of subsidies, artificially low water prices may also lead to increased consumption (see World Bank 2005 for a summary of price elasticity of demand), with knock-on impacts associated. We do not consider these impacts here, because they should be associated with the regulator not the corporate.

Table 19: Summary of ground water depletion societal impacts calculation methodology

6.1 Quantify environmental outcomes		6.2 Estimate societal impacts
Groundwater resource depletion impact pathway		
Methods	• The environmental outcome is not required for this calculation.	• The subsidy cost is the difference between the water sector revenues (from water prices) and financial costs of supply.

7.1. Environmental outcomes

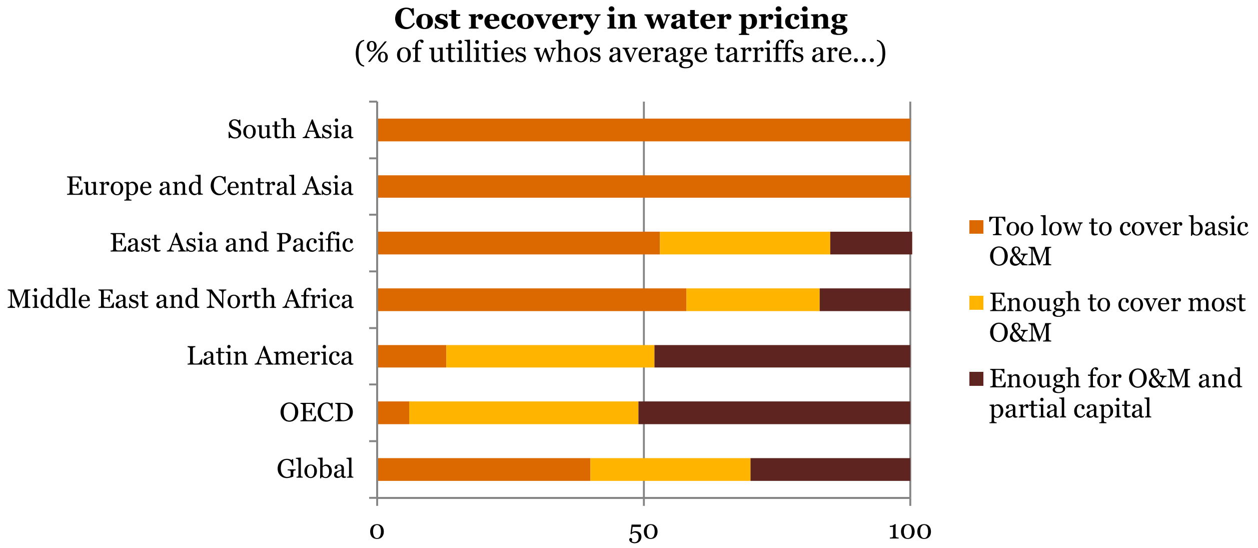
As per the other impact pathways, the primary environmental outcome of corporate water consumption is an increase in water scarcity. These values are not required for the calculation of subsidy costs, however.

7.2. Societal outcomes

The World Bank's (2005) review of average water tariffs in 132 major cities worldwide found that on average 40% were not even sufficient to cover basic operation and maintenance (O&M) costs (Figure 8) (see also UN/DESA, 2008). Subsidies were particularly prevalent outside of the OECD. However, even within the OECD only 50% were deemed to cover at least O&M and only "some" capital costs. The EU's Water Framework Directive has full cost recovery as a core principle, but it recognises that achieving it is still some way off (EEA, 2013).

As a result of only partial cost recovery, corporate water use puts a financial burden on tax payers who are supporting the subsidies. The methodology presented here provides a simple means of estimating those impacts.

Figure 8: Subsidies are prevalent across all regions of the world



7.2.1. Calculate the subsidy cost of water

The subsidy cost of water can be calculated at an aggregate level for the whole supply sector, or separately for different consumers. Often agricultural, industrial and drinking water will have different pricing regimes. To calculate the subsidy costs, for a given price schedule, it is a simple calculation of revenue from water supply minus financial costs of delivery. This gives the total shortfall in finances, which can then be attributed to water use (withdrawal, not consumption) to give a per m^3 figure. Box 3 presents an example for Cyprus.

Box 3: Calculating the subsidy cost of water for Cyprus.

Water use in Cyprus is subsidised, such that use by corporates drives up the tax payments for others in Cyprus. The Water Development Department (2010) provides data on cost recovery in the water sector. Their analysis shows that in 2010 the financial cost of drinking and agricultural water provision was €1.17 /m^3 and €0.34/m^3 respectively, and that 99% of the financial cost of drinking water was recovered, and 76% for agricultural water (2007 prices). Applying these percentages gives impacts of € 0.01/m^3 and € 0.08/m^3, respectively in 2010 (2007 prices). This represents the additional payments by tax payers in Cyprus as a result of other users' consumption.

Table 20: Data sources for groundwater societal impacts

Data	Explanation
Financial data: Revenue (or price), cost of supply and proportion of cost recovery	To calculate subsidy costs
Quantity of water withdrawal that financial data relates to	To attribute impacts per unit of withdrawal

Table 21: key assumptions for groundwater depletion societal impacts

Assumptions	Comment on purpose and reasonableness
Corporate consumers are not supporting water subsidies through a ring-fenced fund	If companies are supporting subsidies through a ring-fenced fund, and that their contribution is at least proportional to the quantity of water they withdrawal, they are not putting an additional burden on other tax payers. This is assumed not to be the case unless information to the contrary is demonstrated.

8. Detailed methodology: economic opportunity costs

Economic opportunity costs of water consumption occur when the corporates use of water deprives another user of water, and that other user has a higher value for the water, or can create a higher social value from that water. Such mis-allocation of water resources is caused by unequal and incomplete pricing of water resources and other market failures. This chapter presents the general principles for assessing economic opportunity costs in a specific context (an assessment of opportunity costs requires good data availability and is not possible without specific knowledge on the context of water users). Like subsidy costs, opportunity costs will not always be relevant, however if they are, the impacts could be significant.

Table 22 presents a summary of the generalised approach. We focus on the principles of the approach and do not present specific steps or calculations because these will vary depending on the specifics of the context and available data.

Table 22: Summary of ground water depletion societal impacts calculation methodology

	6.1 Quantify environmental outcomes	6.2 Estimate societal impacts
Groundwater resource depletion impact pathway		
Methods	• Identify the quantity of water that users which could deliver higher societal benefits per m^3 are deprived of as a result of corporate consumption.	• Estimate the loss in societal benefits (including private revenues and public gains) as a result of inefficient allocation of water resources, based on the marginal productivity of consumption.

8.1. Environmental outcomes

The environmental outcome, associated with the economic opportunity costs of water use, is the quantity of water which other users are deprived of as a result of corporate consumption. Only users which would have a higher total economic value of water use should be considered. Total economic value includes the private gains from consumption, as well as the social benefits associated. For example, agriculture may have a lower marginal productivity of water consumption associated with direct revenues or value added compared to manufacturing, but can provide an essential source of nourishment with considerable social gains associated.

The WSI provides an indication of the quantity of water which is derived from other users, however, in order to identify instances where an opportunity cost is present, it is necessary to go beyond this and identify specific users who are directly deprived and quantify the volume associated. This will typically require:

1. Watershed level assessment of current and potential users of water
2. Hydrological survey or estimate of the quantity of water identified users are deprived of
3. Economic assessment of marginal benefits to consumption of alternative uses (see Societal outcomes)

8.2. Societal outcomes

The impacts associated with inefficient allocation of water resources is equal to the difference in societal gains between the corporate's use and the most efficient user of the water, for the given quantity of water deprived. These social gains are a sum of the private and public gains to production, less the negative externalities. To identify the optimal allocation the societal gains should be considered at the margin (societal gains per unit of water consumption, at a given level of water provision). The data needs may be significant, and the optimal allocation may be a complex multi-stakeholder reallocation such that simplification is required in practice.

Box 4: Example – Opportunity cost of water for electricity generation in South Africa

Inglesi-Lotz and Blignaut (2012) estimate the opportunity cost of water associated with two planned coal power plants, relative to different electricity generation options. They calculate the net marginal revenue per m^3 (NMR) to compare the technologies, and calculate the opportunity cost of coal per kwh relative to the most efficient technology (with highest NMR/m^3). The NMR is calculated based on a revenue function approach, where revenue is a function of the price of electricity generated, quantity of water consumption, and production costs. The results show that wind provides the greatest NRM/m^3 (921,000R/m^3 higher than coal), given the options will generate different quantities of electricity, this results in an opportunity cost of 1.31R/kwh of electricity generated.

9. Sensitivity analysis

9.1. General approach to sensitivity analysis

Sensitivity analysis refers to a process of testing the robustness of a methodology, and its outputs, to changes in the inputs. This is in order to identify those parameters with the greatest potential to drive the results, and to then focus attention towards those drivers.

There is no single approach to conducting sensitivity analysis, and the approach can vary based on the needs of the analysis. Our approach focuses on understanding the inputs which have greatest influence on the results and which we consider to have the most uncertainty surrounding them. It does not consider the outputs (i.e. what would the input need to be to give a pre-defined conclusion) because this depends on the context within which the approaches are being applied.

We focus the sensitivity analysis on the two health impacts, excluding resources, subsidy costs, economic opportunity costs and the impacts of the water supply sector because we have not specified precise calculations or inputs for these impacts. They are generic approaches which need to be adapted to a given context.

9.2. Impact-specific sensitivity analysis

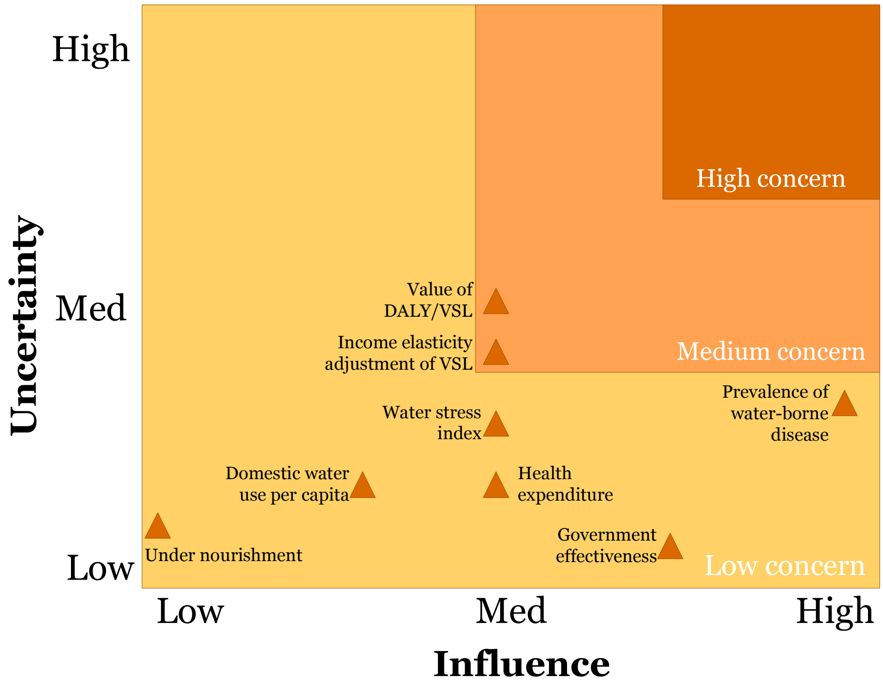
9.2.1. Overall summary and considerations for model use

The following sections provide detail on the materiality of different health impacts, and provide assessments of both the influence and uncertainty of the parameters in the calculations. This summary section highlights those conclusions.

Figure 9 below maps the model parameters on an influence/uncertainty matrix. Those variables towards the top right hand side of the figure (towards the high influence/high uncertainty area) are areas where caution is required during application of the model.

Given the relatively higher materiality of disease, relative to malnutrition, the sensitivity analysis focuses on this. The greatest uncertainty lies in the valuation of the health impacts themselves. The input with the greatest influence on the results is the prevalence of water-borne disease because this affects which quantile regression results are applied. The example of China is discussed below; it’s average DALYs per capita puts it just below the threshold above which domestic water use is a significant driver of disease impacts, however the country average data masks high local variability. This highlights the need to apply the analysis at a local level where possible, and conduct sensitivity analysis based on the different quantiles.

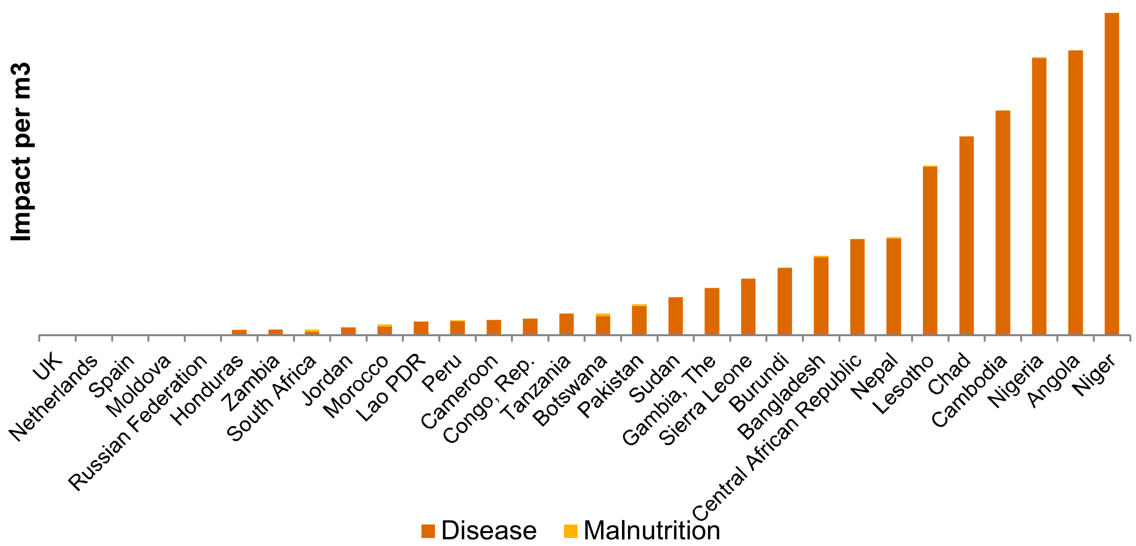
Figure 9: Influence/uncertainty matrix summarising the sensitivity assessment summary for key variables and decisions



9.2.2.Materiality

In terms of the two health impacts, disease is by far the most dominant in the results (Figure 11). Inputs to this calculation are therefore the primary focus of our sensitivity analysis.

Figure 11: Proportions of disease and malnutrition impacts of water consumption in a selection of countries, based on a global country-level data



9.2.3. Parameter impact

Table 23 presents an analysis of how sensitive the results are to the input parameters are in the disease calculation. For consistency with the other methodology papers we present results for the China, the US and Nigeria.

Both the US and China had average DALYs per capita below the 30^{th} percentile in both our regressions, indicating that domestic water use is not a significant variable in explaining the level of water-borne disease in these countries. Most of the variables are therefore not applicable.

Further examination shows that the prevalence of disease in the US is very low, and would need to be increased by 1500% and 4400%, for non-diarrhoeal and diarrhoeal disease, respectively, before it crosses into the 30^{th} to 70^{th} percentile group. China however, is much closer to the threshold. For China, country level analysis is problematic as there is considerable demographic variation, and the average DALY per capita rate is likely to be made up of highly polarised populations and geographies. This highlights the need to apply this analysis at a more locally specific level.

