Best Practice Guide for Mid-Point Life Cycle Impact Assessment in Australia

ALCAS Impact Assessment Committee

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CONTENTS

GLOSSARY OF TERMS AND ACRONYMS

AE	Accumulated exceedance
ALCAS	Australian Life Cycle Assessment Society (www.alcas.asn.au)
AusLCI	Australian Life Cycle Inventory Database Initiative
Area of protection	The issue of concern for the environment or society that can be impacted by activities in the life cycle of goods and services.
BEES	Building for Environmental and Economic Sustainability, a software tool developed by the US National Institute of Standards and Technology (NIST)
BOD	Biochemical oxygen demand, a unit of organic matter in polluted water
Bq	Bequel, a unit of radiation
Characterisation	The step in LCA for calculating a potential impact from an elemental flow (in the LCI) by multiplying the flow by a characterisation factor.
Characterisation factor	Factor applied to an input or output flow in the LCI which converts the flow into common units reflecting the contribution of the flow to a specific impact.
Characterization model	Model used to calculate characterization factors (e.g. USETOX)
CML	Institute of Environmental Sciences (CML) in the Netherlands (http://cml.leiden.edu/research/industrialecology)
CML IA	An impact assessment method developed by CML (Guinee et al., 2002). The Guide refers to the latest version at the time of publication (CML-IA, Version 4.8, August 2016), characterisation factor for which can be sourced from http://cml.leiden.edu/software/data-cmlia.html.
C14	Carbon isotope with molecular weight of 14
C_2H_{4-}	Ethylene
CO	Carbon monoxide

CTU	Comparative toxicity units (for humans or ecosystem), being the additional disease cases per kg of chemicals emitted to the freshwater environment.
DALY	Disability adjusted life years
$1,4\text{-}DB_{\text{-eq}}$	1,4, dichloro-benzene
EC-JCR	See JCR
Eco-indicator 99	An impact assessment method developed by (Goedkoop and Spriensma, 2001)
EDIP	Environmental Design of Industrial Products. An impact assessment method (Hauschild and Wenzel, 1998), (Hauschild and Potting, 2005)
ELU	Environmental load unit
EPD	Environmental Product Declaration. Voluntary declaration providing quantified environmental data using predetermined parameters and, where relevant, additional qualitative or quantitative environmental information. A Type III environmental label.
EPS	Environmental Priority Strategies. An impact assessment method (Steen, 1999b, Steen, 1999a)
Fe	Iron (chemical symbol)
GHG	Greenhouse gas
GWP	Global warming potential
ILCD	International reference Life Cycle Data system – a series of detailed technical documents providing guidance for good practice in LCA in business and government. The Guide refers to the latest version at the time of publication (ILCD, Version 1.0.9, May 2016.
IMPACT 2002+	An impact assessment method developed by (Jolliet et al., 2003).
Impact method	Integrated set of impact assessment methods
IPCC	Intergovernmental Panel on Climate Change
ISO	International Standards Organisation
JRC	Joint Research Centre (of the European Commission)

kg	kilogram
LCA	Life cycle assessment
LCI	Life cycle inventory. The step in LCA associated with the quantification and aggregation of exchanges with the environment (inputs and outputs) for a give product system through its life cycle.
LCIA	Life cycle impact assessment. The step in LCA associated with evaluating the magnitude and significance of potential environmental impacts of a product system.
Life cycle inventory	See LCI
Life cycle impact assessment	See LCIA
LIME	Life cycle Impact Assessment Method. An impact assessment method developed for Japan (Itsubo et al., 2004)
LUCAS	An impact assessment method (Toffoletto et al., 2007)
Mid-point indicator	A point in the cause-effect pathway that is midway between the elementary flow or stressor (eg. the release of a substance) and the resulting impact.
MJ	Megajoule
ODP	Ozone depletion potential
N	Nitrogen
NMVOC	Non-methanic volatile organic compounds
P	Phosphorous
PAF	Potentially affected fraction of species
PEF	Product Environmental Footprint
PM	Particulate matter
PM2.5	Particulate matter up to 2.5 micrometres in diameter
PM105	Particulate matter up to 10 micrometres in diameter
ReCiPe	An impact assessment method developed by RIVM, CML, PRé Consultants, and Radboud Universiteit Nijmegen (Goedkoop et al., 2009). The Guide refers to the latest version at the time of publication (ReCiPe 2012)

Sb	Antimony (chemical symbol)
SETAC	Society of Environmental Toxicology and Chemistry
SOA	Secondary organic aerosols
Standard	Refers to the International Standard for Life Cycle Assessment, ISO14040 (ISO, 2006a)
TRACI	Tools for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI). An impact assessment method developed for the US (Bare, 2002). The Guide refers to the latest version at the time of publication (TRACI, Version 2.1).
U235	Uranium isotope with molecular weight of 235
UNEP	United Nations Environment Programme
USETox	A scientific consensus model endorsed by the UNEP/SETAC Life Cycle Initiative for characterizing human and ecotoxicological impacts of chemicals (www.usetox.org)
VOC	Volatile organic compounds
WMO	World Meteorological Organisation
WSI	Water stress index

FOREWORD

ALCAS is delighted to present the second version of the ALCAS Best Practice Guide for Impact Assessment, which is a significant update to the original version prepared in 2008 (Grant et al., 2008).

The field of impact assessment looks to (international) best practice to inform how we can best account for the environmental impacts that are caused by elementary flows measured in Life Cycle Assessment (LCA) studies. While the impact assessment models behind some indicators are fairly stable (e.g. ozone layer depletion), others are regularly updated (e.g. global warming potentials for climate change) or even completely redeveloped (e.g. toxicity) based on new scientific insights.

Therefore, ALCAS considers it as one of its core duties to keep the Australian LCA community and its stakeholders informed about what is current best practice in the moving field of impact assessment. As the peak body for LCA related matters, ALCAS can draw on experts from its membership with experience in the application of different impact assessment methods and an understanding of their use in the Australian context. I want to acknowledge and thank those who contributed to this Guide and contributed to the transparent and informed discussions.

The Best Practice Guide is intended to be a living document, and future updates are both necessary and expected. Our ability to facilitate updates will depend on new and ongoing support for ALCAS. If you are not a member yet, then we would welcome you on-board.

Rob Rouwette, ALCAS President (September 2013 - September 2015)

1 HOW TO USE THIS GUIDE

The aim of this document, hereafter referred to as the Guide, is to provide up-to-date information about life cycle impact assessment (LCIA) methods, and guidance on the selection of methods for LCA studies centred on Australian products and processes. Given the range of LCIA models and factors available, practitioners are often unsure about which one to apply when conducting life cycle assessment (LCA). This guide can help practitioners make more informed choices about **mid-point** impact categorisation in the context of Australia.

For each of the commonly assessed impact categories, a critique is provided on available LCIA methods leading to a recommendation of best practice for LCAs undertaken in Australia.

The guide will help the reader select a mid-point method (and characterisation factors), but does not give guidance on the selection of impact categories to include in a study. This will depend on the objective of the study, the processes being assessment, and the environmental sensitivities of regions in which the processes occur, and is best determined through a screening study.

The best practice recommendations represent the expert judgement of the authors at the time of writing. Methods selection is ultimately up to the practitioner. As best practice also evolves over time to reflect international developments, the reader is advised to ensure they use the latest version of this document from the Resources section of the ALCAS website (www.alcas.asn.au).

For screening studies, the reader may wish to refer directly to the 'Summary of Recommended Mid-point Methods' in Section 3.1. For more detailed studies, the reader may wish to consult Sections 3.2 onwards, which discusses each impact category in more detail.

Methods can refer to either to the underlying characterisation model or the integrated LCIA method (as used in LCA software). This document will generally refer to the latter because this is how methods are most commonly known, but will make reference to the underlying models. For brevity, LCIA methods are only described briefly, but references to sources of further information are provided.

The characterisation factors for the best practice impact assessment methods are provided in an accompanying ANNEX spreadsheet, which can be downloaded from the Resources section of the ALCAS website (www.alcas.asn.au).

2 INTRODUCTION

Life cycle impact assessment (LCIA) is the third stage of life cycle assessment (LCA), after goal and scope definition, and life cycle inventory (LCI) development (Figure 1). It converts LCI data into more meaningful indicators by assigning inventory items to impact categories, and converting and aggregating them to indicators of impact using characterisation factors (Figure 2).

Various organisations have developed and are continuing to develop integrated LCIA methods consisting of characterization factors based on underlying scientific models. There is no single agreed 'one size fits all' method for LCIA (ISO, 2006a, ISO, 2006b), and practitioners often have limited guidance as to which method to apply in a given study. Method selection can influence results so method choice can be important.

The methods and characterisation factors underlying impact assessment methods commonly reflect the environmental conditions of the region in which they were developed, and may not always translate accurately to all regions. This means that for LCAs conducted in Australia, uncertainty can result when characterisation factors developed in the European context (for example) are applied to processes occurring in Australia. There are a few examples where geographically-regionalised characterisation factors have been developed, including for Australia (for example for toxicity, acidification, water stress). However this process of regionalisation is still ongoing, and it will be some time before characterisation factors aligned to Australian receiving environments are available for all impact categories.

The drive toward international harmonisation of methods also needs to be recognised (Jolliet et al., 2014), the intent of which is to provide a common playing field when comparing LCA results across products and continents, and for consistency in certification programs based on LCA. So the previously noted desire for regionalisation needs to be in the context of international consistency. The end goal is to have internationally accepted models which delivers regionalised characterisation factors, where regionalisation is important.

2.1 Context

The guide has been developed within the context of processes and initiatives occurring within the international LCA community, which are described here briefly.

2.1.1 International standards

The International Standards Organisation (ISO) provides a procedure for LCIA (ISO, 2006a, ISO, 2006b), which has also be adopted as an Australian / New Zealand standard (the Standard). It has mandatory and optional elements (Figure 2). This guide focuses on the mandatory elements, i.e. the characterisation of impacts.

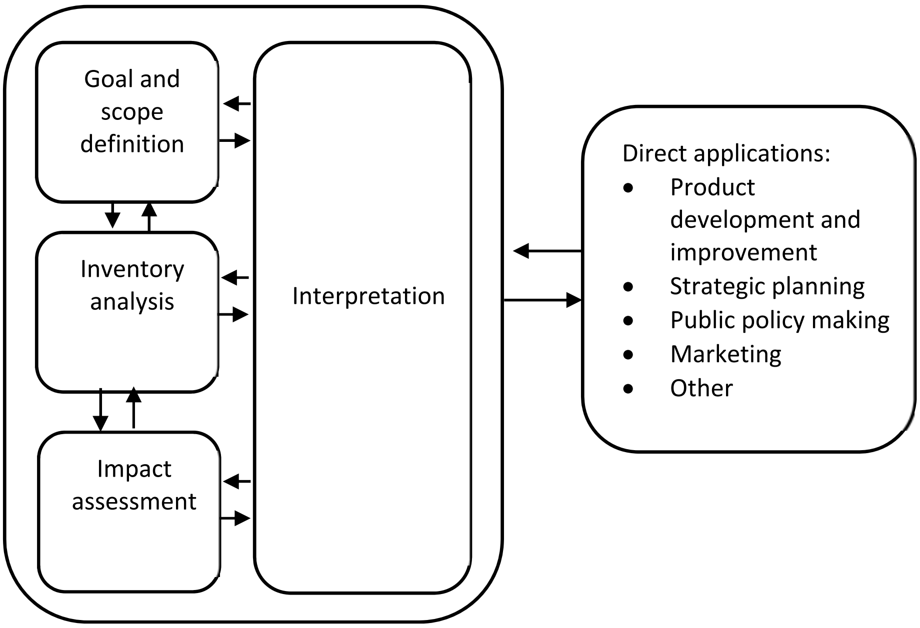

Figure 1 Procedure for life cycle assessment (LCA) according to the International ISO 14040 (ISO, 2006a)

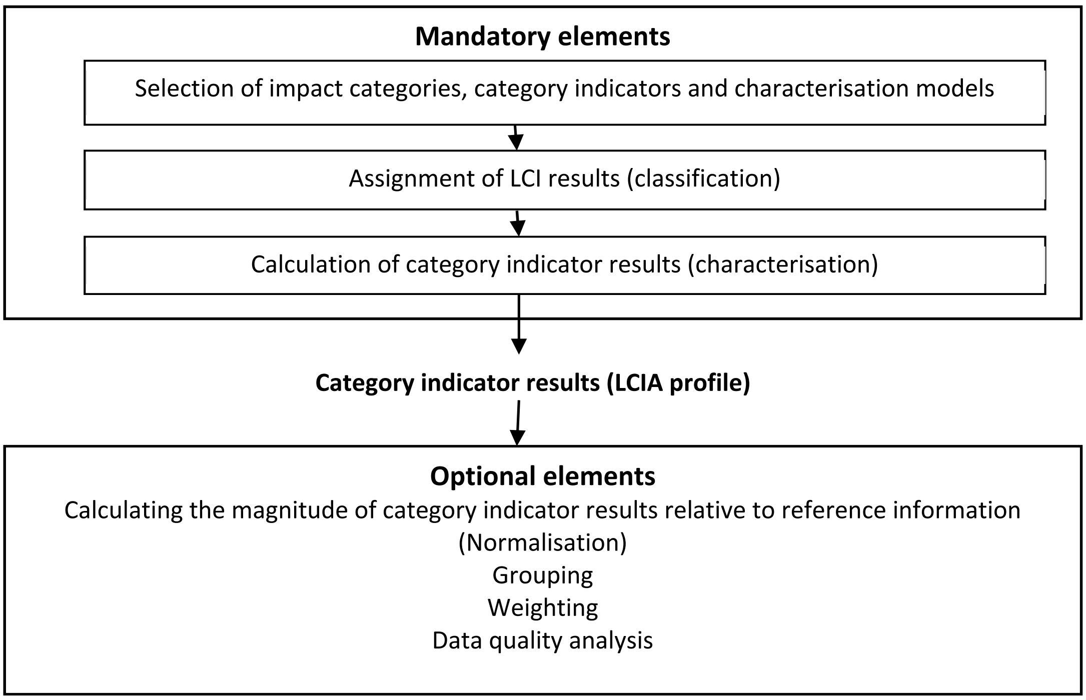

Figure 2 Elements of the life cycle impact assessment (LCIA) phase, according to the International Standard (ISO, 2006a)

2.1.2 Australian life cycle inventory (AusLCI) initiative

The Australian Life Cycle Inventory (AusLCI) initiative (http://alcas.asn.au/AusLCI/) has developed protocols for the consistent development of life cycle inventory (LCI) data suitable for various applications in Australia and overseas. In the future this guide will inform the AusLCI protocols, so data submitted to AusLCI can be consistent with best practice impact assessment.

2.2 Scope

The focus is on impact characterisation of mid-point indicators. Mid-point indicators represent effects midway along the impact pathway (see

Figure 3), and are commonly used as proxy indicators for environmental impacts at the end-point areas of protection. Normalization, grouping and weighting to generate end-point indicators are not covered.

This document covers all the impact assessment categories commonly recognised within the scope of LCA. It mostly adopts the indicator descriptors and definitions of the International Reference Life Cycle Data System (ILCD) Handbook (EC-JRC, 2011) (

Figure 3). Other categories such as noise, nuisance and indoor air quality are not covered, as they are not well developed for use in LCA.

