Value choices in life cycle impact assessment

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Chapter 1

Introduction

1.1 Life cycle assessment

Life cycle assessment (LCA) is the compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle (ISO, 2006a). The product's life cycle includes all processes which can be related to the production, use and disposal of the product (see figure 1.1). All resources, land area and emissions that are used or released during the whole life cycle (also called inventory data) are collected and serve as a basis to assess the potential environmental impacts. As it is impossible to make an inventory of all emissions over the whole production chain of a product, it is necessary to define system boundaries and how to allocate emissions from by-products. This is done in the inventory. Once all the required emission and resource data is collected in an inventory list, a life cycle impact assessment (LCIA) is performed to calculate the potential environmental impact of the inventory data. The outcomes of the assessment (the impact score) can be interpreted and further analyzed to reduce uncertainties from uncertain inventory data, data gaps and important assumptions taken during the data collection and impact assessment (ISO, 2006b).

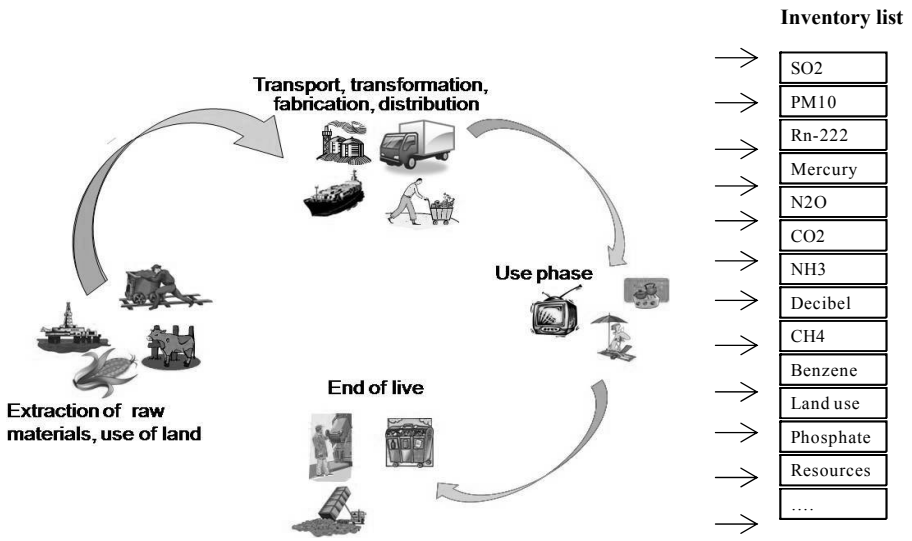

Figure 1.1. Example of a product life cycle, from extraction to end of life. All emissions, land use and resource use are collected in an inventory list.

Several methodologies have been developed to assess the potential environmental impact of a product, also defined as life cycle impact assessment methodologies (e.g., Heijungs et al., 1992, Steen, 1999, Goedkoop and Spriensma, 1999, Hauschild and Potting, 2005, Goedkoop et al., 2008, Frischknecht et al., 2008, Itsubo, 2008). An LCA methodology is a collection of individual characterization methods which together address different environmental impacts (defined as impact categories) covered by the methodology (see figure 1.2). Each characterization method applies its own cause-effect pathway (see

figure 1.3) and impact indicator, to produce so called characterization factors (CFs). CFs are used as weighting factors to aggregate life cycle emissions. The impact score for impact category c (IS_c), also defined as impact category indicator result, equals:

$$IS_c = \sum_x CF_{x,c} \cdot m_x$$

where $CF_{x,c}$ is the CF of intervention x within impact category c (e.g. CO_2-equivalents/kg^{-1}) and m_x the amount of intervention *x* (e.g., the mass of substance x emitted) as collected in the inventory list. When the same units are applied, the impact scores over different impact categories can be added. A simplified example of a cause-effect pathway is presented in figure 1.3.

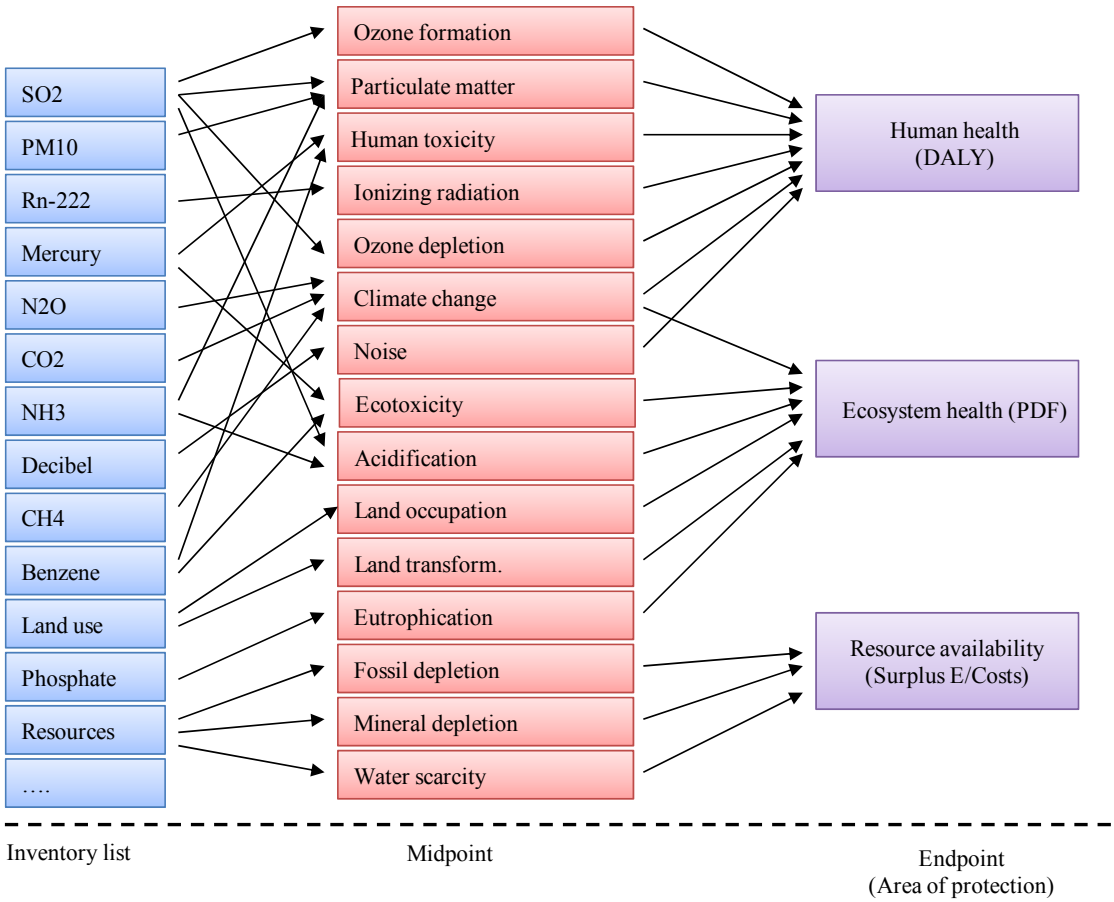

Figure 1.2. List of impact categories for characterization at midpoint and endpoint level (adapted from JRC, 2010a). In this case, the endpoints refer to three areas of protection: human health (expressed in disability-adjusted life years; DALY), ecosystem health (expressed in potentially disappeared fraction of species; PDF) and resource availability (expressed in surplus energy or costs).

The type of impact indicator used in a characterization method to measure the effect will influence the uncertainty in the indicator and thus also the impact score. In general, characterization models can use two types of impact indicators, midpoint indicators and endpoint indicators. Midpoint indicators (e.g., SO_2-equivalents, m^2 land occupied) quantify the effect somewhere along the cause-effect pathway at a

stage where the level of uncertainty is relatively low. Midpoint indicators of individual impact categories are typically expressed as equivalent values and relate to a reference intervention. For each impact category a different reference intervention is chosen. Examples are kg CO_2-equivalents for climate change, SO_2-equivalents for acidification and MJ-equivalents for resource use (Heijungs et al., 1992, Gürzenich et al., 1999, Hauschild and Potting, 2005). This type of indicator reflects the relative importance of emissions or extractions within an impact category.

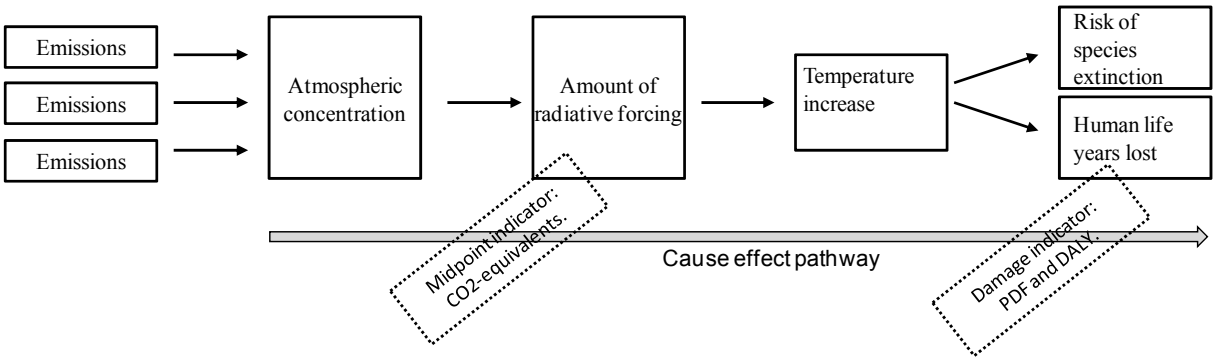

Figure 1.3. Example of a simplified cause-effect pathway for climate change. Different impact indicators are presented along the cause-effect pathway.

Endpoint or damage indicators (e.g. potential disappeared fraction of species) quantify the effect at the end of the cause-effect pathway. Endpoint (or damage) indicators commonly refer to three areas of protection: human health, ecosystem health and resource availability (e.g., Goedkoop et al. 2008). Damages to human health, caused by various types of environmental stressors, are quantified by changes in both mortality and morbidity (JRC, 2010b). The indicator commonly used to quantify human health is disability-adjusted life years (DALY; Hofstetter, 1998). Hofstetter (1998) introduced the DALY-concept in LCA, as inspired by the work of Murray and Lopez (1996a) for the World Health Organization (WHO). The WHO reported DALYs for a wide range of diseases, including various cancer types, vector-borne diseases and non-communicable diseases. The area of protection "ecosystem health" refers to the natural ecosystems around the world in terms of their quality. For damages on ecosystem health, the changes in quality of natural ecosystems as a consequence of exposure to chemicals or physical interventions are quantified (JRC, 2010b). To quantify ecosystem damage, Muller-Wenk (1998b) proposed the potential disappeared fraction of species (PDF) as a damage indicator. This indicator measures the change in species biodiversity over a certain time and area. Resource availability refers to the extraction of scarce resources, such as mineral deposits, fossil energy carriers, fish, trees, and water. The area of protection "resource availability" covers the concern about limited resource availability and the future possibilities to enjoy the resources we have today (JRC, 2010b). The damage from resource depletion can be expressed in surplus costs or surplus energy, based on the extra cost or energy for future mining of lower grade resources (Muller-Wenk, 1998, Goedkoop and Spriensma, 1999). Compared to the midpoint indicator, an endpoint indicator has a relatively high

environmental relevance, allows aggregation of different effects and is considered to be more understandable to decision makers, but is also inherently more uncertain (Bare et al., 2000, UNEP-SETAC, 2003, Jolliet et al., 2004, Reap et al., 2008, Bare, 2009, JRC, 2010b).

1.2 Uncertainties in life cycle assessment

Sigel et al. (2010) defines uncertainty as the lack of confidence about knowledge related to a certain question, whereby confidence about knowledge may range from 'being certain' to 'admitting to know nothing' (of use). Uncertainty in the outcome of an LCA can derive from several sources, such as the lack of spatial and temporal variability, the lack of knowledge about the true value of a parameter or the form of a model (Reap et al., 2008). Different types of uncertainties can be distinguished, which all influence the total uncertainty in the outcome of an LCA. This section describes the typology used in this PhD thesis, together with different ways to handle uncertainties. Particular attention is given to uncertainties derived from value choices and the use of the Cultural Theory as a framework to quantify these uncertainties.

Uncertainty typology

Several researchers have put forward different typologies that attempt to provide a framework for describing different types of uncertainty (Morgan and Henrion, 1990, Hofstetter, 1998, Huijbregts et al., 2001, Walker et al., 2003, Ascough et al., 2008, Refsgaard, 2007, van der Sluijs et al., 2005). Ascough et al. (2008) gives an overview of different classifications of uncertainty according to various literature sources. Each framework distinguishes different types, location and/or natures of uncertainties. Types of uncertainty can relate to e.g., norm-related and fact-related uncertainty (e.g., Hofstetter, 1998, Ascough et al., 2008, Sigel et al., 2010); location of uncertainty relate to e.g., uncertainty within parameters and/or model development (e.g., Huijbregts et al, 2001, Walker et al., 2003); while natures of uncertainty can relate to e.g., epistemic and stochastic uncertainty (e.g., Walker et al., 2003). However, as mentioned by Sigel et al. (2010) "a conceptual framework for comprehensively perceiving and describing uncertainty is still lacking". The typology presented here is a broad framework that differentiates between three types of uncertainty, namely measurement uncertainties, assumptions and ignorance (see table 1.1). Each type of uncertainty can appear on parameter and/or model level, can be norm and/or fact related, can have a different level of uncertainty (from known to unknown), and so on.

- Measurement uncertainties derive from imprecise measurements of parameter values (Huijbregts et al., 2003, Lewandowska et al., 2004, Geisler et al., 2005, Hung and Ma, 2009).

- Assumptions are made to simplify parts of the calculations or when knowledge is uncertain or lack in knowledge occur (Kloprogge et al., in press). Most often, assumptions are not value free and based on scientific facts alone. Some assumptions derive mainly from lack in knowledge, whereby the choice of one option above another can be influenced by personal values such as, commonly acceptance or familiarity. Hertwich et al. (2000) describes these values as contextual values. On the contrary, some assumptions can be mainly driven by personal believes and values that reflect what we care about, without any science being involved. A typical example is the equity of different age groups or species. Hertwich et al. (2000) defined the values driving these assumptions as preference values. Nevertheless, an assumption can be driven by both contextual values and preference values. An example is the choice for time horizon to consider in the modeling, which is based on personal preferences but at the same time includes uncertainty in future developments.

- Ignorance is the lack of awareness that knowledge is wrong or lacking (Refsgaard, 2007). Two kinds of ignorance can be singled out, namely recognized and unrecognized ignorance. In the latter case, there is unawareness about the uncertainty.

Table 1.1. Different types of uncertainty and their effect on parameters and model development.

Uncertainty types	**Data/parameters**	**Model**
Measurement uncertainty Random uncertainty	- Uncertainty in emission measurements - Uncertainty in atmospheric degradation rate of chemicals - ….	Not applicable
Assumptions Can be driven by personal values	- Spatial and temporal approximations of inventory data -Technological approximations of inventory data -…	- Certainty of effects - Model linearity - Time horizon - Age weighting - …
Ignorance Recognized and unrecognized imperfect knowledge	- Lack of quantitative emission data - Lack of quantitative effect data - ….	- Influence of albedo on global warming impacts - ….

How to handle uncertainty

Both the credibility and transparency of an LCA can be enhanced by giving more attention to quantifying uncertainties (Geisler et al., 2005, Reap et al., 2008, Hung and Ma, 2009, Rosenbaum et al., 2009). The ISO 14044 standard presents two techniques to analyze uncertainties during the interpretation of LCA results (ISO, 2006b), i.e. uncertainty and sensitivity analysis.

Uncertainty analysis investigates uncertainties by applying uncertainty ranges and performing a Monte Carlo analysis. This is also called probabilistic analysis. Some LCA software platforms, such as SimaPro (see www.pre.nl) and Umberto (see www.umberto.de), provide the ability to calculate uncertainty ranges of inventory outcomes using Monte Carlo analysis (Lloyd and Ries, 2007). This

allows practitioners to assess uncertainties in inventory data of an LCA (e.g., Geisler et al., 2005, Benetto et al., 2009, Humbert et al., 2009). However, a survey on quantitative uncertainty approaches demonstrates that this is only applied in a limited number of LCA studies (Lloyd and Ries, 2007). Uncertainty ranges of CFs, deriving from the input data of impact assessment models, are not always quantified by the method developer (e.g., Heijungs et al., 1992, Goedkoop and Spriensma, 1999). Moreover, software platforms do not always support the inclusion of uncertainty distribution for CFs. Only a few studies provide rough guidelines and rules of thumb to quantify uncertainty ranges of CFs (Huijbregts et al., 2003, Geisler et al., 2005, Hung and Ma, 2009).

Sensitivity analysis investigates the uncertainties related to choices, such as the effect of changing assumptions made in an LCA study, by running different scenarios. Scenario analysis is widely used to assess possible future situations that reflect different perspectives (EEA, 2000, Cousens et al., 2002, Postma and Liebl, 2005, Van Notten et al., 2003, Ayer et al., 2007). However, uncertainties from value choices in impact assessment modeling are mostly neglected within LCA studies. Most impact assessment methodologies embed value choices without giving practitioners or decision makers the opportunity to assess the difference in result when applying a distinct world view (Jolliet et al., 2003, Hauschild and Potting, 2005). Only a few studies do handle uncertainties arising from value choices made in impact assessment modeling by applying a structured framework (Janssen and Rotmans, 1995, Hofstetter, 1998, Goedkoop and Spriensma, 1999, Frischknecht et al., 2000). The Cultural Theory is generally used as a framework to define different scenarios (for details see section 1.3).

Because the ISO14044 document provides limited guidance, several studies present more detailed procedures and methodologies to improve and quantify uncertainties within LCA calculations (Huijbregts et al., 2001, Fukushima and Hirao, 2002, Björklund, 2002, Lewandowska et al., 2004, Geisler et al., 2005, Benetto et al., 2006, Lloyd and Ries, 2007, Hung and Ma, 2009). Nevertheless, a missing operational framework together with the image of being time consuming is probably why most current life cycle studies do not clearly address uncertainty. If studies address uncertainty, the analysis is typically limited to uncertainties of parameters (Rosenbaum et al., 2009, Finnveden et al., 2009). The next section gives an overview of how proper scenarios can be defined and help to quantify the uncertainties arising from value choices in model development.

1.3 Value assessment

Personal values, such as personal believes, attitudes and risk perceptions, are seen as the criteria people use to evaluate actions and events, and eventually determines the value choices people make (Douglas and Wildavsky, 1982, Schwartz 1992). Perspectives are used as a tool or framework to cluster different personal values (see figure 1.4). The uncertainties from value choices can be quantified by developing two or more scenarios which reflect different perspectives.

Figure 1.4. Graphical representation of the meaning of personal values, value choices and perspectives.

Within LCA, the Cultural Theory of risk has been used to define different modeling scenarios (Janssen and Rotmans, 1995, Hofstetter, 1998, Goedkoop and Spriensma, 1999, Frischknecht et al., 2000). The Cultural Theory is developed by the anthropologist Mary Douglas (1982) and is originally a societal social anthropology approach, based on the structure and functioning of groups within societies. Douglas and Wildavsky (1982) proposed a grid-group theory to help identifying and comparing the different ways of life. Their theory assumes that societies can be characterized along two axes, labeled "group“ and "grid". The “group axes” represents the extent to which an individual is incorporated into a group. The “grid axes” denotes the degree to which an individual's life is circumscribed by externally imposed prescriptions and gives a measure of structure. Each combination of extremes along the two dimensional presentation results in 5 archetypes of people, namely the individualist, the hierarchist, the egalitarian, the hermit and the fatalist. Each archetype reflects a composition of ideologies, cultural biases, social relationships, moral beliefs, concerns or interests. The individualist is characterized by weak group cohesion and regulations for social relations, and considers nature to be stable and able to recover from any disturbance. This coincides with the view that humans have a high adaptive capacity through technological and economic development. This view considers known damages as the most reliable basis for decisions and emphasizes present effects over future gains or losses. The hierarchist is characterized by strong group cohesion with binding regulations for social relations (a society in which roles are described) and considers nature to be in equilibrium. This perspective coincides with the view that impacts can be avoided with proper management and the search for a balance between manageability and the precautionary principle. The egalitarian has strong group cohesion (relationships) and considers nature to be fragile and unstable. This vision gives high priority to the precautionary principle and equal importance to present and future effects. The fatalist is characterized by weak group cohesion and binding regulations, and considers nature as uncountable. This perspective acts on his own, feels a victim of exogenous rules which cannot be influenced, and experiences the world as being governed by chance. The hermit (also defined as autonomist) escapes any influence from society or social level and detaches him/her from what happens in the world.