Nigeria is already in the highest quantile group (70 to 100^{th}). The percentage change in results, following a 10% change in the input variables, matches what we would expect based on the coefficients in our regression. As discussed in Chapter 5 it is government effectiveness and health expenditure which are the most influential variables, followed by domestic water use.

The valuation of DALYs has a directly proportional impact, while a 10% increase in the income elasticity of the VSL (if the income adjustment is applied) leads to a 15% decrease in the values for Nigeria. This is because moving from a factor of 0.6 to 0.66 increases the effect of the income adjustment, and Nigeria has a lower income relative to the base value from the OECD. Section 4.2.5.1 discusses the importance of careful consideration of income adjustments to the VSL. It is important to present results with and without income adjustments (elasticity of 0) to ensure effect of income in the numbers is clear to decision makers.

Table 23: Influence of changes to input parameters on the results

Variable	Flex	Influence rating[11]	US (% change to disease)	China (% change to disease module)	Nigeria (% change to disease)
DALYs non-diarrhoeal/cap	Apply higher quantile	High	DALYs **1500%** higher move from 0-30th quantile (insignificant relationship – zero impact) to 30-70th quantile	DALYs **115%** higher to move from 0-30th quantile (insignificant relationship – zero impact) to 30-70th quantile	N/A in highest (70-100th) quantile already
DALYs diarrhoea/cap	Apply higher quantile	High	DALYs **4400%** higher move from 0-30th quantile (insignificant relationship – zero impact) to 30-70th quantile	DALYs **170%** higher to move from 0-30th quantile (insignificant relationship – zero impact) to 30-70th quantile	N/A in highest (70-100th) quantile already
Water stress index	10%	Med	N/A	N/A	7%
Value of a DALY/VSL	10%	Med	N/A	N/A	10%
Income elasticity adjustment of VSL	10%	Med	N/A	N/A	-15%
Domestic water use per capita	10%	Med	N/A	N/A	-4%
Health expenditure	10%	Med	N/A	N/A	-6%
Under nourishment	10%	Low	N/A	N/A	1%
Government effectiveness index	10%	Med	N/A	N/A	-8%

9.2.4.Parameter uncertainty

On the whole the parameters which are used in the analysis have low levels of uncertainty due to the rigour behind the data collection and aggregation. The level of representativeness of country level parameters to local issues is the most significant driver of uncertainty. Country level analysis provides only an indication of the potential average impacts of corporate water use in a country. Application of more locally specific data will significantly reduce the uncertainty associated with representativeness, but may increases the chance of uncertainty due to accuracy in the local data. Parameter uncertainty should therefore be reconsidered when applying different datasets.

[11] Low = average response for overall cost for three countries is less than 1%

Med = average response for overall cost for three countries is between 1% and 10%

High = average response for overall cost for three countries is greater than 10%

Table 24: Assessing the uncertainty of key parameters based on the reliability of the measurement and the variance in attempts to measure the parameter

Variable	Uncertainty rating	Reliability/quality of measurement	Variance of the number measured
DALYs non-diarrhoea/cap DALYs diarrhoea/cap	Low	WHO gathers the data but recognises there are challenges associated with this in many parts of the world. Conversion to DALYs also has uncertainties associated.	<25%
Water stress index	Low	Data is available at a water basin level, but how closely it matches actual level of competition over time and space may vary	<25%
Value of a DALY / VSL	Med	Estimated, method used is peer reviewed and broadly accepted	<50%
Income elasticity adjustment of VSL	Med	Estimated, method used is peer reviewed and broadly accepted where income adjustment is appropriate	<50%
Domestic water use per capita	Low	World Bank gathers the data but recognises there are challenges associated with this in many parts of the world.	<25%
Health expenditure	Low	World Bank gathers the data but recognises there are challenges associated with this in many parts of the world.	<25%
Undernourishment	Low	World Bank gathers the data but recognises there are challenges associated with this in many parts of the world.	<25%
Government effectiveness index	Low	World Bank estimated index. Estimated, method used is peer reviewed and broadly accepted	<25%

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Appendices

Appendix I: Summary of Motoshita et al.'s methodology for estimating water-borne disease

Motoshita *et al.* develop a two-step multiple regression approach to calculate the impact in DALYs/m^3 of water consumed.

The first stage is a regression with household connection to freshwater supply (this includes everything from piped water supply to access to a well with clean water) as the dependent variable and domestic water consumption, GDP/capita and gross fixed capital formation expenditure/capita as explanatory variables. Using this regression, they calculate the partial differential to explain the relationship between domestic water consumption and the household connection to water supply. The two variables have a positive relationship meaning that as household connection to water supply increases, domestic water consumption does likewise.

The second stage is a multiple regression designed for each of four examined diseases: Ascariasis, Trichuriasis, Hookworm disease and Diarrhoea. The dependent variable in each case is DALYs caused by each infectious disease. The explanatory variables include household connection to freshwater supply for all diseases and selection of: annual average temperature; house connection rate to sanitation; average dietary energy consumption; undernourished population rate; Gini coefficient of dietary energy consumption and health expenditure per capita. With this second regression they calculate a partial differential to explain the relationship between DALYs caused by a disease and the household connection to water supply. There is a negative relationship meaning that as household connection to water supply increases the number of DALYs caused by disease decreases.

Combining these partial differentials indicates that as domestic water use increases, the amount of DALYs caused will decrease. The inverse of that means that, as domestic water consumption fall, DALYs associated with those infectious diseases will increase. It is assumed that there is a fixed amount of water that it is possible to consume in a country and, if a business consumes a specified amount of water that will be taken away from the domestic water use and therefore cause and increase in DALYs.

This methodology is not ideal for our purposes for a few reasons. Firstly, because the damage functions are not calculated for all relevant diseases. Secondly, the damage functions are only calculated at a country level not at a location specific level, which will potentially limit the accuracy of our results. Thirdly, the analysis is only completed on a limited number of countries. The final reason is that the methodology doesn't take into account the water stress level of a location, which is a driver of the extent to which a company's water consumption will impact domestic users' ability to consume water. An update to this research is expected in the near future. When it is published, it will be examined and our methodology updated accordingly.

We feel a simpler, clearer methodology that incorporates a measure of water stress is more appropriate. This is one that can be recalculated based on the raw data available both at a national and location specific level.

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Valuing corporate environmental impacts: Water pollution

PwC methodology paper

Version 4.5

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Contents

Table of Tables

Table of Figures

Table of Equations

Abbreviations and acronyms

Abbreviation	Full name
BAF	Bioaccumulation Factor
CPDB	The Carcinogenic Potency Database
DALYs	Disability Adjusted Life Years
DEFRA	Department for Environment, Food and Rural Affairs
E P&L	Environmental Profit and Loss
E. coli	*Escherichia coli*
EEA	European Environment Agency
EC	European Commission
ED50	Effective dose 50
EEIO	Environmentally extended input-output modelling
EF	Effect Factor
EPA	Environmental Protection Agency
EPI	Estimation Programs Interface
FF	Fate Factor
g	grams
GHG	Greenhouse gas
GNI	Gross national income
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
IRIS	Integrated Risk Information System
kg	kilogram
l	litre
LCA	Life cycle assessment
LCIA	Life Cycle Impact Assessment
LDH	Lactate Dehydrogenase
LOAEC	Lowest Observed Adverse Effect Concentration
LOAEL	Lowest Observed Adverse Effect Level
mm	millimetre
mol	mole

N	Nitrogen
NOAEC	No Observed Adverse Effect Concentration
NOAEL	No Observed Adverse Effect Level
NO_x	Nitrogen oxide
^oC	Degrees Celsius
OECD	Organisation for Economic Co-operation and Development
P	Phosphorus
PPP	Purchasing power parity
REACH	Registration, Evaluation and Authorisation of Chemicals
s	seconds
SETAC	Society of Environmental Toxicology and Chemistry
SGOT	Serum Glutamic Oxaloacetic Transaminase
SPAROWW	SPAtially Referenced Regressions On Watershed Attributes
UNEP	United Nations Environment Programme
UK	United Kingdom
US	United States
VSL	Value of a statistical life
WHO	World Health Organisation
WTP	Willingness to pay
XF	Exposure factor
YLD	Years Lost due to Disability
YLL	Years of Life Lost
yr	year

1. The environmental impacts of water pollution

1.1. Introduction

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 consumption of energy or resources). Despite improvements in some developed countries, water pollution is on the rise globally. Pollution and degradation of water bodies can adversely affect human wellbeing, and thereby carries a societal cost. In this paper, we set out a methodology for identifying and valuing the costs of water pollution in monetary terms.

1.2. Overview of impact area

The impacts of water pollution are principally local or regional and highly dependent on the physical environment and the local demographic exposure. For example, the change in concentration of arsenic following a release depends on the size of the water body and 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.

The most significant water pollutant categories (in societal cost terms) are listed below, sub-divided into 'toxic pollutants,' 'nutrient pollutants', 'pathogens' and 'thermal'. There are numerous individual pollutants that can be categorised into the key areas listed below.

1.2.1. Toxic pollutants

- **Selected toxic substances:** Both organic and inorganic substances, including heavy metals and chemical compounds which may persist or cause undesirable change in the natural environment, bioaccumulate in the food web, and cause adverse effects to human health.

1.2.2. Nutrient pollutants

- **Nitrogen (N) and phosphorus (P):** Both are basic building blocks of plant and animal proteins, which in elevated concentrations can cause a range of negative effects including algal blooms leading to a lack of available oxygen in the water[1].

1.2.3. Pathogens

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

1.2.4. Thermal pollution

- **Thermal:** Discharge of water above or below the ambient temperature of natural water bodies can change the ecological balance.

1.3. Environmental and societal outcomes

The discharge of pollutants to water bodies increases their concentration in the water body, directly reducing water quality and causing secondary phenomena such as eutrophication. These changes can adversely affect people in several ways:

[1] A phenomenon commonly known as eutrophication.

- **Human health impacts:** The build-up of toxins in the human body due to prolonged ingestion of contaminated water or food can cause acute illness, cancer and a host of other conditions.
- **Impaired recreation value:** The nutrient enrichment of waters can cause excessive macrophyte growth leading to eutrophication. This can affect the recreational use of the water body due to health impacts from toxic blooms, water congestion from excessive vegetative growth, unfavourable appearance, and/ or unpleasant odours.
- **Property values:** Eutrophication of water bodies can affect the potential sale value of local property (Krysel *et al.* 2003). The literature 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 yield and changes in species composition. Annual losses to the commercial fishing and shellfish industry from nutrient pollution – attributable to lower yields from oxygen-starved waters and fluctuations in consumer confidence of tainted seafood – are estimated in the United States (US) to be over $40 million annually (Hoagland and Scatasta, 2006).
- **Livestock:** Changes in the toxic concentration of certain chemicals in potable water can negatively impact the health of livestock, leading to reduced production or quality of meat.
- **Agriculture:** Changes in the toxic concentration of certain chemicals in irrigated water can negatively impact the growth of crops, leading to reduced yields.
- **Other ecosystem services:** Reduced water quality due to build-up of toxins or nutrients in an ecosystem can lead to the loss of regulating and supporting services.
- **Environmental impacts of wastewater treatment sector:** Treatment of wastewater is associated with additional environmental impacts including greenhouse gases (GHGs), air emissions and waste.

1.4. Impact pathways

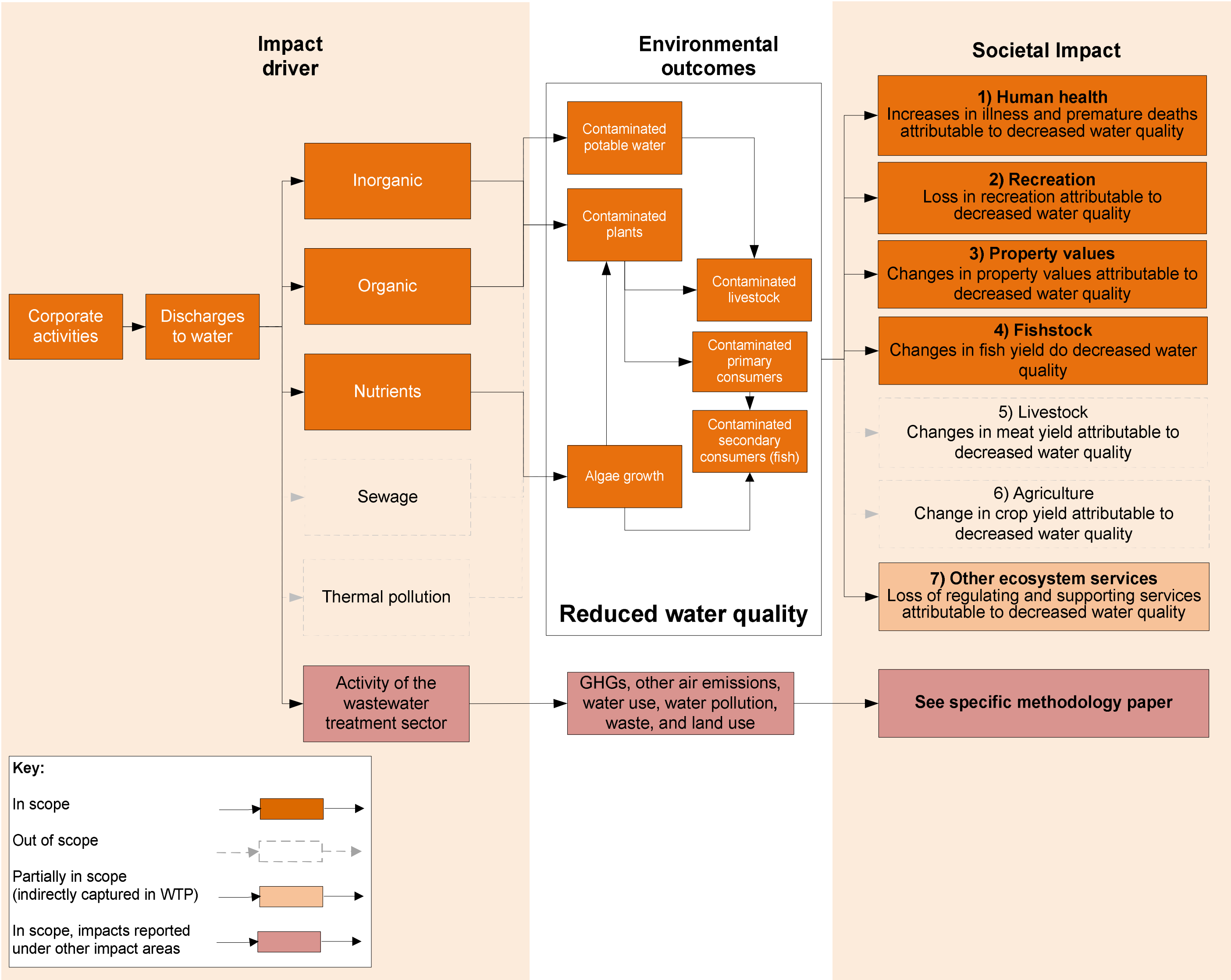
In order to value environmental impacts, we need to understand the causal links between corporate discharges of pollutants to water bodies and the affect these have on human populations. Therefore, we define impact pathways that describe the links between corporate activities, the environmental impacts from those activities, and the resultant societal outcomes. Our impact pathway framework consists of three elements:

- **Impact driver:**
  - *Definition:* These drivers are expressed in units which can be measured at the corporate level, representing either an emission to air, land, or water; or the use of land or water resources[2].
  - *For water pollution:* The release of different types of chemicals and compounds to water.
- **Environmental outcomes:**
  - *Definition:* These describe actual changes in the environment which result from the impact driver (discharge or resource use).
  - *For water pollution:* The changes in the environment as a result of discharges of water pollutants. Primarily these are identifiable as increased concentrations of pollutants and associated reductions in water quality, but secondary effects include the bioaccumulation of pollutants in the food web.
- **Societal impacts:**
  - *Definition:* These are the actual impacts on people as a result of changes in the environment (environmental outcomes).
  - *For water pollution:* The impacts are principally related to health but also include impacts on amenity values, recreation and the market economy.