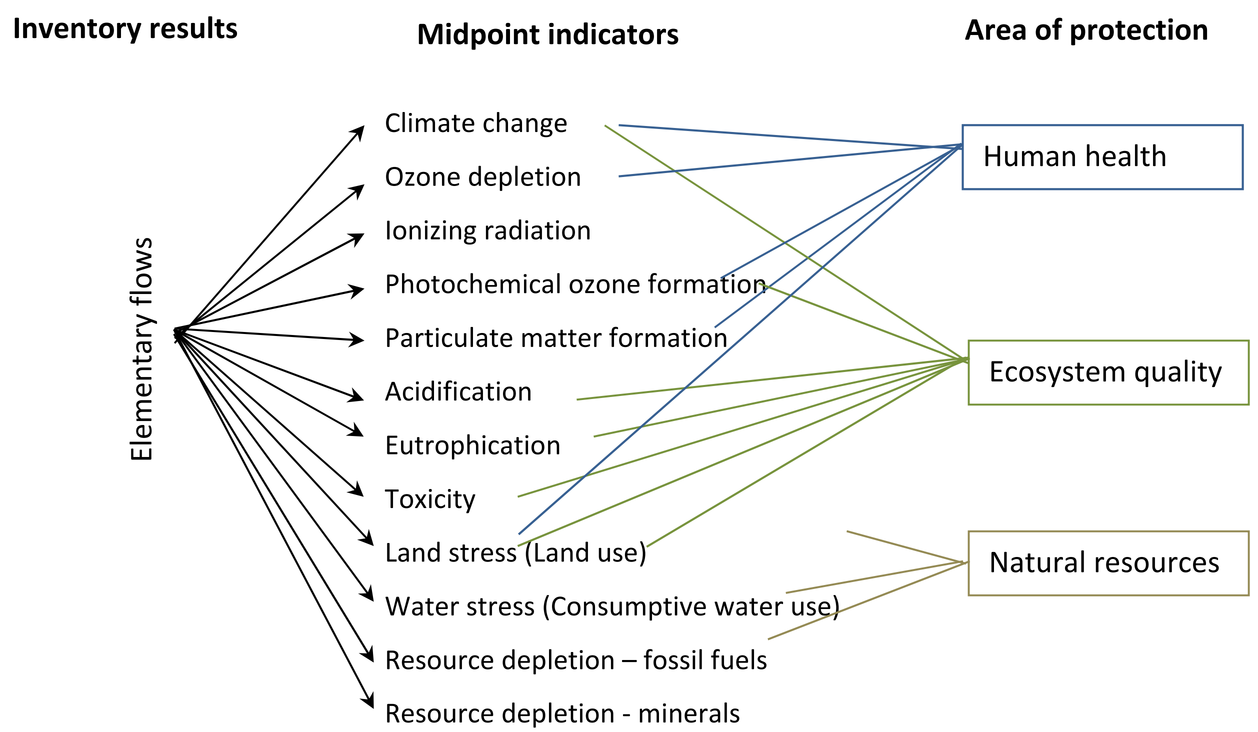

Figure 3 Framework of mid-point and end-point indicators commonly considered in LCA (based on impact pathways described in the LC-Impact method (Huijbregts et al., 2014)

The category of land use has been sub-divided into the biodiversity and ecosystem service aspects of land use, as proposed by land use impact assessment framework developed by the UNEP/SETAC

Life Cycle Initiative (Koellner et al., 2013b). .However as characterization of the eco-system services aspects of land use has not been developed enough to offer guidance, it has not been included in this version of the guide. Only the biodiversity aspects are covered.

2.3 Process

The selection of best practice LCIA methods has been guided by international initiatives and certification schemes that aim to establish harmonisation of methods or recommend best practice.

International consensus / harmonisation efforts:

– **UNEP / SETAC Life Cycle Initiative**

  Development of international consensus on environmental LCIA indicators (Jolliet et al., 2014), including category-specific working groups such as the Water Use LCA (WULCA) working group (Kounina et al., 2013).

Best practice guidance:

– **European Commission's Joint Research Council (EC-JCR) Life Cycle Impact initiative**

  Development of technical guidance that complements the ISO Standards for LCA and provides the basis for greater consistency and quality of life cycle data and methods (Hauschild et al., 2013a), through the International Reference Life Cycle Data System (ILCD) Handbook - Recommendations for Life Cycle Impact Assessment in the European context (EC-JRC, 2011)

– **Impact World+ Framework**

  Regionalized impact assessment covering the whole world, by implementing state-of-the art characterization modelling approaches developed as a joint major update to IMPACT 2002+, EDIP, and LUCAS. Includes characterization models for local and regional impact categories, each of them based on an appropriate spatial scale.

– **LC-IMPACT from a European Commission FP7-funded project**

  This consortium provides a harmonized LCIA methodology and is an outcome of the FP7-funded project LC-IMPACT. It included spatially differentiated information wherever necessary and feasible (http://www.lc-impact.eu/).

The international consensus-building initiatives (UNEP/SETAC and EC-JCR) are Europe-centric, and their recommended methods don't always translate well for Australian processes. However they offer a well-resourced and considered critique of currently available impact assessment methods, and offer a degree of harmonisation which is beneficial in many applications. These initiatives recognise the need for regionalisation to suit different continents, but there has been very limited regionalisation of methods for Australia, and frameworks for enabling regionalised impact assessment for different phases across a product supply chain are yet to be established. Therefore regionalisation is a consideration, but is not yet an overriding requirement for best practice.

Wherever appropriate, the selection of best practice is informed by any international consensus established by the UNEP / SETAC Life Cycle Initiative (Jolliet et al., 2014), and best practices defined in the ILCD Handbook (EC-JRC, 2011). Preference has been given to the former. However as work on the UNEP / SETAC initiative is still ongoing and consensus has not yet been reached on all impact categories, best practices recommended by the EC-JRC process have also been considered.

The guiding principles for selecting best practice are:

1. If there is clear international consensus on a particular method for an impact category of global relevance, then it is prioritised as best practice.
2. Beyond this, recommendations of best practice are based on the judgement of the authors.
3. Consideration is given to the capacity for methods to be regionalised for Australia. For methods that can be regionalised, guidance is provided on the availability of regionalised characterisation factors, and the extent to which regionalisation would influence uncertainty in the results.

3 IMPACT CHARACTERISATION (AT MIDPOINT)

This section provides a summary of best practice mid-point methods, followed by more details of available methods for each impact category, and the rationale for selection of best practice. Characterisation factors for the best practice methods are available from the Resources section of the ALCAS website (www.alcas.asn.au). The order in which the individual impact categories are presented does not reflect the relative importance of the impact categories.

The 'best practice' should be considered as recommended default methods, and aren't intended to replace good LCA practice of selecting the impact assessment method that most suits the study's objectives and scope. It is good practice to compare results of different impact assessment models, so the influence of method choice can be understood.

3.1 Summary of best practice methods

Note: Impact categories with an asterisk (*) are included in EN 15804 and the Australian EPD Programme.

Impact Category	Underlying characterisation model	Unit	Method(s) in which used	Sect-ion
Climate change * (Global warming)	Global Warming Potentials (GWP) for a 100 year time horizon, as per IPCC Forth Assessment Report (IPCC, 2007) [1].	$kgCO_{2\text{-eq}}$	Australian national greenhouse gas assessment methods	3.2
Resource (abiotic) depletion – minerals *	Abiotic depletion of minerals based on concentration of currently economic reserves and rate of de-accumulation (Guinee et al., 2002)	$Sb_{\text{-eq}}$	CML-IA V4.8 August 2016	3.3
Resource (abiotic) depletion – fossil fuels *	Abiotic depletion of fossil fuels based on energy content (lower heating value) (Guinee et al., 2002)	MJ	CML-IA V4.8 August 2016	3.3
Water scarcity	Method of Ridoutt and Pfister (2010), with water stress indices of Pfister et al. (2009)	$m^3\ H_2O_{\text{-eq}}$	NA	3.4
Eutrophication *	Eutrophication potentials based on Heijungs et al. (1992), adopted in CML-IA method (Guinee et al., 2002), which assumes both N- and P-species contribute.	$kgPO_{4\text{-eq}}$	CML-IA V4.8 August 2016	3.5
Acidification *	If assessed, use the change in critical load exceedance, currently based on European characterisation factors (Huijbregts, 1999)	$kg\ SO_{2\text{-eq}}$	CML-IA V4.8 August 2016	3.6
Toxicity – human and freshwater eco-toxicity	USEtox- with regionalised characterisation factors rof Australia, derived based on regionalisation approach of Kounina et al. (2014)	CTUh CTUe	ALCAS Best Practice	3.7
Photochemical ozone formation (oxidation) *	If assessed, use Photochemical Ozone Creation Potentials (POCP)	$C_2H_{4\text{-eq}}$	CML-IA V4.8 August 2016	3.8
Particulate matter formation	Fate and exposure based on Wolff (2000), using the CALPUFF model	$kgPM_{2.5\text{-eq}}$	TRACI V2.1	3.9

(respiratory effects)				
Land use – biodiversity	No best practice identified	-	-	3.10
Land use – ecosystem services	No best practice identified	-	-`	3.11
Ozone layer depletion *	Ozone Depletion Potential (ODP) factors published by the World Meteorological Organisation (WMO, 2011)	kgCFC-11$_{\text{-eq}}$	All methods CML	3.12
Ionizing radiation (human health)	Human health impact model of Frischknecht et al (Frischknecht et al., 2000)	kBq U235$_{\text{-eq}}$	ILCD V1.0.9 May 2016	3.13

1. More recent greenhouse gas characterisation factors have been published in the IPCC Fifth Assessment Report (IPCC, 2013a). However the earlier factors from the Forth Assessment Report are recommended to be consistent with those required in Australian National Greenhouse Gas accounting agreements.

3.2 Climate Change

The impact category of 'climate change' (sometimes referred to as Global Warming[1]) quantifies the impacts of human activities on the climate. The primary impact pathway for human induced (i.e. anthropogenic) climate change through the release of greenhouse gases (GHGs) to the atmosphere. Although climate can also be affected by release of aerosols or black carbon (soot), altering of the surface albedo or changes to cloud cover, LCA studies rarely include climate impacts other than those due to GHG emissions. The GHGs of most importance, and most commonly accounted for, are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). Various hydrocarbon GHGs are also included when data are available. Human activities can also affect climate through the uptake of carbon dioxide into biomass and soils, countering the global-warming effect.

The anthropogenic release of GHGs leads to accumulation of these compounds in the atmosphere, increasing the rate at which energy from the sun is absorbed in the atmosphere and re-emitted as heat. According to the Intergovernmental Panel on Climate Change's (IPCC) Fifth Assessment Report, the atmospheric concentration of CO_2 has increased by 40%, and average temperate has increased by about 0.85°C over the period 1880 to 2012 (IPCC, 2013b). Rising global temperature generates several flow-on effects, such as melting of glaciers and polar ice, sea level rise due to expansion of water in the oceans, changed rainfall patterns causing droughts and flooding, increased incidence of cyclones and other extreme weather events, disruption to ecosystem functions, heat stress in humans and livestock, and ultimately damage to human health and infrastructure, and damage to ecosystems and loss of biodiversity (Figure 4). The areas of protection that climate change relates to are Human Health and Ecosystem Quality (See

[1] We discourage use of this term to describe the impact category as it is the term applied to the characterisation factor used to quantify the impacts.

Figure 3).

The main source of GHG emissions is fossil fuel combustion, which is associated with almost all human activities. It is particularly relevant for electricity and heat generation, transport, agriculture and mining. Thus, climate change is a relevant impact category for all sectors. Furthermore, as climate change is acknowledged to be a critical issue for society, it is the most commonly assessed impact category in LCAs.

The Global Warming Potential (GWP) method developed by the IPCC is widely applied in LCA to assess climate change impact. The IPCC GWP factors are “based on the most up to date and scientifically robust consensus model, with described and calculated uncertainties” Hauschild et al (2013b). GWP factors integrate the radiative forcing of a GHG over a given time frame compared with that of carbon dioxide, and are periodically reviewed and updated by the IPCC. Those published in the IPCC’s Fifth Assessment Report (IPCC, 2013a) are currently applied as midpoint characterisation factors in many characterisation models used in LCA software. However these latest values are not yet reflected in many of Australia’s national greenhouse gas assessment agreements and methods, which current adopt factors from the IPCC’s Forth Assessment Report (IPCC, 2007). Consequently, to be consistent with many Australian protocols, factors from the Forth Assessment Report are recommended. The IPCC publishes GWP factors for 20, 100 and 500 year time horizons, although the 100-year time horizon, which is used for National Inventory Reporting, is used most commonly in LCA.

Global Temperature Change Potential (GTP) is an alternative metric for quantifying climate effects of GHG emissions. GTP is the ratio of change in global mean surface temperature at a chosen point in time due to the GHG in question, relative to that from CO_2. Myrhe et al (2013) provide values for GTP of all the GHGs, over 20, 50 and 100 years. GTP is further along the impact pathway than GWP, so provides results that may be more readily interpreted. GTP values are lower than GWP for GHGs with short lifetimes, such as methane. GTP has greater uncertainty than GWP.

Once emitted, GHGs mix in the atmosphere and the resulting climate change impact is not affected by the location of emissions. Therefore, the same GWP factors can be applied consistently regardless of location. Consequently, there are no challenges to adopting them in Australia.

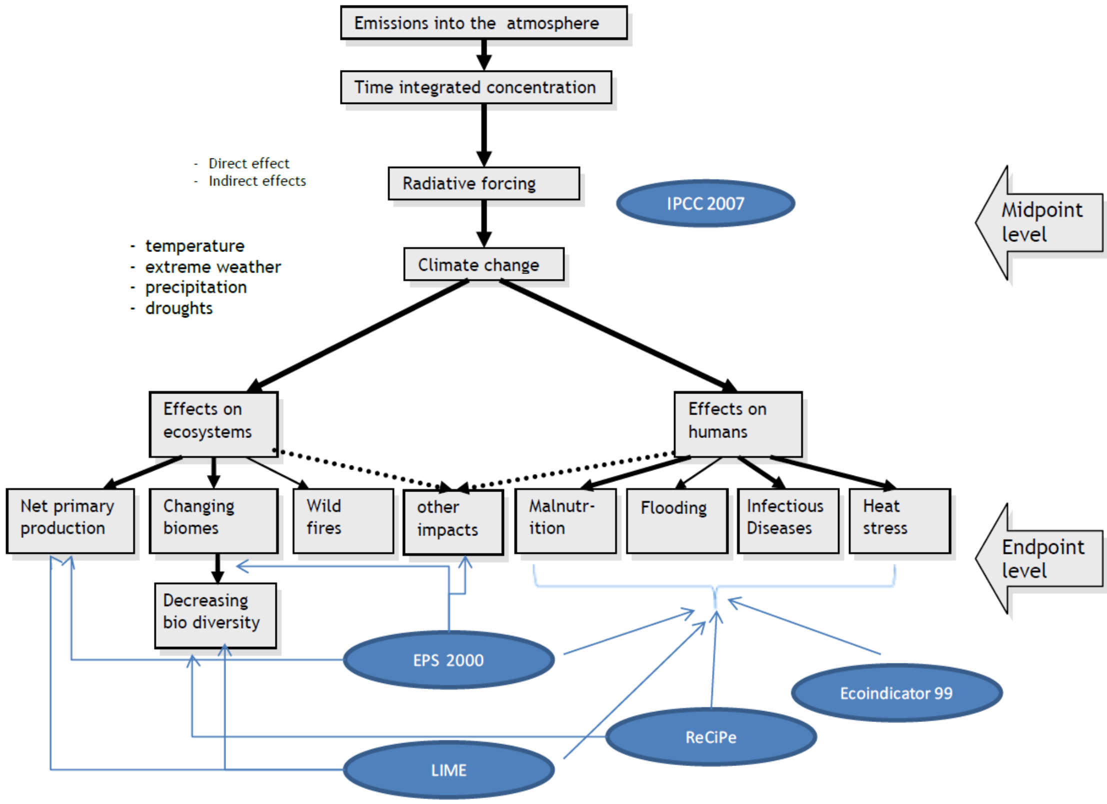

Figure 4 Impact pathways for climate change. Taken from ILCD Handbook (EC-JRC, 2011).