Later, the theory has been further extended by other researchers, who added more attributes each linked with the different archetypes. For example, Thompson et al. (1990) integrated Holling‘s ‘myths of nature’ (Adams, 1995) and different management styles, while Jager et al. (1997) incorporated the

dilemma between benefit and risk, and global and local. Figure 1.5 gives a graphical presentation of the different perspectives, each with their own vision on nature and society.

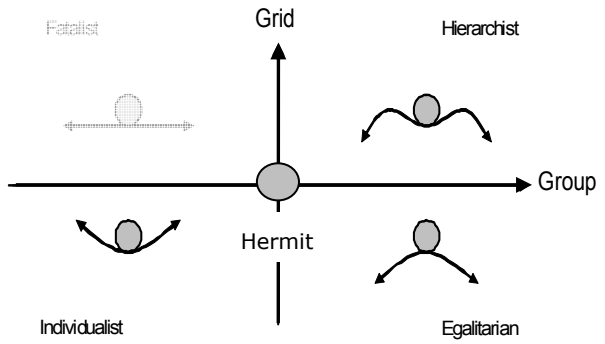

Figure 1.5. Graphical representation of the different perspectives (Douglas and Wildavsky, 1982, Thompson et al., 1990). The x-axis represents the extent to which an individual is incorporated into a group; the y-axis denotes the degree to which an individual's life is circumscribed by externally imposed prescriptions. The grey circle represent the natural system, as being inherently stable for the individualist, stable and manageable within limits for the hierarchist and vulnerable for the egalitarian perspective.

Van Asselt et al. (1996) proposed to use these insights and reasoning with respect to various perspectives as an organizing framework to address subjective judgment and cultural bias in modeling efforts. Different ethical attitudes are used to investigate alternative model routes for decision making. In general it is assumed that only the first three perspectives play part in environmental decision making and thus LCA: the individualist, hierarchist and egalitarian perspectives (Hofstetter, 1998, Hofstetter et al., 2000, Thompson, 2002). Each perspective represents a hypothetical practitioner, stakeholder or decision maker with differences in moral beliefs, concerns or interests that correspond to a specific set of preferences and contextual values that explains one's view on society and nature (Van Asselt and Rotmans, 1996, Hofstetter, 1998). Table 1.2 gives an overview of the visions related to the individualist, hierarchist and egalitarian perspectives. Both the fatalist and the hermit are considered to have no influence in environmental decision making.

Table 1.2. Overview of the different visions among the individualist, hierarchist and egalitarian perspectives (Thompson et al., 1990, Van Asselt and Rotmans, 1996, Jager et al., 1997, Hofstetter, 1998).

	Individualist	**Hierarchist**	**Egalitarian**
Vision on nature	Considers nature robust	Considers nature tolerant	Considers nature vulnerable
Level of knowledge	Only considers certain (proven) effects	Considers likely effects	Considers all known effects
Time horizon	Emphasizes present and short-term effects	Balanced time perspective	Current and future effects are considered equal
Vision on society	Economic output is market driven	Developments within limits of nature	Equality and social driven
Manageability	Adaptive management style	Preventive and comprehensive management style	Controlling and limited management style

1 Introduction

The Cultural Theory is attractive to be applied in LCA, as it both reflects visions on society and views on nature. Furthermore, several studies applied the Cultural Theory in LCA context, showing the practicability of the approach (e.g., Janssen and Rotmans, 1995, Jager et al., 1997, Hofstetter, 1998, Goedkoop and Spriensma, 1999, Frischknecht et al., 2000). The value of applying the Cultural Theory is that it (i) provides an organizing framework to implement a set of value choices, compatible with a perspective or world vision reflecting one's view on nature and society, (ii) stimulates consistent implementation of choices throughout the whole decision process, (iii) provides the practitioner an idea to what extent value choices affect the results of an LCA, and (iv) motivates decision makers and analysts to include value choices in a transparent way. However, the use of the Cultural Theory is often criticized. A common misunderstanding of the Cultural Theory is that every individual fully fits into a certain perspective. Most people would at best fit somewhere between two or different perspectives and switch perspective depending on their task, and role in society (Janssen and Rotmans, 1995, Thompson et al., 1990). As indicated by Hofstetter (1998), there are other typologies next to the Cultural Theory, which can be applied in the context of environmental modeling as well. Below a short overview is given of other theories which indentify values as precursors to environmental beliefs and actions.

Schwartz Value Theory identifies a comprehensive set of 10 different types of values as cognition representation of three universal requirements across cultures: biological needs, interactional requirements for interpersonal coordination, and societal demands for group welfare and survival (Schwartz, 1992). These values are mapped according to their conflicts, using two dimensions, and places tradition versus openness to change, and self transcendence versus self-enhancement Each dimension encompasses several value orientations that people in all cultures recognize. Another theory which looks at the influence of different factors on the perception of risk is the psychometric paradigm (developed by Slovic, Fischhoff and Lichtenstein; in Marris et al., 1997). This theory is an individualistic psychology approach which (initially) sought to identify the numerous factors responsible for individual perceptions of risk. The approach links risk perceptions to various cognitive and social mechanisms and found that 'dread' and 'novelty' are the two major factors explaining the variance of risk perception (Sjöberg, 2003). There are more approaches available, such as the work of Ulrike Beck who offers insights into the social and political basis of risk perception (Beck, 1992) and the work of Dake (1991) which combines the two theories described above by linking different perceptions of risk to the world views described by the Cultural Theory. Several studies give an overview or compare these different theories (e.g., Sjoberg, 1996, Hofstetter, 1998, Roser-Renouf and Nisbet, 2008).

1.4 Aim of this thesis

The uncertainties in LCA sketched above, can have a major influence on LCA outcomes. However, uncertainties in life cycle impact assessment are barely recognized and addressed so far. Therefore, this thesis focuses on uncertainties within life cycle impact assessment modeling.

The goal of this PhD thesis is to assess uncertainties in life cycle impact assessment. The focus is on (i) a number of impact categories assessing ecosystem health with relatively high uncertainty, namely climate change, land use and ecotoxicity, and (ii) uncertainties from value choices in impact categories addressing human health.

Within these topics the following three research questions are tackled:

1. What are the sources of uncertainty in impact assessment modeling of ecosystem health, in particular for ecotoxicity, land use and climate change?
2. What are the uncertainties deriving from value choices made in impact assessment modeling of human health and how can the Cultural Theory be applied to quantify these uncertainties?
3. What are the practical implications of value choices within impact assessment modeling?

The background of these topics is further discussed below.

Ecotoxicity

Ecotoxicity refers to the potential of chemical emissions to affect ecosystems. In terms of biodiversity these effects can be expressed using as indicators potentially affected fraction (PAF) or potentially disappeared fraction (PDF) of species (Klepper, et al., 1998, Muller-Wenk, 1998). The effects of ecotoxicity are not commonly addressed in LCA due to the expected high level of uncertainties in the impact assessment (van Zelm et al., 2009, 2010). An overview of the uncertainties in ecotoxicity, most recent developments and recommendations will help practitioners to better implement this impact category in LCA.

Land use

Land occupation is defined as the use of a certain area for human activities such as storing materials or waste and production of agricultural products or resources (Muller-Wenk, 1998). The level of damage is measured against a chosen reference or baseline land, for example the natural state of an area without human interactions (Milà i Canals et al., 2007). The effects at endpoint level can be quantified using the PDF as indicator for biodiversity loss (Muller-Wenk, 1998). The use of this indicator is developed and implemented in LCIA for land use by using the species area relationship: $S=c*A^z$ with S the number of species, A the area occupied or transformed, c the species richness factor and z the species accumulation factor (Koellner, 2000, 2008, Koellner and Scholz, 2007, Schmidt, 2008). Several

parameter uncertainties and key assumptions are incorporated in the species area relationship and the calculation of CFs for land use. There is a need to quantify these types of uncertainties on the CF level.

Climate change

It is expected that species can become extinct due to changing temperature, precipitation and seasonality (Thomas et al., 2004). Concerning human health impacts, several studies show that climate change results in an increase of various diseases, such as malaria and diarrhea (Mcmichael et al., 2003, Mcmichael and Woodruff, 2006, Patz and Campbell-Lendru, 2005). Only a few methods assess the damage of greenhouse gas emissions towards humans and ecosystems in life cycle assessments (Steen, 1999, Goedkoop and Spriensma, 1999, Tol, 2002). However, these methods contain several limitations in addressing the influence of greenhouse gas emissions at the endpoint level: (i) only a limited number of human health impacts are included, (ii) ecosystem health is neglected or handled in a very simplistic way and (iii) uncertainties are hardly addressed. Therefore, there is a need to develop new CFs for greenhouse gas emissions at the endpoint level for effects on both human health and terrestrial ecosystems. At the same time, uncertainties in the modeling procedure need to be quantified.

Human health

Human health is affected by a range of environmental impacts, e.g., toxicity, climate change, ozone depletion, water scarcity and particulate matter formation. The impact on human health per unit intervention is generally assessed using an environmental impact indicator that allows aggregation of different health effects, expressed as disability-adjusted life years or DALYs (Murray and Lopez, 1996b). The calculation of endpoint indicators involves several uncertainties and assumptions. Some impact assessment methodologies do handle uncertainties arising from value choices by applying the Cultural Theory, but in a limited and not always consistent way (e.g., Goedkoop and Spriensma, 1999, Goedkoop et al., 2008). Therefore, a broader implementation of the Cultural Theory in LCIA of human health is required.

1.5 Outline

After the introduction (Chapter 1), the PhD thesis starts with providing insights in the uncertainties that arise from assessing the impacts from land use and ecotoxicity (Chapter 2 and 3). Chapter 2 presents the cause-effect pathways for both land use and ecotoxicity and gives an overview of the different developments taking place in these research areas. Furthermore, it outlines the various uncertainties that arise when calculating these impacts, such as missing information on cause-effect relationships and double counting with other environmental impacts. Chapter 3 tackles in more detail

the uncertainties in parameters and models within the calculation of the endpoint indicator for land use and applies the Cultural Theory as a framework to quantify uncertainties from value choices.

Chapter 4 presents the differences in CFs due to different value choices within the impact modeling of climate change via application of the Cultural Theory. New CFs for 63 greenhouse gas emissions at the endpoint level are developed. The new CFs are suitable to compare the impacts of greenhouse gases with other types of stressors for both human health and biodiversity. Particularly for impacts on biodiversity, greenhouse gas emissions have not been commonly included in LCA case studies up to now.

Chapter 5 investigates the influence of value choices for seven human health impact categories: water scarcity, tropospheric ozone formation, particulate matter formation, human toxicity, ionizing radiation, stratospheric ozone depletion and climate change. Existing models are consistently adapted to the described set of value choices and new CFs are calculated using the three cultural perspectives.

In Chapter 6 the three sets of CFs developed and presented in Chapter 5 are used to calculate the human health impact score for more than 700 products. The aim of this work is to investigate the consequences of value choices within impact assessment modeling on a range of products covering different product groups.

Chapter 7 presents the overall synthesis of the PhD thesis. Special attention is paid to a critical evaluation of the defined scenarios using of the Cultural Theory. It outlines the overall consequences of the results within this PhD thesis for policy and decision making, and provides recommendations on how to handle uncertainties from value choices in LCA.

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Chapter 2
Addressing land use and ecotoxicological impacts in life cycle assessments of food production technologies

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Abstract

Effects of land use and ecotoxicity are not commonly addressed in life cycle assessment on agricultural food production due to the expected high level of uncertainties in the impact assessment and a lack of available inventory data. This chapter provides an overview of the cause-effect pathways related to the release of toxic chemicals and physical land use practices caused by food production practices. It also discusses the background and application of several life cycle impact assessment methods that produce so-called characterization factors (CFs) to quantify the environmental effects of the agricultural activity occurring along the cause-effect pathways. Particular attention is paid to advances in the data and modeling of ecotoxicological and land use impacts that resulted in the development of a consensus model to calculate CFs for aquatic ecotoxicity and several models to calculate CFs for land use. Finally, for both ecotoxicity and land use modeling, a number of uncertainties are discussed and several requirements for improvement are proposed.

Keywords land use • ecotoxicity • characterization factors • life cycle impact assessment • midpoint • endpoint • cause-effect pathway • physical land use changes • pesticides

2.1 Introduction

Maintaining ecosystems is a great challenge for the agricultural sector. Compared to other economic activities, the agricultural sector has the drive to change the whole ecosystem it uses in order to optimize productivity or yield. This is achieved by four different routes: the application of toxic chemicals (herbicides and pesticides) to eliminate unwanted species, the application of fertilizers to change the nutrient level and acidity in the soil, the control of soil humidity, and physical activities to change the land. These activities have a multitude of wanted and unwanted consequences. Some of the unwanted consequences can be modelled, while others are very difficult to model and can only be monitored by using observational data. In this chapter we analyze the consequences of toxic chemicals to the ecosystem and the influences of physical changes, usually referred to as land use. The first effect is addressed with models while the second is addressed with observational data.

As the world population continues to grow from 6 billion in 2000 to 8.1-9.6 billion by 2050, ecosystem change is expected to prolong to meet the demand of food production (Sarukhán et al., 2005). One way to meet these demands is by land transformation through deforestation and loss of grassland. Of the total terrestrial surface 24% is taken by grassland and cropland (Sarukhán et al. 2005), and by 2030 it is expected to rise by a further 16% (OECD, 2008). However the deforestation rate is slowing down by restoration and replanting initiatives, still a net loss of 7.3 million hectares per year takes place (FAO, 2006). Another way to meet productivity demands is by increasing the yield through intensive use of pesticides. However, both land transformation and use of pesticides tend to reduce biodiversity. Therefore, tradeoffs between land use and ecotoxicity should be considered when analyzing the environmental impacts of various food production technologies over their entire life cycle (Bengtsson et al., 2005, Mansvelt et al., 1995, De Boer, 2003). This can be done with help of Life Cycle Assessment (LCA).

Land use and ecotoxicity are important impact categories within the agricultural production step of LCAs on food products (De Boer, 2003, Mattsson et al., 1998b), even more when intensive farming is compared to less intense farming (Mattsson, 1999, Williams et al., 2006). In the Life Cycle Impact Assessment (LCIA) of agricultural food production it is important to include both the acre and quality of the land used, together with the pesticide application. Next to pesticides, metal pollution from energy use in food production and processing can also have a large contribution to ecotoxicity (Berlin, 2002, Milà i Canals et al., 2006, Mouron et al., 2006, Zabaniotou and Kassidi, 2003). When the effects of land use or ecotoxicity are considered, in several case studies simply the amounts of pesticides, or land occupied or transformed are taken as an indicator (Blonk, 2006, Williams et al., 2008, Mattsson et al., 1998a, Basset-Mens and van der Werf, 2005, Berlin, 2002, Cederberg and Mattson, 2000, Stern et al., 2005, Weidema et al., 2008). Some specific LCA studies attempt to implement land use and ecotoxicity in a more complete way, however, each by applying a different methodology (Mattsson et

al., 2000, Brentrup et al., 2004a, Blonk, 2006, Cordella et al., 2008, Milà i Canals et al., 2006, Mouron et al., 2006). This indicates a high need for consensus and guidelines regarding the implementation of land use and ecotoxicity into LCAs for food production and processing.

This chapter provides an overview of methods to address land use and ecotoxicity in LCIAs of food production and processing. Hereby, guidelines on the application of methods in case studies are presented. Section 1.3 focuses on effects of land use and section 1.4 on effects of ecotoxicity. Both sections outline the potential related impacts, how these are identified in LCAs and presents available tools together with a number of case studies on food and food related products. For ecotoxicity we focus on environmental impacts outside agricultural fields as the direct effects of application of pesticides on agricultural soils are already included in land use indicators. Finally, fields of further research are identified and summarized in section 1.5. Section 1.6 will conclude with sources of further information.

2.2 Life cycle impact methods for land occupation and transformation

Cause-effect pathway

Land use refers to the occupation or transformation of a certain area for human activities, such as storing materials or waste and production of agricultural products or resources (Muller-Wenk, 1998). The physical consequences are multiple, such as fragmentation, direct loss in biodiversity, altered vegetation, soil degradation, changes in water regimes (discussed in Chapter 3) and differences in reflection capability of the earth surface (albedo). Figure 2.1 gives an initial insight in the complexity of the different consequences of land occupation and transformation, and how they are interrelated. Important effects of agricultural land occupation and transformation are reduced soil quality (Oldeman, 2000), and direct species loss (Mace et al., 2005, Sarukhán et al., 2005). Soil degradation results from compactation, erosion, salinization, depletion of minerals, nutrients, and organic matter. The removal of natural vegetation and deforestation are the main causes of soil degradation (43%), resulting in the main erosion routes being water erosion (55%) and wind erosion (28%) (Oldeman, 2000). After long term land clearance or extensive land occupation, it alters vegetation, water regulation, and agricultural productivity (Mantel and Engelen, 1997, Lal, 2001). The Food and Agriculture Organization (FAO) estimated a global loss of 5 to 7 million ha productive land every year due to soil degradation. In 2008, the land surface degraded was 24% (Bai et al., 2008), 3100 million hectare, reflecting an area 3.2 times larger than Europe. Regarding terrestrial biodiversity, land transformation due to agricultural activities is currently considered the main driver for species loss (Sarukhán et al., 2005, Clay, 2004, Mace et al., 2005, Schleuning et al., 2009). Furthermore, not all species have the same value for the ecosystem. Some species have key functions (keystone species)

and by their disappearance or extinction, the loss of the function of the ecosystem may be disproportionally higher compared to the disappearance of other species (Benedek et al., 2007).