[2] A note on language: In this report, the measurement unit for any 'impact driver' is an 'environmental metric.' Therefore, water pollution is the impact driver, and tonnes of pollutant (e.g., nitrogen, benzene, e tc.) are the environmental metrics.

The three stages of the impact pathway are shown in Figure 1 overleaf. Water pollution exhibits a complex pathway, with multiple pollutants each playing a role in multiple environmental and societal outcomes. The label 'out of scope' identifies elements of the impact pathway which are not addressed in this PwC Valuation methodology. The reasons for any such limitations of scope are explained at the end of this chapter.

Figure 1: Impact pathways for water pollution



1.5. Prioritising which impacts to quantify and value

This section outlines the key water pollution impact areas and pathways that will be valued and identifies those that are beyond the scope of this paper. There are no pre-existing comprehensive studies which compare the relative societal costs of the different pollutants and pathways upon which we can base our materiality assessment. We therefore seek to cover as many pollutants and pathways as possible, only excluding areas where there is particularly strong evidence of immateriality, insufficient data, or a compelling case on other grounds.

1.5.1. Prioritising pollutants

For nutrient pollution we model and value the impacts of nitrogen and phosphorous which are widely recognised to be the most significant industrial and agricultural causes of excess nutrients in waterways (EPA2, 2013).

For the other broad categories of pollutants identified above, prioritising pollutants is more complex as there are a diverse range of different specific pollutants. The severity of the potential impacts resulting from discharges of these specific pollutants are equally diverse. For example, the heavy metal ED50 (effective dose that results in an outcome for half the exposed population, see Chapter 4 for discussion) for cancer through inhalation of Mercury is 1.36 kg/lifetime and 0.062 kg/lifetime for Arsenic, with some heavy metals having no proven cancer effects through inhalation (Rosenbaum *et al.,* 2008).

In order to value the impacts of water pollutants the analysis therefore needs to consider specific pollutants[3]. A top down analysis using country level data on point source emissions in the Netherlands (CBS 2011) and the US (EPA 2010, 2011) identifies heavy metals to be the most significant source of human toxicity, representing about 85%[4] of the total impacts (Table 1). However, for any given industry the most material pollutants should to be assessed in the scoping phase of the project. For example, an assessment for a cotton t-shirt needs to include the chemicals used in farming (organic and non-organic pesticides, herbicides, fertilisers), bleaches, dyes and solvents used in manufacturing, as well as use-phase detergents, which are not in this list.

Table 1: Pollutants identified as the most material sources of human toxicity impacts in the US and the Netherlands

• Antimony	• Mercury
• Arsenic	• Molybdenum
• Barium	• Nickel
• Benzene	• Polycyclic aromatic hydrocarbons
• Cadmium	• Thallium
• Chromium	• Selenium
• Copper	• Vanadium
• Lead	• Zinc

In many cases the limiting factor will be identifying the specific pollutants and quantifying the level of discharges. Chapter 3 briefly considers methods to quantify pollutants, but it is not the focus of this paper. Data on the potential toxicity of specific chemicals can also be challenging. In theory, any toxic pollutants with direct

[3] Although due to the proliferation of complex organic compounds it may be necessary to group similar compounds based on their toxicity and value a representative proxy which can be applied to discharges of the specific pollutants in the group.

[4] The process for calculating percentage coverage is as follows: EPA 2010 and 2011point source pollutants were mapped to the USEtox database to get average characterization factors. These characterization factors were then multiplied by the quantity in EPA based on point source loads. The percent of toxicity covered by the 16 prioroity pollutants was about 85%.

impacts on human health documented with credible ED50 data can be valued using this methodology. The USEtox database is a good starting point for this information. It includes about 3,000 organic and inorganic chemicals; about 1,250 of these have ED50s. Information from chemical databases managed by the WHO and EPA, as well as academic literature can be used to address gaps.

1.5.2. Valuation priorities

Health impacts

Corporate water pollution poses a notable risk to human health, particularly in the developing world. Industry is responsible for releasing an estimated 300-500 million tonnes of toxic pollutants into waters every year (WWAP4, 2012). Point-source water pollution from industry has been addressed in many developed countries, but it remains an issue in developing countries where it is estimated that 70% of industrial wastes are dumped untreated into water bodies (WWAP4, 2012). Long-term exposure to low levels of chemical pollutants can lead to chronic health effects such as cancer, increase the risk of adverse pregnancy outcomes, and reduced mental and central nervous function. Emissions of toxic pollutants are considered using detailed chemical fate and exposure modelling.

Nutrient pollutants can also affect health. Damage to human health from excessive nutrients is covered within the methodology on eutrophication.

Recreation, property values, fish stocks, livestock, agriculture and ecosystem services

Impacts to recreation, property values, fish stocks, livestock, agriculture and ecosystem services that occur as a result of excessive nutrient loads are captured in the eutrophication methodology. The impacts are not considered separately due to limited data availability for different locations globally, we therefore consider peoples' preferences to avoid eutrophication in general, and all the associated impacts.

1.5.3. Limitations of scope

Health impacts of pathogens

Inadequate sanitation facilities, improper wastewater disposal and animal wastes are the major sources of microbial pollution (WWAP3, 2009) and there is no doubt that harmful pathogens result in significant societal costs in the form of impacts on human health and wellbeing.

However, microbial pollution is not considered within the scope of this methodology paper for two principal reasons:

1) Human wastes are not typically directly linked to corporate activities. The majority of human wastes do not come from corporate premises and those that do are more likely to be subject to effective wastewater treatment than those which come from private dwellings or other sites.

2) The impacts of consumption of water containing harmful pathogens (driven by lack of available clean water) are captured in water consumption methodology and therefore the risks of double-counting of impacts would be high.

Ecotoxicity

Research by academics into the subject is still in its preliminary stages. The European Commission (EC) has argued that substantial work still needs to be carried out before toxicity effects on biodiversity—and consequently recreation, property values, fish stocks, livestock, agriculture and other ecosystem services—can be considered in a robust manner (European Commission Joint Research Centre, 2011).

Thermal pollution

The impacts of thermal pollution are highly localised and there is no consistent data collected on thermal pollution such that a clear articulation of the causation in an impact pathway is challenging. In many cases there is likely to be overlap with eutrophication. However, we recognise that thermal pollution is an issue for some industries; we address it on a case by case basis and will produce a generalised methodology in due course.

Ground water contamination

Contamination of ground water has been shown to pose a health risk to humans in specific contexts. However, significantly more research is needed to evaluate ground water quality worldwide and no suitable model for understanding the relationships between discharges, changes in groundwater quality and human consequences has been identified. For these reasons, ground water pollution is not addressed in this methodology document.

2. Summary of methodology

2.1. Introduction

Our valuation framework is structured to follow the impact pathway shown in Chapter 1. In aligning the two, we are able to demonstrate the causal links between corporate activities (which result in water pollution) and societal costs. To understand the value of environmental impacts associated with corporate activities, it is necessary to:

1. **Obtain environmental metric data:** The starting point for this methodology is data on the mass of pollutants discharged to water. These metric data are based on an understanding of the corporate activities which they result from. The data can come from a variety of sources which are subject to their own distinct methodologies[5], including life cycle assessment (LCA) or environmentally extended input-output modelling (EEIO). The assumed starting point for this methodology is metric data in the form specified in Table 2 below.

Table 2: Metric data for water pollution

Pollutant	Environmental metric data
Sixteen primary water pollutants	Mass of pollutant emitted to water from corporate activities (kilograms, (kg)

2. **Quantify environmental outcomes:** We quantify physical changes in the environment resulting from corporate pollutant releases or resource use (as measured by the metric data). This is discussed further in Table 3, column 2.
3. **Estimate societal impacts:** We estimate the societal cost (impact on people) resulting from environmental changes which in turn are the result of corporate activities. This is discussed further in Table 3, column 3.

It is not always necessary or appropriate for economic valuation of the environment to go through each of these steps explicitly. A single methodological step may cover some or all steps at once. However, developing each valuation methodology by following a clearly defined impact pathway helps to retain a causal link and ensure rigor, transparency, and consistency.

2.2. Summary of methodology

Environmental metric data on water pollution are the starting point for this methodology paper and hence the methods for collecting or estimating these data are not exhaustively covered. However, for the purposes of valuation it is important to understand any additional characteristics of the metric data that are likely to be available (e.g. the location of the emission and the population density nearby). For this reason, likely sources of metric data across a typical corporate value chain are summarised in Chapter 3, Table 4.

Our methodology for taking the metric data on water pollutants and quantifying and valuing the associated impacts on society is summarised in Table 3, overleaf considering first the valuation module for toxic pollutants and subsequently the valuation module for nutrient pollution.

[5] The sources of metric data are outlined in Chapter 3.

Table 3: Summary of water pollution societal impacts calculation methodology, key variables and assumptions

	Quantify environmental outcomes	Estimate societal impacts
Toxic pollutants valuation module		
Methods	• The potential impacts of effluents on human health are modelled based on the chemical fate as the pollutant travels through different media (water, soil, air, food products), and the likelihood of human exposure. • The model considers the physical characteristics of pollutants, the geophysical characteristics of locations, and the demographics in the location of interest. • Dose-response functions describe the likelihood of different health impacts occurring given a specified level of exposure. Chemical and impact specific functions estimate health outcomes for populations exposed to pollutants.	• The severity of health impacts are assessed using Disability Adjusted Life Years (DALYs) and valued using the Organisation for Economic Co-operation and Development (OECD) methodology for valuing changes in health and life.
Key variables	• Geophysical characteristics of locations: land and water area, temperature and rainfall. • Physical characteristics of pollutants: solubility, partitioning coefficients, and degradation rates. • Dose response functions.	• DALYs for relevant adverse health effects.
Assumptions and justification	• The chemical fate and exposure modelled using the USEtox (Rosenbaum *et al.*, 2011) is an acceptable simplification of reality. USEtox was developed by the Task Force on Toxic Impacts under the United Nations Environment Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) Life Cycle Initiative to include the best elements of available LCA multi-media models. • Geophysical characteristics are specified using a number of simplified parameters (average temperature, average rain rate, average fresh water depth etc.) which can be defined at a level of detail consistent with the resolution of the input data on toxic discharges. The model currently does not have capabilities to calculate chemical fates with more time sensitive information	• Value of DALYs derived from Value of a statistical life (VSL) based on an OECD meta-analysis of studies.

Quantify environmental outcomes		Estimate societal impacts
	(for instance daily rain rates). Time sensitive improvements would require geoprocessing capabilities that are not built into the USEtox model. • We assume steady state conditions when calculating chemical fate. This modelling technique is well established in the literature. • Where ambient concentrations are not known we apply linear dose response functions. This assumes that pollutant concentrations are already above any damage threshold, such that any addition of pollution in the environment causes an impact. Linear functions are standard in academic and government analysis in situations where direct on site measurements are not available. However, where ambient concentrations are known to be below safe thresholds this can be accounted for.	
Nutrient valuation module		
Methods	• To determine the eutrophication potential of P in freshwater, we use Helmes' fate factors (FF) based on advection, retention and water use. Fate factors were derived for a 0.5° x 0.5° grid covering the globe. • For all nutrient emissions to marine waters, we use the Redfield ratio (one kg of P has seven times more eutrophying potential than one kg N).	• To determine the cost of eutrophication to society, we use values based on estimations of WTP. • These damage values were based on structural benefit transfer from contingent valuation studies.
Key variables	• Environmental data: type of water. • Fate factors. • Income in the country/location (to adjust for PPP).	• The WTP per kg is derived from a number studies using transfer functions. Values were adjusted to account for differences in income, but not explicitly for differences in environmental preferences by country.
Assumptions and justification	• Helmes' fate factors present an acceptable simplification of reality. • The Redfield ratio is considered the standard, as defined in the Handbook on Life Cycle Assessment, the operational guide to the ISO standards (Guinée et al., 2002).	• Fate factor calculations are used as a proxy for eutrophiccation potential and these are applied to scale WTP estimates. This is the best available proxy we identified. The effect factor (EF) (on ecosystems) calculations from the Helmes' model, which would bring the model from mid-point to end-point, were not included as they were deemed

Quantify environmental outcomes	Estimate societal impacts
	immature by the EC. • Preferences for environmental quality vary in line with income. In instances where primary surveys of environmental preferences are not possible it is common practice for value transfer purposes to rely on income adjustment.

3. Data requirements

3.1. Introduction

The availability of high quality data on company effluent discharges and resource use across the value chain, as well as the accessibility of relevant contextual information, are key determinants of the viability of different impact quantification and valuation techniques and will affect the ultimate level of uncertainty surrounding any results.

Gathering appropriate data is a precursor to valuing the environmental impacts from water pollution. Therefore, this chapter discusses the types and potential sources of data required to value emissions to water.

There are four types of data considered here:

- **Metric data:** These relate to companies' release of effluents to water.
  - Pollutant type.
  - Pollutant quantity.
- **Substance data:** These relate to the characteristics of the pollutants themselves.
  - Physical characteristics of pollutant.
  - The impacts of the pollutant on human health.
- **Contextual data:** These relate to the context in which pollutants are released.
  - Wastewater treatment levels.
  - Stringency and enforcement of regulation.
  - Immediate destination of emission.
  - Characteristics of receiving water body.
  - Characteristics of exposed population.
- **Other coefficients:** These are factors derived from the academic literature or other credible sources which are used to convert metric and contextual data into value estimates.

Substance and contextual data play a key role in estimating the human health impacts. To understand dosing, we execute our model that includes substance properties and regional context to determine the amount of substance that remains in the water course and how much would be taken in directly or indirectly by humans. Substance data are also used to determine the potential damage a substance may cause to humans.

While methods for the collection or estimation of basic metric data are not the subject of this paper, the data generation methods used are nonetheless relevant to the likely availability of contextual data and therefore the viability of different potential valuation approaches. This chapter therefore has two purposes: firstly, it describes the most likely sources of metric data across a typical corporate value chain and the implications for contextual data availability; secondly, it sets out key substance, contextual and other coefficient data requirements and the preferred sources for these.

3.2. Environmental metric data

This section discusses the availability of metric data for water pollutants. The metric data required are the masses of each pollutant emitted to water from a given source location in a given year.

Measurement of effluent discharges to water is best done on-site using direct in-line measurement. However, aside from large regulated facilities in developed countries this is rarely a practical data source, and instead the drivers of pollution to water can be measured to estimate discharges indirectly. 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 discharge released to water 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 give 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 these data. This is likely to vary across a company's value chain. In Table 4 we list examples of the likely metric data availability across the corporate value chain and implications for appropriate contextual information.