When developing or using life cycle inventories (e.g. AusLCI), care should be taken to separate biogenic carbon emissions (CO_2 and CH_4 from biomass and soil) and carbon emission from fossil sources. Inventories may include substantial emissions of biogenic carbon (e.g. CO_2 from land transformation), which can have a large bearing on the LCIA results. CO_2 emissions from biogenic sources (e.g. those from biofuels) are often assumed "carbon neutral" in LCA studies because they are assumed to be offset by carbon sequestered as the biomass regrows. However, according to international LCA and carbon-footprinting standards (BSI, 2011, ISO, 2013) biogenic GHG flows shall be included in the carbon footprint and also reported separately from the fossil based GHG flows (see Figure 5).

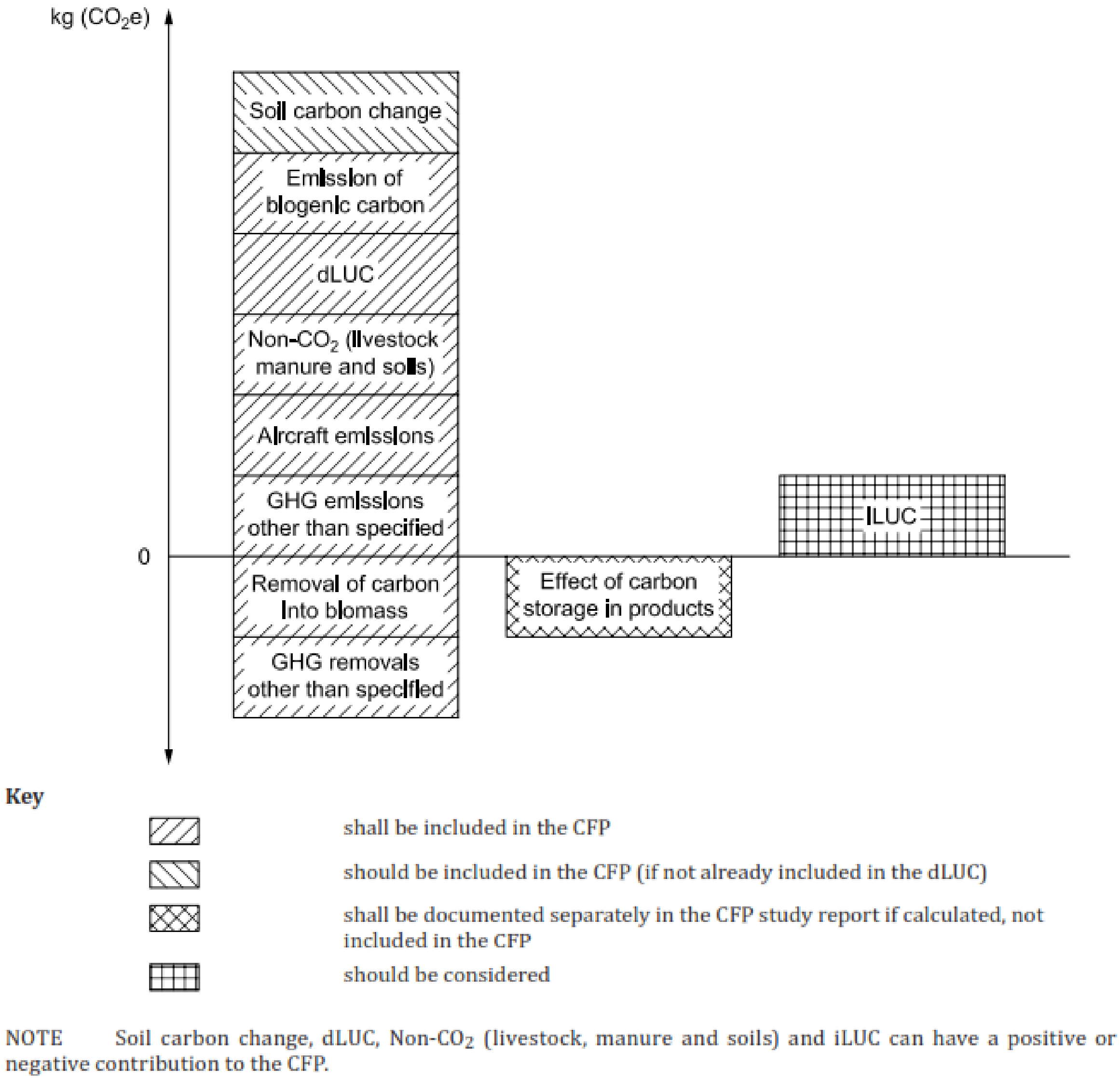

Figure 5 –GHG emissions and removals included in the carbon footprint and reported separately (ISO, 2013)

Conventionally in LCA timing of emissions has not been considered – the emissions are simply summed across the entire life cycle. However, several recent methods quantify the climate benefit of temporary sequestration (such as in wood products) (eg. (Brandão et al., 2013)), or the effect of emissions that are later offset (such as $CO_2$from forest biomass used for bioenergy and subsequently regrown) (eg. (Cherubini et al., 2011)). Carbon footprinting standards (PAS2050 (BSI, 2011) and ISO/TS 14067 (ISO, 2013)) do not permit consideration of timing of emissions in the calculation of carbon footprint, but do allow the effect of temporary storage to be reported as a supplementary figure. PAS2050 provides a method for this based on Clift and Brandão (2008). ISO/TS 14067, while not providing a method, does require that the inventory specifies the timing of

GHG emissions and removals relative to the year of production of the product, to enable the calculation of a supplementary figure that reflects timing of emissions.

Best practice

The current best practice for climate change characterisation at the mid-point is to use the GWP (100 year) factors defined in the IPCC's Forth Assessment Report as default (IPCC, 2007), to be consistent with those applied in many Australian national greenhouse gas assessment agreements and methods.

When the product system includes biogenic carbon flows, ensure that biogenic carbon flows are inventoried and reported separately from fossil based carbon flows (including CO2 from land transformation). Use ISO/TS 14067 for reporting guidance (ISO, 2013).

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

3.3 Resource depletion (fossil and mineral)

The impact category of 'resource depletion', also referred to as 'abiotic depletion' characterises the depletion of natural resources from the Earth. This is usually separated into biotic and abiotic resources. Biotic resources are not covered here but are dealt with under land use where biotic resources are produced, or in biodiversity, where biotic resources are damaged. This leaves abiotic resources which can be broken down into minerals and fuels or minerals and fossil fuels (depending on how you view nuclear fuels). Therefore these aspects of resource depletion can be referred to as 'Abiotic Resource Depletion (ADP)". The area of protection that fossil and mineral resource depletion relates to is Human Natural Resources (See

Figure 3). Methods that characterise resource depletion at the mid-point are summarised in Table 1. Two main approaches used in the different methods are:

- the scarcity approach, which ranks depletion based on the reserve and the current rate of use;
- the damage to reserves approach which is based on the marginal cost increase from depletion of the quality of the reserves

Research is currently underway to extend the marginal cost of mineral depletion after accounting for substitution effects between different minerals. This involves detailed analysis of the uses of rare minerals and how those product systems can change with reduced availability.

Table 1 Summary of resource depletion impact assessment methods

Category name	Method	Units	Approach
abiotic depletion (fossil fuels)	CML-IA	MJ	Based on energy content of the fuel
abiotic depletion (minerals)	CML-IA	g Sb_{eq}	Based on concentration of reserves and rate of deaccumulation.
abiotic depletion (minerals) -economic reserve base	CML-IA	kg Sb_{eq}	Based on concentration of currently economic reserves and rate of deaccumulation.
energy resources	ecological scarcity 2013	UDP	The ratio of annual production to available reserves is used as the basis for the characterization.
mineral resources	ecological scarcity 2013	UDP	The ratio of annual production to available reserves is used as the basis for the characterization.
resources	EDIP 2003	PR2004	Based on primary resource value in 2004
Depletion of reserves	EPS	ELU	economic value of depletion of reserves- (ELU is economic unit)
metal depletion	ReCiPe (now superseded)	kg Fe_{eq}	Damage to resource stock based on increased cost of extraction

fossil fuel depletion	ReCiPe (now superseded)	kg oil_{-eq}	Damage to fossil fuel stock based on increased cost of extraction
Mineral, fossil & ren resource depletion	ILCD method	kg Sb_{-eq}	Based on concentration of reserves and rate of deaccumulation.

There is currently no international consensus on best practice for resource depletion. In the European context the ILCD Handbook (EC-JRC, 2011) recommends the mineral and fossil depletion based on scarcity of resources and the rate of de-accumulation which is in turn recommended by the Product Environmental Footprint (PEF) scheme. Most certification organisations recommend the CML-IA methods.

Best practice

For Australian LCA community best practice recommendation is the CML-IA method as this aligns best with the requirements the Australian EPD scheme. In instances where data is to be submitted to the PEF the ILCD method could be considered.

Aspirational practice is to look at the damage to resource stocks taking account of the substitution of minerals based on major uses.

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

3.4 Water scarcity

Consumptive water use is abstracted water that is no longer available for other uses because it has evaporated, transpired, been incorporated into products and crops, or consumed by man or livestock (EEA, nd). The water scarcity that it can cause is a problem of international concern (WWAP, 2012). Globally, water use has been increasing at more than twice the rate of population growth (Water, 2012), and most withdrawals are in watersheds already experiencing water stress (Ridoutt and Pfister, 2010). The extent of the problem is such that planetary environmental boundaries for freshwater use have been proposed to avert irreversible environmental change (Rockström et al., 2009). The pressure on global freshwater resources arises from the demand for everyday goods and services which use water in their production. The interconnected nature of global economic systems means that water abstraction can occur far from where final consumption occurs. Solutions must be more than improving local water resource management to include sustainable consumption and production. A local initiative to reduce water use can lead to shifting of burden to another location where production increases to meet demand and this can lead to an overall exacerbation in water stress (Huang et al., 2014). The areas of protection that water use relates to are Human Health, Environmental Quality and Natural Resources (See

Figure 3).

The assessment of water use in LCA is guided by a new international standard, ISO 14046 (ISO, 2014), which builds on other core documents in the ISO 14040 series. This standard features two important aspects. Firstly it underscores the importance of regional variations in freshwater availability and the impacts related to local consumptive water use. As such, it advises against the aggregation of water use on the inventory level where different water sources or different geographical contexts are concerned. The scale of geographical resolution should be consistent with the goal and scope of the study. Secondly, in regards to land-based production systems (e.g. agriculture, forestry, reservoirs used for hydropower production), it clarifies that water consumption in these instances is the <u>change</u> in evaporation caused by production-based land-use, not the <u>absolute flows</u> associated with evaporation or evapotranspiration. ISO 14046 (2014) does not prescribe the use of a particular impact assessment model or characterisation factors.

Models to assess consumptive water use and water scarcity has been an important recent development in LCA (Kounina et al., 2013) and continues to be a priority (Jolliet et al., 2014). The range of available models is now rather broad, and includes both midpoint and endpoint assessment methods. Some models are very narrow in scope, such as the model addressing thermal pollution in aquatic freshwater environments (Verones et al., 2010), or the model addressing the effects of consumptive water use on biodiversity in wetlands of international importance (Verones et al., 2013). It could be said that many of the new models have not yet undergone significant testing and application and could possibly be best described as still in the experimental phase of development.

To date, the most widespread approach has been to use the Water Stress Index (WSI) of Pfister et al. (2009). The WSI can be used as a characterisation factor to calculate a midpoint indicator Water Deprivation (Pfister et al., 2009). Alternatively, local WSI values can be divided by the global average WSI and used as characterisation factors whereby the result is reported relative to an equivalent volume of water consumption at the global average WSI (i.e. H_2O equivalent; Ridoutt and Pfister (2010)). This latter approach has been adopted by the European Food Sustainable Consumption and Production Roundtable in the ENVIFOOD Protocol (EC-Food SCPRT, 2013).

The WSI is a function ranging from 0.01 to 1 based on the local long-term freshwater withdrawal-to-availability ratio with adjustments for monthly and annual variability of precipitation and water storage capacity. The preference is to use WSI values at the watershed level, which are available in Google Earth (www.ifu.ethz.ch/ESD/downloads/EI99plus). If the specific watershed is not known then WSI values at the country level could be used for initial screening of significance (contained in Pfister et al (2009)). The thinking behind the WSI is that greater potential for environmental harm exists when consumptive water use occurs in locations of high water stress compared to locations of low water stress. Various alternative indices have been proposed (e.g. Boulay et al. (2011) and Hoekstra et al. (2012)); however, they have been less widely adopted in LCA and have been shown to be highly correlated with the Pfister et al. (2009) index (Boulay et al., 2015b).

One criticism of the WSI-based approach to impact assessment is the lack of coherence with indicator results obtained using endpoint models which assess the impacts on human health and ecosystem quality separately. The differences are largely due to the absence of socio-economic factors in the WSI. In some regions, the human health impacts of water stress are moderated by the importation of water intensive goods, especially food, as well as investments in alternative water supply infrastructure (e.g. desalination) and technologies that increase water use efficiency. For example, Belgium and Peru have almost identical WSI values (0.715 and 0.716 respectively) but differ significantly in terms of endpoint characterization factors for water use impacts on human health (000E+00 and 6.53E-7 DALY/m^3 respectively; Pfister et al., (2009)). It is now widely accepted that there is no common midpoint in the cause-effect chains of damages to human health and ecosystem quality from consumptive water use. One solution would be to develop separate indicators for human health and ecosystem quality, as is the case with human- and eco-toxicity. However, this approach is not consistent with the market demand for a single indicator result relating to consumptive water use and the resulting water scarcity. Another solution has been proposed by Ridoutt and Pfister (2014), whereby a regionalized WSI dataset ($WSI_{HH,EQ}$) was developed based on the normalization and weighting of endpoint model results (www.ifu.ethz.ch/ESD/downloads/WSI_HH_EQ.kmz). However, this approach is limited by the current state of maturity of endpoint models and the limited acceptability of weighting by some stakeholders. A task group working under the UNEP SETAC Life Cycle Initiative is seeking to develop a consensual water scarcity indicator for use in routine assessment and in situations leading to environmental labels and declarations (Boulay et al., 2015a). However, this consensus indicator is not expected to be finalised until 2016.

Best Practice

Until such time as a new consensus water scarcity indicator emerges and becomes established in practice, it is suggested to follow the recommendation of the European Food Sustainable Consumption and Production Roundtable by adopting the method of Ridoutt and Pfister (2010). Earlier collective efforts to characterise best practice LCA impact assessment models are too old to capture recent model developments and practices related to consumptive water use and are considered to be outdated in this regard (EC-JRC, 2011, Finnveden et al., 2009). In applying the method of Ridoutt and Pfister (2010), it is important to recognize limitations of the underpinning the WSI of Pfister et al. (2009). This index is based on the long-term withdrawal-to-availability ratio covering the period 1961 to 1990. As such, the index may over- or under-report water stress in regions where water demands have changed significantly. Following on from this, the index does not take account of future potential impacts of climate change on water availability. The index also does not differentiate different sources of freshwater within a watershed (such as ground and surface water, which may differ significantly in the level of sustainable use). In addition, the index is dependent on the level at which watersheds are aggregated. That said, the equations used to compute the WSI are transparently reported and there is no barrier to the calculation of customised WSI values based on local data parameters. This could be justified in LCA studies where water use impacts are highly significant and the value of the WSI is influential in determining study conclusions. An updated WSI dataset for Australia, based on national water use statistics, could also be produced. A global normalisation value can be found in Ridoutt et al. (2014).

Characterisation factors

Characterisation factors for the best practice method, i.e., using the method of Ridoutt and Pfister (2010) with water stress indices (WSI) of Pfister et al. (2009), are provided in the accompanying ANNEX spreadsheet. In this method WSI values for a particular catchment / region are divided by the global average WSI value to generate characterisation factors that are relative to an equivalent volume of water consumption with average global water stress (H_2O equivalent). A characterisation factor less than 1 indicates a relative water use stress that is less than the global average, and greater than 1 indicates a relative water use stress that is greater than the global average. Characterisation factor are provided for different spatial scales - global, national, Australian states and Australian regional catchments. The underlying WSI values at the sub-catchment level were sourced from spatial WSI values provided as Google Earth data layer by ETH, Zurich (www.ifu.ethz.ch/ESD/downloads/EI99plus). The global, national and state factors were derived directly from Pfister et al. (2009). Factors for regional catchments were derived for catchment boundaries which provide an appropriate level of resolution (Figure 6). WSI values for sub-catchments were derived from spatial WSI values provided as Google Earth data layer by ETH, Zurich (www.ifu.ethz.ch/ESD/downloads/EI99plus).