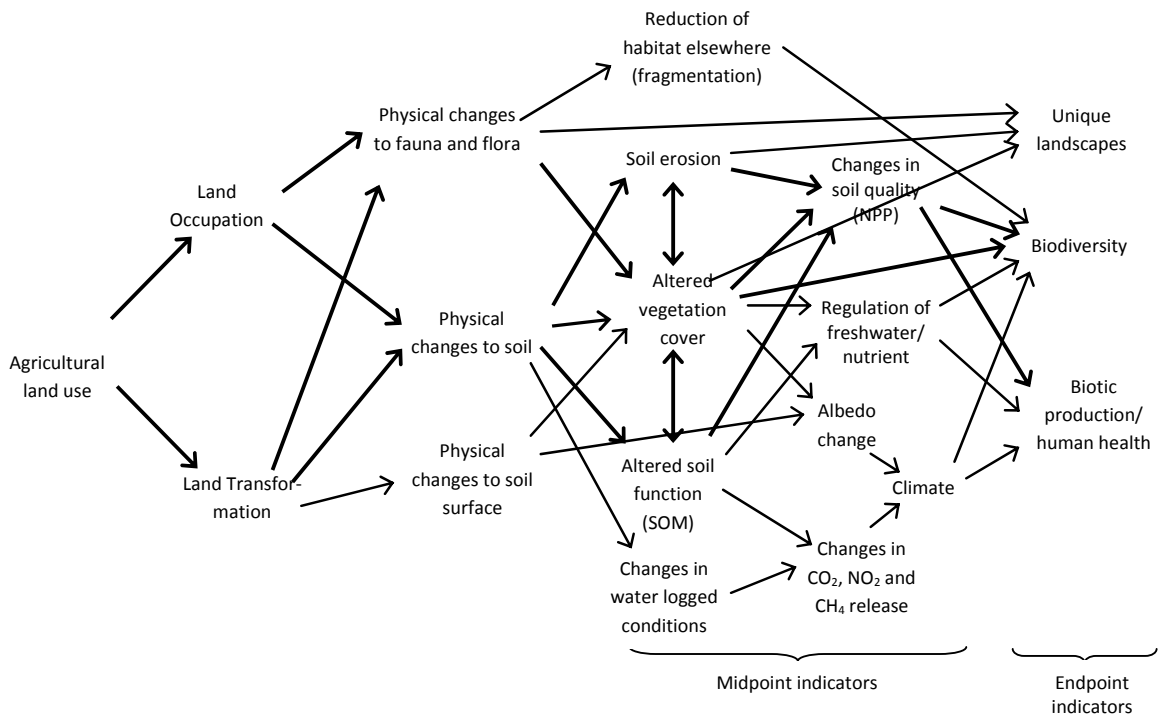

Figure 2.1. Overview of the cause-effect pathways of land use (adapted from Guinée et al., 2006, Hauschild et al., 2009). Midpoint indicators refer to indicators in the middle of the cause-effect pathway, while endpoint indicators refer to the actual damage resulting at the end of the cause-effect pathway. The thick arrows present the main pathways related to agricultural land use.

Framework

Within the framework of LCIA, the effects of land use can be divided in three activities: transformation, occupation, and restoration of land (figure 2.2). All three activities can be combined whereby occupation follows transformation and results in restoration. As a consequence of each activity nature is modified in a way that is defined as damaging. The level of damage is measured against a chosen reference or baseline land what refers to the non-use of the area, for example the natural state of an area without human interactions (Milà i Canals et al., 2007a).

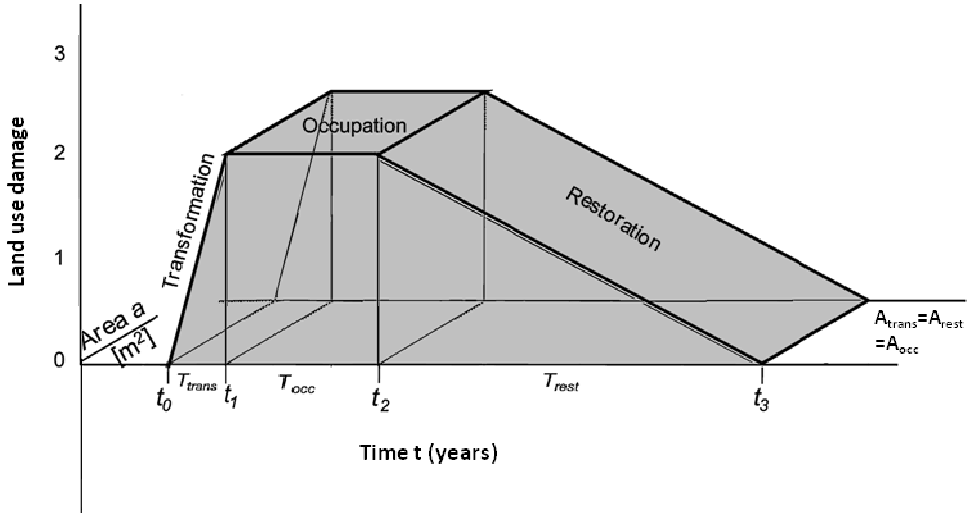

Figure 2.2. Land use activity subdivided into transformation, occupation and restoration. The grey volume represents the total damage score of a land use activity and is the integral over area A and duration T of each land use activity. At which level the damage starts and ends is determined by the chosen baseline (adapted from Koellner and Scholz, 2007a, 2007b).

The LCIA score for land occupation (IS_{occ}) can be expressed as:

$$IS_{occ} = CF_{occ,i} \cdot A_i \cdot t_i$$

where $CF_{occ,i}$ is the characterization factor (CF) for land occupation with land use type i; and A_i .t_i the area occupied (m^2) multiplied with the time of occupation by land use type i (yr). The LCIA score for land transformation or restoration ($IS_{trans/rest}$) is expressed as:

$$IS_{trans/rest} = CF_{trans/rest,i} \cdot A_i$$

where $CF_{trans/rest,i}$ is the CF for land transformation into land use type i or restoration to natural land; and A_i the area transformed or restored (m^2). The time needed for transformation or restoration is included in the CF and usually based on an estimate of the transformation and the restoration time. The CFs for occupying or transforming land surfaces quantify the physical consequences of the human activity, using one or more quality indicators chosen in the middle or at the end of the cause-effect pathway (see figure 2.1). Mila i Canals et al. (2007a) presents a list of possible midpoint and endpoint quality indicators that cover direct and indirect effects of land occupation and transformation.

In the midpoint approach, quality indicators, such as soil pH, soil organic matter, and net primary production, are applied. Midpoint indicators are well-suited for the comparison of different land use activities, but do not provide the possibility to compare the environmental impact of land use with other terrestrial ecosystem related impacts, such as acidification or eutrophication. The midpoint CFs for land occupation ($CF_{occ,i}$(midpoint)) and land transformation ($CF_{trans/rest,i}$(midpoint)) can be expressed as:

$$CF_{occ,i}(\text{midpoint}) = Q_b - Q_i$$

$$CF_{trans/rest,i}(\text{midpoint}) = Q_b - Q_i \cdot t_{trans/rest} / s_{i-o}$$

with Q_i, Q_o and Q_b the midpoint quality indicator for land use type i, the original land use type o and the baseline land use type b, $t_{rest/trans}$ the time needed for transformation or restoration and s_{i-o} the slope factor to reflect that restoration appear gradually. Note that the original and baseline land use type can be the same and that not every midpoint method considers the impact from transformation or restoration.

The endpoint approach refers to quality indicators for species richness (biodiversity), loss of unique landscapes, and reduction in biotic production. The change in species richness, or biodiversity, is commonly used as endpoint indicator within LCA and allows to integrate or compare the direct effects of land use with other environmental impacts (Muller-Wenk, 1998). The endpoint CF for land occupation ($CF_{occ,i}$(endpoint)) is expressed as the potentially disappeared fraction of species (PDF):

$$CF_{occ,i}(\text{endpoint}) = PDF = 1 - \frac{S_i}{S_b}$$

with S_i and S_b the number of species at land use type i and the number of species at baseline land use type b. The endpoint CF for land transformation or restoration ($CF_{trans/rest,i}$ (endpoint))is expressed as:

$$CF_{trans/rest,i}(\text{endpoint}) = (1 - \frac{S_i}{S_o}) \times t_{trans/rest} / s_{i\text{-}o}$$

with S_o the number of species at original land use type o. The number of species depends on the size of the area, also defined as species area relationship (SAR). The area size can be considered in the PDF calculations (Koellner and Scholz, 2008, Goedkoop et al., 2008). The use of biodiversity as endpoint indicator covers only part of the cause-effect pathway of land use (Hauschild et al., 2008a). For example, the loss of unique landscapes or the effects on human health through albedo climate regulation are not covered by this indicator.

Methods

Each method has its own quality indicators and thereby covers a certain part of the cause-effect pathway (figure 2.1). To be able to choose the preferred indicator we need to know what we want to preserve. Method developers can choose for indicators which reflect the naturalness of the land or for indicators which reflect the service of the land. In figure 2.3 we position the different methods to the extent they focus on impacts on “naturalness” or “system service”. For several methods we use ovals to position them on the axes because they clearly focus in one direction but it is vague to what extent

they consider the other vision. Three main clusters can be identified, (i) methods that apply indicators that focus on the naturalness of the system, such as PDF, (ii) methods that focus on system services, and (iii) the indicator 'area size' that does not have a focus on naturalness or system service.

Table 2.1 provides an overview of available land use methods. Case studies are introduced that apply the various methods and their main conclusions regarding land use are presented. The most simple way of considering land use in the life cycle of food production is by simply adding up all land occupation and transformation area size (Heijungs et al., 1992). This way of implementing land use is simple and robust, but lacks environmental relevance. Other methods give a more complete overview of the damage from land use by using multiple midpoint indicators (Bos and Wittstock, 2008, Baitz et al., 1998, Oberholzer et al., 2006, Mattsson et al., 2000, Michelsen, 2007). The use of several quality indicators next to each other requires substantial data input that is not always present. Furthermore, it creates multiple results, which cannot be easily aggregated. Mila i Canals (2007b) argues that the quality indicator soil organic matter (SOM) can be used as a single midpoint indicator for agricultural land use, and covers different impacts such as soil fertility, climate regulation and water regulation. However, several land use impacts are excluded by the SOM indicator, such as erosion, compactation and salinization. Endpoint methods use the loss of species diversity, expressed as potentially disappeared fraction of species (PDF), as indicator to assess the physical effects of land use (Muller-Wenk, 1998, Schmidt, 2008, Lindeijer, 2000, Weidema and Lindeijer, 2001, Koellner, 2000, Goedkoop et al., 2008, Koellner and Scholz, 2007, 2008, Itsubo, 2008).

Recently, eight different land use characterization methods with readily available CFs for land occupation and transformation were evaluated for the European Commission (Hauschild et al., 2008a). The method of Milà i Canals (2007b), which uses Soil Organic Matter as quality indicator, is identified as the best applicable midpoint approach, while ReCiPe2008 (Goedkoop et al., 2008) is preferred as endpoint approach. However, the most recent work of Koellner and Scholz (2008) contains new elements and data, such as the use of target species, what brings the method at least on the same level as ReCiPe2008.

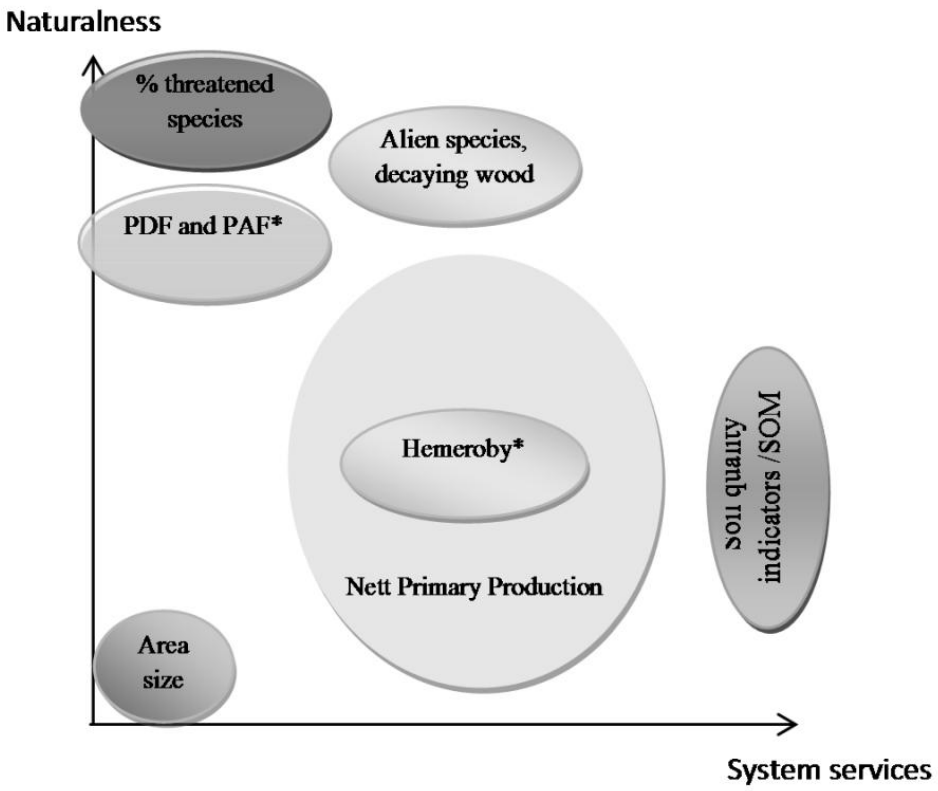

Figure 2.3. Different models scaled on level of naturalness and system service. *Measures the degree of human interventions by indicators such as the share of species and physical and chemical soil features.

Table 2.1. Land use models applied in LCIA together with food production and processing case studies performed with the models. When no case studies on food products were available others are introduced. M=Midpoint, E=Endpoint, O=Occupation, TR=Transformation and Restoration.

Model	Characteristics	M/E	O/TR	Implemented in	Region	Case studies	Main outcomes
Area size	Adds up land occupation and transformation separately (m^2) (Goedkoop et al. 2008; Heijungs et al. 1992)	M	O/TR	CML 92/2002 (Guinée et al. 2002) ReCiPe 2008 (Goedkoop et al. 2008)	-	Pig production (Basset-Mens and van der Werf 2005)	Organic pig production has highest land use score. Crop and feed production stage are dominant contributors to land use (>80%).
						Bread (Blonk 2006)	Most land use takes place in the wheat production.
Soil quality indicators	Combination of five to seven indicators: Emission filtering; Physical and Chemical Filtration; Ecosystem stability and biodiversity; Erosion stability; Filter, buffer function for water; Groundwater availability/protection; Net Primary production; Water permeability; soil organic matter; soil structure. Soil pH, accumulation of heavy metals, high soil content of phosphorus and potassium (Baitz et al. 1998; Bos and Wittstock 2008; Mattsson et al. 1998)	M	O/TR		Not specified	Vegetable oil crops (Mattsson et al. 2000)	The indicators erosion, SOM, soil structure, soil pH, P and K status and biodiversity provided a good picture of land use damage. The different indicators make it difficult to draw conclusions. The loss of SOM was the most serious threat for soybean. Soil compactation is a problem for rapeseed. Soybean and palm oil production give highest thread to biodiversity.
Hemeroby	Degree of human interventions (Natural Degradation potential, NDP) measured by the share of species, physical and chemical soil features and land use types (Brentrup et al. 2004b; Brentrup et al. 2004a)	M/E	O		Not specified	Wheat (Brentrup et al. 2004b)	Level of NDP decreases in relation to higher yield from fertilizer use. Applying more than 144kg N/ha didn't affect the yield and thus NDP. Aggregation with other impacts was difficult
General quality indicators and forest specific indicators	Biodiversity is measured by species richness, ecosystem scarcity and ecosystem vulnerability. Amount of decaying wood, areas set aside and introduction of alien tree species scaled from 0 to 3. Rotation time= restoration time (Michelsen 2007)	M	O/TR		Not specified	Logging of spruce (Michelsen 2007)	Quality indicators determined for taiga forest, and coastal forest. Logging coastal forest gives 40% more impact than logging taiga forest.
Soil organic matter (SOM)	(Milà i Canals et al. 2007)	M	O/TR		Not specified	New method, no case studies yet	
Biodiversity and Net Primary Production (NPP)	Transformation is not included yet, but can easily be incorporated in the existing method (Lindeijer 2000)	M/E	O		Global	Construction materials (Lindeijer et al. 2002)	Case study on brick, stone and wood, to test the method. Wood can score favorable or unfavorable, depending of the chosen baseline.
	Biodiversity measured by species richness, ecosystem scarcity and ecosystem vulnerability. Exchange of chemical substances added as extra factor. Transforation time is included in the occupation factor, together with a slope factor s_{i-o} (Weidema and Lindeijer 2001)	M/E	O/TR		Not specified	No case studies	
	Biodiversity measured as species loss due to local and regional effects, based on Japanese red species list. This is combined with net primary production data from the Chikugo model (Itsubo 2008).	M/E/	O/TR	LIME (Itsubo and Inaba 2003)	Japan	New method, no case studies yet	

Biodiversity	% of threatened plant species in a region. Land restoration is considered by adding a fixed restoration time of 30 years to the original occupation time (Muller-Wenk 1998).	E	O/ TR	Eco-indicator 99 (Goedkoop and Spriensma 1999), IMPACT 2002+ (based on Eco-indicator 99) (Jolliet et al. 2003)	Central Europe	Beer production (Cordella et al. 2008)	EI99 was used. Barley cultivation was main contributor for land use. Although the uncertainties for this impact were high.
						Meat products (Blonk et al. 2007)	EI 99 was used. Farming and feed production stages are dominant contributors to land use. However, the effects of cheap grazing in natural areas should be considered and animal welfare is close related to the area occupied. Both could not analyzed in a quantitative way and thus worst case results are presented.
	Potential disappeared fraction of vascular plant species combined with fraction of land available (Koellner 2000). Smidt used this approach to develop CFs for Malaysian and Indonesian forest systems (Schmidt 2008)					Meat and dairy products (Weidema et al. 2008)	IMPACT2002 was used. The impact of land occupation is dominant in the consumption of meat and dairy and contributes most for diary and beef.
	Potential disappeared fraction of all vascular plant species, threatened plant species, mosses and molluscs. Considering species area relationship. Restoration/transformation time is considered with the inclusion of a slope factor $s_{i\text{-}o}$ (Koellner and Scholz 2007, 2008)	E	O	Ecological Scarcity 2009 (Frischknecht et al. 2008)	Central Europe	Biofuels (Scharlemann and Laurance 2008)	The benefits from ethanol production from sugarcane diminishes when the total environmental impact is considered, including biodiversity loss and soil erosion. For 50% of 26 biofuel crops the total environmental impact is higher than fossil fuels.
		E	O/ TR	ReCiPe 2008 (Goedkoop et al. 2008)	Central Europe + Great-Britain	New method, no case studies yet	

Uncertainty

Several uncertainties arise within the application of land use methods. First, the applied quality indicators cover only part of the land use cause-effect pathway (figure 2.1). Most midpoint methods only refer to soil quality, while endpoint methods only take into account direct species loss. Several effects, like fragmentation, the loss of unique landscapes, or albedo climate regulation are not covered by these indicators.

Second, even though endpoint indicators allows to aggregate several ecosystem effects, such as land use, ecotoxicity and eutrophication, a certain risk in double counting environmental impacts occurs (Hauschild et al., 2008a). The loss of species does not only reflect the consequences of land use but also other impacts caused by farming, such as the effects of pesticide or fertilizer use (Goedkoop and Spriensma, 1999).

Third, the calculation of land use CFs requires the choice of a baseline. The historical land use state or potential land use state after restoration can be chosen but does not consider land evolution (Milà i Canals et al., 2007a). The average species richness of the region (Koellner, 2000), the maximum species richness (Weidema and Lindeijer, 2001), or another alternative system can also be considered as baseline (Milà i Canals et al., 2007a).

Fourth, often occupation occurs in an area that is already in use. Therefore, land transformation and restoration are mostly excluded in LCA of products. However, including these two land use activities within the impact category land use is mostly relevant when new conversions of natural land take place. For food production this is the case within continents where agriculture still expands, such as the soy bean production in South America.

Fifth, for each region the number of species differs. This makes the species richness indicator region dependent, which significantly influences the results. While the work of Koellner and others (Koellner and Scholz, 2007, , 2008, Koellner, 2000, Muller-Wenk, 1998) is developed for Central Europe, ReCiPe 2008 (Goedkoop et al., 2008) uses a combination of Swiss and British data, LIME is based on Japanese species (Itsubo, 2008), and Schmidt (2008) introduced Malaysian land use types. To make the LCIA of land use globally applicable more region specific CFs need to be developed.

Sixth, the species area relationship applied to calculate the endpoint CFs makes the calculations area dependent. Smidt (2008) calculates CF for an area size of 100m^2, Koellner and Scholz (2008) apply an area size of 1m^2, and Goedkoop et al. (2008) an area of 10,000m^2. Which area size to use is not yet standardized.