Table 4: Likely metric data availability across a corporate value chain

	Metric data	Implications for contextual data
Own operations	Effluents released to water may be available for manufacturing facilities in company management information, particularly if the company is regulated. The other estimation techniques detailed for the supply chain can also be used if direct data are unavailable.	Based on knowledge about the location of the company and supplier, it should be possible to source geophysical information from public sources, if not from the company and their suppliers themselves.
Immediate suppliers	Supplier questionnaires can be directed to areas of high materiality or those with limited quality data from other sources. Most companies do not measure pollutants to water by substance, unless regulated. If regulated, wastewater discharge figures can be found in company management information. The other estimation techniques detailed for the supply chain can also be used if direct data is unavailable.	
Upstream/ supply chain	EEIO can be used to give 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 this is deemed appropriate.	Depending on the visibility of the supply chain location, information may or may not be available for some suppliers. Tracing raw material flows can be a good method of determining the location of different activities and processes in the supply chain. Multi-region EEIO models and trade-flow data bases can be used to approximate this, or supplier questionnaires where feasible.

	Metric data	Implications for contextual data
Downstream/ use phase	It is necessary to estimate the probable emissions associated with a product or service over its lifetime. For a laundry detergent, this may relate to direct chemical emissions to water after use. For other products, it may relate to indirect water pollutant emissions as a result of electricity consumption.	Depending on the product, the location of sale could be used as an estimation of the use and disposal phase location. In some cases it may be necessary to consider trade flows using data bases, or multi-region EEIO.
End of life/ re-use impacts	Different products are disposed in different ways. Some may be recycled or upcycled (in which case allocation of emissions needs consideration). Others will be sent to landfill or incinerators. This needs to be estimated based on the type of product and location of disposal.	

3.2.1. Regional adjustment of metric data

Where direct data are not available, estimating the likely emissions across a global supply chain is challenging, particularly as similar processes will have different levels of emissions in different countries. To apply industrialised models to developing countries we adjust metric outputs to account for the different scale of likely impacts relative to developed countries.

- **Step 1:** Adjust for likely quantity of pollutant discharge based on the underlying level of regulatory stringency and enforcement. This serves as a proxy to capture the nature of the technologies used in industrial production and the level of control over effluents.
- **Step 2:** Adjust for likely waste water treatment efficiency by assessing the average removal rates at each treatment stage.

3.3. Substance data

Substance-specific input parameters are required to model the pollutants behaviour and are shown in Table 5 and 6. The physical characteristics of the chemicals are widely available in the literature. Data was obtained from the USEtox database.

Table 5: Physical characteristics of pollutant

Information	Purpose	Default metrics
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 a low fate factor in water, due to rapid volatilisation.	l/kg (USEtox database)
Degradation rate in water, air, soil, 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 were set at 1.10-20/s, indicating no degradation of inorganics in the environment.	per 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/kg_{fish}, l/kg_{biota} (USEtox database)

Additional data are needed to calculate the effects of a substance once it has been ingested by a human. Data on dose response data was obtained from the Integrated Risk Information System (IRIS) and The Carcinogenic Potency Database (CPDB) databases, while data on critical effect value were obtained from the global burden of disease (See Table 6).

Table 6: Human health characteristics of pollutant

Information	Purpose
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.
Critical effect value	Specific type of adverse health impact related to the dose response. The model uses this value to convert from number of cases to DALY.

3.4. Contextual data

The fate of emitted pollutants to water is highly variable depending on geophysical parameters, and the impacts on people are dependent on local demographic characteristics. For a company's own operations, or that of its closest suppliers, specific locations may be known. However, emissions data from the rest of the value chain may only be available at a country level, which is considered the minimum acceptable level of geographical specificity.

Once the location is determined to the highest level of detail available, matching contextual information can be found from public sources. The tables below (Table 7 and Table 8) summarise the data required for estimating and valuing the environmental impacts of emissions to water.

Table 7: Geophysical data

Information	Purpose
Land, freshwater and coastal area	Defines the area within which the pollutant could potentially disperse. Typically set at a country level but can be defined locally.
Temperature, wind speed, average participation	Weather conditions influence the amount of substance remaining in the water course. 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 marine) to which the pollutant is directly emitted. Ratio is based on coastal population concentration at a country level, but could be defined locally.

Table 8: Exposure data

Information	Purpose
Exposed population	For indirect ingestion of pollutants, the exposed population is based on a production-based intake scenario. For direct ingestion of pollutants, the inverse of calculations on access to treated freshwater determines the number of people within region drinking contaminated water.
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.
Consumption of meat, dairy, fruits, vegetables, grains, fish and sea food	Dietary habits influence the amount of pollutant intake by humans. Daily intake is set at a country level.

3.5. Model coefficients

The total cost associated with an emission is determined using estimates of WTP to protect human health. Economic estimates and data are necessary to convert impacts to costs. See Table 9 for a list of key coefficients.

Table 9: Model coefficients

Data input required
WTP to avoid mortality and morbidity
Elasticity of WTP with respect to income
Gross national income per capita, adjusted for purchasing power parity
Inflation
Value of a DALY (Used to put a monetary value on the damage function calculated in DALYs/kg. DALY value is calculated based on OECD estimate of the VSL)

4. Detailed methodology: Toxic pollutants valuation module

This chapter covers the valuation of human health impacts from toxic pollutants emitted to water. The valuation module for toxic pollutants traces the pollutant from release to ingestion to induced health harms and ultimately values those health harms. For a summary of steps, see Table 10. 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 (of evaporated pollutants that were initially emitted to water). Once ingested (or inhaled), the health harms depend on the individual pollutant and its dose. We assign value to those health harms using published data on what individuals would pay to avoid those harms, ultimately reaching a total societal cost of water pollution.

Table 10: Summary of toxic pollutants societal impacts calculation methodology

	4.1 Quantify environmental outcomes	4.2 Estimate societal impacts
Toxic pollutants valuation module		
Methods	• The potential impacts of effluents on human health are modelled based on the chemical fate as the pollutant travels through different media (water, soil, air, food products), and the likelihood of human exposure. • The model considers the physical characteristics of pollutants, the geophysical characteristics of locations, and the demographics in the location of interest. • Dose-response functions describe the likelihood of different health impacts occurring given a specified level of exposure. Chemical and impact specific functions estimate health outcomes for populations exposed to pollutants.	• The severity of health impacts are assessed using Disability Adjusted Life Years (DALYs) and valued using the Organisation for Economic Co-operation and Development (OECD) methodology for valuing changes in health and life.

4.1. Environmental outcomes

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 output of this model is the pollutant-specific 'characterisation factor' which gives the number of health harms per unit of pollutant emitted. This modelling draws on a body of work known as LCA multimedia modelling. For background on this type of modelling, please see Appendix I.

Our preferred model for the calculation of characterisation factors is USEtox (Rosenbaum et al., 2011). Among the model options, it offers the largest substance coverage with more than 1,250 substances, and reflects more up to date knowledge and data on effect factors than other approaches (please see Appendix I third party results comparing multimedia models). It was specifically designed to determine the fate, exposure and effects of toxic substances. Additionally, it has the ability to consider spatial differences with the addition of country specific parameters.

USEtox has been adopted for regulatory assessments in e.g. the European Union's EUSES in 2004 and for persistence screening calculations as recommended by bodies such as the OECD (Klasmeier et al., 2006). This type of model is already widely used in Life Cycle Impact Assessment (LCIA) and was 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 only addressing emissions to water (to avoid double-counting with our other valuation methodologies e.g., air pollution). These modifications do not change any of the underlying calculations of the model.

In USEtox, substances that have a 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 with characterization models – in this case USEtox – which represent the mechanism of a 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 1):

Equation 1: Characterisation factors for human health

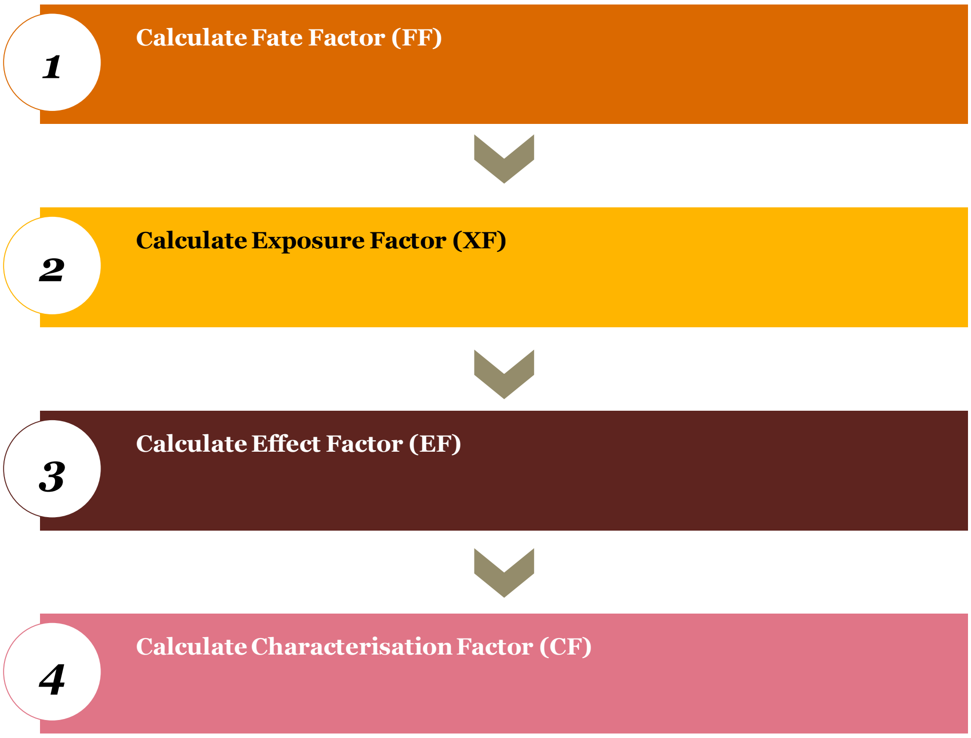
$$\boldsymbol{CF = FF \times XF \times EF}$$

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

The methodological steps to move from a register of health harms to societal value are shown in Figure 2.

Figure 2: Process steps for estimating health harms from pollutant emissions



4.1.1. Step 1: Calculate fate factor

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

There are four processes that 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). Intermedia transfer rates and removal rates depend both on the conditions in an environmental compartment and on the properties of a substance. A fate matrix calculates the intermediate transfer rates and removal rates processes against the substance specific parameters, as well as 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 will be persistent 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. For additional detail on the structure of the fate factor matrix, please see Appendix II.

Fate factors can be defined for an individual water system if data is available. If individual water system data is not available, fate factors can be defined at a country level due to data limitations.

4.1.2. Step 2: Calculate exposure factor

The exposure factor is the rate of intake of a substance (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 exposure they receive.

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

The output for this step is a substance- and country-specific exposure factor for freshwater and for marine water.

Direct ingestion

Direct ingestion occurs in our model 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 the percentage of the population with reasonable access to an adequate amount of water from an improved source, such as a household connection, public standpipe, borehole, protected well or spring, and rainwater collection. This metric is used as a proxy to determine what percentage of the population will drink untreated surface water.

The model also includes variables to cover the amount of polluted water ingested by the population at risk. Please see Appendix III for more detail on the specific equation from USEtox to calculate the exposure factor.

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

Direct inhalation

Direct inhalation occurs in our model via human intake of polluted air. In order to avoid double counting with impacts considered in our air emissions valuation model, we only consider the inhalation of pollutants that were initially emitted to water, but subsequently evaporated and became airborne.

Indirect ingestion

Indirect ingestion occurs in our model via 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 plant and from air to plant. However, for our use of the model, only the mass of substances that have been transferred from the freshwater or marine water environmental compartments to air or soil are considered.

Ingestion through meat and milk is estimated using the Travis and Arms (1988) biotransfer factor models for cows adapted for animal fat content and respective animal intake rates. The biotransfer considers both consumption of contaminated plants and drinking water. Again for plants only the masses of substances that have been transferred from the freshwater or marine water environmental compartments to air or soil are considered.

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

Food consumption patterns are varied 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 sub-populations (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 where food is produced (and contaminated) and not necessarily the location of where the population lives. 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).

Please see Appendix III for more detail on the specific equation for calculating the exposure factor.

4.1.3. Step 3: Calculate 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 change in life time 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-response, we have chosen the linear model as most appropriate for our purposes. For more detail on linear dose response functions and their advantages and disadvantages, please see Appendix IV.

The effect factor for each pollutant is based on a linear dose response function. The output for this step is a substance and country specific effect factor for freshwater and for marine water. USEtox calculates separate effect factors for non-carcinogenic effects and carcinogenic effects using the same equation. Equation 2 shows the calculation steps. For more information on dose response functions and calculating effective dose 50 (ED50), please see Appendix V.

Equation 2: 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

4.1.4. 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 kg of substance released. The basic calculation is (Equation 3):

Equation 3: Characterisation factors for human health

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

To move from 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.

Key assumptions and data

The key assumptions underlying this step are listed in Table 11. Data types requires for the model are listed in Table 12.

Table 11: Assumptions required for determining 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 is able to be defined locally where exact emission source location is known. It is therefore a necessity 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.
We assume steady state conditions when calculating substance fate	This modelling technique is well established in the literature.
A linear dose response function is assumed when determining ED50	A linear function assumes that emission concentrations are already above any damage threshold, such that any addition of pollution in the environment causes an impact. Determining whether pollutants are below any damage threshold requires data on ambient concentration and biogenic emissions data which are not globally available. Linear functions are therefore the standard in academic and government analysis.

Table 12: Data required for determining environmental outcomes

Data input	Description
Substance	
Molecular weight	Sum of atomic weights of all atoms in the compound's molecule.
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 a low fate factor in water, due to rapid volatilisation.
Degradation rate in water, air, soil, 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 were set at 1.10-20/s, indicating no degradation of inorganics in the environment.
Bioaccumulati on factor in fish/biota	Ratio of the chemical concentration in fish to the chemical concentration in the water body where the fish are exposed.
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.
Context	

Data input	Description
Land, freshwater and coastal area	Defines the area within which the pollutant could potentially disperse. Typically set at a country level but can be defined locally.
Temperature, wind speed, average participation	Weather conditions influence the amount of substance remaining in the water course. 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 marine) to which the pollutant is directly emitted. Ratio is based on coastal population concentration at a country level, but could be defined locally.
Exposed population	For indirect ingestion of pollutants, the exposed population is based on a production-based intake scenario. For direct ingestion of pollutants, the inverse of calculations on access to treated freshwater determines the number of people within region drinking contaminated water.
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.
Consumption of meat, dairy, fruits, vegetables, grains, fish and sea food	Dietary habits influence the amount of pollutant intake by humans. Daily intake is set at a country level.

4.2. Societal impacts

In the previous section, we established the number and type of health harms. Now we must value those harms to reach the societal impact of water pollution. The severity of health harms is approximated using DALYs. See Appendix VI for more information on DALYs. We then apply monetary values to those DALY totals based on willingness to pay (WTP) estimates. There are alternative approaches to this valuation including the cost approach, but we feel that WTP is the most complete valuation approach. See Appendix VI for more information on valuation approaches including WTP)

DALYs are typically used by health economists and policy makers to understand the relative severity of health conditions. They often use them to compare the cost effectiveness of investment (cost per avoided DALY), but do not typically value the welfare loss associated with a DALY. However, a methodology to do so was developed in Lvovsky et al's (2000) publication for the World Bank on valuing the health effects of various pollutants. In their paper they show that the value of a DALY can be derived from the VSL and the number of DALYs lost associated with that lost life. This approach has subsequently been applied in a government policy context by Pearce *et al.* (2004) who used it to help evaluate the EU's Registration, Evaluation and Authorisation of Chemicals (REACH) policy.