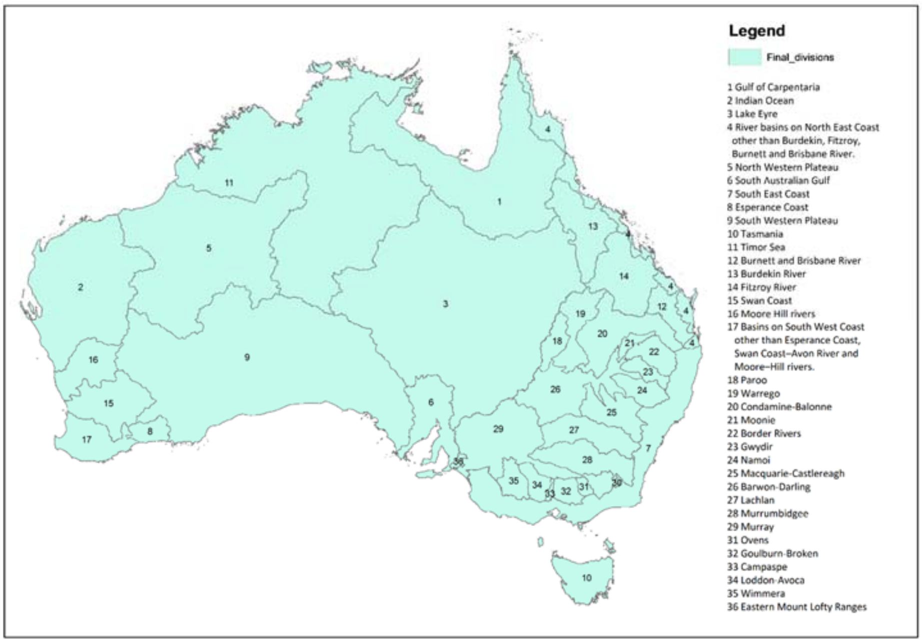

Figure 6– Catchment boundaries for water scarcity impact characterisation

3.5 Eutrophication

The impact category of 'eutrophication' characterises the eutrophying impacts when macro-nutrients are released to air, water and soil. The macro-nutrients most commonly accounted for are nitrogen (N), phosphorous (P) and organic compounds (BOD[2]). Eutrophication (also known as nutrification) can occur in aquatic and terrestrial environments, but the former is more commonly a problem. When macro-nutrients find their way to water (aquatic eutrophication) it can lead to accelerated algae growth, reduced sunlight infiltration and oxygen depletion, which can ultimately lead to changes in species composition. Releases to land (terrestrial eutrophication) can increase susceptibility of plants to diseases and pests potentially also leading to changes in species composition, for example encouraging weeds. The end-point area of protection that it relates to is the Ecosystem Quality (see

Figure 3).

Eutrophication is an important impact category for processes involving the use or mobilisation of nutrients (agricultural cropping and pastures) or disposal of wastes with high content of organic compounds (livestock production, food processing, pulping, urban solid waste and wastewater treatment and disposal etc.). However fuel combustion (for electricity and transport) and other processes that release nitrogen compounds to air (eg. nitrogen oxides from fuel combustion, volatilisation of ammonia from fertiliser and manure, etc.), also contribute to eutrophication. So it is of relevant to most production systems.

Terrestrial eutrophication is considered less important than aquatic eutrophication in the Australian context, due to the generally nutrient-limited status of Australian soils, and low population densities. However there are some hotspots where soils have become overloaded with nutrients

[2] Organic compounds are measured and expressed as biochemical oxygen demand (BOD) or chemical oxygen demand (COD).

through intensive agriculture. For Australia-centric supply chains there may be justification for only considering aquatic eutrophication. However for more global supply chains its inclusion is warranted if processes known to contribute to terrestrial eutrophication are present.

Eutrophication impact assessment considers the cause-effect chain that leads to the eutrophying effects of macro-nutrients (see Figure 7). Other impact pathways for nutrients are captured by other impact categories, such as when converted to a greenhouse gas contributing to climate change (eg. nitrous oxide N_2O), or when contributing to toxicity, health impacts or acidification (eg. nitrogen oxides NO_x and ammonia NH_4).

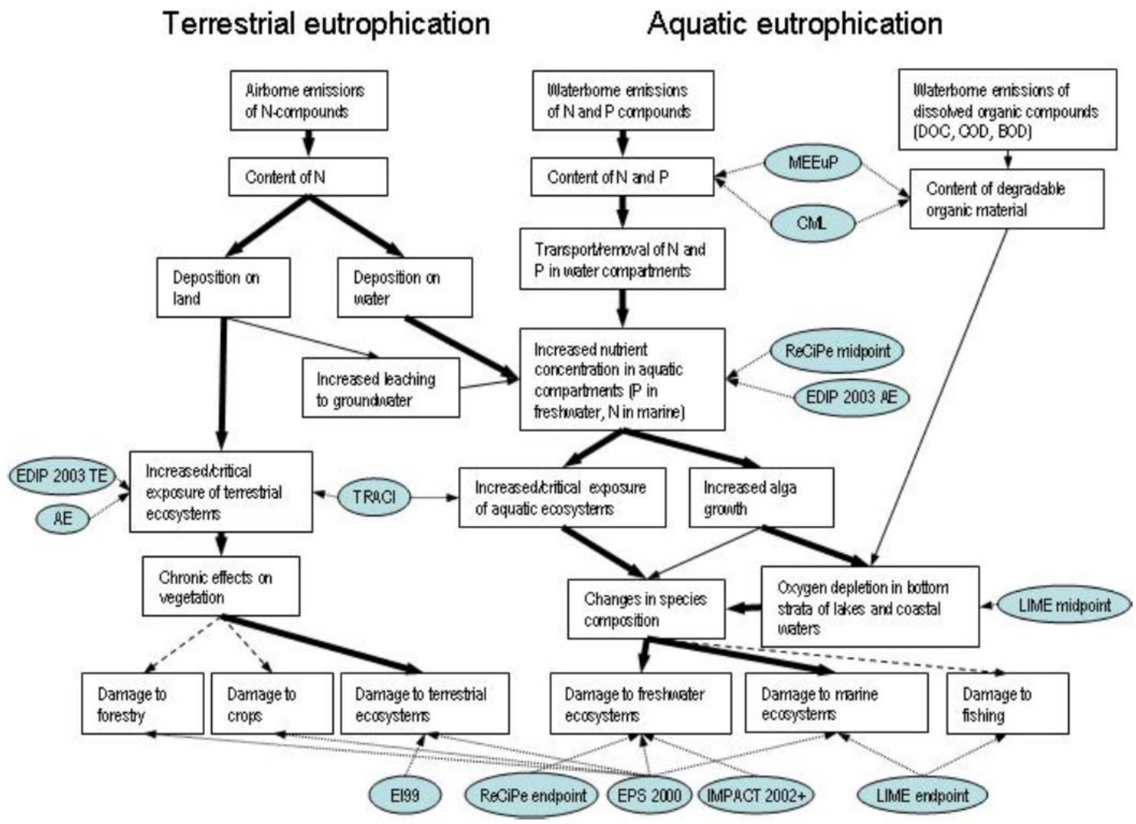

Figure 7 Impact pathways for eutrophication. Taken from ILCD Handbook (EC-JRC, 2011)

Methods that characterise aquatic and terrestrial eutrophication at the mid-point are summarised in Table 2. Many are derived from ‘stoichiometric nutrification potentials’, in which the potential of an emission to nutrify is based on the stoichiometry of macro-nutrient uptake in biomass (algae). This model is applied in its most basic form in the CML-IA and EDIP97 methods to capture both aquatic and terrestrial eutrophication under a single ‘eutrophication’ indicator, with no distinction between N-limited (marine) and P-limited (freshwater) receiving environments. IMPACT2002+ refines this slightly by focusing on aquatic eutrophication only and distinguishing between P-limited impacts and N-limited impacts[3]. Distinction between freshwater (generally P-limited) and marine (generally N-limited) waters is seen as important for aquatic eutrophication (EC-JRC, 2011) because responses to nutrient influxes are different so the degree of impact may differ depending on whether nutrients are emitted to fresh or marine catchments.

Other methods have refined characterisation factors for aquatic eutrophication by applying fate-exposure factors to reflect the probability of transport to an aquatic environment to which it is the limiting nutrient. These have been developed in the context of Europe (EDIP2003, ReCiPe Midpoint – now superseded), North America (TRACI and LUCAS) and Japan (LIME).

Table 2 Summary of eutrophication impact assessment methods (adapted from (EC-JRC, 2011))

Context	Method	Aquatic eutrophication model	Terrestrial eutrophication model
General	CML-IA	Stoichiometric nutrification potentials applied to represent both aquatic and terrestrial eutrophication, assuming the receiving environment is both P- and N- limited	
	IMPACT2002+	Stoichiometric nutrification potentials, distinguishing between P-limited and N-limited impacts.	
Europe	-		Accumulated exceedance (AE) model
	EDIP	Based on stoichiometric nutrification potentials, but with fate / exposure factors from CAMEN (for waterborne emissions) and RAINS (for airborne emissions), and distinguishes between P-limited and N-limited impacts.	Increase in area of terrestrial ecosystem exposed above critical load for N using the RAINS model
	ReCiPe Midpoint (now superseded)	Based on stoichiometric nutrification potentials, but with fate / exposure factors from EUTREND, and distinguishes between P-limited and N-limited impacts.	
Japan	LIME	Oxygen depletion due to increase in nutrients and BOD/COD for Japanese closed marine environments, not freshwater.	
North America	TRACI / LUCAS	Based on stoichiometric nutrification potentials, but with transport factors.	

There is currently no international consensus on best practice for eutrophication. The UNEP/SETAC process has noted freshwater eutrophication as a good candidate for harmonization, but final

[3] In LCA software only the P-limited (freshwater) characterisation factors may be included

outcomes may not evolve until 2017 (Jolliet et al., 2014). In the European context the ILCD Handbook's recommendation for aquatic eutrophication is the midpoint method used in ReCiPe based on a model developed by Struijs et al., (2009), which distinguishes freshwater and marine eutrophication (EC-JRC, 2011). For terrestrial eutrophication it recommends the accumulated exceedance model (Seppälä et al., 2006, Posch M, 2008), which is based on the amount of emissions above an estimated critical load rather than the total amount of emissions.

The method most commonly adopted by certification organisations is the CML-IA method, which is based on the afore-mentioned 'stoichiometric nutrification potentials'. It captures emission to air, water and soil, but does not distinguish between aquatic and terrestrial eutrophication, and hence generates a single indicator of eutrophication potential ($kgPO_{4eq}$), which assumes that both N- and P-species contribute to eutophication. The Product Environmental Footprint (PEF) Guide is guided by the ILCD Handbook and hence recommends the ReCiPe mid-point method for aquatic (freshwater and marine) eutrophication, and the accumulated exceedance model for terrestrial eutrophication, effectively resulting in three separate indicators (kgN, kgP and AE).

The challenge for applying any of these models to Australian processes is that underlying characterisation factors have not been validated in the Australian context. It is not clear whether they adequately represent the eutrophication responses of nutrients in aquatic systems outside Europe. Hence there is currently no capacity for regionalising impact assessment in Australia.

Best practice

Current best practice is to generate a single indicator of eutrophication potential ($kgPO_{4eq}$) based on the 'stoichiometric nutrification potentials' as applied in CML-IA, which assumes both N- and P-species contribute to eutrophication. For Australia-centric supply chains this is appropriate in the absence of regionalised factors based on fate-exposure models. However results should note the limitations of the method and estimate the uncertainty that non-regionalisation creates. For instance, if it is know that the receiving environment is N-limited, yet both N- and P-species are accounted for, the potential for overestimation of impacts should be noted. For more global supply chains (not dominated by European processes) then a similar reasoning applies as regionalisation has not occurred in many regions. An exception is North America, where there has been regionalisation of stoichiometric nutrification potentials in the US (TRACI method) and Canada (LUCAS method). For supply chains dominated by European processes, the practitioner may consider applying the ILCD Handbook recommendations noted earlier. Aspirational practice would be to regionalise the best practice methods recommended in the ILCD Handbook with characterisation factor representative of nutrient responses in Australian aquatic systems.

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

3.6 Acidification

The impact category of 'acidification' quantifies the acidifying impacts when acid precursor compounds are released to air and subsequently deposited on land or water. The substances most commonly accounted for are nitrogen oxides (NOx), sulphur oxides (SOx), sulphuric acid and ammonia. When these are emitted to air they react with moisture in the atmosphere to form acidic compounds (such as nitric acid, sulphuric acid, etc.), and subsequently deposit in terrestrial and aquatic environments.

When acidic compounds deposit on land (terrestrial acidification) it can reduce soil pH (making it acidic) which leads a decline in richness of vascular plants (Huijbregts et al., 2014). The end-point area of protection that is effected is Ecosystem Quality (see Figure 3). Most impact characterisation models focus on terrestrial acidification.

Terrestrial acidification precedes aquatic acidification, and so inland waters are only acidified after the acid neutralising capacity of the watershed has been depleted (EC-JRC, 2011). Only a few LCIA methods (EDIP97 and the CML-IA) also cover waterborne emissions. However the ILCD notes that these methods are not sufficiently developed, and in fact consider that aquatic acidification should be a separate impact category (EC-JRC, 2011).

There are a number of acidification processes that not part of the impact pathway captured by this category:

- Ocean acidification caused by the uptake of carbon dioxide (CO_2) from the atmosphere leading to the decrease in ocean pH;
- Aesthetic impacts caused by acid corrosion on human-built structures (e.g. buildings and statues;
- Acidification of agricultural soils caused by imbalances in the soil N chemistry caused mostly by the addition of ammonia-based synthetic fertilisers (Peters et al., 2011).

Acid precursor substances based on N that are part of the acidification impact pathway (i.e. NOx) are also associated with the impact pathways of eutrophication (see Section 4.4) and climate change (as NOx can be re-released as nitrous oxide (N_2O) after deposition) (see Section 4.1). The acidification impact category only considers the impact pathways that lead to the acidification (Figure 8).

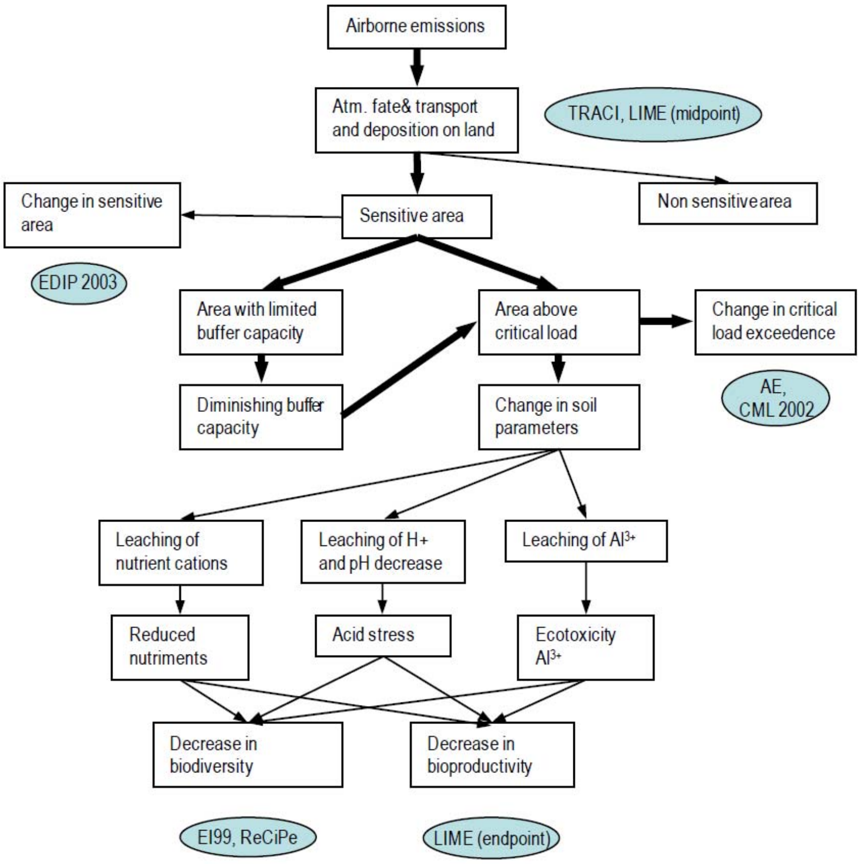

Figure 8 Impact pathways for acidification. Taken from ILCD Handbook (EC-JRC, 2011)

Acidification is a relevant impact for processes releasing NOx and SOx, acidic gases and ammonia. NOx and SOx, the species that tend to contribute most of acidification impacts, are released from the burning of fossil fuels in electricity, steam and heat generation. So any fossil-energy consuming process in a product life cycle may have acidification implications. For example, acidification has been a problem in regions with intensive industrial activities releasing high levels of acid precursors to surrounding air sheds, and causing what has been referred to as 'acid rain'. There have been examples of these impacts on the forests of northern Europe and North America.