Finally, by using the overall species loss as indicator, no differentiation between species who contribute more to the ecosystem than others (keystone species) is made. To differentiate between vulnerable species, the loss of target species can be considered as species richness indicator. Both the LIME method (Itsubo, 2008) and the work of Koellner and Scholz (2008) give the possibility to follow this approach.

2.3 Life cycle impact methods for ecotoxicity

Cause-effect pathway

Ecotoxicity refers to the potential for biological, chemical, or physical stressors to affect ecosystems. The term was first outlined by Truhaut (1977), who defined it as "the branch of toxicology concerned with the study of toxic effects, caused by natural or synthetic pollutants, to ecosystems, animals (including human beings), plants, and microbial communities". Research within ecotoxicology is being used to set environmental regulations. Legal environmental quality criteria are set based on generic risk limits for toxic compounds for water, sediment and soil, derived in ecological and human risk assessments (Sijm et al., 2002). Although the pollution peaks in surface waters in the 1970s have now largely subsided in the western world due to strict toxic-chemical regulations the problem of pollutants is still with us today. Environmental quality standards are being exceeded at many sites (Posthuma et al., 2008). In addition to well-identified spots, a diffuse pollution, defined as the chronic presence of mixtures of toxic chemicals over large surface areas, in concentrations exceeding generic and protective environmental quality standards for one or more compounds, covers vast areas of land, vast volumes of sediment, and many surface water bodies. The apparent magnitude of ecotoxicological effects (in terms of increased, diffuse contamination levels) creates major problems for policymakers, due to concerns in the general public, due to an array of regulations that suggest a need to manage ecological stressor impacts and pollution risks, for example in relation to protected species, and due to the uncertainty on the magnitude of local impacts (Kapo et al., 2008, Posthuma et al., 2008).

In intensive agricultural practice, pesticides can cause substantial impact on ecosystems. Previous research, focusing on mixture toxicity assessment, showed that there is a large variation among pesticides regarding their impact on freshwater ecosystems (De Zwart, 2005, Henning-de Jong et al., 2008). It is therefore important for agricultural production processes to find and apply alternatives to some of the currently used pesticides.

Figure 2.4 shows the cause-effect pathway for ecotoxicity impacts. Emissions to air, vegetation, water, or soil will affect a variety of species. Starting from cold-blooded organisms, chemicals can be accumulated along the food chain. The whole ecosystem is affected by toxic compounds, which, in

terms of biodiversity, can be expressed as the potentially affected fraction (PAF) or potentially disappeared fraction (PDF) of species. The ecotoxic effects that chemicals cause on the environment can be assessed up to the PAF or PDF, which is called the endpoint. All points earlier on the cause-effect pathway are referred to as midpoints.

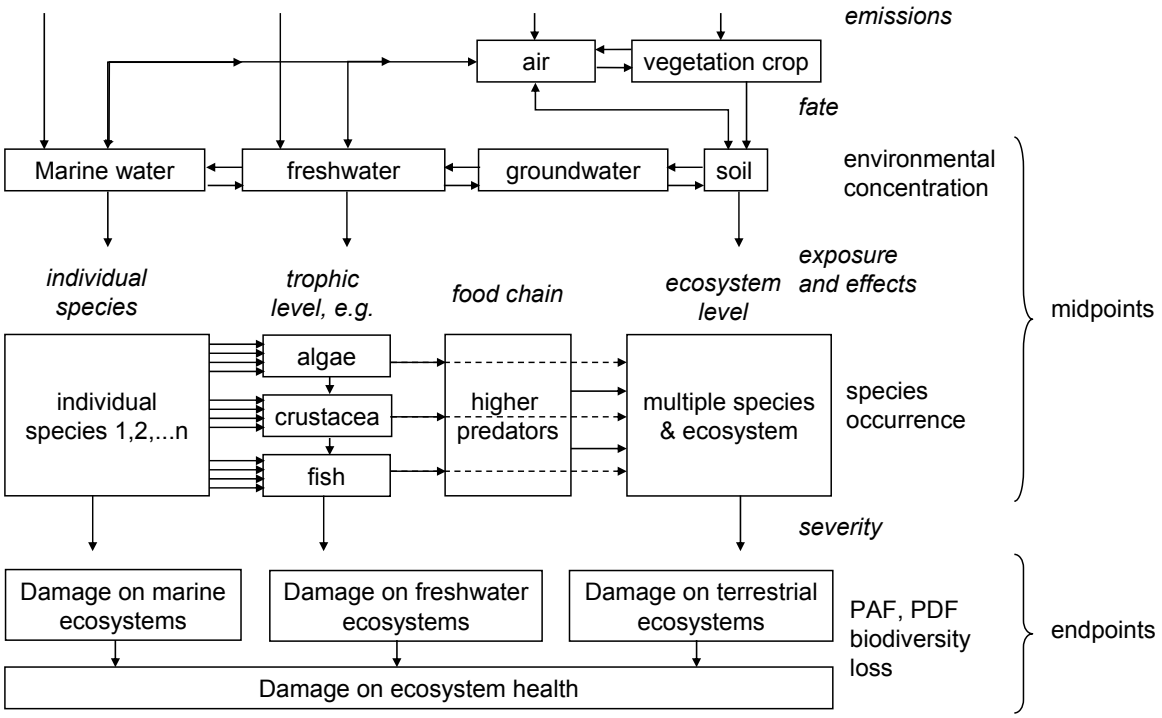

Figure 2.4. Cause-effect pathway for ecotoxicity (adapted from Hauschild et al., 2008).

Framework

The LCIA score for ecotoxicity of a chemical x in compartment j ($IS_{x,j}$) equals the emission of a chemical x to compartment i ($M_{x,i}$) multiplied by the CF:

$$IS_{x,j} = M_{x,i} \cdot CF_{x,i,j}$$

where $CF_{x,i,j}$ is the ecotoxicological CF of chemical x emitted to compartment i and transported to compartment j (e.g. in $m^3.yr.kg^{-1}$). To estimate pesticide emissions from field application, models can be used, such as PestLCI (Birkved and Hauschild, 2006).

The CF accounts for the environmental persistence (fate), and ecotoxicity (effect) of a chemical:

$$CF_{x,i,j} = FF_{x,i,j} \cdot EF_{x,j}$$

$FF_{x,i,j}$ represents the compartment-specific fate factor that accounts for the transport efficiency of substance x from compartment i to, and persistence in environment j (FF in yr), and $EF_{x,j}$ is the effect factor of chemical x in compartment j.

The environmental fate factor is defined as the change in the steady state concentration in an environmental compartment due to a change in emission (e.g. Huijbregts et al., 2005):

$$FF_{x,i,j} = \frac{V \cdot dC_{x,j}}{dM_{x,i}}$$

in which V is the volume of environment j (m^3), $dC_{x,j}$ is the change in the steady state dissolved concentration of substance x in environment j ($kg \cdot m^{-3}$), and $dM_{x,i}$ is the change in the emission of substance x to compartment i ($kg \cdot yr^{-1}$). Emission compartments commonly included in ecotoxic evaluations within LCIA are urban and rural air, freshwater, seawater, and agricultural and industrial soils. Environmental receptors generally identified are terrestrial, freshwater, and marine environments (Margni et al., 2002, Rosenbaum et al., 2008). FFs are generally calculated by means of 'evaluative' multimedia, multi-pathway fate and exposure models, such as CalTOX (McKone, 1993), IMPACT 2002 (Pennington et al., 2005), and SimpleBox (Den Hollander et al., 2004).

The effect factor is defined as the change in potentially affected fraction of species (PAF) due to a change in concentration in compartment j:

$$EF_x = \frac{dPAF}{dC_x} = \frac{dPAF}{dTU_x} \cdot \frac{dTU_x}{dC_x} = S_{PAF} \cdot \frac{1}{10^{\mu_x}}$$

where EF_x represents the effect factor of substance x ($m^3 \cdot kg^{-1}$); and dTU is the change in toxic units. The PAF-value expresses stress on ecosystems due to the presence of a single chemical or a mixture of chemicals. A PAF reflects the fraction of all species that is expectedly exposed above a certain effect-related benchmark, such as the Effect Concentration for 50 percent of species (EC_{50}) (De Zwart and Posthuma, 2005). S_{PAF} is the slope factor of the potentially affected fraction of species.

Two main classes of methods are currently identified for the calculation of the slope factor S: (a) methods assuming linear concentration-response relationships, and (b) methods accounting for the non-linearity in concentration-response relationships (Larsen and Hauschild, 2007, Pennington et al., 2004, Van de Meent and Huijbregts, 2005). In the non-linear methods S depends on the toxic mode of action of the chemical (Van de Meent and Huijbregts, 2005, Van Zelm et al., 2007). Figure 2.5 shows the linear and non-linear approach to derive S_{PAF}.

The chemical-specific part of the effect factor equals $1/10^{\mu}$ and reflects the inherent toxicity of a chemical, defined as the inverse of the average toxicity of a chemical, which is the concentration of substance x where 50 percent of the species is exposed above an acute or chronic toxic value ($kg \cdot m^{-3}$). μ_x is the average sensitivity of species to pesticide x ($g \cdot l^{-1}$), with sensitivity being expressed as an EC50 or another ecotoxicity test endpoint.

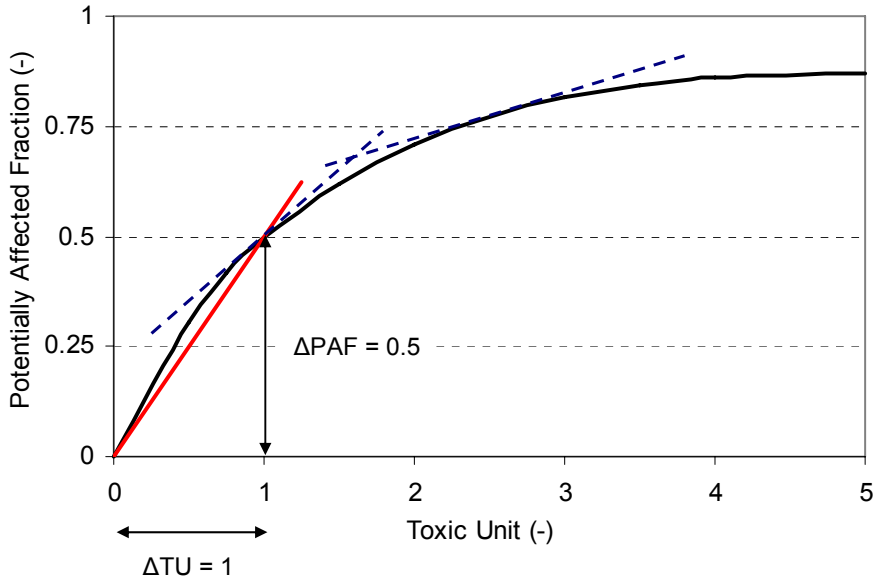

Figure 2.5. The linear and non-linear approaches for deriving the slope factor for potentially affected species (S_{PAF}). The straigth line refers to the linear approach in which ΔPAF/ΔTU is always 0.5, while the dotted lines refer to the non-linear concept (in which dPAF/dTU depends on the ambient TU of the chemical or mode of action under consideration).

Midpoint indicators are referred to ecotoxicity potentials and express the relative impacts of chemicals towards each other. The midpoint CFs can be used in comparison studies to understand which alternative(s) cause(s) most ecotoxicity. Different pesticide applications can for example be compared to see the environmentally best option to apply on a certain crop. Midpoints cannot, however, be used to compare different impact categories with each other. There is still debate going on regarding the differentiation between midpoint and endpoint characterization for ecotoxicity and the best damage assessment to be applied (Larsen and Hauschild, 2007, Rosenbaum et al., 2008). The PAF, based on EC50 data, may be regarded as the endpoint level. Posthuma and De Zwart (2006) showed for responses of fish species assemblages that the observed loss of species that can be ascribed to mixture toxicity closely matches the predicted risks based on EC50-data, at least in a relative sense (slope 1:1), and with a maximum observed fraction of lost species equal to the EC50-based ecotoxicity predictor variable. Due to these relationships, PAF as predictor parameter may have the diagnostic properties required to assess ecological conditions. Explicitly modeling further up to potentially disappeared fraction of species via a damage approach is possible as well (Larsen and Hauschild, 2007).

Methods

Ecotoxicity assessment models, namely BETR (MacLeod et al., 2001), CalTox (McKone, 1993), EDIP (Tørsløv et al., 2005), IMPACT2002 (Jolliet et al., 2003), USES-LCA (Van Zelm et al., 2009b), and Watson (Bachmann, 2006) all work with (part of) the framework mentioned above. The Task Force on ecotoxicity and human toxicity impacts, established under the LCIA program of the United

Nations Environment Program (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC), aimed at making recommendations for CFs for toxicity that incorporated broad scientific consensus. The method has to be simple enough to be used on a worldwide basis for a large number of substances. After a comprehensive comparison of the existing human and ecotoxicity characterization models mentioned above, the scientific consensus model USEtox was constructed. USEtox consists of a multi-media fate and exposure model and includes the linear method in the effect calculations (see Rosenbaum et al., 2008). Figure 2.6 shows the compartment setup of USEtox. As USEtox results from a consensus building effort amongst related modellers, the underlying principles reflect common agreed recommendations from these experts. Moreover, the model addresses the freshwater part of the environment problem and includes the vital model elements in a scientifically up-to-date way. For example, for LCA comparative reasons average toxicity among species is taken as a basis and not the most sensitive species. Furthermore, chronic EC50 data are prioritized. USEtox can be considered an endpoint model as the factors express the potentially affected fraction of species integrated over time and volume per unit mass of a chemical emitted (PAF $m^3 \cdot day \cdot kg^{-1}$).

Table 2.2 provides an overview of ecotoxicity methods available, with their characteristics and in which methodology they are implemented. Furthermore, case studies are listed that apply the various methods, including their main conclusions regarding ecotoxicity.

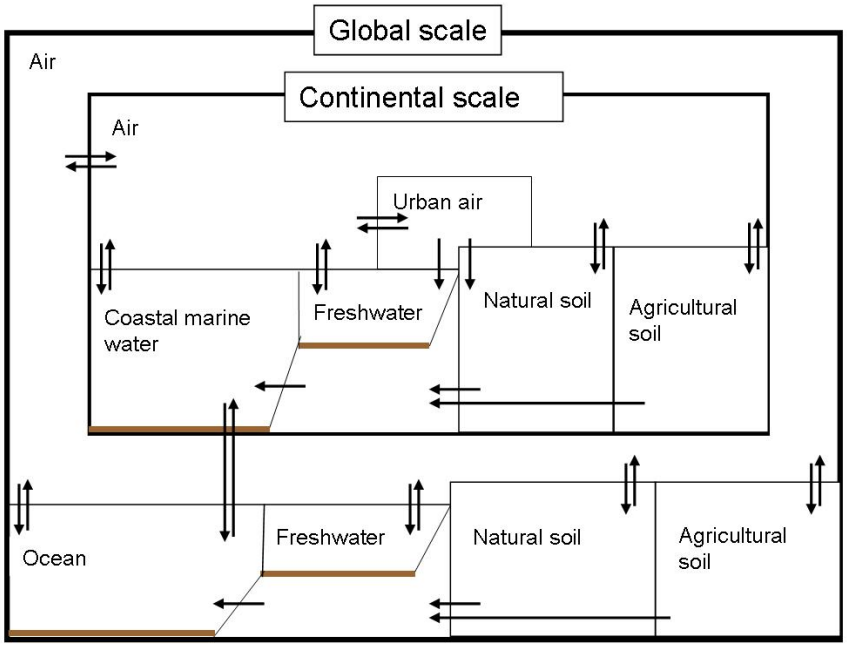

Figure 2.6. Compartment setup of the USEtox consensus model (adapted from Rosenbaum et al., 2008).

Table 2.2. Ecotoxicity models applied in LCIA, together with food production and processing case studies performed with the models. M= Midpoint, E= Endpoints.

Model	Characteristics	M/E	Region	Implemented in	Case studies	Main outcomes
CalTOX	Multi-media fate and exposure model. Linear effect method based on NOECs (McKone, 1993)	M	US	TRACI (Bare et al., 2002)		
EDIP	The fate assessment is simplified, i.e. no intermedia transfer processes are included. Linear effect method based on the geomean of trophic levels, applying chronic data. Spatially differentiated exposure factors for Europe. Marine compartment excluded. (Tørsløv et al., 2005)	M	World	EDIP 1997/2003 (Hauschild and Potting, 2005, Hauschild and Wenzel, 1998)	Apples (Milà i Canals et al., 2006)	Comparison of 5 orchards. Pesticides and metals in water included. Aquatic ecotoxicity dominated by emissions of heavy metals related to energy consumption
					Fish (Thrane, 2006)	One tuna fish. Marine ecotoxicity of biocides from anti-fouling agents. Fishing stage in life cycle dominant contributor to ecotoxixity.
IMPACT 2002	Multi-media fate and exposure model. Linear effect method based on EC50s. Uncertainty estimates and normalization factors included (Pennington et al., 2005)	M/E	Europe	IMPACT 2002+ (Jolliet et al., 2003)	Apples (Mouron et al., 2006)	Variability of impacts between fruit farms. Pesticides and heavy metals cause large impacts.
					Meat and dairy (Weidema et al., 2008)	Environmental impacts of consumption in EU-27. Copper emissions to soil from pig and dairy farming need to be reduced.
			Japan	Lime (Itsubo and Inaba, 2003)		
			Canada	LUCAS (Toffoletto et al., 2007)		
USES-LCA	Multi-media fate and exposure model. Linear effect method based on EC50s. Possibility to apply non-linear effect method for freshwater ecotox (Van Zelm et al., 2009b)	M/E	Europe	Eco-indicator 99 (Goedkoop and Spriensma, 1999)	Beer (Cordella et al., 2008)	Case study applying endpoint indicators. Ecotoxicity minor contribution.
					Tuna (Hospido and Tyedmers, 2005)	Marine ecotoxicity important impact category. Metals in diesel and anti-fouling paint are polluters.
					Fish (Thrane, 2006)	Eco-indicator 99 verified conclusions obtained with EDIP (see above)
				CML2002 (Guinée et al., 2002)	Cane sugar (Ramjeawon, 2004)	Herbicide loss during cane cultivation sole contributor to aquatic toxicity.
					Industrial milk (Høgaas Eide, 2002)	Comparison of three Norwegian dairies. Emissions of heavy metals in waste management phase most important for ecotoxicity. Small dairy greatest environmental impact.
				ReCiPe 2008 (Goedkoop et al., 2008)	New method, no case studies yet	
BETR	Multi-media fate and exposure model. No effect part (MacLeod et al., 2001)	M	Europe/ World			
Watson	Multi-media fate and exposure model. No effect part (Bachmann, 2006)	M	Europe			
USEtox	Multi-media fate and exposure model. Linear effect method based on (chronic) EC50s. Developed from the 6 above mentioned methods, this consensus model receives broad scientific agreement (Rosenbaum et al., 2008)	M	World		New method, no case studies yet	

Uncertainty

Previous LCA case studies show a relatively large uncertainty range for freshwater ecotoxicity, compared to other (nontoxic) impact categories (Geisler et al., 2005, Huijbregts et al., 2003). Geisler et al. (2005), in their study on plant protection products, state that before the freshwater ecotoxicity impact scores are used in decision support, measures to reduce uncertainty have to be taken. Uncertainty in ecotoxicological CFs is taken into account in various researches (Huijbregts et al., 2000, Payet, 2004, Van Zelm et al., 2009a, Van Zelm et al., 2010). These researches show that the main uncertainty is related to the effect factor, and specifically in the availability of reliable species toxicity effect data and the choice of the slope factor. Concerning the latter point, it is considered debatable, whether the linear slope factor (S_{PAF}) of 0.5 is the best option to apply (Larsen and Hauschild, 2007). Van Zelm et al. (2009a) provided effect factors with non-linear slope factors for 397 pesticides with a focus on the toxic mode of action. From a conceptual point of view, the nonlinear slope factor can be preferred as it allows for addressing nonlinear concentration–response relationships. However, the nonlinear method is more complex than the linear method and has a high data demand. More research in this area is therefore still needed.