An alternate approach would be to take direct estimates of the value of the negative health cases via WTP figures, but limited data makes that approach untenable. Therefore, we use DALYs as an interim step.

The methodological steps to move from a register of health harms to societal value are shown in Figure 3.

Figure 3: Process steps for valuing the health impacts of toxic pollutants

1 **Estimate DALY value for each health harm**

2 **Apply monetary value of DALY**

3 **Compute the total societal cost via impacts on human health**

4.2.1. 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, a weighted average was applied. Average values for cancer and non-cancer 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 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 somewhat more difficult for non-carcinogenic effects where uncertainty factors are much greater. This increased uncertainty is for a myriad of reasons, including 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), and that DALYs are currently not available for all relevant non carcinogenic health effects potentially caused by chemical exposure.

Despite these concerns, we propose the use of 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 13 shows a small selection of the DALYs associated with individual pollutant negative health cases including the use of average values (e.g., 2.7 for non-cancer) when other published data is not available.

Table 13: 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, 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

4.2.2.Step 2: Applying a monetary value to a DALY

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

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

In Lvovsky *et al.'s* (2000) paper, they derive the value of the DALY from the value of statistical life (VSL) based on the number of DALYs lost associated with that lost life (Equation 4). 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). The discussion below presents our application of this approach. The values used are consistent with the values used for the VSL in the other environmental impact methodologies.

Equation 4: Value of a DALY

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

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 early teens to mid-fifties (Figure 4: Age weighting for DALYsFigure 4); the DALYs are therefore age weighted (Prüss-Üstün *et al.*, 2003).

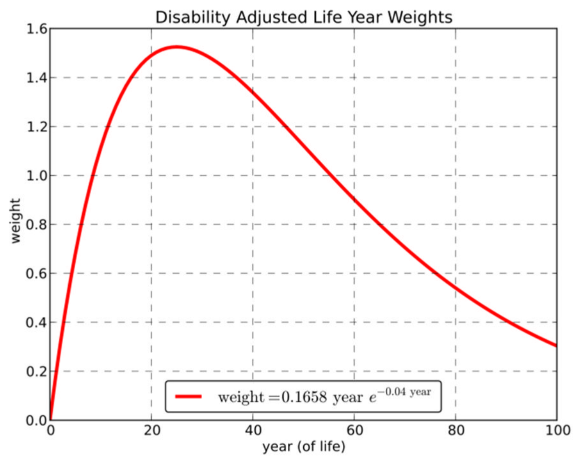
Prüss-Üstün et al. (2003) provide a formula and suggested coefficients to calculate the relative weighting of each year of life (X_w), which is set out in Equation 5 .

Equation 5: Age weighting formula for calculating DALYs

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

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 year of the 78 years of life expectancy associated with the OECD 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 Equation 6 below.

Equation 6: Discount age weighting for DALYs

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

The discounted, age–adjusted, proportion of life lost (PLL_{wd}) is calculated using Equation 7. This represents the proportion of life lost for a person who expected to live to 78 but died prematurely at 47.

Equation 7: 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. Table 14 contains the steps of the calculation that result in the value of DALY of $185,990 (in 2011USD).

Table 14: Value of a DALY

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

The value of a DALY for OECD nations is transferred to other countries. If an income adjustment is to be included (see section 4.2.1) differences between income per capita adjusted for PPP can be accounted for in accordance with Equation 8). An income elasticity of 0.6 is recommended as a central estimate of the values presented in OECD (2010).

Equation 8: Income adjustment transfer function

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

Where:

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

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

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

Equity considerations

Most countries operate a principally market-based economy, where the allocation of resources is determined largely by the forces of supply and demand, which also establish prices in the economy. In this context, an individual's income determines the quantity of marketed goods that they can obtain. When estimating the monetary value of goods (or 'bads') which are not currently traded in markets, the income constraint must therefore be considered.

As peoples income changes, their level of demand for a good usually changes, and the amount they would pay for each unit of the good also changes. Empirical evidence for environmental goods (or avoidance of 'bads') suggests that this 'income effect' is positive – people are prepared to pay more as their income increases (Pearce, 2003). For this reason, if values estimated in one location are to be used in a different location, they need to be adjusted to take account of differences in the income constraints of people in each location.

This is best illustrated using an example. Suppose a survey of people living beside a lake in the USA finds that they value the leisure time they spend around the lake at $1,000 per year. This represents about 2% of their average annual income. Combining this with the number of people who live in close proximity to the lake allows for an estimate of the value of the lake for leisure purposes to be produced. This non-market value estimate can be taken into account when decisions which might affect the future of the lake (e.g. new developments) are considered.

Now suppose we wish to estimate the value of a similar lake in Uganda. Resources to conduct a new survey aren't available but the number of people living near to the lake can be estimated, and it is known to be a popular recreation area. However, the average per capita income in Uganda is 1/100th of the average per capita income in the USA[6]. So assigning the same value of $1,000 per person in the Ugandan context would clearly be inappropriate; suggesting that local people would pay twice their average annual income for a year's worth of leisure at the lake. In order to estimate the value that local people place on the lake, relative to their other priorities, it is necessary to adjust for the differences in income constraints.

This central concept of income effects in non-market valuation of environmental goods is relatively uncontroversial, as is the practice of adjusting for differences in income and purchasing power when transferring value estimates between countries. However, when valuing goods (and bads) relating to human health, equity considerations become more apparent.

As with environmental goods, empirical evidence demonstrates that the amount individuals' would pay to maintain good health and to reduce risks to life increases with income (Viscusi and Aldy, 2003; Scotton and Taylor, 2010; OECD, 2010). This is reflected in estimates of the Value of a Statistical Life (VSL)[7]. When applying a VSL estimate calculated in one location to health outcomes in another location, it is common practice in the health literature (see for example: OECD, 2012; Hammitt and Robinson, 2011) to adjust the VSL to reflect the income differential between those locations, as described above.

These differences in preferences for life and health between locations may reflect a genuine acceptance of greater health risks, particularly in the context of other priorities such as economic development or employment. However, because preferences of this nature are often considered to be constrained by the limited choices available in low income contexts, the use of differing VSLs is contentious where decisions may relate to inter-regional resource allocations. In recognition of these concerns, the OECD (amongst others) recommend that where decisions may relate to allocations between regions a single VSL estimate should be used in policy analysis across those regions.

Given the range of possible decision-making contexts where E P&L results may be considered[8] it is important that the decision maker is aware of this potential issue and is in a position to make an informed decision. Whether the primary presentation includes or excludes income adjustments to health related values is therefore a decision for the ultimate user.

Either way we suggest that the effect of differing income levels on the results of an EP&L is assessed through sensitivity analysis.

Where the decision context has implications for inter-regional allocations, two sets of results should be presented: one which reflects equity concerns without any income adjustment to health related values, and a second which does take into account income differentials.

The decision maker will still need to consider a range of factors beyond pure environmental or health impacts. For example, a study which does incorporate income adjustments across a range of countries could provide incentives to shift polluting activities to lower income countries where the implied cost of impacts would be

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[6] Even after accounting for differences in purchasing power the ratio is 1/40th.

[7] *"Value of a Statistical Life (VSL), … represents the value a given population places ex ante on avoiding the death of an unidentified individual. VSL is based on the sum of money each individual is prepared to pay for a given reduction in the risk of premature death, for example from diseases linked to air pollution."* OECD, 2012

[8] For example, some decision contexts will be confined to a single country and could involve comparing environmental values to other factors (outside the E P&L) determined by prices or incomes within that country; while others could require prioritisation of impacts across many countries.

lower – this may be undesirable. However, a similar study which does not adjust for differences in income may deter foreign investment in lower income countries; investment which could have created improvements in well-being in excess of any health related losses.

For this reason decision makers may also wish to consider a more holistic decision making framework such as PwC's Total Impact Measurement and Management (TIMM) which values environmental impacts alongside economic, fiscal and social impacts[9].

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

Once we have established characterisation factor (which establishes the number of negative health outcomes), DALY impact per health outcome and the value of a DALY, computing the total value of pollutants simply becomes a matter of arithmetic.

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

For more information on the arithmetic calculation steps, please see Equation 9, Equation 10, and

Equation 11 below.

Equation 9: Country specific pollutant cost for human toxicity

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

Where:

$Metric\ quanitity_{c1,fw,z}$ is the mass of the substance released to freshwater in a given country

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

$Metric\ quanitity_{c1,mw,z}$ is the mass of the substance released to marine water in a given country

$Characterization\ factor_{c1,mw,z}$ is the mass of the substance released to marine water in a given country

$DALYs_z$ is the number of DALY associated with the critical cancer and non-cancer effects of the substance

$DALY\ value_{c1}$ is the PPP adjusted value a DALY in monetary terms

Equation 10: Global pollutant cost

$$\boldsymbol{Global\ impact_z = \sum (impact_{c1,fw,mw,z}, impact_{c2,fw,mw,z}, impact_{c3,fw,mw,z}, \dots impact_{cn,fw,mw,z})}$$

Equation 11: Global water pollution cost

$$\boldsymbol{Global\ impact_{total} = \sum (Global\ impact_z, Global\ impact_y, Global\ impact_x, \dots Global\ impact_n)}$$

[9] See "Measuring and managing total impact: A new language for business decisions", PwC 2013: http://www.pwc.com/gx/en/sustainability/publications/total-impact-measurement-management/assets/pwc-timm-report.pdf and: http://www.pwc.com/totalimpact for more information.

Key assumptions and data

The key assumptions underlying this method are listed in Table 15. The data required to estimate the human health impacts are summarised in Table 16.

Table 15: Assumptions required for valuing human health impacts from water pollution

Assumption	Comment on purpose and reasonableness
Average cancer value used when DALYs are not available for specific disease incidences.	Average DALYs when other data is not available were covered in academic literature by Huijbregts at all 2005.
Countries with lower GNI have a higher appetite for risk to their health, such that as GNI increases by 1 unit the VSL increases by 0.6 units.	VSL based on an OECD meta-analysis of studies and PPP adjusted using an income elasticity of 0.6 as per OECD guidance.

Table 16: Data required for valuing human health impacts from water pollution

Data
Critical effect value.
WTP to avoid mortality and morbidity.
Elasticity of WTP with respect to income.
Gross national income per capita, adjusted for purchasing power parity.
Inflation
Value of a DALY (Used to put a monetary value on the damage function calculated in DALYs/m3. DALY value is calculated based on OECD estimate of the VSL and PPP adjusted to each country.)

5. Detailed methodology: Nutrient valuation module

This chapter covers the valuation of societal costs of emitting excess nutrients to water. The valuation module for nutrients estimates the eutrophication potential of nutrients in fresh and marine water and then estimates the value based on published data on what individuals would pay to avoid those harms. For a summary of our method, see Table 17.

Table 17: Summary of nutrients societal impacts calculation methodology

Quantify environmental outcomes		Estimate societal impacts
Nutrient valuation module		
Methods	• To determine the eutrophication potential of P in freshwater, we use Helmes' fate factors (FF) based on advection, retention and water use. Fate factors were derived for a 0.5° x 0.5° grid covering the globe. • For all nutrient emissions to marine waters, we use the Redfield ratio (one kg of P has seven times more eutrophying potential than one kg N).	• To determine the cost of eutrophication to society, we use values based on estimations of WTP. • These damage values were based on structural benefit transfer from contingent valuation studies.

5.1. 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 discussed in Box 1. We model the eutrophication potential of P in freshwater using Helmes' fate factor model. In the absence of a similarly detailed model for marine eutrophication, we leverage the Redfield Ratio to assess the eutrophication potential of N in marine waters. We have relied on leading approaches wherever possible including those of the ISO handbook on Life Cycle Assessment.

The methodological steps to calculate environmental impacts are show in Figure 5.

Box 1: Summary of limiting nutrient theory

In different environments algal growth is limited by different nutrients. If more of the limiting nutrient is introduced into the system, this will promote an increase in growth. However, an 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 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 considered to contribute to eutrophication in marine waters (Howarth & Marino 2006).

In application to impact assessment, most models adopt these general rules, acknowledging that it is a simplification as other nutrients can be limiting in specific conditions (Finnveden & Potting 1999).

Figure 5: Process steps for valuing the environmental impacts of nutrients

1 **Determine environmental outcomes of phosphorus in freshwater using Fate Factor**

2 **Determine environmental outcomes of nitrogen and phosphorus in marine water using Redfield Ratio**

5.1.1. Step 1. Determine the environmental outcomes of phosphorus in freshwater

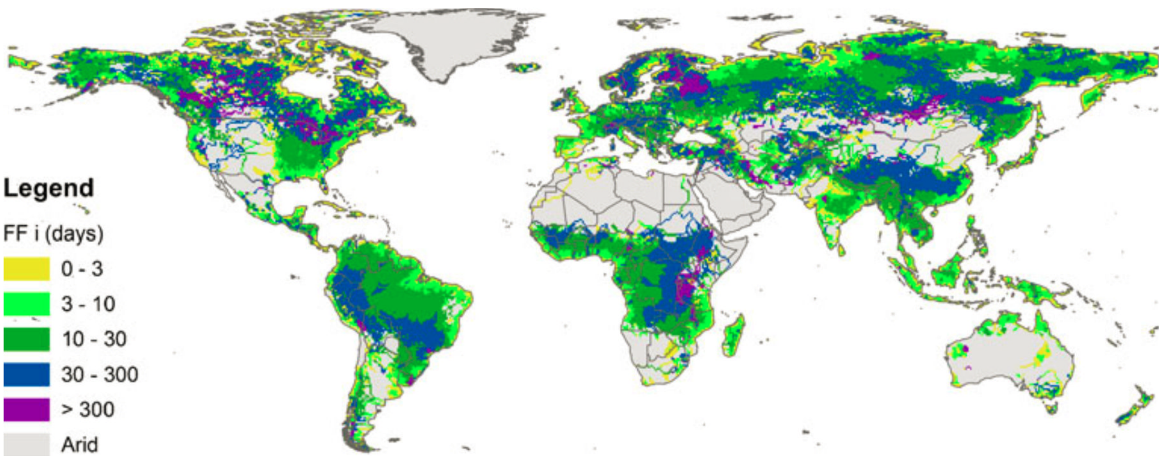
To determine the eutrophication potential of P in freshwater we use a model described by Helmes *et al.* in their 2012 paper. The model is the only P model we identified that can derive spatially explicit fate factors for P emissions to freshwater on a worldwide scale. It was created as part of Life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

Helmes' phosphorus model

The fate factor calculated by the Helmes model, essentially calculates the eutrophication potential of one kg of P released into freshwater. A higher fate factor means a higher cumulative persistence, implying P will be available longer for algae growth and the negative effects caused by 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 country level was done by a weighted average of fate factors. For a list of P fate factors used, see Appendix VII.