Acidification has not been considered an important impact category for processes occurring in Australia, as the accumulated releases of acid precursors has not been high enough to exceed critical loads and cause impacts. However for supply chains with fossil-energy consuming processes occurring elsewhere, the assessment of terrestrial acidification is likely warranted.

Methods that characterise acidification at the mid-point are summarised in

Table 3.

Table 3 Summary of acidification impact assessment methods (adapted from (EC-JRC, 2011))

Context	Method	Acidification model
Europe	Accumulated exceedance (AE) model	Change in critical load exceedance method based on the increase in area of terrestrial ecosystem exposed above critical load for N (Posch M, 2008, Seppälä et al., 2006). It provides European country-specific characterisation factors.
	CML-IA	Change in critical load exceedance method based on the Hazard Index (HI) method, in which the factors represent the marginal change in HI from the actual load relative to a critical load. It provides European country-specific characterisation factors (Huijbregts, 1999)
	EDIP	European country-specific characterisation factors, using the Unprotected Area (UA) method.
	ReCiPe (now superseded)	This is a base saturation method which considers soil sensitivity, and is an alternative to critical load based method.
	MEEUP	A model employed in EC legislation that considers acidification potential in terms of H+ releases without addressing the fate of chemicals in air and soil.
Japan	LIME	Increase in H+ deposition per unit area due to an acidifying chemical.
North America	TRACI	Provides generic and spatially differentiated characterisation factors for the US, based on the fate model ASTRAP

There is currently no international consensus on best practice for acidification. The UNEP/SETAC process has noted acidification as a good candidate for harmonization, but final outcomes may not evolve until 2017 (Jolliet et al., 2014). In the European context the ILCD Handbook recommends the accumulated exceedance (AE) model (Seppälä et al., 2006, Posch M, 2008), because it is used in European policy and distinguishes between sensitive and non-sensitive areas. It is based on the amount of emissions above an estimated critical load rather than the total amount of emissions. The CML-IA method is also a critical load exceedance method, but the data and models are not as up to date as the AE model.

Most certification organisations recommend the CML-IA method. The Product Environmental Footprint (PEF) scheme adopts the recommendations of the EC-JRC by requiring use of the Accumulated Exceedance (AE) method.

The challenge for applying any of the noted models to supply chains containing Australian processes is that is that their underlying characterisation factors are not necessarily valid in

Australian context. As noted earlier, the critical loads above which acidification is problematic may not be reached for releases occurring in Australia. So methods based on critical loads in Europe and elsewhere may overestimate impacts in Australia. There is currently no capacity for regionalising acidification methods for Australia. Impact World I Plus is working on regionalised impact method which likely show would how little impact Australian acidification emissions have. However this method is still under development.

Best practice

For Australia-centric supply chains acidification could be excluded due to the lack of any clear evidence of this issue in Australia. Acidification impacts from airborne emissions required air shed modelling and models for receiving environment condition. This has not been undertaken for Australia, but the empirical evidence is that there is no significant acidification from airborne emissions at this point. Where LCA studies involve global supply chains the acidification should be included using the CML-IA method. This is considered appropriate in the absence of regionalised characterisation factors for Australia, and it aligns best with the requirements the Australian EPD scheme. In instances where data is to be submitted to the PEF the ILCD method using cumulative exceedance could be considered.

Aspirational practice would be to use the Accumulated Exceedance method once it was more accessible in impact assessment methods.

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

3.7 Toxicity – human- and eco-toxicity

Life cycle impact assessment (LCIA) of toxicity takes into account the fate, route of exposure and toxicity impact of toxic substances when released to air, water or land. Categories of chemical substances commonly accounted for are pesticides, heavy metals, hormones and organic chemicals.

The human toxicity impact category captures the adverse effects of chemicals on human health, including both carcinogenic and non-carcinogenic impacts. The impact of a chemical will vary significantly depending on impact pathway, fate and exposure to humans (Hauschild et al., 2011). Eco-toxicity includes the effects of emissions on the environment in particular on individual species. Eco-toxicity uses the same fate modelling framework as human toxicity however the exposure pathways and endpoints are different as shown in Figure 8.

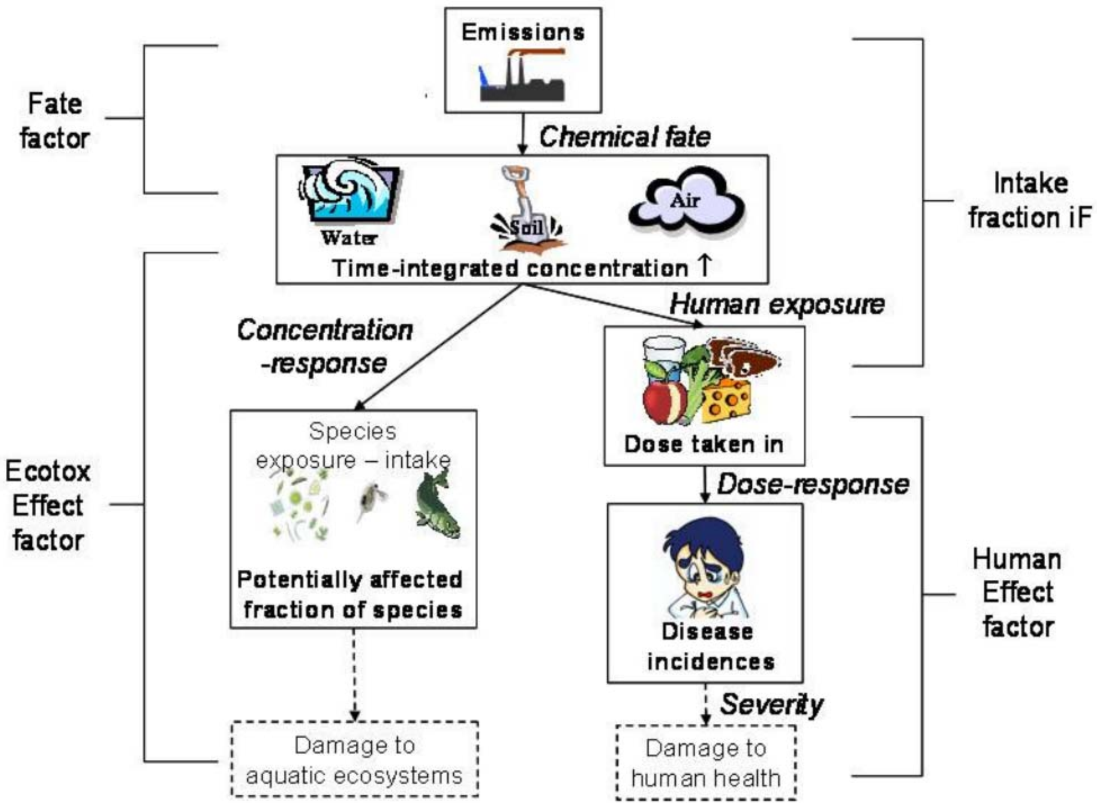

Figure 9 Fate and effect modelling steps for human and eco-toxicity (USEtox manual (Hjubregts et al, 2010))

These fate-exposure models make it possible to estimate ecotoxic effects on freshwater, marine and terrestrial environments. However, the majority of available ecotoxicological effect data is for freshwater species, hence these are traditionally used in the ecotoxicity models for all three environments (freshwater; marine; terrestrial). Some methodological development research is underway to reduce the uncertainty created by this assumption that freshwater ecosystem effects are appropriate for the other environments. Nonetheless, limitations remain to using the available marine and terrestrial ecotoxicity impact categories.

There are three general sources of uncertainty to consider when choosing a toxicity impact assessment model for use in LCA:

1. The inherent uncertainty in the data and biophysical relationships underpinning the fate-exposure-effect models.
2. The high variability of results across different chemical species, introducing uncertainty if the chosen model does not include appropriate coverage of the species of concern to any particular study.
3. The sensitivity of the fate-exposure results to geographic and demographic variability across different regions of the world.

Some examples of models include TRACI, IMPACT 2002+, LIME, USES-LCA, USES-LCA 2.0, CalTOX and USEtox. USEtox (Hjubregts et al, 2010) is recommended by the ILCD Handbook as the default LCIA method for characterisation of human and eco-toxicity impacts, and with foundations within the UNEP-SETAC Life Cycle Initiative it is the only model endorsed by an authoritative international body (EC-JRC, 2011).

Prior to the relatively recent implementation of USEtox, the previous versions of this guide and the Building Product Innovation Council (BPIC) LCIA method (Bengtsson and Howard, 2010) had recommended the USES-LCA 2.0 model with Australian specific characterisation factors adapted by Lundie et al. (2007) for human toxicity and ecotoxicity impact assessment. The benefit of these adapted characterisation factors was increased geographic specificity to models primarily developed for the European climate, environment and population densities. A report by Hjuibregts and Lundie (2002) highlights the uncertainty of toxic impacts from substance emissions in differing locations with finding that substances emitted to agricultural soil in Australia were 160 times less likely to lead to human exposure than those in Western Europe. The human exposure factor for substances emitted to air, freshwater and seawater were found to be 20 times lower.

Despite the obvious importance of geographic specificity when applying human exposure factors, the use of such an Australian specific characterisation leads to a trade-off due to reduced comparability with international LCA studies. The application of a geographically specific characterisation is also only relevant when substance emissions are known to occur within the characterised region. With modern global production systems and a shrinking Australian manufacturing industry, it is likely that substance emissions within an Australia products lifecycle occur outside Australia or in an unknown location. According to the ILCD Handbook recommendations, spatial variations in toxicity impact may be "partially cancelled out by other factors, such as having multiple sources of emissions or may be negligible relative to other sources of uncertainty/variation for many contaminants" (EC-JRC, 2011).

While the ILCD Handbook states that USEtox has almost full compliance with their scientifically-based compliance criteria, they note that ReCiPe (based on USES-LCA 2.0) and IMPACT 2002+ LCIA methods complied with essential criterial and TRACI (based on CalTOX) had a good science-based criteria compliance (EC-JRC, 2011). Yet although these models offer an alternative, USEtox should be preferred as it is a global consensus model which involved key toxicity model developers of CalTOX, IMPACT 2002, USES-LCA, BETR, EDIP, Watson and EcoSense (Rosenbaum et al., 2008).

This scientific consensus model resulted from careful comparison of several existing toxicity characterisation models (Rosenbaum et al., 2008). With over 1, 250 human toxicological characterisation factors, it offers the most up-to-date and extensive coverage of chemical substances available (Hauschild et al., 2013a). Despite being the most accurate LCIA characterisation method available the ILCD Handbook (EC-JRC, 2011) notes LCIA results for human toxicity and ecotoxicity have high uncertainty relative to other impact categories, such as global warming potential with low uncertainty at the midpoint characterisation level.

The main strength of USEtox characterisation modelling is the focus on getting the relative importance of substances right (Hauschild et al., 2013a). Unlike some methods attempting to quantify only endpoint impact, for example in Disability Affected Life Years (DALY), the Comparative Toxic Unit for humans (CTU_h) USEtox indicator gives a midpoint impact assessment for relative potential human toxicity. Although classified as a midpoint impact assessment method, Hauschild et al (2013) note that there is no true midpoint to assess with toxicity impacts due to the heterogeneity of impact pathways and effect factors across substances. The arbitrary selection of a toxicity impact pathway midpoint can be seen below in Figure 10, showing the position of LCIA output metrics through the human toxicity impact pathway.

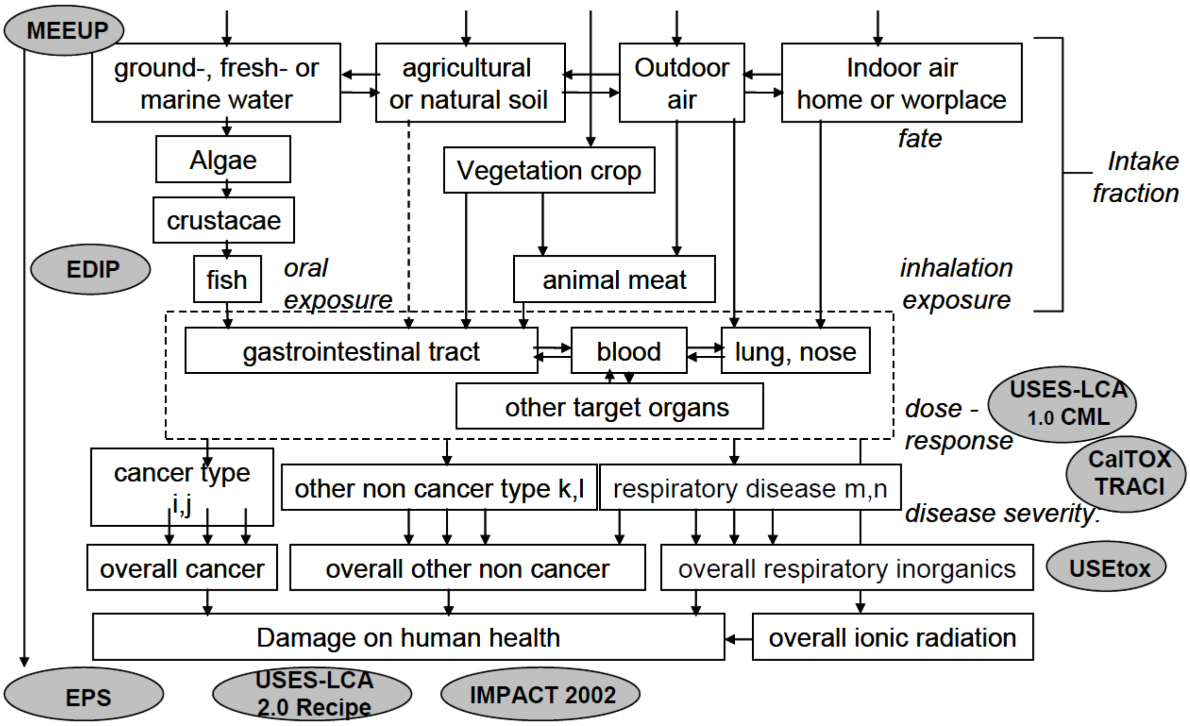

Figure 10 - The human toxicity impact pathway. Taken from ILCD Handbook (EC-JRC 2011)