Midpoint indicators are calculated in all food case studies, except in a case study on beer by Cordella et al. (2008). There is a need to apply common agreed endpoint indicators that express the actual damage, such as the potentially affected fraction of species, caused by the chemicals (Brentrup et al. 2004a). With the recent developments in endpoint models and ongoing research to the slope factor and damage indicators, application of endpoint CFs will become more common.

Exposure to higher predators due to bioaccumulation of chemicals along the food chain has not been addressed in LCA so far. As chemicals can accumulate in food chains causing other impacts on warm-blooded organisms than on cold-blooded, there is uncertainty attached to the exclusion of bioaccumulation and research in this area is needed as well.

Main ecotoxicity pollutants in food production and processing are pesticides and metals, which have their own specific qualities and properties that lead to uncertainties in LCA modeling. In current LCIA ecotoxicity models, degradation of a chemical is taken into account by following disappearance of the parent compound only. Many pesticides, however, are known to transform in the environment to degradation products that are also harmful to the environment, in some cases even more than their parent compounds (Fenner et al., 2000, Gasser et al., 2007). Van Zelm et al. (2010) quantified uncertainty attached to the exclusion of transformation products of a number of pesticides in freshwater ecotoxicological effect factors. They show that for several pesticides transformation products cannot be disregarded as they can damage the aquatic environment to a large extent. The fate modeling of metals is still an unresolved issue and a source of large uncertainties (Rosenbaum et al., 2008). Strandesen et al. (2007) developed a new concept to include speciation in the fate modeling of metals. They concluded that

multi-species models need to be used to characterize the potential ecotoxicological impacts of metals, since the behaviour of metals cannot be addressed by a single-species model that assumes a fairly uniform behaviour of metals in very different model regions. This indeed, increases the need for spatially differentiated fate and exposure modeling (Strandesen et al., 2007).

2.4 Future trends

A number of future research trends are envisaged. First, endpoint indicators focusing on species disappearance allow to aggregate land use and ecotoxicity effects with each other, but also with climate change, eutrophication and acidification. However, there is a risk in double counting environmental impacts as the CFs for land use are derived from empirical data (species counts) which can also include other environmental impacts

Second, several specific methods are available to analyze land use effects. However, the regional dependency of land use, reduces the validity of applying these methods in local case studies. Within the existing land use methods the following improvements are required:

- Investigate the sensitivity of land use CFs towards the choice of baseline, input parameters of the species area relationship, and the application of target species
- Define land use types and the inclusion of different land use practices in more detail
- Derive CFs for developing countries as large food production takes place here (cassava, rice, palm oil)
- Improve insight in the influence of uncertainty in parameters and choices within the species area relationship of endpoint land use models
- Develop quality indicators that cover other parts of the cause-effect pathway than commonly considered, such as changes in unique landscapes

Finally, for the impacts of food production and processing on freshwater ecotoxicity, specific attention in further developments should be given to:

- Increase pesticide coverage
- Include transformation products for pesticides with harmful daughter products
- Modeling of metals in a more precise way
- Include parameter uncertainty in the estimates of the CFs
- Define, and model up to, an endpoint level that receives consensus among researchers

2.5 Conclusions

This chapter presented an overview of method developments that allow the assessment of environmental impacts caused by land use and ecotoxicity. Over the last years, large improvements have been made to enhance the methods and their way of interpretation. Progress in defining recommended practice is also made, particularly for aquatic ecotoxicity with the USEtox™ consensus. However, further testing of the methods with case studies is necessary for both land use and ecotoxicity models, including the assessment of uncertainties in the estimates of the CFs. It is remarkable that only a few case studies were found that consider both impact categories (see table 2.1 and table 2.2). Especially for agricultural products, it is important to compare and aggregate land use and ecotoxicity effects with special attention to avoid double counting of environmental impacts.

2.6 Sources of further information and advices

More information on land use in LCIA can be found in:

- Milà i Canals et al. (2007a), describing a framework for LCA of land use
- Milà i Canals (2007b), describing the Soil Organic Matter concept as midpoint indicator
- Koellner and Scholz (2007, , 2008), addressing a state-of-the art endpoint modeling method for land transformation and occupation
- Hauschild et al. (2008a), chapter 'Land use' (p.101-110), describing the evaluation and recommendation of land use models
- http://fr1.estis.net/sites/lciatf2/, describing task force 2 on natural resources and land use of the LCIA program within the UNEP-SETAC life cycle initiative. The task force focuses on improvements and consensus within land use characterization methods.

More information on ecotoxicity modeling in LCIA can be found in:

- Hauschild et al. (2008a), chapter 'Ecotoxicity' (p.94-101), describing evaluation and recommendation of ecotoxicity models
- Hauschild et al. (2008b), describing the consensus building process of the USEtox consensus model
- Rosenbaum et al. (2008), describing the USEtox consensus model
- http://fr1.estis.net/sites/lciatf3/, describing task force 3 on toxicity impacts of the LCIA program within the UNEP-SETAC life cycle initiative. The task force focuses on improvements and consensus within human and ecotoxicity characterization methods.
- More information on pesticide modeling in LCIA can be found in:
- Birkved and Hauschild (2006), describing how to estimate field emissions of pesticides

- Van Zelm et al. (2009a), addressing ecotoxicity endpoint modeling of pesticides in LCIA

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Chapter 3
Uncertainties in the application of the species area relationship for characterization factors of land occupation

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Abstract

Purpose. Uncertainties in land use damage modeling are recognized, but hardly quantified in life cycle assessment (LCA). The objective of this study is to analyze the influence of various key assumptions and uncertainties within the development of characterization factors (CFs) for land use in LCA. We assessed the influence on land use CFs of (i) parameter uncertainty, and (ii) the choice for a constant or land use specific species accumulation factor z and including or excluding regional effects.

Methods. A model framework was developed to analyze the uncertainties of CFs for six land use types and three agricultural practices. The CFs are expressed as potential disappeared fraction (PDF) of vascular plant species based on the species area relationship ($S=c.A^z$). The species area relationship describes the relation between the species number and area size, with help of the species accumulation factor z and the species richness factor c. A dataset representative for Great Britain was used to quantify both modeling choices and parameter uncertainty. Modeling choices were analyzed by defining three coherent scenarios, based on Cultural Theory perspectives. The parameter uncertainties of average species number and species accumulation factor z were quantified using Monte Carlo simulation.

Results and discussion. Pair-wise comparison of the CFs shows that 68-85% of the CFs significantly differ from each other within each perspective. It is found that the ranking of organic, less intensive and intensive land practices of each land use type is unaltered by the chosen model scenario. However, the absolute values of the CFs can change from negative to positive scores with an average difference of 0.8 PDF between the two extreme perspectives, i.e., individualistic and egalitarian. The difference between these scenarios is for 40% explained by the choice in z and for 60% by the choice in including regional effects. Within the egalitarian and hierarchist perspective the species accumulation factor z is for more than 80% responsible for the parameter uncertainty.

Conclusions. Modeling choices and uncertainties within the species area relationship hardly change the ranking of the different land practices but largely influence the absolute value of the CFs for land use. The absolute change in the land use CFs can change the interpretation of land use impacts compared to other stressors such as climate change.

Keywords uncertainty analysis • life cycle assessment • land use • biodiversity • perspectives • species area relationship

3.1 Introduction and objective

Human land use activities are one of the dominant stressors for terrestrial species. Currently, 24% of the earth terrestrial surface is occupied by cultivated systems including cropland and grassland used to produce food, feed and fiber (FAO 2000; Sarukhán et al. 2005). By the year 2100, land use change is projected to have the largest global impact on species richness (Sala et al. 2000).

The impact of land use activities is also an important element to consider in the Life Cycle Assessment (LCA) of products. Within the framework of LCA, the effects of land use can be divided in three conceptual activities: transformation, occupation and restoration of land (Mila i Canals et al. 2007). Land occupation is defined as the use of a certain area for human activities such as storing materials or waste and production of agricultural products or resources; while land transformation and restoration are described as the processes which require transforming one land type into another (Muller-Wenk 1998). As a consequence of occupying or transforming land surfaces, ecosystems are modified in a way that is generally judged as damaging, such as loss of biodiversity or reduction in soil quality. For each activity characterization factors (CFs) can be calculated, based on a chosen quality indicator that describes the potential damage to the ecosystem. Mila i Canals et al. (2007) present a list of possible quality indicators that cover most direct and indirect effects of land use. Soil quality (Baitz et al. 1998; Mattsson et al. 2000; Oberholzer et al. 2006; Milà i Canals et al. 2007; Bos and Wittstock 2008), scaling (Sleeswijk et al. 1996; Jeanneret et al. 2006) and thermodynamic (Wagendorp et al. 2006) indicators work well for the comparison of different land use activities, but do not provide the possibility to compare the environmental impact of land use with other terrestrial ecosystem related impacts, such as acidification or eutrophication.

Aggregation across impact categories can be done by using indicators positioned at the end of the cause-effect chain. For ecosystem damage, Muller-Wenk (1998) proposes the potentially disappeared fraction (PDF) of species as endpoint indicator. This indicator measures the change in species diversity and is integrated over a certain time and area presented by the life cycle inventory. This approach is further developed and implemented by using the species area relationship with the species richness factor c and species accumulation factor z ($S=c*A^z$) for land occupation and transformation (Koellner 2000; Koellner and Scholz 2007; Koellner 2008; Schmidt 2008). Koellner and Scholz (2008) provide uncertainty estimates for CFs caused by empirical variation in the species richness data and limited sample size (parameter uncertainty). They also compare the results of a linear and non-linear calculation model (model uncertainty) and analyze the differences in results between the species groups plants, threatened plants, moss and mollusks (choice uncertainty). Koellner and Scholz (2008) and Smidt (2008) use a constant z value of 0.21 or 0.23 to calculate CFs for land use. However, the value of the species accumulation factor z is widely discussed (Rosenzweig 1995; Crawley and Harral 2001; Collins et al. 2002) and depends on the type of habitat (Hannus and von Numers 2008; Kallimanis et al. 2008), the taxa

(Humphreys and Kitchener 1982; Collins et al. 2002) and the size of the area (Lomolino 2001; Crawley and Harral 2001; Losos and Schluter 2000; Kallimanis et al. 2008; Harner and Harper 1976). A range of land use type and spatial scale specific z values are published (Crawley and Harral 2001; Manhoudt et al. 2005; Dolnik and Breuer 2008).

The objective of this study is to analyze various key assumption and uncertainties in the species area relationship and how they influence the CFs for land occupation. We focus on the influence due to applying a constant or a variable species accumulation factor z and the choice of including or excluding regional effects. Furthermore, we assess the parameter uncertainties caused by the uncertainty in the species accumulation factor z and in the average species number using Monte Carlo simulation. The CFs are expressed as PDF of vascular plant species. Three types of management practices of cropland, fertile grassland, infertile grassland, tall grassland, moorland and woodland are analyzed using the data of Countryside Survey 2000 (Defra 2000). Results are presented for three scenarios, quantifying the influence of coherent sets of value choices in the modeling procedure. These scenarios include the choice for (i) species accumulation factor z, and (ii) including or excluding regional effects.

3.2 Methodology

Model approach

Framework. The paper focuses on occupation of land. Land occupation causes a change in species richness within the occupied area compared with the baseline land (Mila i Canals et al. 2007). The baseline to which we measured the actual damage of land use activities was chosen to be the species richness on the type of land that will arise without human distortion (Koellner 2000; Vogtlander et al. 2004). Within continental Europe this is forest for 80-90% of the land (Stanners and Philippe 1995). During the occupation of land two effects are observed:

- The land quality on the occupied area itself changes, defined as local damage;
- The area size of surrounding baseline area and the occupying land use type changes, described as regional damage.

The total damage score can be defined as:

$$DS_{tot,i} = DS_{loc,i} + DS_{reg,i} \quad (1)$$

with $DS_{tot,i}$, $DS_{loc,i}$ and $DS_{reg,i}$ the total, local and regional damage score (PDF·m^2·yr) due to occupation with land use type i. The relative change in species richness for the local and regional situation can be calculated as:

$$DS_i = CF_i \cdot A \cdot t_i \quad (2)$$

where CF_i stands for the CF of land use type i; and $A_i .t_i$ the area occupied (m^2) multiplied with the time of occupation by land use type i (yr), as collected in life cycle inventories. In our study, the CF to assess the environmental damage of land use, is the potentially disappeared fraction (PDF) of species (Goedkoop and Spriensma 1999; Koellner and Scholz 2007), which is calculated by:

$$CF_i = \frac{\Delta S}{S_b} = \frac{S_b - S_i}{S_b} = 1 - \frac{S_i}{S_b} \tag{3}$$

with S_b the species number on the baseline land use type and S_i the species number on the occupied land use type I (unit less). The species number can be estimated by the species area relationship (Rosenzweig 1995; MacArthur and Wilson 1967). This relationship is described as:

$$S = c \cdot A^z \tag{4}$$

where S represents the species number (unit less), A the size of the area (m^2), c the species richness factor and z the species accumulation factor.

Local damage. The local damage describes the change in species richness on the occupied area compared with the species richness on the baseline land. Implementing equation (4) into equation (3) gives the local CF ($CF_{loc,i}$):

$$CF_{loc,i} = 1 - \frac{S_{i,l}}{S_{b,l}} = 1 - \frac{c_i}{c_b} \cdot A_o^{z_{i,l} - z_{b,l}} \tag{5}$$

with A_o the new area occupied (m^2), c_b and c_i the species richness factor of the baseline land use type and land use type i, $z_{b,l}$ and $z_{i,l}$ the species accumulation factor of the baseline land use type and land use type i on the local scale l.

Regional damage. The regional damage describes the marginal species change outside the occupied area. Occupying part of the baseline land can reduce the species richness in the region defined as the surrounding baseline area which can be occupied with other land use types but in our case is assumed to be forest area (regional effect I). Next to this, when the area occupied with land use type i gets connected with already existing land of the same type, the area of land use type i is enlarged (regional effect II) and may create a rise in species number on land use type i. The regional damage score for land use type i, considering both effects, is given by:

$$DS_{reg,i} = DS_{regI} + DS_{regII} \tag{6}$$

with DS_{regI} and DS_{regII} the damage scores for regional effect I and II. The marginal species loss for regional effect I is:

$$\Delta S_{b,r} \approx A_o \cdot z_{b,r} \cdot c_b \cdot A_r^{z_{b,r} - 1} \tag{7}$$

where A_r stands for the size of the surrounded region (m^2), c_b the species richness factor of the baseline land use type and $z_{b,r}$ the species accumulation factor of the baseline land use type with area size $A_r - A_o \approx A_r$ (m^2). More details can be found in appendix 3. Integrating equation (7) into equation (2), but reformulated for regional damage score (DS_{regI}), equals to:

$$DS_{regI} = A_o \cdot \frac{c_b \cdot z_{b,r} \cdot A_r^{z_{b,r}-1}}{c_b \cdot A_r^{z_{b,r}}} \cdot A_r \cdot t = z_{b,r} \cdot A_o \cdot t \tag{8}$$

The CF for regional effect I (CF_{regI}) equals to:

$$CF_{regI} = z_{b,r} \tag{9}$$

Regional effect II is calculated using the same approach as described by equation (7) and (8), but considering an enlargement taking place what results in a negative damage score. The regional CF ($CF_{regI+II}$), when considering both effects, is given by:

$$CF_{regI+II} = z_{b,r} - z_{i,r} \tag{10}$$

with $z_{b,r}$ and $z_{i,r}$ the species accumulation factor of the baseline land and land use type i, on the regional scale r.

Implementation

Land use types and management practices. The dataset used in the study is the Countryside Survey 2000 (Defra 2000). The survey gathered vascular plant species data within Great Britain in 1998 by detailed field observations in randomly selected 1 km squares. Altogether, 569 sample squares were visited, which contain over 18,000 vegetation plots. For each vegetation plot, the corresponding land use type and vascular plant species number was collected, and depending on the location classified into a specific broad habitat (Smart et al. 2003). We derived median species richness per land use type and calculated the 95% confidence level (CL) with the standard error (table 3.1). The land use types arable land, fertile grassland, infertile grassland, moorland grass and tall grassland were considered to be man-made and included in this paper. The land use type 'upland wooded' was considered the most natural woodland type and therefore used as baseline. An overview of the different land use types, plot types, amount of vegetation plots and the percentage of plots located in each broad habitat is listed in appendix 3 (table 3.4). A description of the land use types and broad habitats can be found in appendix 3, table 3.5.

The Countryside Survey 2000 (Defra 2000) presents species richness figures for three types of land use plots: (i) the species richness of the field core, (ii) the species richness at the inner margin and (iii) the species richness within the crop edge. Based on the fact that the species richness on arable land is strongly influenced by land use intensiveness (Wilson et al. 1999) and crop edges can act as refuge for species

disappearing on the crop field (Fried et al. 2009) we linked the species richness of the three different plot types to three types of land use intensiveness using the CORINE land-cover classification (EEA 1995). We assumed that the species richness of field cores corresponds with intensive fields without edges, the species richness of inner field margins corresponds with less intensive used areas that contain only small borders and the species richness of field edges corresponds with organic arable areas with plenty of edges, and small natural plots.

Species accumulation factor z. The species accumulation factor z is required to calculate the species richness factor c (equation 4) and the local and regional CFs (equation 5 and 7). In this paper, the calculations were done by using (i) variable z values as derived by Crawley and Harral (2001), and (ii) a constant z value of 0.25 (Crawley and Harral 2001). The variable z values are land use type specific and applied to spatial scales ranging from 10 m^2 to 10,000 m^2 (figure 3.1). For arable land, variable z values were available for a spatial scale of 100 to 200 m^2 only. The variable z values used to calculate the c values and CFs for each land use type are presented in the table 3.6 of appendix 3.

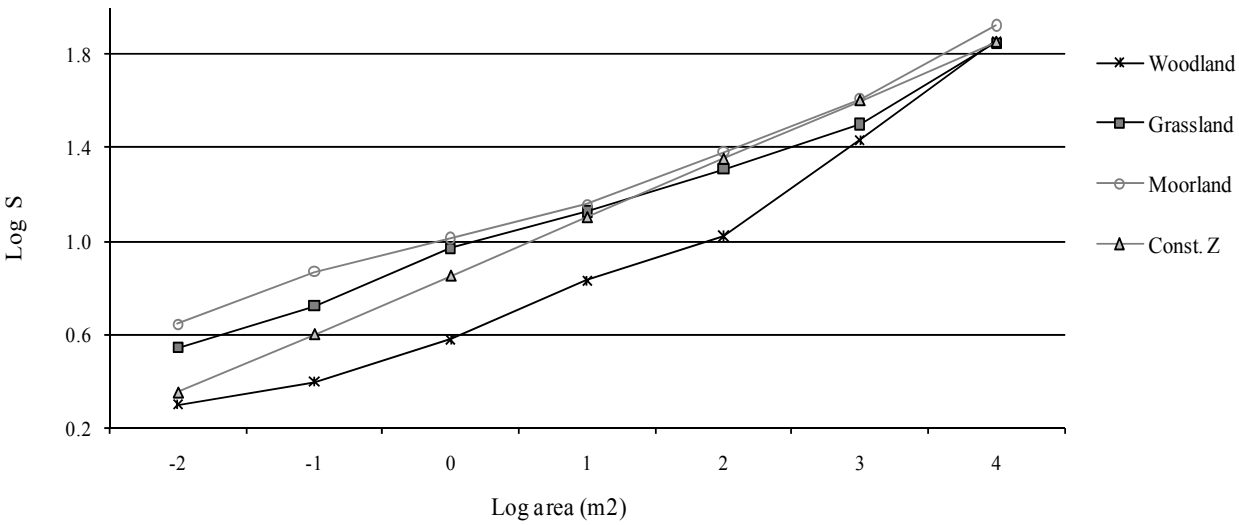

Figure 3.1. Scale and land use type dependency of the species area relationship. The z value determines the slope of the curve. With a log-log presentation a constant z value of 0.25 results in a straight line, while scale dependent z values results in area specific slopes. Data derived from Crawley and Harral (2001).