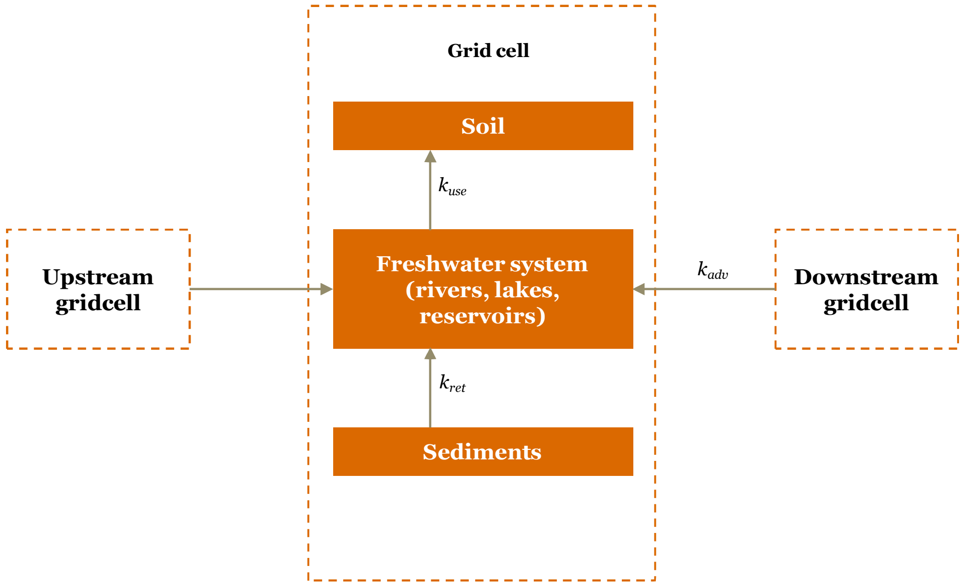
Figure 6: Fate factors for phosphorus emissions to freshwater



Source: Helmes et al. 2012

The Helmes model traces the persistence of one kg of P through grids (Figure 6). The persistence of P is based on three processes: use, advection and retention. For more information on calculating the fate factor via use, advection, and retention, please see Appendix VII.

Figure 7: Freshwater phosphorus fate model



Where:

k_{use} is use, the removal of phosphorus from the system when water is taken for domestic, industrial and agricultural purposes.

k_{ret} is retention, the uptake of phosphorus by biomass and its adsorption to suspended solids.

k_{adv} is advection, the flow of water out of the grid.

The fate factors generated by Helmes are calculated in days. See Table 18 for some sample country-level fate factors. For a complete list of country-level fate factors, please see Appendix VIII.

In the Societal outcomes section we outline how WTP estimates are applied to emissions of P. The fate factors needs to be scaled to the base countries where the valuation estimates are based, because these studies measure WTP to avoid the eutrophication associated with emissions of one kg of P, which implicitly includes the eutrophication potential of P in that location.

Table 18: A sample of country-level Fate Factor outputs

Country	Average Fate Factor (days)
Afghanistan	64.51
Albania	163.00
Algeria	228.53
Andorra	8.76
Angola	16.07

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

No equivalent models to Hermes' are available for modelling eutrophication potentials 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 kg of P has seven times more eutrophying potential than one kg N in marine water. This relationship concurs with the Redfield ratio (Redfield, 1963). These weights were used for assessing the eutrophication potential of nutrients to marine waters.

In the future, we may be able to leverage a marine-specific eutrophication model, when one is appropriately mature. A spatially differentiated fate factor model is currently being developed through the LC-IMPACT programme and tested on industry cases. This model looks at marine eutrophication caused by N emissions spatially differentiated on 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. Due to limited testing and a lack of peer review, however, this methodology was not used to model the fate of N to marine waters. As spatially differentiated global fate models for marine eutrophication become peer reviewed, results should be considered to replace the allocation method for nutrients.

Key assumptions and data

The key assumptions underlying this method are listed in Table 19. The data required to execute the methodology are listed in Table 20.

Table 19: 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 (on ecosystems) calculations from the life cycle assessment model, which would bring the model from mid-point to end-point, were not included as they were deemed immature by the EC. The EC recommends no life cycle assessment based end-point calculations for eutrophication.
Use of the Redfield ratio to scale eutrophication potential of N and P.	The Redfield ratio is considered the standard set in the Handbook on Life Cycle Assessment, the operational guide to the ISO standards (Guinée, 2002).

Table 20: Data required for determining environmental outcomes from excess nutrients

Data
Fate factors for phosphorus to freshwater

5.2. Societal impacts

There are numerous impacts caused by excessive nutrients including decreased 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 use WTP to estimate damage values per kg of N or P. This approach makes best use of the somewhat limited literature on valuation of eutrophication impacts. We convert the published values to cover other countries using Benefit Transfer (see below).

The methodological steps to calculate societal costs are show in Figure 5.

Figure 5: Process steps for valuing the societal costs of excess nutrients

1. **Value eutrophication in freshwater via Williness To Pay**
2. **Value eutrophicatoin in marine water via Willingness To Pay**
3. **Calculate overal societal impact of excess nutrients**

5.2.1. Step 1: Valuing eutrophication in freshwater

Ahlroth presents an approach to use WTP estimates for reduced eutrophication impacts to calculate a generic damage value per kg of P released to freshwater in Sweden. Studies in other parts of the world are currently limited. The benefit transfer approach presented below is based on Ahlroth's values, but could be applied to other source data where available. In applying values from a benefit transfer approach, such as this, it is important to consider the applicability of these values to other areas. For example, it is questionable as to whether values derived from a study in Sweden could be applied to developing countries.

Ahlroth 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 kg of P in Sweden, using a structural benefit transfer of eight studies to calculate total WTP and annual deposition amount. For further details on Ahlroth's work and the structural benefit transfer method applied see Appendix IX.

The underlying studies were similar in design and valued a quality change. Respondents were presented with different water quality scenarios, described using a water quality ladder. The ladder presented 5 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 to move 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 assumes a constant marginal WTP, which results in a price of $136 per kg 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 1.

Box 1: Benefit transfer of WTP

Conducting primary research on WTP is expensive and time-consuming, particularly at the global scale. A more time and cost-effective alternative to primary valuation studies, widely used in policy, is benefit transfer. This involves applying estimates of WTP 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 21.

Table 21: Types of benefit transfer

Method	Description
Value transfer	• The value from the primary study is adjusted for PPP and inflation in order that the value accurately reflects the real value of money.
Function transfer	• Values are estimated based on a number of other characteristics, based on econometric analysis of the determinants of WTP in the primary study so that the econometric function, rather than simply the value, is transferred. • Allows greater adjustment for context and improving accuracy and reliability.
Meta-analysis	• Econometric analysis 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, there is only limited primary research on WTP values across cities and countries and those studies which does exist often use inconsistent approaches. Benefit transfer can help overcome this lack of consistent primary work by providing a single value or set of values which can be applied and adjusted consistently to different geographical and socioeconomic contexts.

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

5.2.2.Step 2. Valuing eutrophication in marine water

Our approach to valuing marine water nutrients similar to that for freshwater nutrients.

For coastal areas, Ahlroth analysed existing valuation studies that estimated the value of improving water quality in marine water. As per the approach taken for freshwater, Ahlroth calculates a per kg WTP value for phosphorus and nitrogen, using a structural benefit transfer method.

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

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

5.2.3. Step 3. Sum to societal impacts of all excess nutrients

Once we have established the eutrophication potential and damage value (via WTP) for N and P in fresh and/or fresh and marine water, calculating the total societal cost of excess nutrients is 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 cost associated with excessive nutrients emissions in country. Equation 12 summarises the matrix multiplication to create the societal cost figure for each country.

Equation 12: Country specific pollutant cost for eutrophication

$$\boldsymbol{Impact_{c1,fw,mw,N,P} = Metric\ quanitity_{c1,fw,P} \times Eutrophication\ potential_{c1,fw,P} \times WTP_{c1,fw,P},} +$$

$$\boldsymbol{Metric\ quanitity_{c1,mw,N} \times Eutrophication\ potential_{c1,mw,N} \times WTP_{c1,mw,N}}$$

$$\boldsymbol{Metric\ quanitity_{c1,mw,P} \times Eutrophication\ potential_{c1,mw,P} \times WTP_{c1,mw,P}}$$

Where:

$Metric\ quanitity_{c1,fw,P}$ is the mass of phosphorus released to freshwater in a given country

$Metric\ quanitity_{c1,mw,N}$ is the mass of nitrogen released to marine water in a given country

$Metric\ quanitity_{c1,mw,P}$ is the mass of phosphorus released to marine water in a given country

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

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

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

$WTP_{c1,fw,P}$ is the PPP adjusted willingness to pay for one kg of phosphorus in freshwater in any given country

$WTP_{c1,mw,P}$ is the PPP adjusted willingness to pay for one kg of phosphorus in marine water in any given country

$WTP_{c1,mw,N}$ is the PPP adjusted willingness to pay for one kg of nitrogen in marine water in any given country

The cost of excessive nutrient water pollution globally is the sum of country specific costs, as shown in Equation 13.

Equation 13: Global excessive nutrient cost

$$\boldsymbol{Global\ impact_{N,P}} = \sum(\boldsymbol{impact_{c1,fw,mw,N,P}, impact_{c2,fw,mw,N,P}\ , impact_{c3,fw,mw,N,P}, \ldots impact_{cn,fw,mw,N,P}})$$

Key assumptions and data

The key assumptions underlying this method are listed in Table 22. Key data for the model are listed in Table 23.

Table 22: Assumptions required for determining societal impacts from excess nutrients

Assumptions	Comment on purpose and reasonableness
The WTP per kg is derived from a number of studies using transfer functions. Values were 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 to develop a more sophisticated function which included preferences for the environment, this approach is considered an acceptable approximation.
The Redfield ratio is appropriate to scale the eutrophication potential of N and P in marine water	The Redfield ratio is considered the standard 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	Using the effect factor (on ecosystems) calculations from the LCA model would bring the model from mid-point to end-point. However, these were not used as they were deemed immature by the EC. The EC recommends no LCA based end-point calculations for eutrophication.

Table 23: Data required for determining societal impacts from excess nutrients

Data
Damage values per kg of nutrient
Gross national income per capita, adjusted for purchasing power parity
Inflation

6. Sensitivity analysis

6.1. General approach to sensitivity analysis

Sensitivity analysis refers to a process of testing the robustness of a methodology, and its outputs, to changes in the inputs. This is in order to identify those parameters with the greatest potential to drive the results, and to then focus attention towards those drivers.

There is no single approach to conducting sensitivity analysis, and the approach can vary based on the needs of the analysis. Our approach focuses on understanding the inputs which have greatest influence on the results and which we consider to have the most uncertainty surrounding them. It does not consider the outputs (i.e. what would the input need to be to give a pre-defined conclusion) because this depends on the context within which the approaches are being applied.

6.2. Sensitivity analysis

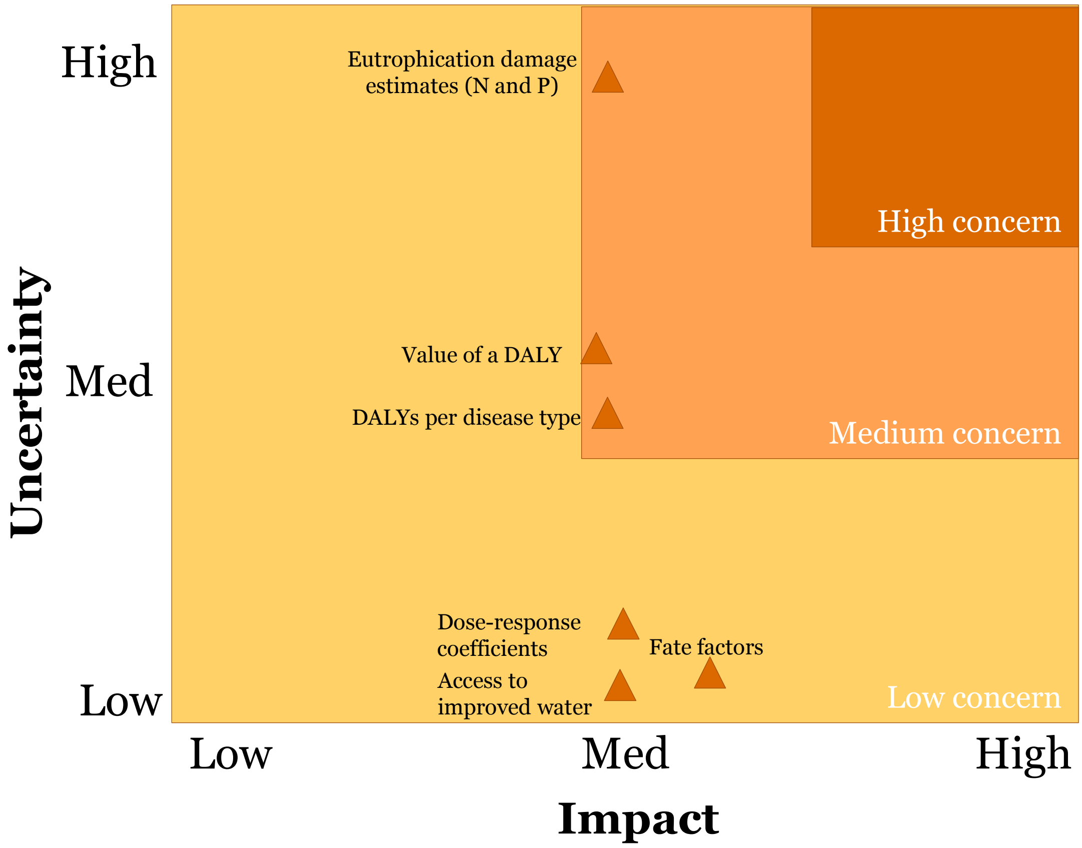
6.2.1. Overall summary and considerations for model use

The following sections provide detail on the materiality of different modules, and provide assessments of both the impact and uncertainty of the parameters that form this particular model. This summary section highlights those conclusions. Figure 6 below maps the model parameters on an impact/uncertainty matrix. Those variables towards the top right hand side of the figure (towards the high impact/high uncertainty area) are areas where we intend to focus our attention in future model development.

The water pollution module has a wide variety of important variables but none that are disproportionately powerful. Water pollution modelling is very challenging, and we have used the best available approaches. But it is important to monitor the space as new tools and studies emerge, as the current state of the field feels more like a mid-point than maturity.

In cases where there is a very high ratio of benign vs. high impact pollutant emissions, it will be worth considering additional non-human health ecosystem impacts.

Figure 6: Impact/uncertainty matrix summarising the sensitivity assessment summary for key variables and decisions



6.2.2.Materiality

There are two important dimensions which influence materiality; i) the quantity of each pollutant emitted, and ii) the scale of impacts associated with one kg of emission in a given location.

We cannot make assertions about the quantity emitted because this will depend on the corporate activities in question. We therefore focus our sensitivity analysis on a subset of pollutants: arsenic and mercury for human toxicity, as indicative of the sensitivity of the model, as well as nitrates and phosphates for eutrophication.

The three human toxicity pollutants selected here are amongst those with the highest per kg impacts. We acknowledge that it is possible that an application could have orders of magnitude differences in impacts between relatively benign human health emissions (e.g., alcohol) and potent ones (e.g., mercury), but based on application experience, we believe focusing on the highest impact parameters gives the best picture of model robustness.

6.2.3.Parameter impact

All of the parameters show a moderate impact level. This shows that most of these parameters are directly proportional to the result, for example increasing the DALYs per illness functions as a scalar on the total human health module. In this case, we can feel relatively confident, but it is important to seek the best data available in each context.

Access to improved water is one of a large number of input variables which describe the environmental and socio-economic context in the model, the impact of these other input parameters is also considered 'medium'.

For the variations in human health, the changes are calculated as the average of As and Hg societal costs.