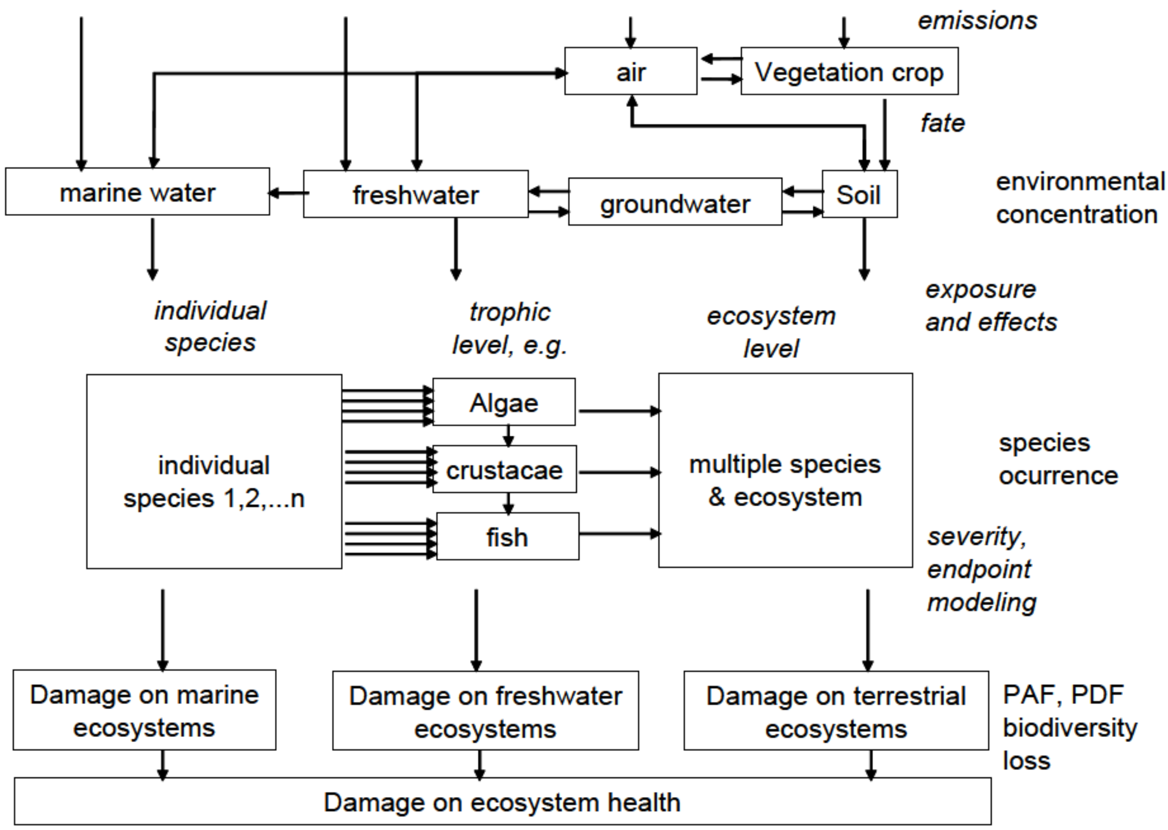

Figure 11 - The eco-toxicity impact pathway. Taken from ILCD Handbook (EC-JRC 2011)

Similarly, for eco-toxicity a midpoint indicator, comparative toxicity unit equivalent (CTUe/kg), provides an estimate of the potentially affected fraction of species (PAF) integrated over time and volume per unit mass of a chemical emitted (PAF m3.day/kg) (Henderson et al., 2011). Recent developments in the Impact World methodology have produced regionalised fate models that accounts for both local emission exposures and transportation of pollutants between different regions. As part of this, a set of USEtox factors have been developed for the Australian region (Kounina et al., 2014). Similar factors are available for all other regions of the world and are the beginnings of a move to regionalised impact assessment in LCA. While this is a great step forward for Australian LCA, it is unclear whether the level of geographic disaggregation possible in USEtox will be able to adequately represent Australia's extreme spatial heterogeneity. If not, this will limit the usefulness of USEtox based analysis that considers the relative significance of emissions in different parts of Australia.

For the characterisation of endpoint impacts the ILCD recommendations are to adapt the USEtox midpoint characterisation by applying average severities for cancer (11.5 DALY/case) and non-cancer (2.7 DALY/case) diseases. These factors are however classified as interim and not ready for recommendation (class I, II or III) classification (Hauschild et al 2013).

Best Practice

For LCA studies in with the majority of processes occur in Australia we recommend the use of USEtox method with regionalised factors for Australia. This is based on the impact regionalisation approach of Kounina et al (2014), which was used to generate the regionalised factors for Australia developed by Tim Grant and Olivier Jolliet . This method includes impact characterisation for freshwater ecotoxicity, human toxicity – cancer and human toxicity – non-cancer.

For LCA studies where the majority of processes are outside Australia, or their location is not known, we recommend the use of the global USEtox factors published by (Hauschild et al 2013). For LCA studies where the inclusion of marine and/or terrestrial ecotoxicity indicators is important, we recommend a two-step process:

(i) The global version of USES-LCAv2 be used as the default for all four toxicity-related impact categories (human toxicity; freshwater, marine & terrestrial ecotoxicity)

(ii) The human toxicity and freshwater ecotoxicity results are also calculated using USEtox models, to check whether this would change the conclusions that are drawn.

For increased transparency we recommend the use of these indicators at the midpoint level as the modelling through to the endpoints is poorly developed at this stage.

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

3.8 Photochemical ozone formation (photochemical oxidation)

The impact category of 'photochemical ozone formation' (also referred to as photochemical oxidation), quantifies the impacts from increases in ozone concentrations in the troposphere (12-20km above the Earth's surface), which is formed as a secondary contaminant from the oxidation of the primary contaminants (volatile organic compounds (VOC) or carbon monoxide) in the presence of nitrogen oxides (NOx) and under the influence of light. It can be given a number of different names, including ozone formation, photochemical ozone formation or creation, photo oxidant formation, photo smog, or summer smog. Ozone in the troposphere can be called 'ground level ozone' to distinguish it from stratospheric ozone, which is the focus of the ozone depletion impact category (Section 3.12).

Ozone is a powerful oxidizing agent readily reacting with other chemical compounds to make many possibly toxic oxides. Photochemical and chemical reactions involving ozone occur naturally in the troposphere. However at high concentrations, brought about by human activities, it is a pollutant and a constituent of smog. The primary contaminant most commonly accounted for are volatile organic compounds (VOC), carbon monoxide (CO) and nitrogen oxides (NOx), with the most common source being incomplete combustion of fossil fuels, such as gasoline, diesel, in internal combustion engines.

This impact category considers the impact pathways that lead to the effects of increased tropospheric ozone concentrations on humans and vegetation (Figure 12). At high concentrations it is hazardous to human health, including irritation of the respiratory system and aggravation of asthma. At lower concentrations it causes damage to vegetation. Therefore the areas of protection it relates to are human health and ecosystem quality (see

Figure 3).

Tropospheric ozone (and the smog it causes) can be an important problem at a regional scale in densely-populated or highly-industrialised areas (such as large cities in Asia, Europe, and North America), or where the topography traps pollutants. However it is not considered an important impact category for processes occurring in Australia, as accumulation of ozone is not commonly very high and lower population densities means hence exposure is lower. However for supply chains involving processes with significant fuel combustion in impact-prone regions, its assessment may be warranted.

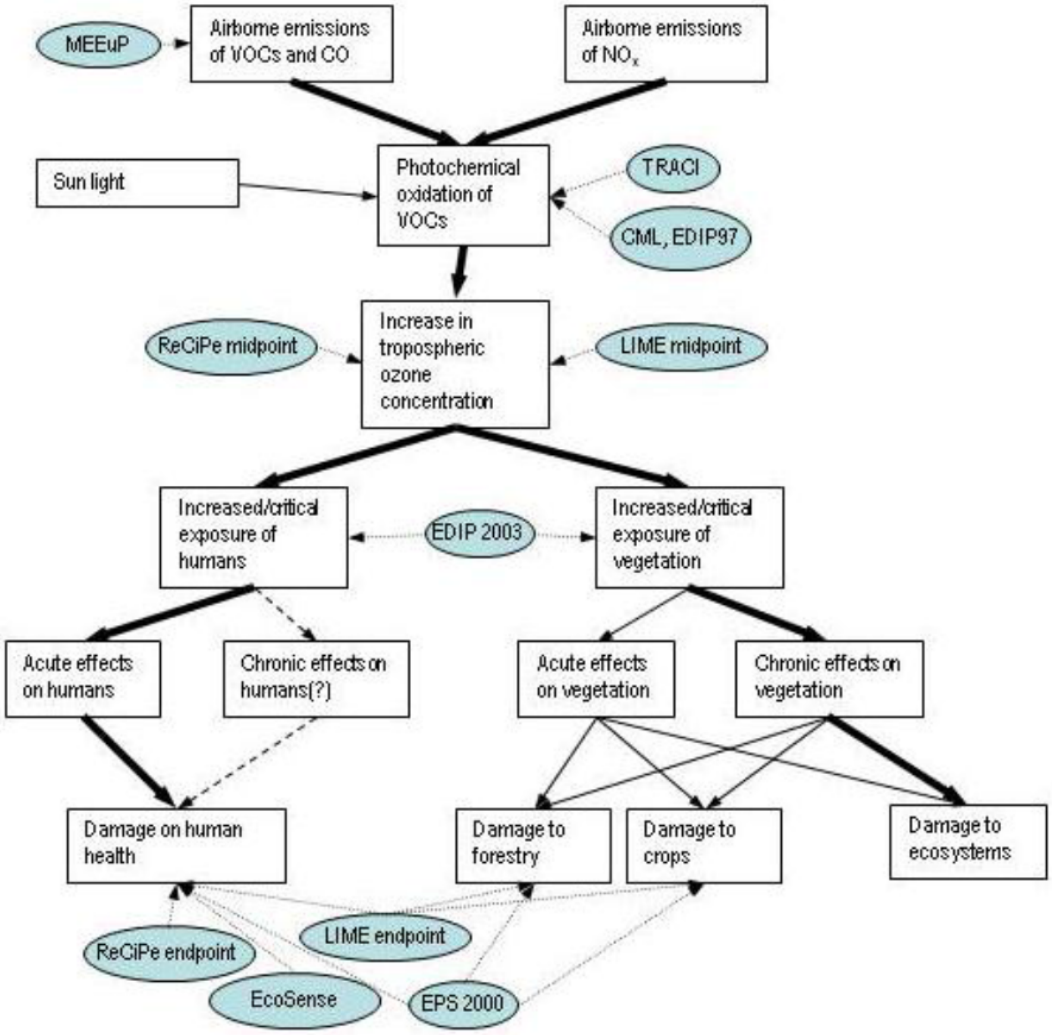

Figure 12 - The photochemical impact pathway. Taken from ILCD Handbook (EC-JRC 2011)

Methods that characterise acidification at the mid-point are summarised in Table 4.

There is currently no international consensus on best practice for photochemical ozone formation. In the European context, the ILCD Handbook recommends the LOTOS-EUROS method (Van Zelm et al., 2008) as applied in ReCiPe, because it is able to supports spatial differentiation which they note to be important, especially for human health impacts.

The challenge for assessing photochemical ozone formation impact for supply chains containing Australian processes is that there are currently no spatially differentiated characterisation factors for Australia.

Table 4 Summary of impact assessment methods for photochemical ozone formation (adapted from (EC-JRC, 2011))

Context	Method	Human health impacts	Vegetation impacts
General	CML-IA	Photochemical Ozone Creation Potential (POCP) based on UK AEA model using a simplified description of the atmospheric transport	
Europe	EDIP	Country-specific factors (for Europe) based the Eulerian EMEP model, with impact on humans modelled as number of people exposed in excess of WHO guidance value for chronic effects times duration (as pers·ppm·hrs)	Country-specific factors (for Europe) based the Eulerian EMEP model, with impacts on vegetation modelled as ecosystem area exposed above threshold for chronic effects times duration (m2·ppm·hrs)
	ReCiPe Midpoint (now superseded)	Models marginal increase in ozone formation due to emissions of NMVOC or NOx, applying the LOTOS-EUROS spatially-differentiated model to calculate European factors	
Japan	LIME	Models ozone formation from 8 archetypes of VOCs (in C_2H_4 equivalents) using a Japanese modification of the Photochemical Box Model from US EPA to produce Ozone Conversion Equivalency Factors (OCEF) which are geographically differentiated for seven Japanese regions	
North America	TRACI / LUCAS	Maximum Incremental Reactivity (MIR) model from Carter, 2000 for characterisation factors, average factors for US based on weighting according to population density patterns, characterisation factor for NOx based on national influence relative to NMVOCs.	

Best practice

For Australia-centric supply chains there would commonly be a strong argument for excluding photochemical ozone formation. For LCA studies involve global supply chains then it should be included using the CML-IA method. This generic method is considered appropriate in the absence of spatial differentiation and regionalised characterisation factors for Australia, and it aligns best with the requirements the Australian EPD scheme.

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

3.9 Particulate matter formation (respiratory effects)

The classification of emissions to air with effect on human health has seen a range of approaches. The class of emissions most commonly accounted for in the category are particulates and so-called particulate precursors that give rise to secondary (inorganic) aerosols (SIA) via atmospheric chemistry. Most current methods characterise them under various separate categories with different names in different associated methods (see Table 5). In the CML-IA method, they are characterised under "human toxicity" along with the other emissions that affect human health.

Table 5 Impact methods and their indicator category terminology / approach to emissions of primary particulate matter and precursors (secondary) as well as related substances

Method	Primary PM	SIA precursors	SOA precursors (VOCs) and NOx	Toxic substances
Eco-Indicator 99	Respiratory inorganics	Respiratory inorganics	Respiratory organics	Carcinogens
CML-IA	Human toxicity	Human toxicity	POXx	Human toxicity
ReCiPe	Particulate matter	Particulate matter	POX	Human toxicity
IMPACT world+	Respiratory inorganics	Respiratory inorganics	Respiratory organics	Cancer / Non-cancer
ILCD recommended	Particulate matter	Particulate matter	POX	Cancer / Non-cancer
AUS V3.01 (Simapro)				Cancer / Non-cancer
BEES	Air pollution	Air pollution	Smog	Cancer / Non-cancer
TRACI	Respiratory effects	Respiratory	Smog	Cancer/ Non-cancer

POX stands for Photochemical Oxidation (photchemical ozone creation potential, see section 3.8)

Sometimes this impact category is referred to as 'respiratory inorganics'. The use of the term 'inorganic' should be avoided because primary particles may be organic as well as inorganic, and the term 'respiratory' can be confusing because photochemical oxidants (smog) (discussed in Section 3.8) also have respiratory effects. Therefore, the term of 'particulate matter' (PM) is most adequate to capture the essence of this category and distinguish it from others.

While PM concentrations in Australia are not a general problem, as they are in other countries, there are local air pollution issues. The main contributors to primary aerosol emissions are industrial operations and power generation. However PM emissions from vehicle exhaust can contribute significantly to health damages because they are emitted in high density areas and at low elevation. Secondary aerosol precursor emissions in many areas are due to vehicle exhaust and domestic wood heaters. Ammonia emissions from agriculture are a major contributor to secondary PM in Europe and the USA and presumably also in Australia. In the context of Australian LCA, this category may therefore be important in processes and supply chains that include domestic heating, transport, and power generation.

As mentioned above, both primary emissions of particles and formation of secondary particles due to atmospheric chemical reactions contribute to resultant particle concentration. The environmental mechanism for the category is represented in Figure 13. The end-point area of protection that it relates to is Human Health (see

Figure 3).