Species richness factor c. The species richness factor of the baseline and occupied land use type is required in the calculation of the local CF (equation 5). For each land use type two versions of the c value were derived by applying equation 4, using a species accumulation factor z corresponding to the respectively area size of the surveyed plot (variable z) and a constant z value of 0.25 (Crawley and Harral 2001). The size of the survey area and the counted species numbers are given by the Countryside Survey 2000 dataset (Defra 2000). Table 3.1 presents, per land use type and plot type, the c values for a constant and variable z value.

Table 3.1. Land use type specific values for species richness factor c, calculated with variable (var.) values for species accumulation factor z (hierarchic and egalitarian perspective) and a constant z value of 0.25 (individualistic perspective). The area sizes and median species numbers derive from Defra (2000).

Land use type	**Number of plots[a]**	**Plot size (m^2)**	**Species number**		**c value with var. z values**		**c value with const. z value**	
			Median	**95% CL**	**Median**	**95% CL**	**Median**	**95% CL**
Organic arable land	26	10	7.3	5.8-9.1			4.1	3.2-5.3
Less intensive arable land[b]	206	100	11.4	11.0-11.9			3.6	2.8-4.6
Intensive arable land[b]	465	200	5.1	4.8-5.4			1.4	1.0-1.8
Organic fertile grassland	212	10	11.5	10.8-12.2	8.1	7.5-8.7	6.5	5.6-7.3
Less intensive fertile grassland	33	100	13.1	10.5-16.1	5.0	3.9-6.4	4.1	3.0-5.8
Intensive fertile grassland	445	200	8.6	8.0-9.2	3.2	2.9-3.6	2.3	1.7-3.0
Organic infertile grassland	353	10	15.2	14.5-16.0	10.8	10.1-11.4	8.5	7.6-9.7
Intensive infertile grassland	458	200	19.1	18.3-19.9	7.2	6.5-8.0	5.1	3.9-6.7
Organic moorland grass	63	10	12.0	10.1-14.2	9.1	7.7-10.8	6.8	5.5-8.2
Intensive moorland grass	366	200	18.5	17.5-19.6	5.6	4.9-6.4	4.9	3.7-6.4
Organic tall grassland	646	10	10.9	10.6-11.2	7.8	7.4-8.1	6.2	5.4-7.0
Less intensive tall grassland	253	100	11.4	10.9-11.9	4.4	4.0-4.9	3.6	2.9-4.5
Intensive tall grassland	125	200	7.3	6.3-8.4	2.8	2.3-3.3	2.0	1.4-2.7
Intensive woodland	206	200	10.8	9.9-11.7	1.1	0.8-1.4	2.9	2.2-3.8
Base: Semi-natural woodland	41	10	11.4	9.8-13.3	6.6	5.6-7.6	6.5	5.3-7.8

[a]Number of plots used to calculate the median species number and 95% confidence level of the average species number
[b]No z value presented by Crawley and Harral (2001); a z value for field crops of 100 m^2 is taken from Manhoudt et al. (2005)

Perspectives. To handle value choices in a consistent way the Cultural Theory can be applied (Thompson et al., 1990, Hofstetter, 1998). Three cultural perspectives are generally used, i.e., the individualistic, the egalitarian and the hierarchist perspective (Goedkoop and Spriensma, 1999, Goedkoop et al., 2008). The individualist coincides with the view that mankind has a high adaptive capacity through technological and economic development, and only proven effects should be considered. The egalitarian coincides with the view that nature is strictly accountable and precaution is required. The hierarchical perspective coincides with the view that impacts can be avoided with proper management, and that the choice on what to include in the model is based on the level of (scientific) consensus. Table 3.2 gives an overview of the value choices included in our study and how we link them to the three perspectives we refer to in the discussion.

Dependency of the z value towards land use type and area size is widely acknowledged in the literature on species area relationships (e.g., Rosenzweig 1995; Crawley and Harral 2001). Therefore, we considered a variable z value for the hierarchist and egalitarian perspective. An area size of 10,000 m^2 is used, as this is largest available in the dataset. The use of a constant z value is a simplification of the model what makes it more robust, independent of the life cycle inventory data, and is assumed for the individualist perspective. A second value choice relates to the inclusion of the regional effect I and II. The regional effects are zero when a constant z value is used (see equation 10) and therefore not applicable for the individualist perspective. The egalitarian perspective followed worst case scenario what coincides with excluding environmental benefits and thus regional effect II, while the hierarchist included both regional effect I and II.

Monte Carlo simulation. The parameter uncertainty within the CFs derives from both the uncertainty in c values and z values. For the c values a t-distribution for the uncertainty in species number was applied and standard deviations were derived from the Countryside Survey 2000 (Defra 2000). For the species accumulation factor z a bounded normal uncertainty distribution (ranging from 0 to 1) was used. The uncertainty in area-dependent z values was quantified with a coefficient of variation of 0.05 (Crawley and Harral 2001). A constant z value was assumed to have a higher uncertainty (coefficient of variation of 0.1) due to the relative high variation in generic z-values reported in the literature. Monte Carlo simulations were performed in Crystal ball (Crystal ball 1998), applying 10,000 iterations for each simulation.

The difference between CFs was statistically tested by taking the covariance between the CFs into account in the Monte Carlo simulation. Depending on the perspective, covariance in the CFs can occur due to equal species accumulation factors z and an equal baseline land use type (table 3.1, 3.3 and 3.4). For instance, in the individualistic perspective the constant z-value was varied simultaneously for all land use types, as we assume in this case the same z-value for all land use types and practices included. For the hierarchic and egalitarian perspective, the same is true for the regional z-value of the baseline land use 'semi-natural woodland' and the z value of the different grassland land use types, which were simultaneously varied. The covariance in the CFs was accounted for by calculating the difference between pairs of CFs in the Monte Carlo simulations. If the difference between the pair of CFs was in > 95% of the runs negative or positive, we consider the CFs to be significantly different from each other ($\alpha = 0.1$, two-sided confidence interval).

Table 3.2. Combination modeling choices and uncertainty for the species accumulation factor z and the regional effect, expressed in three different cultural perspectives.

Value choice	**Individualist**	**Hierarchist**	**Egalitarian**
Species accumulation factor z	Constant z value of 0.25	Variable z (10,000 m^2)	Variable z (10,000 m^2)
Regional effect	No regional effect	Regional effect I+II	Regional effect I

3.3 Results

Table 3.3 shows the CFs for the individualist, hierarchist and egalitarian perspective. For arable land z values for a spatial scale of 10,000 m^2 were not available, preventing the calculation of CFs for arable land within the hierarchist and egalitarian perspective. Independent from the perspective chosen, we observe that intensive land use practice has the highest CF, followed by less intensive and organic land use practice of each land use type (table 3.3, 3.5, 3.6 and 3.7). Note that the occupation of land can result in a combination of a both local and regional effect and therefore the total CF (and PDF value) can become higher than one as referred to the land occupation.

Table 3.3. CFs for the individualist, hierarchist and egalitarian perspective, for six different land use types and three levels of land use intensiveness. The 95% confidence level (CL) is calculated using a t-distribution for the average species number S (table 3.1) and a bounded normal uncertainty distribution for the species accumulation factor z.

	CF Individualist		**CF Hierarchist**		**CF Egalitarian**	
Land use types	Median	95% CL	Median	95% CL	Median	95% CL
Organic arable land[a]	0.36	0.15-0.51				
Less intensive arable land[a]	0.44	0.31-0.54				
Intensive arable land[a]	0.79	0.73-0.83				
Organic fertile grassland	-0.01	-0.18-0.15	0.55	0.13-0.83	0.90	0.49-1.16
Less intensive fertile grassland	0.36	0.14-0.52	0.75	0.45-0.95	1.10	0.82-1.29
Intensive fertile grassland	0.65	0.56-0.72	0.87	0.68-1.02	1.22	1.04-1.35
Organic infertile grassland	-0.33	-0.56- -0.13	0.37	-0.17-0.72	0.72	0.20-1.05
Intensive infertile grassland	0.21	0.02-0.37	0.61	0.22-0.86	0.96	0.59-1.19
Organic moorland grass	-0.05	-0.32-0.16	0.65	0.28-0.90	0.97	0.62-1.21
Intensive moorland grass	0.23	0.04-0.39	0.83	0.59-1.00	1.15	0.93-1.31
Organic tall grassland	0.04	-0.12-0.18	0.58	0.17-0.84	0.93	0.54-1.17
Less intensive tall grassland	0.44	0.32-0.54	0.80	0.54-0.97	1.15	0.91-1.30
Intensive tall grassland	0.70	0.61-0.77	0.91	0.72-1.04	1.25	1.09-1.37
Intensive woodland	0.55	0.44-0.65	0.84	0.78-0.88	1.28	1.21-1.34
Baseline: Semi-natural woodland	0.00		0.00		0.00[b]	

[a]No z value presented by Crawley and Harral (2001)
[b]For the baseline, considering regional effect I is always in combination with regional effect II

We observe that the absolute values of the CFs differ depending on the perspective chosen. The individualist and egalitarian perspective present the lower and upper range of the three perspectives respectively, with an average difference of 0.8 PDF and a maximum difference of 1.05 PDF, i.e., the CF of organic infertile grassland varies from -0.33 PDF for the individualist perspective to 0.72 PDF for the egalitarian perspective. The hierarchist perspective falls between the individualist and egalitarian perspective. The median CFs for the hierarchist perspective (including regional effect I+II) range from 0.37 PDF to 0.91 PDF for the land use types other than the baseline. Compared to the hierarchist perspective, the exclusion of regional effect II within the egalitarian perspective results in CFs that are 32-44% higher (table 3.7 in appendix 3). When using a constant z value, the individualist perspective always results in lower CFs, in some cases leading to negative median CFs.

The calculated parameter uncertainty results in an uncertainty range of on average 0.20 PDF (95% confidence interval) for land use types other than the baseline, and vary between 0.04 and 0.54 PDF (table 3.3). For the hierarchist and egalitarian perspective the uncertainty derives for more than 85% from the regional variable z values, while the parameter uncertainty in median species numbers per land use type mainly clarifies the uncertainty within the individualist perspective. Accounting for the positive covariance between the CFs of the various land use types, we found that for the individualist perspective 85% of the CFs are significantly different, while for the egalitarian and hierarchist perspective respectively 68% to 80% of the CFs are significantly different ($\alpha = 0.1$, two-sided confidence interval). Tables 3.5, 3.6 and 3.7 in the appendix present for each perspective a matrix of ranked land use types (sorted constitutively from small to large CFs), together with the corresponding confidence levels of the difference in CFs.

3.4 Discussion

Within the calculations of the CFs for 15 different land use types the sensitivity towards the z-values and regional effects in the damage model was analyzed. The effects of land occupation was quantified as disappeared fraction of species to allow aggregation of land use damage with other ecosystem impacts such as climate change (De Schryver et al. 2009). Value choices in the damage model were assessed by establishing three scenarios following the Cultural Theory (Thompson et al. 1990; Hofstetter 1998) and parameter uncertainty was quantified through Monte Carlo simulation.

Perspectives. The results show that independent of the perspective, the intensive land use practice has the highest CF, followed by less intensive and organic land use practice of each land use type. However, the absolute values of the CFs can substantially differ, depending on the perspective chosen. The differences in results are caused by the choice of a generic or a land use specific species accumulation factor z, and the choice of including or excluding regional effects. The difference in CFs between the hierarchist and individualist perspective mainly derives from the choice in applying a variable or constant z value and is land use type specific. The choice of applying a constant z value is a way of simplifying reality. The constant z value can range from 0.12 for large human induced areas to 1.00 or even more for isolated islands, depending on several factors, such as the level of isolation and diverse habitats (Humphreys and Kitchener 1982; Rosenzweig 1995). Comparing the egalitarian and the hierarchist perspective, the difference in CFs is land use type independent and determined by the choice to include or exclude regional effects.

For the hierarchist and egalitarian perspective, the baseline z-value is higher or equal to the z-value of land use type i. This result in positive CFs, even when the corresponding species richness factor c of land use type i is higher than the c value of the baseline. The individualist perspective applies a constant z value, resulting in CFs that are determined by the difference in c_b and c_i. This is obviously also the case when the area-dependent z values z_i and z_b are approximately equal. Due to data limitations, this study only calculated CFs for an area size of 10,000 m^2. For the land use type arable land, an area-dependent z value for 10,000 m^2 was not available and thus CFs could not be calculated for the hierarchist and egalitarian perspective. An option for deriving area-dependent z-values per land use type could be by developing a mathematical relationship, mechanistic or empirical, between z and area size. Such a mathematical relationship is readily available for the United Kingdom (Crawley and Harral 2001), although not further specified per land use type. Further research in deriving land use specific general relationships between z and area size would be of added value and could also be used to derive CFs for other area sizes than 10,000 m^2.

In this study the Cultural Theory is used a framework to develop different scenarios and with this to evaluate effects from different visions on including regional effects and the use of variable z values. However, the developed scenarios are suggested default scenarios and can be adapted depending on the

vision of the practitioner. Furthermore, other value choices could be implemented as well, such as the choice of baseline and the type of species to consider.

The choice of baseline may also influence our results and can be linked to cultural perspectives as well. In this study we used 'semi-natural woodland' as default baseline without considering alternative baseline settings. However, the average regional species richness (Koellner and Scholz 2008) and the maximum species richness (Lindeijer 2000; Weidema and Lindeijer 2001) are also proposed as a baseline for land use in LCA context. The species richness of our baseline "natural land" equals the average species number for Great Britain, which ranges from 5.0 till 7.5 species/m^2 (AFE 2001; Crawley and Harral 2001). The maximum species richness in our dataset is found on the land use type 'organic infertile grassland', what corresponds to 8.5-10.8 species/m^2 and a z value of 0.35 instead of 0.44. The lower z value results in reduced CFs up to a factor of 2, when using the maximum species richness in the egalitarian perspective. We did not consider the choice of a maximum baseline in our model scenarios as it falls outside the scope of our study. In case of applying the cultural perspectives, we suggest to use (i) the average regional species richness for the individualist perspective as this is most local, (ii) 'semi-natural' land for the hierarchist perspective as this is most consensus, and (iii) the maximal species richness for the egalitarian perspective who believes any land occupation is damaging.

Another aspect that can be linked to the perspectives is the inclusion of threatened species or all species in the calculation of CFs (De Schryver et al. 2009). For instance, Koellner and Scholz (2008) provide information on the effect of land use on respectively all species and threatened species only. This allows differentiation between land use types with original species richness and species richness induced by human interference, for example natural woodland and artificial meadow. A subdivision in threatened species was not possible within our dataset, but is considered a relevant modeling step in a further specification of the perspectives.

Parameter uncertainty. The presented parameter uncertainty differs per perspective. For the individualist perspective the parameter uncertainty only derives from uncertainty in the species richness factors c and is higher than the uncertainty from local damage for the hierarchist and egalitarian perspective. This is due to the higher uncertainty range for the constant z value applied in the individualist perspective compared to the variable z values for the other two perspectives. However, when adding the area dependency and regional damage, the parameter uncertainty of the total CFs for the hierarchist and egalitarian perspectives becomes higher than for the individualist perspective that excludes these effects. The exception is, however, for intensive woodland with zero regional effects in all perspectives, because occupation with the same land use type as the baseline removes regional effects and excludes the area dependency due to the same z values (z_i and z_b) applied.

For the hierarchist and egalitarian perspective the parameter uncertainty is for more than 80% explained by the uncertainty in the regional species accumulation factor z. However, the uncertainty in the applied z

value can be larger than presented when the choice in area size would be reflected in the parameter uncertainty instead of considered as modeling choice.

Within each model scenario, the CFs of the different land use types do not always significantly differ. On average, two to four land use types with consecutive CFs do not significantly differ. The individualist shows the highest number of CFs which significantly differ due to the covariance in the applied constant z value.

Model uncertainty. Apart from model choices and parameter uncertainty, also model uncertainties can influence the outcomes of the study. First of all, land use CFs are highly region dependent. The use of data from Great Britain makes the results not applicable on a global scale but still adequate to analyze different model uncertainties, as done in this study. Second, it is important to note that the use of variable z values requires the size of land area under study. This is not always known in LCA, what includes extra model uncertainty and can largely influence the outcome of the CF. Third, within the formulas A_r-$A_o \approx A_r$ and $A_i+A_o \approx A_i$ we assumed the newly occupied area A_o to be small compared to the size of the region. When the occupied area A_o is less than 1% of both A_r and A_i, the total regional effect ($CF_{regI+II}$) only marginally increases, i.e < 5% of the regional PDF. Fourth, our assumption of field edges, representing the amount of species present on low intensive or organic fields, is debatable. Hald (1999) compared the species density of field margins with field centers for both organic and conventional farming. This research indicates that the species density of field margins of organic farming is representative for the whole field, while the species density of the field center of conventional fields is 30-40% lower than of the field margins (Hald 1999). While it is generally recognized that organic farming has positive effects on the species richness compared to conventional farming, in some cases field studies indicate negative or mixed effects (Hole et al. 2005). Hole et al. (2005) analyzed 76 comparative studies on organic and conventional farming, and indentified three broad management options largely used (but not exclusive) in organic farming and assumed to be mainly advantageous to biodiversity: (i) prohibition/ reduced use of chemical pesticides and inorganic fertilisers; (ii) establishment of non-crop habitats and field margins; and (iii) preservation of mixed farming (i.e., arable fields in close juxtaposition with pastoral elements). This indicates that focusing on field margins alone is not sufficient to define land use intensiveness. We suggest the inclusion of mixed farming together with pesticides and fertilizer use as two extra indicators on both inventory and impact assessment level to better assess the impact on biodiversity due to land use intensiveness.

Comparison with other life cycle impact assessment methods. In figure 3.2, we compare our CFs with the CFs of Koellner and Scholz (2008) representing all plant species. The CFs of the individualist perspective are closest to the CFs of Koellner and Scholz (2008), as for five out of the seven land use types the CFs differ less than 0.2 PDF and two land use types show a difference of 0.3 PDF. The choice of a fixed z value and excluding regional effects, as employed in the individualist perspective,

corresponds with the modeling approach of Koellner and Scholz (2008). For the other two perspectives the differences are larger, i.e., up to 0.50 PDF for the hierarchist and 0.87 PDF for the egalitarian perspective. Note that Koellner and Scholz (2008) also present parameter uncertainties but solely deriving from uncertainty in species number and therefore smaller than the uncertainties presented in our study.