***Table 24:* Assessing parameter impact by assessing the change to the relevant module in one of three countries after flexing the parameter**

Variable	Flex	Impact summary[10]	US (% change)	China (% change)	Nigeria (% change)
Human toxicity					
Access to improved water	10%	Med	1%	3%	6%
Dose-response coefficients from the literature	10%	Med	-10%	-10%	-10%
DALYs per disease type	10%	Med	10%	10%	10%
Value of a DALY	10%	Med	10%	10%	10%
Eutrophication					
Fate factors	10%	Med	7%	7%	7%
Societal cost per tonne of phosphorus in fresh water estimates	10%	Med	10%	10%	10%
Societal cost per tonne of nitrogen in marine water	10%	Med	10%	10%	10%

6.2.4.Parameter uncertainty

Most of the relevant parameters are taken from peer reviewed literature and cover exactly the impacts we are seeking to address, which keeps uncertainty low. The notable exception is Ahlroth's Swedish damage figure for eutrophication. These are values from a single study and require large assumptions to transfer the values to address all eutrophication in other countries. We recommend identifying more locally specific estimates depending on the application.

[10] Low = average response for overall absolute impact for three countries is less than 1%

Med = average response for overall absolute impact for three countries is 10% or less

High = average response for overall absolute impact for three countries is greater than 10%

***Table 25:* Assessing the uncertainty of key parameters based on the reliability of the measurement and the variance in attempts to measure the parameter**

Variable	Uncertainty rating	Reliability/quality of measurement	Variance of the number measured
Human toxicity			
Access to improved water	Low	Consensus best approach and peer-reviewed / endorsed by respected 3rd party	<1 orders of magnitude
Dose-response coefficients from the literature	Low	Consensus best approach and peer-reviewed / endorsed by respected 3rd party	1-2 orders of magnitude
DALYs per disease type from Huijbefts	Med	Equal quality to other possible approaches, but other benefits (e.g. better coverage)	<1 orders of magnitude
Value of a DALY	Med	Equal quality to other possible approaches, but other benefits (e.g. better coverage)	<1 orders of magnitude
Eutrophication			
Fate factors	Low	Consensus best approach and peer-reviewed / endorsed by respected 3rd party	<1 orders of magnitude
Societal cost per tonne of phosphorus in fresh water estimates	High	Clearly imperfect but no viable alternative	<1 orders of magnitude
Societal cost per tonne of nitrogen in marine water	High	Clearly imperfect but no viable alternative	<1 orders of magnitude

6.2.5. Other relevant considerations

There are a number of key assumptions in how we create our model to approximate the impact pathways that are important to consider. Modelling the pathway from emissions to human intake is a challenging task with many uncertainties. Although we are confident in our selection of the USEtox model as respected by third parties and well suited to our purposes, we acknowledge that there is still significant uncertainty in its outputs. The scope is also an issue, as the USEtox model does not address intake from groundwater, only surface water.

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Appendices

Appendix I: Life cycle assessment multimedia (LCA) models and the selection and modification of USEtox

LCA multimedia models consider the fate of emissions to water, based on the substance's persistence and how readily it could travel through water, soil and air (hence multimedia) in a given context. This physical assessment is combined with local demographic information to estimate an exposure potential, and substance toxicity information to estimate the likely impacts to health.

The physical, demographic and toxicity assessments are summarised into context and substance-specific characterisation factors (Pennington et al. 2004; Udo de Haes et al. 2002; Assies 1997; Hogan et al. 1996). The mass of each substance emitted is multiplied by the characterisation factor to provide environmental outcomes (ISO 2006; Pennington et al. 2004; Udo de Haes et al. 2002).

Characterisation factors can be used outside of traditional product LCA, as they are derived based on the emission of one kg of a substance to water. Characterisation factors are therefore the selected method used to determine the environmental outcomes related to human health from water pollutants.

There are a number of multimedia models available. In 2011, the European Commission-Joint Research Centre published Recommendations for Life Cycle Impact Assessment, analysing eight of the most widely used models. The analysis supported USEtox as the strongest overall. USEtox is a multimedia fate model, meaning that it traces a substance through different environmental media. The model classifies five distinct environmental media (air, freshwater, marine water, agricultural soil and natural soil) at two geographical scales (global and continental). The continental scale contains the following environmental compartments: urban air, rural air, freshwater, marine water, natural soil and agricultural soil; while the global scale contains the following compartments, air, freshwater, ocean, natural soil and agricultural soil. The continental scale is nested in the global scale. 'Nested' means that substances can be transported from one scale to a higher scale and vice versa.

Adaptations to the model

We have modified the USEtox model from its original form for use in the E P&L. These modifications do not change any of the underlying calculations of the model. Modifications include the addition of geographic specificity, and limit the entry point of pollutants to emissions to water. Both modifications are discussed further below.

Geographic specificity

Traditionally LCIA methods have mostly relied on generic or non-spatial multimedia environmental models. It is widely acknowledged that differences in fate and exposure mechanisms and differences in sensitivity and background levels for effect can vary significantly depending on different geographical contexts. While the USEtox model has principally been used at a global scale to date, the model is set up to be able to be calculated at a more geographically specific scale.

We sourced country specific parameters from GLOBACK, a comprehensive dataset for country-specific parameters. These include geophysical data on water balance, and a set of human exposure related parameters, based on country-specific consumption patterns.

Table 26 shows the results of the analysis for all eight models.

Table 26: Comparative analysis of multimedia models by the European Commission-Joint Research Centre

Criteria	USEtox	ReCiPe	IMPACT2002+	TRACI	EDIP 2003	CML 2002	MEEuP	EPS 2000
Completeness of scope	A/B	B	A	B/C	B/C	A/B	E	C
Environmental relevance	B	B	B	C	C	B	E/D	D
Scientific robustness & Certainty	B	B	B	B	C	B		C
Documentation, Transparency & Reproducibility	A	A/B	A/B	A	A	A		B
Applicability	A	A/B	A/B	B	B	A/B		C
Science based criteria overall evaluation	A	B	B	B/C	C	C	E	C
Stakeholders acceptance: Overall evaluation	A/B	B	C	B	C	B		C

Where a score was assigned to each model reflecting the compliance of the model with the criterion:

a. Full compliance

b. Compliance in all essential aspects

c. Compliance in some aspects

d. Little compliance

e. No compliance

Limiting model to cover only emissions to water

Although the USEtox model has the capacity to trace substances through compartments starting in any compartment, the approach used for this methodology focuses on the impacts associated with the release of effluents to freshwater and marine water (e.g, excluding emissions released to air which condense into water). This decision was made to draw a distinction between the cost of emissions to air (covered by the air pollution methodology) and emissions to water (covered by this methodology). Therefore, USEtox is only used to consider effluents initially released to freshwater or marine water. Effluents released to other environmental compartments (air or soil), which may eventually enter the watercourse are not considered.

Appendix II: Detail on USEtox fate matrix modelling

The USEtox fate model accounts for ability of a substance to persist and move between environmental media (e.g. air, water, soil, etc.), using a matrix framework established in Rosenbaum et al. (2007). In Rosenbaum's matrix (Equation 14) a column denotes the source compartment (where the substance was initially released) and a row denotes the destination compartment (where the substance is transferred into).

Equation 14: Fate factor matrix

$$\boldsymbol{FF} = \begin{pmatrix} \boldsymbol{FF_{fw,a}} & \boldsymbol{FF_{fw,fw}} & \boldsymbol{FF_{fw,mw}} & \boldsymbol{FF_{fw,s}} \\ \boldsymbol{FF_{mw,a}} & \boldsymbol{FF_{mw,fw}} & \boldsymbol{FF_{mw,mw}} & \boldsymbol{FF_{mw,s}} \end{pmatrix}$$

Where air (a), freshwater (fw), marine water (mw), and soil (s).

Appendix III: Equations for calculation of exposure factor from direct and indirect ingestion

We calculate the exposure factors for direct and indirect ingestion using the following equations.

Equation 15: Exposure factor for direct ingestion

$$XF_{xp,fw}^{direct} = \frac{IR_{xp,fw} \times P}{\rho_{fw} \times V_{fw}}$$

Where:

$IR_{xp,fw}$ symbolises the direct intake of water, polluted at a certain level, by the exposed population [kilograms/day]

P is the exposed population head count

ρ_{fw} is the bulk density of the water [kilograms/m^3]

V_{fw} is the volume of water [m^3]

Equation 16: Exposure factor for indirect ingestion for freshwater

$$XF_{xp,fw}^{indirect} = \frac{BAF_{xp,fw} \times IR_{xp} \times P}{\rho_{fw} \times V_{fw}}$$

Where:

$BAF_{xp,fw}$ is the bioaccumulation factor corresponding to the exposure pathway

IR_{xp} is the individual ingestion rate of a food substrate corresponding to the exposure pathway [kilograms/day]

P is the exposed population

ρ_{fw} is the bulk density of the water [kilograms/m^3]

V_{fw} is the volume of water [m^3]

While the exposure factors related to marine water indirect ingestion are expressed as:

Equation 17: Exposure factor for indirect ingestion for marine water

$$XF_{xp,mw}^{indirect} = \frac{BAF_{xp,\mathrm{mw}} \times IR_{xp} \times P}{\rho_{mw} \times V_{mw}}$$

Where:

$BAF_{xp,mw}$ is the bioaccumulation factor corresponding to the exposure pathway

IR_{xp} is the individual ingestion rate of a food substrate corresponding to the exposure pathway [kilograms/day]

P is the exposed population

ρ_{mw} is the bulk density of the water [kilograms/m3]

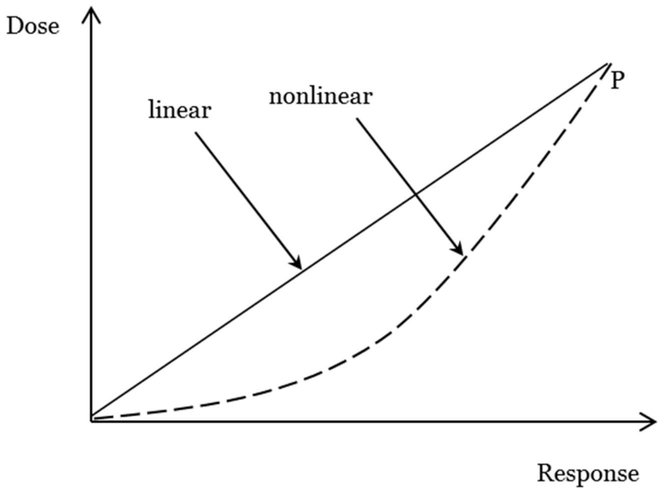
V_{mw} is the volume of water [m3]

Appendix IV: Linear dose-response functions

There are many possible dose-response function forms, Figure 7 provides a basic illustration of two.

The simplest functional form for the dose-response relationship is linear, whereby the number of cases of cancer or non-cancer increases in proportion to an increasing concentration of water pollutants, as shown in Equation 18 where dH is the change in number of estimated health incidents such as hospital admissions, death, and emergency room visits, b is a response coefficient that describes the change in number of incidents per unit change in concentration, P is the exposed population, and dC is change in ambient concentration.

Figure 7: Forms of dose-response functions



Equation 18: Linear dose-response function

$$dH = b \times P \times dC$$

Using linear dose-response functions has a number of limitations:

- Linear functions gain broad applicability at the expense of local level specificity. Different countries have different baseline incidence rates, access to healthcare and populations. These all affect the likelihood of increased illness (Ostro, 1994).
- Linear dose-response functions are mostly drawn from the US, UK and Canada. As such, any transfer of the dose-response function to other countries implicitly assumes that the relationship between ambient levels of pollution and health effects can be extrapolated across countries (Ostro, 1994).
- However, linear dose-response functions also have several advantages which make them the most suitable approach for this methodology.
- Lower data requirements than alternatives. This is particularly important for an EP&L methodology which needs to be applicable globally. This is because the data needed for more complex functions, such as ambient concentrations and local illness probability, are only available for a small number of countries.

E P&L principally deals with small changes in concentrations which limits the impact of functional form on the final results.

Appendix V: Background information on calculating ED 50

ED50 is calculated using the equation shown in the body of the paper. The USEtox author recommendations for when TD50 was unavailable.

- In case only carcinogenic, low-dose, slope factors are available, the ED50 was calculated by multiplication of 1/q* with the extrapolation factor for 1/q* to ED50, which is a factor of 0.8;
- In case only NOAEL-data or NOAEC-data are available, the ED50 was calculated by multiplication with the extrapolation factor for NOAEL to ED50, which is a factor of 9;
- In case only LOAEL-data or LOAEC-data are available, the ED50 was calculated by dividing by the extrapolation factor for LOAEL to NOAEL, which is a factor of 4, and multiplying by the extrapolation factor for NOAEL to ED50, which is a factor of 9.

Where ED50's are extant for non-human subjects, we convert to human ED50 using the data in Table 27.

Table 27: Extrapolation factor for interspecies differences

Type	CF interspecies (-)	Average bodyweight (kg)
Human	1.0	70
Pig	1.1	48
Dog	1.5	15
Monkey	1.9	5
Cat	1.9	5
Rabbit	2.4	2
Mink	2.9	1
guinea pig	3.1	0.750
Rat	4.1	0.250
Hamster	4.9	0.125
Gerbil	5.5	0.075
Mouse	7.3	0.025

Appendix VI: Background on willingness to pay (WTP) and DALYs

Willingness to pay

Short-term and long-term exposures to pollutants are consistently associated with ill-health effects (Defra, 2011b), which are typically divided into two categories:

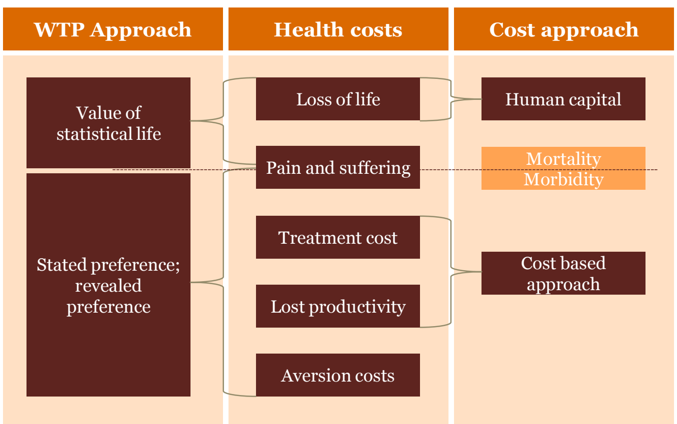
- **Morbidity:** Increased incidence rates of illness. These are measured, for example, by cases of chronic asthma and chronic bronchitis, respiratory hospital admissions and emergency room visits for asthma;
- **Mortality:** Premature death.

There are two broad approaches to estimating the social cost associated with morbidity and mortality:

- Cost approach:
  - **Mortality:** The value of the lost contribution to economic activity due to premature death, known as the Human Capital approach;
  - **Morbidity:** The cost of treatment and lost productivity, a lower-bound proxy for WTP, known as the Cost of Illness approach;
- Willingness to pay:
  - **Mortality:** Stated and revealed preferences for avoided or reduced risk of death, known as the VSL;
  - **Morbidity:** Stated and revealed preferences for avoided or reduced illness.

While both approaches are used in policy making, WTP approach is a more complete measure. WTP values for morbidity encompass direct (medical costs) and indirect costs of illness (i.e. lost productivity) as well as intangible aspects (e.g. pain and suffering). They therefore offer a better representation of individual preferences regarding the likelihood of illness or premature mortality ex ante (OECD, 2006). Figure 8 shows the difference between cost of illness estimates and WTP estimates.