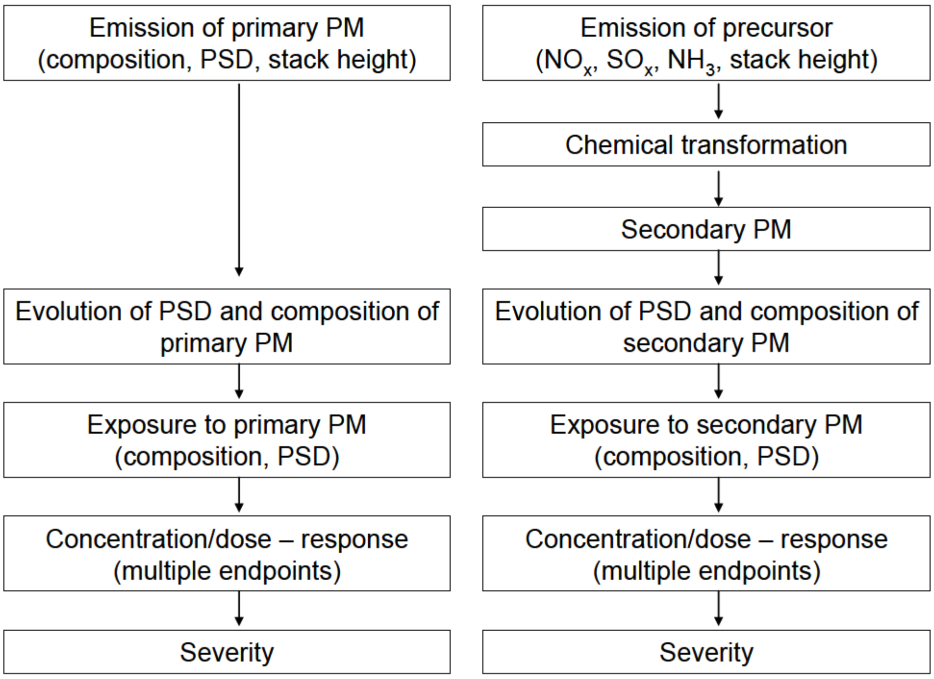

Figure 13 Impact pathways for particulate matter impacts. Taken from the ILCD Handbook (EC-JRC, 2011)

Figure 13 shows that composition does affect exposure-response function, but this is not typically accounted for. Only the particle-size distribution expressed as PM10 or PM2.5 equivalent is used to determine the endpoint damage in disability-affected life years (DALY) (e.g.(Van Zelm et al., 2008)). Midpoint characterization is thus typically in kg PM10-equivalent (ReCiPe) or kg PM2.5-equivalent (TRACI). An exception is the CML-IA method which characterises as kg 1.4DB-eq in keeping with classification of these emissions under human toxicity.

It is in the fate modelling that the major difference between primary and secondary aerosols occurs. The effect range and therefore exposure (different population density) differs significantly between primary and secondary particles for a given emission location. Primary particles typically have their largest effect at the local scale and therefore characterization factors differ by orders of magnitude between rural and urban sources. This applies especially to low-level sources such as vehicle exhaust. Secondary particles, on the other hand, take time to form from the precursors and lead to regional effects, on a scale of thousands of kilometres. This means that, depending on stack height and prevailing winds, the local population density may have little influence on the characterization factor.

There are many characterization models that form the basis of impact assessment methods. Well-known examples are USEtox and EcoSense (for further overview see ILCD Recommendations Impact Assessment (EC-JRC, 2011). The main factors that influence results are the inclusion or exclusion of the terrain model, population distribution, weather parameters (wind), and atmospheric chemistry (background concentrations of catalysts). EcoSense covers all of those parameters on small-scale grids but only for locations in Europe. In Australia, TAPM and AERMOD are the main models, but neither appear to include all of the above detail.

The available impact methods are listed in Table 5. The ReCiPe characterization factors are derived from the EUTREND model for average European conditions. Together with TRACI (using CALPUFF, average USA conditions), the resulting factors can be considered to be based on the most detailed and complete modelling available as well as up-to-date epidemiological information regarding dose-effect relations. ReCiPe covers emissions of PM_{10}, sulphur oxides (SO_2/SO/SO_x), and nitrogen species (NO_x, NH_3). A weak point of ReCiPe is that characterization for $PM_{2.5}$ is the same as for PM_{10} which is contrary to epidemiological evidence. $PM_{2.5}$ is considered more for human health damages. TRACI covers PM_{coarse}, $PM_{2.5}$, nitrogen and sulphur species (NH_3, NO_x/NO_2, SO_2), total suspended particulates (TSP) as well as carbon dioxide (CO). None of the methods distinguish high and low elevation of emissions or high and low population density at emission location. IMPACT World+ (based on Riskpoll, USEtox and Greco (2007)) provides global average impact factor. Whether those are more representative for Australia than those derived for Europe or USA is impossible to say. The ILCD recommended method is to use Riskpoll. Impact factors are available in the ILCD 2011 Midpoint impact method. For endpoint, ILCD recommends using ReCiPe. Both IMPACT World+ and ILCD distinguish high and low population density at emission location and IMPACT World+ also distinguishes high and low stack emissions for primary PM. This method covers $PM_{2.5}$, PM_{10}, PM_{coarse}, $PM_{>10}$, NH_3, NOx, SO_2.

There is no consensus regarding the treatment of CO emissions. In principle, primary CO emissions should be classified as contributing to another impact category, photochemical oxidation (Hauschild et al., 2013b) (see Section 3.8), and ReCiPe, BEES and CML-IA methods characterise it as such. Exposure to CO however is considered to contribute to “winter smog” or inorganic particles contributing to respiratory problems, and TRACI and EI99 (superseded) as well as ILCD, characterise CO as a respiratory inorganic, a precursor of secondary PM. Possibly, CO should be classified under both, along with NO_x.

Secondary organic aerosols (SOA) are currently not included in any of the impact methods. They are products of photochemical oxidation processes, i.e. the precursors are largely volatile organic compounds (VOC). ReCiPe does list non-methanic VOCs (NMVOC) under PM formation with a zero characterization factor (implementation Simapro 8.0.3). The reason for excluding SOA-related impacts is presumably that formation and anthropogenic contributions are very uncertain (e.g. Sauter et al. 2012, LOTOS EUROS V1.8 reference guide).

There is no international consensus or even an ILCD recommended impact method for this category. The ILCD recommendation is to develop new characterization factors based on a combination of best available models. For Australia, no characterization factors have been modelled so far. While ReCiPe and TRACI offer robust factors, it is unlikely that those factors, even in a relative sense, are appropriate for Australia. Given the rather unique population distribution of Australia (clustering, insularity, major cities all on the coast), ratios between factors for different substances are likely to be quite different from those on other continents and even between Australia's major population centres.

It is highly recommended to develop mid- and endpoint characterization factors specifically for Australia for an appropriate range of sub-compartments (metropolitan, urban, rural), with the minimum requirements of including secondary inorganic aerosol (SIA) formation (from NO_x, SO_x and NH_3), differentiating between high (stack) and low (traffic) emissions, and differentiating between PM fractions preferably with $PM_{2.5}$ as reference substance.

Best practice

Current best practice is to use methods based on the fate and exposure model of Wolff (2000), using the CALPUFF model, when PM formation is a relevant category. This has been operationalised in the TRACI method. TRACI distinguishes between $PM_{2.5}$ and PM10, and the US background concentrations (O_3, NH_3) are probably more appropriate for Australia than the high background concentrations typical for Europe. Using this midpoint characterization is appropriate given the need for population-specific modelling in endpoint characterization. In the reporting, the limitations of this impact method applied in Australian context should be highlighted. Normalisation is only available for North America.

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

3.10 Land use – biodiversity

There is a general acceptance that the term “biodiversity” encompasses diversity at the three levels: genetic, populations/species, and communities/ecosystems (Redford and Richter 1999), with some authors including a forth level of regional landscape and associated concepts of structure and function (Noss, 1990). There are currently no methods which allow for simultaneous measurement of all four levels of biodiversity. There have been numerous attempts to integrate direct and indirect land use in LCA and its impact on biodiversity (e.g. Koellner and Scholz (2007); Koellner and Scholz (2008); Michelsen (2008); (Schmidt, 2008); Geyer et al. (2010)), but none of the proposed metrics are fully operational or applied globally. Existing methods do not allow for simultaneous measurement of a range of taxa (flora, mammals, birds, frogs and invertebrates) or the ecosystem services they underpin. The characterization factors typically suggested for land use impacts on biodiversity in LCA are local species diversity and functional diversity.

Two types of land use interventions are usually considered in life cycle inventories and impact assessments; land transformation and land occupation (Lindeijer, 2003, Milà i Canals et al., 2007). The areas of both occupied and transformed land are recorded in the inventory flow. In the land use impact assessment framework, impact of land use is often compared to a reference “natural” system. The concept of reversibility of impacts from land use is also important to consider; depending on the nature of the impact, regeneration time exceeds modelling periods typically used in LCAs, and re then classified as “permanent” impacts. When dealing with systems that involve a period of transformation followed by a longer period of occupation, allocation of impacts is required. It has been suggested that a period of 20 years as an allocation period for the transformation stage, as this is considered to be consistent with standards and regulations for land use-derived greenhouse gas emissions allocation (BSI, 2011, IPCC, 2006). The allocation of impacts from land transformation is done at the inventory level, whereas the calculation of land transformation impacts related to the LCIA phase (Koellner et al., 2013a). For land use impact calculation, modelling periods of 20, 100 and 500 years (used by the IPCC for global warming) are used depending on regeneration times (Koellner et al., 2013a). The area of protection for biodiversity is Ecosystem Quality.

Although relevant to all land use activities that have an impact on ecosystems, biodiversity impacts are especially relevant for agriculture, mining and forestry, and new urban development.

The cause –effect chain captured by biodiversity are shown in Figure 14 below. They are closely linked with the impact pathways associated with other ecosystem services (Figure 14). The end-point indicator linked to biodiversity impacts is Biodiversity Damage Potential.

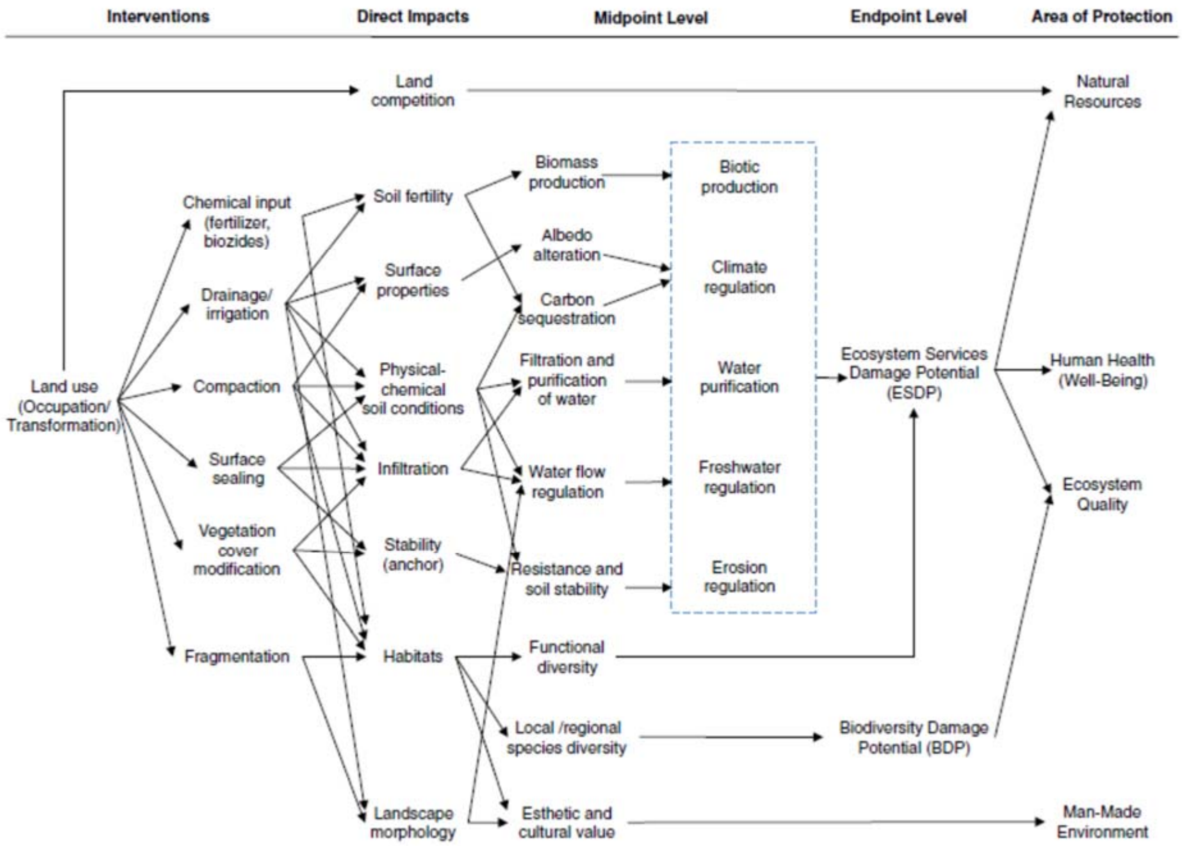

Figure 14 Impact pathways for land use impacts (Koellner et al., 2013b)

Many of the early approaches used net primary productivity (NPP) as a surrogate for biodiversity (e.g. Hampicke (1991); Swan and Petterson (1991); Lindeijer (2000); Weidema and Lindeijer (2001)). However, NPP is not a suitable surrogate for biodiversity worldwide with many systems having a negative relationship between biodiversity and productivity (Wardell-Johnson et al., 2004). The use of NPP as a surrogate for biodiversity has also proven inadequate for different ecosystems; for example, desert systems have low NPP, but extremely high diversity of many groups such as reptiles (e.g., Cogger (2000)). A number of studies have attempted a species-based approach using an estimate of especially vascular plant diversity (mainly due to data availability), primarily species richness (van Dobben et al., 1998, De Schryver et al., 2010, Koellner and Scholz, 2008, Köllner, 2000).

This is problematic as species richness only considers one component of biodiversity, and species richness in one taxonomic group rarely relates to richness in other groups (Michelsen, 2008). In addition, for many areas the true species richness values are largely unknown and attempting to estimate them would likely produce results with high levels of uncertainty. Others have attempted to focus on the potential impacts on threatened species or communities (Weidema and Lindeijer, 2001). Threatened species are often atypical in their response to disturbances and therefore are unsuitable as a surrogate of biodiversity. One paper developed a metric based on species richness, ecosystem rarity and ecosystem vulnerability (Weidema and Lindeijer, 2001); however this is only applicable at the biome level (e.g., rainforest, desert) and would therefore be too coarse for meaningful comparisons for practices within vegetation formations within a biome. Geyer et al. (2010) presented a life cycle inventory method that used a geographic information system (GIS) to calculate elementary flows of habitat types. Although promising, it is expected that if this method were adapted to a global scale, there would be a significant lag time and cost to acquire, process and analyse the remotely sensed data for biodiversity that is currently largely unavailable. Curran et al. (2011) conducted a review of the use of indicators to model biodiversity in LCA. They found serious conceptual shortcomings in the way models are constructed, with scale considerations largely absent, and a disproportionate focus on species richness. In addition, most available models are restricted to one or a few taxonomic groups and geographic regions (Curran et al 2011). Curran et al. (2011) make the point that important drivers of biodiversity loss (overexploitation and invasive species) are completely missing from LCA. More recently, de Baan et al (2013) suggested an approach where species richness of different land use types was compared to a (semi) natural regional reference situation to calculate relative changes in species richness. The authors concluded that the approach may be used as a rough quantification of land use impact on biodiversity on a global scale. This methodology was further developed by Mueller et al (2014) in the assessment of the biodiversity impacts of milk production in Sweden. The work highlighted the fact that high levels of direct land use cannot be assumed to lead to high impacts on biodiversity. Coelho and Michelsen (2014) have proposed a globally applicable model for assessing land use impacts on biodiversity without the use of any taxa as indicators, using kiwifruit production in New Zealand as a case study. In their model, variables such as ecosystem scarcity, ecosystem vulnerability and impact on biodiversity were combined with a “deviation from naturalness” (hemeroby) factor. The authors detail several drawbacks with the method proposed, such as lack of reliable data to support the use of the variables proposed, and the simplistic linear approach associated with the use of hemeroby. The use of a functional diversity index for several taxonomic levels to calculate characterisation factors for land use impacts has been proposed by Souza et al (2014). This approach, based on a series of functional traits, aimed to capture relationships between, redundancy or complementarity between species and the functions they play. The authors describe the challenges in the availability and selection of appropriate functional traits for different taxa. The lack of relevant empirical data is one of the key issues hindering other proposed methods (Coelho and Michelsen, 2014, de Baan et al., 2013, Mueller et al., 2014). A weighting system based on absolute species richness, vulnerability and irreplaceability is proposed; however attempts to incorporate transformation impacts in addition to plant species richness data (occupation effects) were challenged by the lack of empirical data.