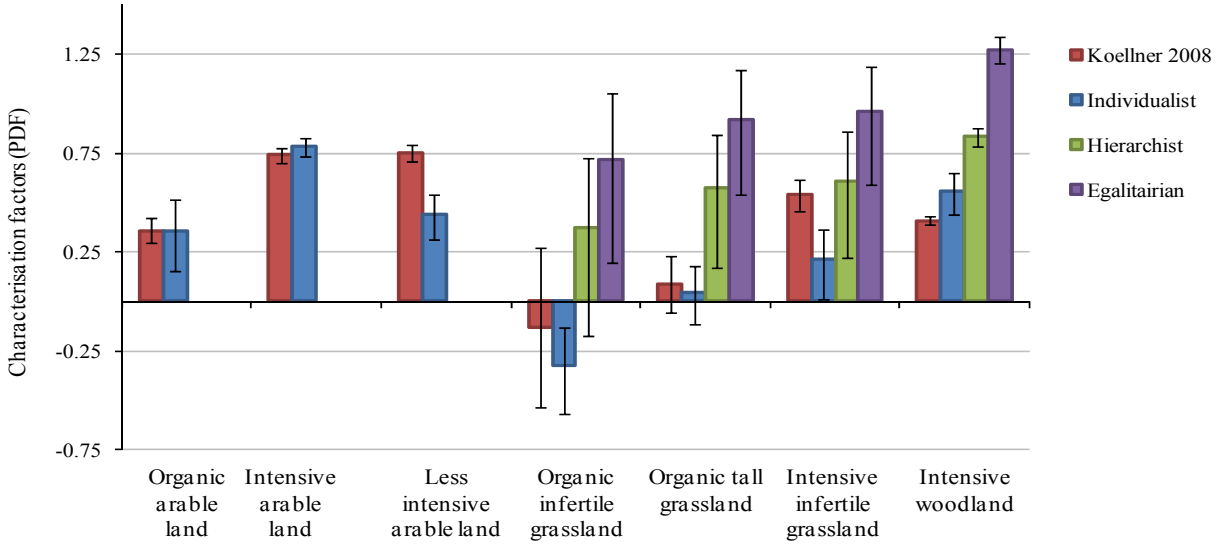

Figure 3.2. CFs for the three cultural perspectives and the local CFs of Koellner and Scholz (2008), for seven land use types. The error bars indicate the 95% confidence level (CL) calculated by using a t-distribution for the average species number S and a bounded normal uncertainty distribution (ranging from 0 to 1) for the species accumulation factor z. Note that no CFs are presented for arable land when using a hierarchist or egalitarian perspective as for this land use type no variable z values were found.

3.5 Conclusion and recommendations

We analyzed the uncertainties in the species area relationship and how they influence the CFs of various land use types. Three scenarios are used to handle the different modeling choices. It is found that the ranking of organic, less intensive and intensive land practices of each land use type is not affected by the chosen model scenario. The absolute values of the CFs are highly dependent on the modeling choices within the species area relationship. Applying a constant z value and excluding regional effects results in CFs which are in the lower range of our results and can even turn negative. The use of a variable z value and including regional effects results in higher CFs. Depending of the chosen scenario, the calculated parameter uncertainty derives mainly from the uncertainty in the variable z value or the uncertainty in average species numbers. Within each model scenario, the majority of the CFs (68% to 85%) significantly differ from each other. Decreasing the parameter uncertainty, in particularly the species accumulation factor z, could further increase the number of significantly different CFs. We consider the use of variable z values as preferable, although the use of CFs on the basis of variable z value may not always be feasible due to the user knowledge of the actual area size of the land occupied and the available land use specific z values. More research towards the development of land use specific general relationships between z value and area size is needed.

3.6 Acknowledgement

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3.8 Appendix 3

Model approach

Local damage. The local CF refers to a change in species richness on the occupied land area compared with the baseline land (figure 3.3).

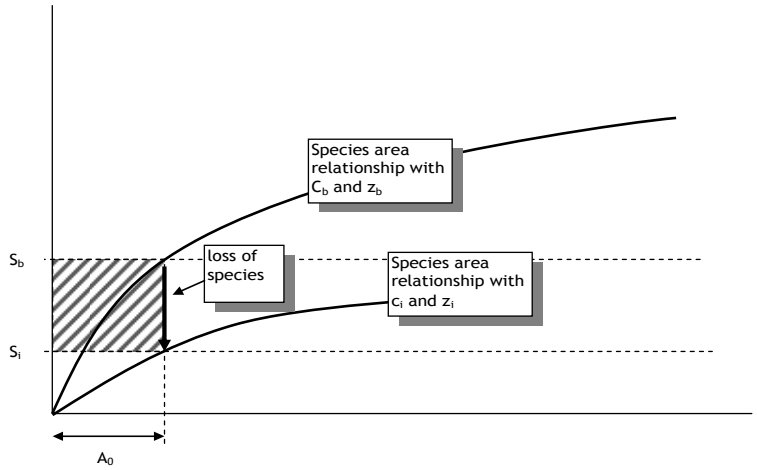

Figure 3.3. Local damage of area A_0 is illustrated as the shaded area. Note that, depending on the baseline species richness an increase in species richness can appear and the local damage can become negative.

Regional damage. The regional damage score I (DS_{regI}) due to a decline in surrounded area size can be calculated as:

$$DS_{regI} = CF_{regI} \cdot A_r \cdot t = \frac{\Delta S_{b,r}}{S_{b,r}} \cdot A_r \cdot t \tag{1}$$

where CF_{regI} stands for the CF of the regional damage I (PDF), A_r the size of the surrounded region (m^2), t the time of occupation (yr), $S_{b,r}$ the species number on the baseline land with area size A_r, and $\Delta S_{b,r}$ the difference in number of species on the total natural land and after occupation with land use type i. The regional damage I is illustrated as the grey area in figure 3.4. The size of the area can be calculated by multiplying $\Delta S_{b,r}$ with A_o. $\Delta S_{b,r}$ can be calculated as:

$$\Delta S_{b,r} \approx A_o \cdot z_{b,r} \cdot c_b \cdot A_r^{z_{b,r}-1} \tag{2}$$

where A_o stands for the area occupied (m^2), c_b the species richness factor of the baseline land use type and $z_{b,r}$ the species accumulation factor of the baseline land use type with area size $A_r – A_o \approx A_r$. Combining equation (2) with equation (1), the regional damage score I (DS_{regI}) equals to:

$$DS_{regI} = z_{b,r} \cdot A_o \cdot t \tag{3}$$

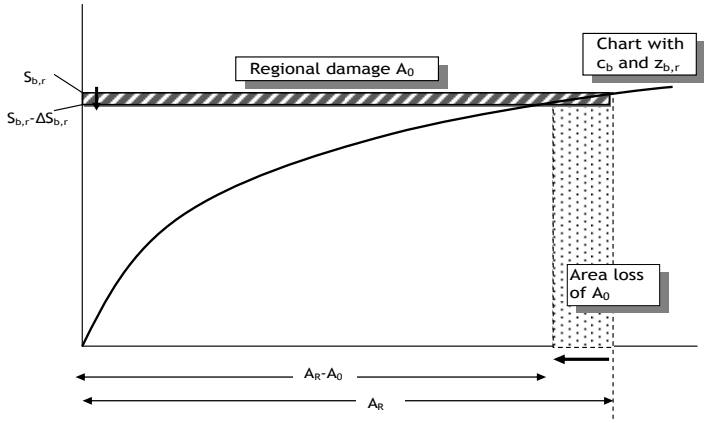

Figure 3.4. Regional damage I is presented as the shaded area in the top of the figure. When the size of the surrounded area drops from A_r to A_r-A_o, the amount of species reduces from $S_{b,r}$ to $S_{b,r}$-ΔS_r.

The enlargement of land type i, due to extra land occupation, can be calculated as:

$$DS_{regII} = CF_{regII} \cdot A_i \cdot t = \frac{\Delta S_{i,r}}{S_{i,r}} \cdot A_i \cdot t \tag{4}$$

where CF_{regII} stands for the CF of the regional damage II, $S_{i,r}$ the species number on land use type i with area size A_i, and $\Delta S_{i,r}$ the difference in number of species on land type i before and after occupation of land use type i. The regional damage II is illustrated as the shaded area in figure 3.5. The size of the area can be calculated by multiplying $\Delta S_{i,r}$ with A_i. $\Delta S_{i,r}$ can be calculated as:

$$\Delta S_{i,r} \approx -A_o \cdot z_{i,r} \cdot c_i \cdot A_i^{z_{i,r}-1} \tag{5}$$

where A_i stands for the area of existing land use type i (m^2), c_i the species richness factor of land use type i, $z_{i,r}$ the species accumulation factor of the baseline land use type with area size A_i+A_o≈A_i. Combining equation (5) with equation (4), the regional damage score II (DS_{regII}) equals to:

$$DS_{regII} = -A_o \cdot \frac{c_i . z_{i,r} . A_i^{z_{i,r}-1}}{c_i . A_i^{z_{i,r}}} \cdot A_i \cdot t = -z_{i,r} \cdot A_o \cdot t \tag{6}$$

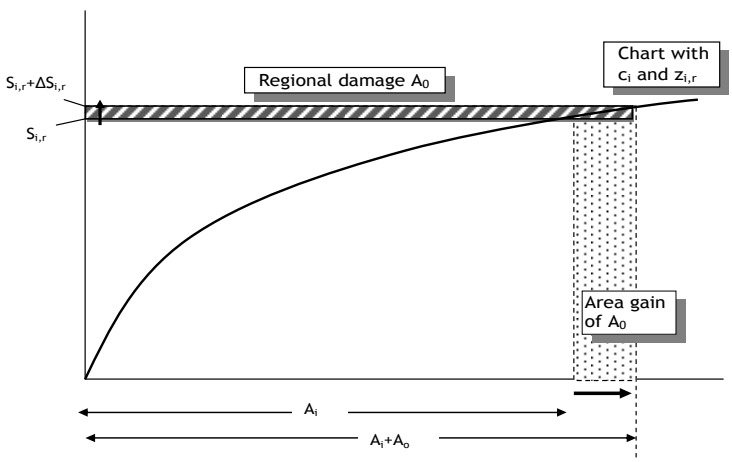

Figure 3.5. Regional damage II is presented as the shaded area in the top of the figure. When the size of land use type i increases from A_i to A_i+A_o, the amount of species rises from $S_{i,r}$ to $S_{i,r}$+$\Delta S_{i,r}$.

The enlargement of land use type i only takes place when the occupied area is considered to be linked with other areas from the same land use type (figure 3.6). When the occupied area is not linked with the area of the same land use type (situation A in figure 3.6) only regional damage I takes place, referring to an area decrease of the surrounded baseline (formula 3). When the occupied area is linked with the area of the same land use type (situation B in figure 3.6) the regional damage score ($DS_{regI+II}$) is a combination of a decrease in the surrounded baseline and an enlargement of land use type i:

$$DS_{regI+II} = (z_{b,r} - z_{i,r}) \cdot A_o \cdot t \tag{7}$$

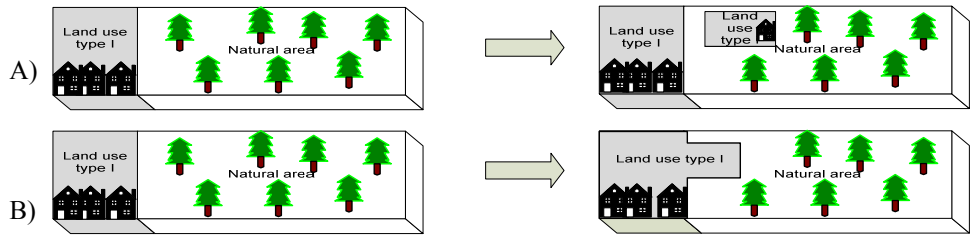

Figure 3.6. Including or excluding regional effect II. A) Land use type i is isolated from other land use types and only regional effect I is considered. B) Land use type i is connected with an area of the same land use type i and regional effects I and II are considered.

Implementation

Land use types and management practices. The Countryside Survey 2000 is a major audit of the countryside from Great Britain carried out in 1998-1999 (Defra 2000). The Broad Habitat classification contains 22 general land cover types, primarily defined on landscape features that allow the whole land surface of Great Britain to be mapped (Smart et al. 2003). Table 3.4 gives an overview of the different land use and plot types, with the amount of plots and the percentage of plots located in a specific broad habitat. Table 3.5 gives a description of the different land use types.

Tabel 3.4. Overview of the Countryside Survey 2000 dataset. Per land use type a number of different plot types are surveyed. Each figure represents the percentage of plots classified into the corresponding broad habitat (Smart et al. 2003). Descriptions of the land use types and broad habitats are presented in Table 3.5.

Aggregated class/ Land use type	Plot type[a]	Number of plots	Acid Grass	Arable and horticultural	Bog	Bracken	Broadleaf, mixed and yew woodland	Conifer woodland	Dwarf shrub heath	Fen, mars and swamp	Improved Grass	Neutral Grass
Arable land	A	206		100								
	B	26		100								
	X	465		95							5	
Fertile grassland	A	33		100								
	B	212		17							83	
	X	445		11							86	2

Infertile grassland	B	353	8	8		4	7			5	61	8
	X	458	5			3	5			6	69	12
Moorland grass	B	63	46						22	32		
	X	366	33		14	6	3	8	22	11	3	
Tall grassland	A	253		100								
	B	646		52			6				38	5
	X	125		71			14				14	
Upland wooded	B	41					61				39	
	X	206	5			15	29	45	6			

[a]A: Field margins; B: Field boundaries; X: Field core

Table 3.5. Description of the different land use types and broad habitats (Defra 2000; Smart et al. 2003). Only the land use types with clear human activity are included in the study.

Land use type	**Description**
Arable land (crop/weed)	Communities of cultivated and disturbed ground. Includes land under arable cultivation.
Fertile grassland	Improved and semi-improved grasslands very common across Great Britain. Usually with a long history of high macro-nutrient inputs and cut more than once a year for silage.
Infertile grassland	Unimproved and semi-improved communities in wet or dry and basic to moderately acidic vegetation. Lowland, species-rich mesotrophic grassland is represented here.
Moorland grass	Extensive, graminaceous upland vegetation, usually with a long history of sheep grazing.
Tall grassland	Most typical of road verges and infrequently disturbed patches of herbaceous vegetation. Includes 'old field' communities of spontaneous, fallow grassland. Usually dominated by perennial grasses and tall herbs.
Upland wooded	Includes upland semi-natural broadleaved woodland and scrub plus conifer plantation. Also includes established stands of Bracken (Pteridium aquilinum).
Broad habitat	Description
Arable and horticultural	All arable crops such as different types of cereal and vegetable crops, together with orchards and more specialist operations such as market gardening and commercial flower growing.
Acid grassland	Vegetation dominated by grasses and herbs on a range of lime-deficient soils which have been derived from acidic bedrock or from superficial deposits such as sands and gravels.
Bog	Wetlands that support vegetation that is usually peat-forming and which receive mineral nutrients principally from precipitation rather than ground water.
Bracken	Vegetation greater than 0.25 ha in extent which are dominated by a continuous canopy cover (>95% cover) of bracken (Pteridium aquilinum) at the height of the growing season.
Broadleaf	Dominated by trees that are more than 5 m high when mature, which form a distinct, although sometimes open, canopy with a cover of greater than 20%.
Conifer woodland	Dominated by trees that are more than 5 m high when mature, which form a distinct, although sometimes open, canopy which has a cover of greater than 20%.
Dwarf, shrub, heath	Vegetation that has a greater than 25% cover of plant species from the heath family or dwarf gorse species.
Fen, marsh, swamp	On ground that is permanently, seasonally or periodically waterlogged as a result of ground water or surface run-off.
Improved grassland	On fertile soils and is characterized by the dominance of a few fast growing species, such as rye-grass and white clover.
Neutral grassland	On soils that are neither very acid nor alkaline.

Species accumulation factor z.

Table 3.6. Land use type and area size specific z values, used to calculate the c factors and CFs for the hierarchist and egalitarian perspective. The z values for grassland are applied for all grassland types. The area size applied for the regional z-values is set to 10,000 m^2.

	z factors to calculate c			z factors to calculate CF (10,000 m^2)	
Land use types	Plot 10 m^2	Plot 100 m^2	Plot 200 m^2	Local effect: $z_{i,l}$	Regional effect: $z_{i,r}$
Arable land[a]		0.44	0.44		
Grassland[b]	0.15	0.21	0.18	0.35	0.35
Moorland Grass[b]	0.20	0.26	0.23	0.32	0.32
Woodland[b]	0.24	0.19	0.44	0.44[c]	0.44[d]

[a]Values derived from Manhoudt et al. (2005)
[b]Values derived from Crawley and Harral (2001)
[c]Also used for the baseline land use type, $z_{b,l}$
[d]Also used for the baseline land use type, $z_{b,r}$

Results

Regional characterization factors

Table 3.7. Regional CFs, considering the decline of the surrounded area only ($CF_{reg\ I}$) or both positive and negative regional effects ($CF_{reg\ I+II}$). The main land use types are presented. For the land use type arable land no regional factors could be calculated.

Land use type	$CF_{reg\ I}$ (Egalitarian)		$CF_{reg\ I+II}$ (Hierarchist)	
	Median	95% CL	Median	95% CL
Grassland[a]	0.44	0.40-0.48	0.09	0.04-0.14
Moorland grass[b]	0.44	0.40-0.48	0.12	0.07-0.17
Woodland[c]	0.44	0.40-0.48	0.00	

[a]Includes the land use types organic, less intensive and intensive fertile grassland, infertile grassland and tall grassland
[b]Includes the land use types organic and intensive moorland
[c]Includes the land use type intensive woodland

Parameter uncertainties. For each land use type, the covariance in the CFs was accounted for by calculating the difference between pairs of CFs in the Monte Carlo simulations. Tables 3.8, 3.9 and 3.10 present a matrix of ranked land use types, ordered consecutively from small to large CF, for the three perspectives. Each value displays the probability of the difference between pairs of CFs, namely the CF of the column land use type minus the CF of the row land use type. If the difference between the pair of CFs was in > 95% of the runs positive or negative, we consider the CF of the column land use type to be significantly different from the CF of the row land use type ($\alpha = 0.1$, two-sided confidence interval).

Table 3.8. Matrix of ranked land use types for the individualist perspective. Each value presents the probability that CF column land use type is significantly larger than CF of row land use type.

Individualist (probability of CF column > CF row)	Organic infertile grassland	Organic moorland grass	Organic fertile grassland	Organic tall grassland	Baseline	Intensive infertile grassland	Intensive moorland grass	Organic arable land	Less intensive fertile grassland	Less intensive arable land	Less intensive tall grassland	Intensive woodland	Intensive fertile grassland	Intensive tall grassland	Intensive arable land
Organic infertile grassland		1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Organic moorland grass	0.00		0.71	0.86	0.65	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Organic fertile grassland	0.00	0.29		0.92	0.50	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Organic tall grassland	0.00	0.14	0.08		0.27	0.99	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Baseline	0.00	0.35	0.50	0.73		0.99	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Intensive infertile grassland	0.00	0.00	0.00	0.01	0.01		0.81	0.94	0.98	1.00	1.00	1.00	1.00	1.00	1.00
Intensive moorland grass	0.00	0.00	0.00	0.00	0.00	0.19		0.90	0.95	1.00	1.00	1.00	1.00	1.00	1.00
Organic arable land	0.00	0.00	0.00	0.00	0.00	0.06	0.10		0.48	0.83	0.84	0.99	1.00	1.00	1.00
Less intensive fertile grassland	0.00	0.00	0.00	0.00	0.00	0.02	0.05	0.52		0.89	0.90	1.00	1.00	1.00	1.00
Less intensive arable land	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.11		0.57	1.00	1.00	1.00	1.00
Less intensive tall grassland	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.10	0.43		1.00	1.00	1.00	1.00
Intensive woodland	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00		1.00	1.00	1.00
Intensive fertile grassland	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.98	1.00
Intensive tall grassland	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02		1.00
Intensive arable land	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	

Table 3.9. Matrix of ranked land use types for the hierarchist perspective. Each value presents the probability that CF column land use type is significantly larger than CF of row land use type.