Figure 8: Types of costs covered by WTP approach and cost approach



Under the WTP approach, the shadow price of mortality is termed the VSL. VSL is an individual-specific value, defined as the marginal rate of substitution between mortality risk and income, i.e. the individual's WTP for a small reduction in mortality risk divided by the risk change (Hammitt, 2002). Similarly, WTP for life encompass values of the loss of life and the intangible aspects of pain and suffering.

DALYs and health outcomes

DALYs measure the severity of disease, combining years lost due to premature death (i.e. Years of Life Lost (YLL)) and 'healthy' years lost to ill health or disability (i.e., Years Lost due to Disability (YLD)). Healthy years lost are calculated by multiplying the length of time the disease occurs and a disability weighting based on the severity of the disease as described in Prüss-Üstün et al's (2003) report for the WHO on assessing the environmental burden of disease.

DALYs have been reported for a number of cancer types (Hofstetter 1998; Frischknecht et al. 2000; Crettaz et al. 2002). However, DALY estimates for non-cancer effect types suitable for LCA are generally scarcer. A comprehensive study by Huijbregts et al. 2005, derives damage factors from the extensive WHO's Burden of Disease and other health statistics provided by Murray and Lopez (1996a, 1996b). A total of 49 non-communicable diseases were evaluated. A sample of these DALY values From Huijbregts work for individual cancer and non-cancer health outcomes is shown in Table 28 as well as the overall averages.

Table 28: Examples of disability adjusted life years for selected diseases

Cancer	**DALY**	**Non-cancer**	**DALY**
Mouth and oropharynx cancer	6.2	Multiple sclerosis	19.7
Oesophagus cancer	17.9	Obsessive-compulsive disorder	0.2
Stomach cancer	13.6	Inflammatory heart disease	5.5
Colon and rectum cancer	8.8	Asthma	0.6
Liver cancer	22.5	Diabetes mellitus	2.2
Pancreas cancer	16.2	Renal agenesis	80
Leukaemia	28.3	Down syndrome	55.9
Cancer average DALY (based on weighted average)	**11.5**	**Non-cancer average DALY** (based on weighted average)	**2.7**

Appendix VII: Phosphorus fate factor for freshwater

The P fate factor in the Helmes model is calculated using the following approach.

Advection

Advection is the flow of water out of the grid, which is determined by the rate of water flow. This value is the ratio between its discharge (Fekete et al. 2002) and the total water volume in the grid cell, which is the sum of the volumes of lakes, reservoirs and rivers. See Equation 19.

Equation 19: Advection of phosphorus

$$k_{adv,i} = Q_i / V_{tot,i}$$

The river volume was calculated from the river width, depth and length. River width and depth were calculated by empirical relationships from Wollheim et al. (2006), while length was determined with the relation between watershed surface area and river length from Vörösmarty et al. (2000). Lake and reservoir volumes were taken from Green et al. (2004). For application of these data to fate factor calculation, the volume of cells in large lakes needed to be adjusted in the 0.5°×0.5° data set to correct for a discrepancy concerning low flows of edge lake cells, as the data set assigned the flow in such edge cells to the centerline of the water body. In the model presented in this manuscript, the volume of edge lake cells has been grouped with the volume of the centreline cells to ensure consistency between the flows and volumes of lake grid cells.

Retention

Retention is governed by two main processes: the uptake of P by biomass and its adsorption to suspended solids and their subsequent physical settling (Hejzlar et al. 2009). The overall retention rate in a grid cell is the volume-weighted average of the removal rates in the separate water bodies.

Equation 20: Retention of phosphorus

$$k_{ret,j} = \sum_{wb} \frac{V_{wb,j}}{V_{tot,j}} \times k_{ret,wb,j}$$

Where:

$V_{wb,j}$ is the total volume of the river, lake or reservoir

$V_{tot,j}$ is the total volume in grid cell j

The fraction reflects the amount of time the P resides in the water body relative to its total persistence in the grid.

$k_{ret,wb,j}$ is the removal rate of phosphorus in the water body

Equation 21: Retention of phosphorus explained in detail

$$k_{ret,j} = \sum_{wb} \frac{V_{wb,j}}{V_{tot,j}} \times k_{ret,wb,j} = \frac{1}{V_{tot,j}} \left(V_{riv,j} \times k_{ret,riv,j} + V_{lak,j} \times \frac{v_f}{D_{lak,j}} + V_{res,j} \times \frac{v_f}{D_{res,j}} \right)$$

$$= \frac{1}{V_{tot,j}} (V_{riv,j} \times k_{ret,riv,j} + A_{lak,j} \times v_f + A_{res,j} \times v_f)$$

$$= \frac{1}{V_{tot,j}} \left(V_{riv,j} \times k_{ret,riv,j} + V_f \left(A_{lak,j} + A_{res,j} \right) \right)$$

Where:

v_f is the phosphorus uptake velocity

$D_{wb,j}$ is the depth

$A_{wb,j}$ is the surface area

Removal rates are from US fate model SPAtially Referenced Regressions On Watershed Attributes (SPARROW) (Alexander *et al.* 2004), in which these rates and other input variables are calibrated to predict P loads in individual streams. Although parameters are derived for the US, the model includes a wide diversity of streams, lake and climatic conditions that can be applied to other continents. SPARROW is the most globally representative model currently available.

Use

Use is defined as the P removed from the system when water is taken for domestic, industrial and agricultural purposes.

Fate factor calculation

The fate factor for P for emission in cell *i* is the sum of the sum of the fate factors for the individual

cell of emission and of all downstream receptor grid cells ($FF_{i,j}$).

Equation 22: Helmes' freshwater phosphorus fate factor

$$FF_i = \sum_j FF_{i,j} = \sum_j f_{i,j} \times \tau_j$$

Where

$FF_{i,j}$ is the partial fate factor of emitting grid *i*

$f_{i,j}$ is the fraction of P from *i* that reaches *j*

τ_j is the persistence defined in the following equation:

Equation 23: Persistence of phosphorus

$$\tau_j = \frac{1}{k_{adv,j} + k_{ret,j} + k_{use,j}}$$

Appendix VIII: Average country level phosphorus fate factors

Country	Average Fate Factor (days)	Country	Average Fate Factor (days)	Country	Average Fate Factor (days)
Afghanistan	64.51	Georgia	117.02	Nigeria	21.98
Albania	163.00	Germany	26.91	Korea, Dem. Rep.	2.72
Algeria	228.53	Ghana	15.29	Norway	258.99
Andorra	8.76	Greece	24.50	Pakistan	66.98
Angola	16.07	Guatemala	46.22	Panama	71.00
Argentina	10.76	Guinea	23.89	Papua New Guinea	6.32
Armenia	117.50	Guinea-Bissau	3.60	Paraguay	20.00
Australia	4.27	Guyana	2.83	Peru	166.71
Austria	77.01	Haiti	57.04	Philippines	22.76
Azerbaijan	78.12	Honduras	19.94	Poland	20.87
Bangladesh	14.10	Hungary	36.86	Portugal	2.33
Belarus	32.90	Iceland	5.99	Puerto Rico	1.91
Belgium	41.17	India	25.08	Romania	22.01
Belize	9.59	Indonesia	7.06	Russian Federation	131.45
Benin	33.11	Iran, Islamic Rep.	1,668.58	Rwanda	2,635.15
Bhutan	10.86	Iraq	309.62	San Marino	3.71
Bolivia	189.60	Ireland	25.06	Senegal	6.51
Bosnia and Herzegovina	23.73	Israel	292.92	Serbia	24.34
Botswana	1.30	Italy	51.67	Sierra Leone	4.17
Brazil	37.67	Jamaica	1.67	Singapore	1.00
Brunei Darussalam	1.51	Japan	27.58	Slovak Republic	36.15
Bulgaria	28.57	Jordan	3,915.02	Slovenia	26.96
Burkina Faso	24.52	Kazakhstan	71.07	Solomon Is.	1.28
Burundi	7,673.41	Kenya	267.94	Somalia	29.92
Cambodia	54.32	Kuwait	2.53	South Africa	67.75
Cameroon	37.20	Kyrgyz Republic	286.54	South Korea	2.88
Canada	470.20	Lao PDR	23.76	Spain	3.24
Central African Republic	26.52	Latvia	24.69	Sri Lanka	3.00
Chad	55.00	Lebanon	7.80	Sudan	558.36
Chile	12.40	Lesotho	31.25	Suriname	1.40
China	34.72	Liberia	5.90	Swaziland	431.75
Colombia	17.96	Libya	14,156.62	Sweden	98.79
Congo, Rep.	8.69	Liechtenstein	833.35	Switzerland	387.49
Congo, Dem. Rep.	913.35	Lithuania	22.85	Syrian Arab Republic	159.65
Costa Rica	3.96	Luxembourg	13.83	Tajikistan	51.73
Cote d'Ivoire	12.19	Macedonia, FYR	431.65	Tanzania	1,888.96
Croatia	24.75	Madagascar	12.57	Thailand	14.67
Cuba	1.27	Malawi	2,818.29	The Gambia	4.35
Cyprus	15.71	Malaysia	2.98	Timor-Leste	2.00
Czech Republic	28.53	Mali	60.77	Togo	44.59
Denmark	22.81	Mauritania	30.89	Trinidad and Tobago	1.55
Djibouti	67.18	Mauritius	1.14	Tunisia	148.32
Dominica		Mexico	25.65	Turkey	194.01
Dominican Republic	8.03	Moldova	38.73	Turkmenistan	578.43
Ecuador	8.13	Monaco	1.55	Uganda	616.26
Egypt, Arab Rep.	9.89	Mongolia	977.66	Ukraine	22.38
El Salvador	23.40	Montenegro	10.41	United Kingdom	5.98
Equatorial Guinea	4.50	Morocco	3.09	United States	118.82
Eritrea	2,019.05	Mozambique	217.67	Uruguay	28.31
Estonia	44.08	Myanmar	9.50	Uzbekistan	72.71
Ethiopia	241.57	Namibia	7.28	Venezuela, RB	27.75
Faeroe Islands	0.67	Nepal	15.32	Vietnam	5.53
Fiji	1.70	Netherlands	33.71	West Bank and Gaza	2,455.06
Finland	227.58	New Caledonia	39.35	Zambia	379.46
France	33.49	New Zealand	6.17	Zimbabwe	30.32
French Guiana	1.82	Nicaragua	206.34		
Gabon	10.62	Niger	39.74		

Appendix IX: Ahlroth's structural willingness to pay estimate methodology

Below are details on the equations used by Ahlroth to derive WTP estimate for environmental improvement (e.g., reduced eutrophying nutrients).

Equation 24: Indirect utility function

$$V = [(P - r(q))^{-\alpha} m]^{K}$$

Where:

P is a relative price that represents the travel costs

r is a valuation function which describes how the environmental quality affects the effective price of a trip

q is an index for environmental quality (e.g. sight depth, pH value, fish catch, etc)

m is income

α, K are parameters

Using Roy's identity, we can derive the demand for trips, X:

Equation 25: Demand for trips

$$X = -\frac{V_P}{V_m} = \frac{\alpha m}{P - r(q)}$$

From travel cost studies, Ahlroth obtains an estimate of the marginal consumer surplus for an environmental improvement.

Equation 26: Marginal consumer surplus

$$\frac{\partial CS}{\partial q} = \frac{\partial}{\partial q}\int_{P_0}^{P_c} X dP = \alpha m \frac{r'(q)}{P_0 - r(q)}$$

Solving for r´(q)

$$r'(q) = \frac{\frac{\partial CS}{\partial q}}{\frac{\alpha m}{(P_0 - r(q))}} = \frac{\frac{\partial CS}{\partial q}}{x}$$

The WTP for obtaining a certain improvement is defined as:

$$\mathbf{V(m, P, Q0, \alpha) \ = \ V(m - WTP, P, Q1, \alpha)}$$

From the specification of the indirect utility function, WTP can be written as:

$$WTP = m - \left(\frac{P - r(q_1)}{P - r(q_0)}\right)^{\alpha} m$$

Solving for α

$$a = \frac{\ln(\frac{m - WTP}{M})}{\ln(\frac{P - r(q_1)}{P - r(q_0)})}$$

The WTP is linked to the experienced change in effective price for using the amenity. Values of r´(q) are obtained from travel cost studies and WTP values are derived from contingent valuation studies. The above equation is calibrated by inserting values on WTP, m, P, q_o and q_1 from a valuation study.

The form for r(q) is a function of the quality index q and some parameter β. It is calibrated by inserting the marginal value per trip from the chosen travel cost study. The functional form for the r(q) function can reasonably be assumed to take a logistic form. In two travel cost studies, Sandström (1996) and Paulrud (2003), use conditional logit (CL) models to estimate willingness to pay. Both displayed a declining marginal utility of quality, so that the WTP for 1-metre improvement of sight depth was smaller at larger sight depths. The model form selected by Ahlroth is as follows:

$$r(q) = P\gamma(1 - e^{-\beta q})$$

Where

P is the travel cost

q is a quality measure

β and γ[11] are parameters

Deriving the equation:

$$r'(q) = P\beta e^{-\beta q}$$

Where

r´ is the increase in consumer surplus per trip attached to an increase in environmental quality

β[12] is calibrated by inserting values of r´, P and q from the selected TC study into the above equation. Since β influences how much r changes in response to changes in quality, q. Together with γ, it influences the curvature of the transfer function, i.e. how much marginal WTP changes between different quality levels.

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[11] γ is an exogenous parameter and should be set to a value equal to or below one so that it will not exceed the cost. This assumes that the respondent will always perceive some positive amount as a cost (Sandström 1996, Paulrud 2004a, Soutukorva 2005).

[12] β cannot be solved for analytically from r´(q), the value of β is derived numerically, using Solver in Excel

To calibrate the γ parameter, Ahlroth uses a model and applies different values of γ with data from two travel cost studies that value water quality: Soutukorva (2005) and Sandström (1996). The resulting functions were compared with the CL functions estimated on the raw data in the original studies.

For both travel cost studies, a γ value of 0.8 was found to best mimic the original functions.

The α values calibrated from the different studies are shown in Table 29. To illustrate the difference in WTP estimates implied by different values of α, WTP for a one-class quality change (from class 3 to class 2) is also shown in Table 29. The values are calculated with mean Swedish income for 2005 (Statistics Sweden, 2007).

Table 29: Parameter values for transfer functions

Study site	α	WTP (SEK)
Vansjö-Hoböl	0.029	1,844
Orre	0.038	2,439
Lagenvassdraget	0.010	658
Ånøya	0.008	497
Steinsfjord	0.008	518
Randers fjord	0.014	895
Lake Oulujärvi	0.013	840
Guestrower-Seen	0.007	442
Ville-Seen	0.008	505

Parameter values for transfer functions calibrated on CV studies. WTP estimate for Sweden for a water quality increase from class 3 to class 2, using transfer function with Swedish mean income level 2005. β = 0.22

The estimates from Orre and Vansjö-Hobol lie considerably higher than the other locations, as has been noted in several benefit transfer studies (e.g., Muthke and Holm-Muller 2004, Methodex 2007a). They both represent large lakes without similar substitutes nearby. The mean α value is 0.015 if Orre and Vansjö-Hobol are included, and 0.010 otherwise. This corresponds to a WTP value for an improvement from class 3 to class 2 of SEK 965 and 645, respectively. Since substitute sites are considered likely to be available in most geographic contexts, the lower α value of 0.010 was selected. Damage values for each region are computed using eutrophication mappings and adjusted for income.