In summary, methods for incorporating biodiversity in LCA have been largely hindered by a lack of information on the relationships between land-use and biodiversity, lack of empirical date and no universal, appropriate metric for biodiversity at alternative scales. A new biodiversity metric (BioImpact) has recently been proposed (Turner et al 2014). It relies on literature review and expert opinions through a series of questions which aim to encapsulate the main issues within a disturbance impact framework. Using a series of semi-quantitative questions, biodiversity impacts are estimated - and scaled to a single measure that can be incorporated into LCA (Penman et al., 2010, Turner et al., 2014). This method is under final stages of development, with planned work including the development of biodiversity scores for a number of relevant production systems.

A land use assessment framework has recently been established by the United Nations Environment Programme (UNEP/Society of Environmental Toxicology and Chemistry (Koellner et al., 2013b) to harmonize practices and provide principles for Life Cycle Inventories on a global scale, provide guidelines for LCIA methods and provide operational sets of characterization factors for impacts on biodiversity and services provided by terrestrial ecosystems.

The methodologies adopted by Eco-indicator 99, Impact 2002+, and ReCiPe (Goedkoop et al., 2009, Goedkoop and Spriensma, 2001, Jolliet et al., 2003) include midpoint and/or endpoint indicators, with the underlying models based on species diversity loss. Typically for endpoint indicators, the species loss from impacts due to a production system are combined for a production period of one year (e.g. EPS 2000, LIME, Swiss EcoScarcity) . With the exception of LIME (valid for Japan), the models for all other methodologies described above are valid only for specific regions within Europe. Mid-point indicators estimate species losses or local extinction rates as caused by a range of separate mechanisms. These methodologies use the Potential Disappeared Fraction of species

(PDF), which is basically a measure of the rate of species loss for a period of time as a result of land occupation and/or conversion or other processes that impact on aquatic ecosystems. PDF can be expressed in different ways; e.g. Eco-indicator 99 uses the rate of species loss per m^2 per year as the endpoint indicator); ReCiPe on the other hand uses actual species lost per year (also as an endpoint indicator); and finally Impact 2002+ applies a normalisation procedure to determine rate of species loss per person per year, as a midpoint indicator. Impacts from land transformation have to be allocated to output (functional units) arising from the new land use.

For Australia, the existing biodiversity impact methods fail to capture complexities associated with the impacts of land occupation and transformation on biodiversity. Although available methods can be regionalised for the Australian context, lack of supporting data, and more importantly, the low level of confidence in the sensitivity and reliability of existing methods for Australian conditions means that the LCA practitioner should exercise caution in the use of such methods.

Best practice

There is currently no agreed best practice for the use of a biodiversity indicator in LCAs globally, for the reasons described above. LCA practitioners need to note the limitations in the use of the existing methods based on single indicators such as species richness or NPP, or methods that seek to combine two or three concepts, as they may lead to inconclusive or unreliable results and may not represent a suitable proxy for biodiversity, especially for more complex ecosystems. The current work by the UNEP/SETAC on global land use impact assessment may result in new improved methods. Key elements in the development of required inventory data to support best practice models include generation of spatial layers (for both the system in question and the reference scenario), collection of data supporting the characterization factors and finally calculation of the land use impact (Koellner et al., 2013b). The UNEP/SETAC guidelines suggest that in the creation of models it should be stated which impact pathways are modelled, which land use/cover typology as well as the biogeographical differentiation level are used for the development of CFs and, in addition to the reference situation, whether relative or absolute quality changes are used for the calculation of land use impacts (Koellner et al., 2013b). This work may be complemented by the development of alternative metrics (e.g. BioImpact in Australia).

Aspirational practice is to generate methods that capture the range of important issues associated with biodiversity, that can be globally applied and that do not require extensive funding for its development.

3.11 Land use – ecosystem services

Characterisation of the eco-system services aspects of land use, as recently developed under the UNEP/SETAC Life Cycle Initiative (Saad et al., 2013, Koellner et al., 2013b, Brandão and Canals, 2013, Milà i Canals et al., 2007) has not been included in this version of the Guide, but will be developed for future versions.

3.12 Ozone Layer Depletion

The impact category 'ozone layer depletion' characterises the reduction in concentrations of ozone in the stratosphere (ozone layer) when ozone depleting substances (ODS) are released to air. Ozone (O_3) is a natural constituent of the Earth's atmosphere and is an extremely reactive substance. Its presence in the stratosphere is the result of a continual cycle of formation and breakdown processes, occurring both chemically and by photo-dissociation. There is strong scientific consensus that anthropogenic emissions of ozone depleting substances caused substantial levels of stratospheric ozone depletion over the latter parts of the twentieth century. The ozone layer plays a critical role in regulating conditions on Earth, but has been substantially depleted by CFC (chlorofluorocarbon) and other halocarbon emissions. This has increased transmission of UVB radiation to the surface, and been implicated in a range of negative human and ecosystem health impacts. The end-point area of protection that it relates to is Human Health. (Lane, 2015)

Net stratospheric ozone concentrations are strongly influenced by a small group of reaction pathways, mostly associated with halogen, NOx, and HOx free radicals. The groups of substances involved in these are chloroflurocarbons (CFC), hydrochloroflurocarbons (HCRC) and halons in refrigerants, solvents and fire extinguisher agents. The Montreal Protocol (1987) regulated the phase-out of these substances. Even though this has been successful in mitigating ozone depletion, there remains a legacy of halocarbons that will continue for many years. (Lane, 2015).

Ozone Depletion Potential (ODP) factors for halocarbons have been the cornerstone of midpoint impact assessment for a long time, and most LCIA methods use steady state ODP values, which are periodically updated by the World Meteorological Organisation (WMO, 2011). Nitrous oxide (N_2O), carbon dioxide (CO_2) and methane (CH_4) are now known to also influence the ozone layer. A proportion of N_2O emissions break down into NO radicals that can initiate catalytic ozone destruction. CO_2 and CH_4 as greenhouse gases, on the other hand, have radiative properties that act to reduce temperatures in the stratosphere, slowing the rate of ozone depletion. However the influence of these substances is not currently included in ODPs (Lane, 2015).

Best practice

Best practice is to use the ODP values published by the WMO, and applied in most impact assessment methods. Aspirational practice would be to adopt evolving methods that account for the effects of N_20, CO_2 and CH_4.

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

3.13 Ionizing Radiation

Ionizing radiation characterises impacts from the release of radioactive species (radionucleides) to air and water. The species most commonly accounted for are the radionucleides of caesium, iodine, radon and uranium etc. Anthropogenic sources are the nuclear fuel cycle, phosphate rock extraction, coal power plants, and oil and gas extraction (Frischknecht et al., 2000). When released to the environment, they can impact both human health and ecosystems. So the end-point areas of protection they relate to are Human Health and the Ecosystem Quality (see Figure 3). Characterisation of human health impacts is more developed than ecosystem impacts.

Release of radioactive material is a consideration for nuclear power generation, and its assessment may be warranted for processes with nuclear energy inputs (for example in Japan, South Korea, France, Germany, Switzerland, United Kingdom, Spain, Sweden, Finland, Belgium, Russia, and the US). However radioactive materials can be of some relevance for other parts of the nuclear fuel cycle (such as uranium mining and milling) and coal power plant (Frischknecht et al., 2000), and so may have some relevance in the Australian context.

Ionizing radiation impact assessment considers the cause-effect chain that leads to the internal accumulation in humans, leading to cancer and hereditary effects (Figure 7) and bioaccumulation and external irradiation in other species (see Figure 16). Methods that characterise ionizing radiation at the mid-point are summarised in Table 6.

For human health impacts, the ILCD Handbook (EC-JRC, 2011) notes that only the model of Frischknecht et al. (2000) meets the requirements of a quantitative approach. It employs fate and exposure assumptions based on assessment of routine atmospheric and liquid discharges in the French nuclear fuel cycle (Dreicer et al., 1995), and is employed in a number of impact assessment methods (Impact 2002+, ReCiPe and ILCD 2011 Midpoint). However there are differences in the reference units – kBq $U235_{eq}$ for ILCD 2011 and ReCiPe, versus $BqC\text{-}14_{eq}$ to air for Impact 2002+.

For ecosystem impact, only one method is reported. It is a screening level ecological risk assessment based on the eco-toxicological effects observed from a gamma irradiation exposure experiment on nine commonly adopted freshwater reference organisms (Garnier-Laplace et al., 2009). However it is only included as an interim method (EC-JRC, 2011) in the ILCD 2011 Midpoint method, and is not included in any other integrated LCIA methods. Consequently impacts of ionizing radiation on ecosystems have not often been included in LCA studies to date.

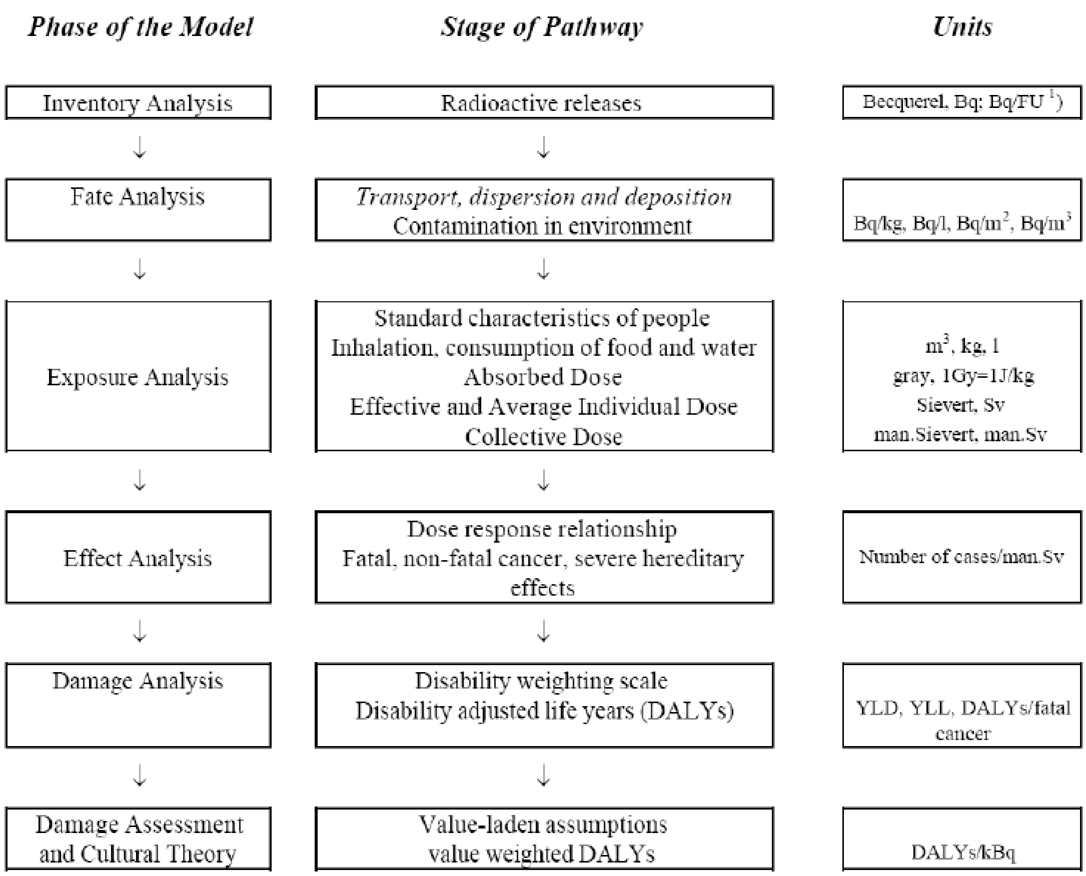

Figure 15 Impact pathways for the human health effects of ionizing radiation (Frischknecht et al., 2000)

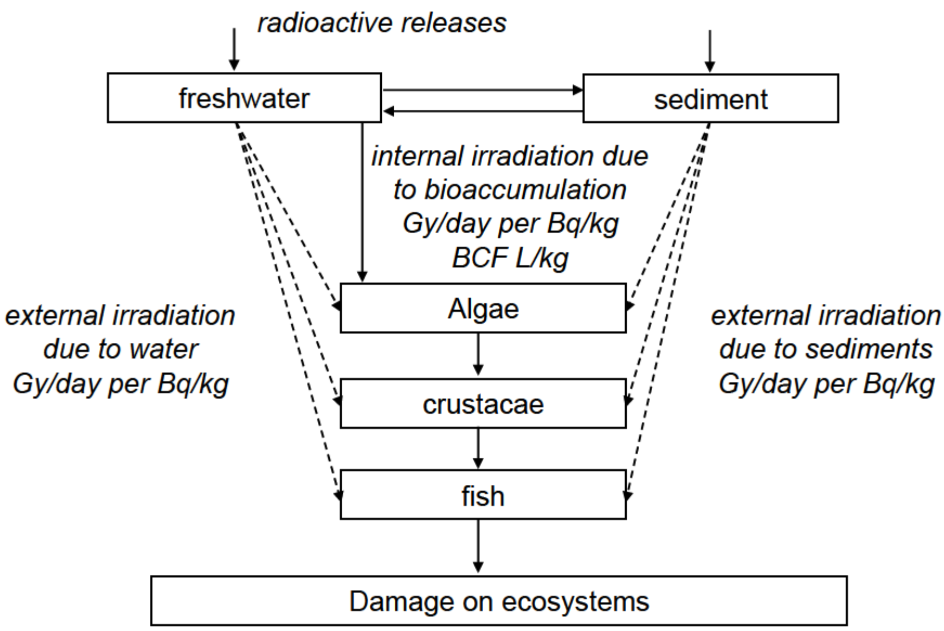

Figure 16 Impact pathways for the ecosystem effects of ionizing radiation. Taken from ILCD Handbook (EC-JRC, 2011)

Table 6 Summary of ionizing impact assessment methods

Method	Human health impact model	Unit	Ecosystem impacts model
Impact 2002 ReCiPe ILCD 2011 Midpoint	All the methods use the model of Frischknecht et al. (2000), which is based on Dreicer et al. (1995).	$BqC\text{-}14_{eq}$ $kBq\ U235_{eq}$ $kBq\ U235_{eq}$	Screening level ecological risk assessment for radioactive releases of Garnier-Laplace et al. (2009) – ILCD 2011 Midpoint method only.

There is currently no formal international consensus on best practice for ionising radiation, and it is not a current priority of the international consensus-building process (Jolliet et al., 2014). However given there is only one recognised method for human health impacts, and one interim method for ecosystem impacts, consensus is in effect implicit.

There are two challenges in the Australian context. The first is that releases of radioactive materials are currently not included in the Australian Life Cycle Inventory (AusLCI) datasets. Therefore LCA studies involving Australian processes cannot rely on existing data sets and may require additional inventory development for radioactive material releases. The second is that characterisation factors have been developed in the context of Europe and not been validated in the Australian where population densities are much lower and hence exposure factors can be expected to be different. This means that if the available methods are applied to releases occurring in Australia the potential uncertainty of the results due to regional differences should be assessed and explained.

Best practice

Current best practice would be to assess the human health impacts of ionizing radiation, using the ILCD 2011 Midpoint methods, for supply chains known to include releases of radioactive materials, and where ionizing radiation is assessed (through a screening LCA) to have some significance. However results should note the limitations of the method and estimate the uncertainty that the aforementioned non-regionalisation creates. Aspirational practice would be the routine inclusion of radioactive material releases in LCI for Australian processes (where relevant) and the regionalisation of the recommended human health impact methods for Australian conditions.

Characterisation factors

Characterisation factors for the best practice method are provided in the accompanying ANNEX spreadsheet.

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