Hierarchist (probability of CF column > CF row)	Baseline	Organic infertile grassland	Organic fertile grassland	Organic tall grassland	Organic moorland grass	Intensive infertile grassland	Less intensive fertile grassland	Less intensive tall grassland	Intensive moorland grass	Intensive woodland	Intensive fertile grassland	Intensive tall grassland
Baseline		0.94	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Organic infertile grassland	0.06		1.00	1.00	0.95	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Organic fertile grassland	0.00	0.00		0.92	0.76	0.97	1.00	1.00	0.99	0.98	1.00	1.00
Organic tall grassland	0.00	0.00	0.08		0.72	0.90	1.00	1.00	0.99	0.97	1.00	1.00
Organic moorland grass	0.00	0.05	0.24	0.28		0.38	0.79	0.90	1.00	0.90	0.98	0.99
Intensive infertile grassland	0.00	0.00	0.03	0.10	0.62		1.00	1.00	0.98	0.95	1.00	1.00
Less intensive fertile grassland	0.00	0.00	0.00	0.00	0.21	0.00		0.90	0.77	0.75	1.00	1.00
Less intensive tall grassland	0.00	0.00	0.00	0.00	0.10	0.00	0.10		0.65	0.63	1.00	1.00
Intensive moorland grass	0.00	0.00	0.01	0.01	0.00	0.02	0.23	0.35		0.50	0.71	0.83
Intensive woodland	0.00	0.00	0.02	0.03	0.10	0.05	0.25	0.37	0.50		0.68	0.80
Intensive fertile grassland	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.29	0.32		0.97
Intensive tall grassland	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.17	0.20	0.03	

Table 3.10. Matrix of ranked land use types for the egalitarian perspective. Each value presents the probability that CF column land use type is significantly larger than CF of row land use type.

Egalitarian (probability of CF column > CF row)	Baseline	Organic infertile grassland	Organic fertile grassland	Organic tall grassland	Organic moorland grass	Intensive infertile grassland	Less intensive fertile grassland	Less intensive tall grassland	Intensive moorland grass	Intensive fertile grassland	Intensive tall grassland	Intensive woodland
Baseline		0.91	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Organic infertile grassland	0.09		1.00	1.00	0.96	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Organic fertile grassland	0.00	0.00		0.92	0.72	0.97	1.00	1.00	1.00	1.00	1.00	1.00
Organic tall grassland	0.00	0.00	0.08		0.68	0.90	1.00	1.00	0.99	1.00	1.00	1.00
Organic moorland grass	0.00	0.04	0.28	0.32		0.48	0.90	0.97	1.00	1.00	1.00	1.00
Intensive infertile grassland	0.00	0.00	0.03	0.10	0.52		1.00	1.00	0.98	1.00	1.00	1.00
Less intensive fertile grassland	0.00	0.00	0.00	0.00	0.10	0.00		0.90	0.71	1.00	1.00	0.99
Less intensive tall grassland	0.00	0.00	0.00	0.00	0.03	0.00	0.10		0.54	1.00	1.00	0.98
Intensive moorland grass	0.00	0.00	0.00	0.01	0.00	0.02	0.29	0.46		0.90	0.96	0.98
Intensive fertile grassland	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10		0.97	0.82
Intensive tall grassland	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.03		0.65
Intensive woodland	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.02	0.18	0.35	

Chapter 4
Characterization Factors for Global Warming in Life Cycle Assessment based on Damages to Humans and Ecosystems

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Abstract

Human and ecosystem health damage due to greenhouse gas (GHG) emissions is generally poorly quantified in the life cycle assessment of products, preventing an integrated comparison of the importance of GHGs with other stressor types, such as ozone depletion and acidifying emissions. In this study, we derived new characterization factors (CFs) for 63 GHGs that quantify the impact of an emission change on human and ecosystem health damage. For human health damage, the disability-adjusted life years (DALYs) per unit emission related to malaria, diarrhea, malnutrition, drowning, and cardio-vascular diseases were quantified. For ecosystem health damage, the potentially disappeared fraction (PDF) over space and time of various species groups, including plants, butterflies, birds and mammals, per unit emission was calculated. The influence of value choices in the modeling procedure was analyzed by defining three coherent scenarios, based on Cultural theory perspectives. It was found that the CF for human health damage by carbon dioxide (CO_2) ranges from $1.1.10^{-2}$ to $1.8.10^{+1}$ DALY per kton emission, while the CF for ecosystem damage by CO_2 ranges from $5.4.10^{-2}$ to $1.2.10^{+1}$ disappeared fraction of species over space and time (km^2.year/kton), depending on the scenario chosen. The CF of a GHG can change up to four orders of magnitude, depending on the scenario. The scenario-specific differences are mainly explained by the choice for a specific time horizon and stresses the importance of dealing with value choices in the life cycle impact assessment of GHG emissions.

Keywords characterization factors • greenhouse gases • endpoint level • ecosystem damage • human health damage • life cycle impact assessment

4.1 Introduction

Climate change, partly caused by anthropogenic emissions, is a global threat to the health of humans and ecosystems. Within the next 50 years it is expected that species can become extinct due to changing temperature, precipitation and seasonality (Thomas et al., 2004). Concerning human health impacts, several studies show that climate change results in an increase of various diseases, such as malaria and diarrhea (Mcmichael et al., 2003, Mcmichael and Woodruff, 2006, Patz and Campbell-Lendru, 2005). In this context, it is important that the environmental impact of greenhouse gas (GHG) emissions is taken into account in the environmental life cycle assessment (LCA) of products. Global warming potentials (GWP) are widely used to convert product life cycle emissions of various GHGs into a global warming score (Steen, 1999). The Intergovernmental Panel of Climate Change (IPCC) regularly updates the GWPs for a wide range of GHGs (Forster et al., 2007).

While the GWP works well for the relative comparison of the importance of GHGs in the life cycle of a product, this concept does not allow comparison of the environmental impact of GHG emissions with other types of environmental impacts, such as human health impacts due to ozone depletion and loss of species diversity due to changes in land use. Aggregation of different health effects can be achieved through the use of indicators positioned at the end of the cause-effect chain. For human health damage, the concept of disability-adjusted life years (DALYs) has been proposed as endpoint indicator (Hofstetter, 1998), while for ecosystem health damage the loss of species diversity has been introduced (Kollner, 1999).

Only a few researchers attempted to assess the damage of GHG emissions towards humans and ecosystems in product assessments. The Eco-indicator 99 methodology quantifies human damage for a number of GHGs by considering the effects of thermal extremes, vector borne diseases and sea level rise with the FUND model (M. and Spriensma, 1999, Tol, 2002). Ecosystem damage was excluded due to lack of data. The Environmental Priority System (EPS), accounts for effects on both human health and ecosystem damage (Steen, 1999). In particular the influence of GHGs on biodiversity are considered in EPS in a relatively simplistic way by simply assuming that the present rate of extinction will be doubled. Current LCA methodologies contain several limitations in addressing the influence of GHG emissions at the endpoint level. Only a limited number of human health impacts are included, while ecosystem damage is neglected or handled in a very simplistic way and uncertainty is hardly addressed.

The goal of this study is to develop new characterization factors (CFs) for 63 GHG emissions at the endpoint level for effects on both human health and terrestrial ecosystems. This makes it possible to integrate or compare the effects with other environmental impacts calculated at the endpoint level. The effects on human health per unit GHG emission are expressed in DALY, caused by an increase in malnutrition, diarrhea, flooding, malaria and heat stress. The effects on terrestrial ecosystems per unit GHG emission are expressed in potentially disappeared fraction (PDF) of species, including plants,

butterflies, birds and mammals. Results are presented for three scenarios, quantifying the influence of coherent sets of value choices in the modeling procedure. These scenarios include the choice for (i) a specific time horizon, (ii) including or excluding indirect negative radiative forcing of ozone depleting chemicals, (iii) including all species or currently threatened species only, (iv) including or excluding the ability to adapt to climate change by humans and ecosystems and (v) considering age weighting or discount rate and the type of future scenario in the Burden of Disease calculations.

4.2 Methodology

Framework. Life cycle impact assessment (LCIA) is the step in LCA where the impact of emissions caused by product life cycles is assessed. Within LCIA, damage to human health and ecosystem health is commonly quantified with endpoint models. These models produce so called CFs that are used as weighting factors to aggregate life cycle emissions into scores for human health damage and ecosystem health damage. As product life cycles commonly contribute to total emissions in a marginal way (Udo De Haes et al., 1999), the endpoint models are applied with small emission perturbations only.

For global warming, we divided the endpoint modeling from emission to damage into four consecutive steps:

$$CF_{x,e} = \frac{dC_x}{dE_x} \cdot \frac{dRF}{dC_x} \cdot \frac{dTEMP}{dRF} \cdot \frac{dIMPACT_e}{dTEMP} \tag{1}$$

where dE_x is the change in emission of GHG x (kg.year^{-1}), dC_x the change in air concentration of GHG x (ppb), dRF the change in radiative forcing (W.m^{-2}), dTEMP the change in global mean temperature (ºC), and $dIMPACT_e$ the marginal change in damage for environmental endpoint e (Potentially Disappeared Fraction of species for terrestrial ecosystems and Disability Adjusted Life Years for human health). Figure 4.1 gives a graphical overview of the framework. We selected the full set of 63 GHGs that have a direct influence on global warming, as assessed in the GWP-calculations by the IPCC in their fourth assessment report (Forster et al., 2007).

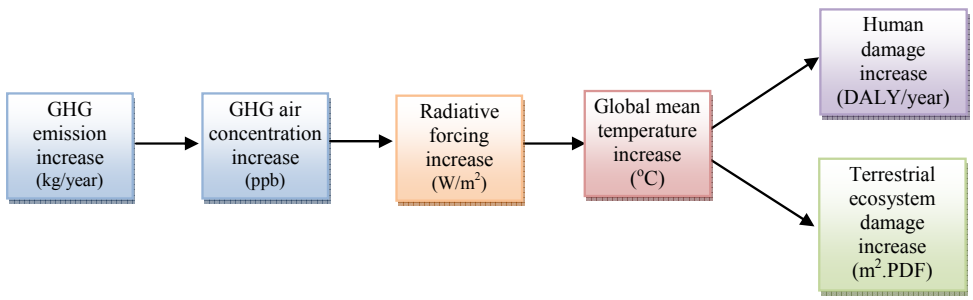

Figure 4.1. Framework used in the development of new endpoint CFs for GHG emissions.

From emission to concentration. The change in concentration caused by the change in emission is calculated by using first order decay rates of GHGs in the atmosphere (Harvey et al., 1997):

$$\frac{dC_{x,t}}{dE_x} = cv_x.LT_x(1-e^{-t/LT_x}) \tag{2}$$

where cv is the substance specific mass to concentration conversion factor (ppb/kg), LT the lifetime of the substance (year) and t is the time horizon after which the concentration change is assessed (year). dC/dE was calculated for the time horizons 20 years, 100 years and infinite. Information on life times of GHGs, except CO_2, was directly taken from Forster et al. (2007). For CO_2, the change in concentration due to an emission cannot be derived using a first order decay rate (2007). Instead, we derived dC/dE values for CO_2 directly from the CO_2 response function in Foster et al. (2007), as based on the Bern carbon cycle model. The mass-to-concentration factor, specific for every GHG, was calculated using the Law of Boyle, taking an average air temperature of 263.7 Kelvin, a tropospheric volume of $7.2.10^{18}$ m^3 and an average air pressure of 49200 Pascal. The life times and mass-to-concentration conversion factors for the GHGs included and the dC/dE factors for CO_2 are listed in appendix 4 (table 4.2).

From concentration to radiative forcing. GHGs cause a change in radiative forcing due to their radiation capacity. Some gases, however, also cause indirect effects such as the change in radiation due to the formation of tropospheric ozone and stratospheric water vapor or the depletion of stratospheric ozone. The latter can result in a global cooling effect. The change in radiative forcing due to a concentration change is therefore equal to:

$$\frac{dRF}{dC_x} = \frac{dRF_{direct}}{dC_x} + \frac{dRF_{indirect}}{dC_x} \tag{3}$$

where dRF_{direct} is the direct change in radiative forcing and $dRF_{indirect}$ the indirect change in radiative forcing (Wm^{-2}).

The change in direct radiative forcing caused by the change in concentration is given by:

$$\frac{dRF_{direct}}{dC_x} = re_x \tag{4}$$

where re_x is the radiative efficiency coefficient of GHG x ($Wm^{-2}ppb^{-1}$), provided by Forster et al (2007) and listed in appendix 4 (table 4.2).

The indirect effect of a GHG on radiative forcing is given by:

$$\frac{dRF_{indirect}}{dC_x} = \frac{dS}{dC_x} \cdot \frac{dRF}{dS} \tag{5}$$

where dS is the change in the situation of a specific 'stressor', i.e. the change in tropospheric ozone, water vapor or stratospheric ozone. The GHGs for which indirect effects are included are methane (tropospheric

ozone and water vapor), CFCs, HCFCs, methylchloride (CCl_4), methylchloroform (CH_3CCl_3) and Halons (stratospheric ozone). Further details on the indirect forcing calculations are given in appendix 4.

From radiative forcing to temperature. The change in global mean temperature caused by the change in radiative forcing is given by:

$$\frac{dTEMP_t}{dRF_t} \approx \frac{\Delta TEMP_t}{\Delta RF_t} \tag{6}$$

The relation between temperature and radiative forcing depends on the climate sensitivity and on the rate of heat absorption by the oceans and does not depend on the type of GHG that is emitted (Randall et al., 2007). The climate sensitivity is a measure of the global surface temperature change for a given radiative forcing. It encompasses the complexity of processes responsible for the way the climate system responds to a radiative forcing, including non-linear feedback processes, for example, clouds, sea ice and water vapor that have a delay over time.

As dTEMP/dRF cannot be derived analytically, we applied the climate model IMAGE 2.2 (Eickhout et al., 2004) to empirically determine the temperature sensitivity factor for the time horizons 20 years, 100 years and infinite. IMAGE uses the Upwelling Diffusion Climate Model (UDCM) developed by Wigley and Raper (1987) for this purpose. We increased the global CO_2 emission with 2.85 Gton per year in scenario A1b (2000)(2000), starting in year 2000. The 2.85 Gton/year is equivalent to 10% of the global CO_2 emission in year 2000 and is added to the yearly global emissions over time. The ratio of the change in radiative forcing and temperature (ΔTEMP/ΔRF) after 20 years and 100 years due to the 10% yearly emission increase was respectively 0.34 and 0.48 $^{o}C.W^{-1}.m^2$. As IMAGE is a dynamic model, ΔTEMP/ΔRF for an infinite time horizon cannot be explicitly calculated. Instead, it was set equal to the ΔTemp/ΔRF that was found to be numerically stable over time, i.e. a yearly change in dTEMP/dRF of smaller than 1%, which was equal to 0.67 $^{o}C.W^{-1}.m^2$.

From temperature to human damage. The climate change damage factor for humans links the change in temperature to a change in Disability Adjusted Life Years (DALY) [yr/yr.°C] and is calculated by:

$$\frac{dIMPACT_{human}}{dTEMP} \approx \frac{\Delta DALY_{tot}}{\Delta TEMP} \tag{7}$$

where $\Delta DALY_{tot}$ stands for the change in the yearly total attributable burden of a population of getting a disease [yr/yr].

The attributable burden is calculated by:

$$\Delta DALY = \sum_r \sum_h \Delta DALY_{r,h} \tag{8}$$

and

$$\Delta DALY_{r,h} = \Delta RR_{r,h} \cdot DALY_{r,h} \quad (9)$$

where $\Delta DALY_{r,h}$ is the change in the yearly attributable burden in region r for health effect h (yr/yr), and $\Delta RR_{r,h}$ is the change in relative risk of health effect h in region r, due to a temperature change (-) and $DALY_{r,h}$ is the yearly burden of disease for region r and health effect h (yr/yr) in the year 2030. The results of $\Delta DALY_{r,h}$ are presented in appendix (table 4.8).

A wide range of health effects related to global warming are reported in the literature, including malnutrition, heat stroke, drowning and a large number of infectious diseases, such as malaria, dengue, cholera and tick-borne diseases (Steen, 1999, Confalonieri, 2007). Due to lack of data, it was not possible to quantify all these human health impacts caused by global warming. We based our assessment on McMichael and Woodruff (2006) and Ezzati et al. (2004), who derived region-specific relative risks related to global warming for malnutrition, diarrhea, malaria, coastal and inland flooding, and heat stress in the year 2030 compared to 1990 (see tables 4.6 and 4.7 in the appendix). The difference in relative risks in 2030 (ΔRR) and temperature rise (ΔTEMP) of two future scenarios, as given in McMichael and Woodruff (2006) and Ezzati et al. (2004), were derived per geographical region and disease type.

DALY estimates for 2030 derived from Mathers and Loncar (2006) for an optimistic, pessimistic and a baseline future scenario. In DALY calculations, value choices are related to whether age weighting and discount rate for future impacts are preferred (Tsuchiya, 2002). We combined the optimistic future scenario with age weighting and a 3% discount rate, the pessimistic scenario with no age weighting and no discount rate, and the baseline scenario with no age weighting and a 3% discount rate. The combination of value choices is further explained in the section on cultural perspectives (see tables 4.6 and 4.7 in appendix 4).

From temperature rise to terrestrial ecosystem damage. The endpoint damage factor for terrestrial ecosystem damage due to climate change links the marginal changes in temperature to marginal changes in disappeared fraction of species [PDF/°C] and can be calculated by:

$$\frac{dIMPACT_{eco}}{dTEMP} \approx \frac{A \cdot \Delta PDF}{\Delta TEMP} \quad (10)$$

where ΔPDF is the average change in potentially disappeared fraction of species due a temperature change ΔTEMP (-) and A is the total surface of (semi)natural terrestrial areas of the world, $10.8.10^7$ km^2. The FAO Global Arable-ecological Zones database gives an overview (percentage) of the main types of land (http://www.fao.org/ag/agl/agll/gaez/index.htm) which was combined with the total land surface on earth (Coble et al., 1987).

Thomas et al. (2004) included in their review 9 studies that link regional extinction risk of various species groups (covering a total of 1084 species) with temperature increase in that region. We used his work as a

basis for our calculations and assumed that extinction risk equals to the disappeared fraction of species. Extinction risks were calculated by Thomas et al (2004) for the species groups plants, butterflies, birds and mammals. They assessed the influence of a number of methodological choices in their calculations, i.e. extinction risks (i) with or without dispersal of species, (ii) for all species or Red list species only (IUCN, 2001), and (iii) by applying three different interpretations of the species area relationship. Depending on the interpretations of species area relationships the calculation method calculates the overall changes in the distribution areas summed across species, the average proportional change averaged across species or the risk of each species in turn. Further details on calculation steps and data use are presented in appendix 4 (table 4.9).

Cultural perspectives. To handle value choices that arose in the modeling procedure in a consistent way, we applied the cultural perspective theory (Hofstetter, 1998). Three cultural perspectives were used, i.e. the individualistic, the hierarchist and the egalitarian perspective. The individualist coincides with the view that mankind has a high adaptive capacity through technological and economic development and that a short time perspective is justified. The egalitarian coincides with the view that nature is strictly accountable, that a long time perspective is justified, and a worst case scenario is needed (the precautionary principle). The hierarchical perspective coincides with the view that impacts can be avoided with proper management, and that the choice on what to include in the model is based on the level of (scientific) consensus (Goedkoop and Spriensma, 1999).

Table 4.1 gives an overview of the value choices we were able to include in our study and how they relate to the three perspectives. Depending on the time horizon chosen, long term or short term processes are emphasized. The 20 year time horizon corresponds with the shortest time horizon employed in the GWP calculations of the Intergovernmental Panel of Climate Change (Forster et al., 2007) and is applied in the individualist perspective. The hierarchist perspective coincides with a 100 year timeframe, which is most frequently used as example in the ISO standards (ISO, 2003) and the Kyoto protocol (UNFCCC, 2006).The egalitarian perspective follows an infinite time horizon that is in line with the precautionary principle.

A second value choice relates to the inclusion of indirect effects of ozone depletion chemicals on radiative forcing which are highly uncertain. A sharp decline in atmospheric concentration can appear by the legal restriction of these substances in products (UNEP, 2000). This will reduce the indirect effects of ozone depleting chemicals near to zero (see appendix 4). We consider the indirect effects of ozone depletion chemicals for the individualist perspective only.

For the relative risk estimates, differences in assumptions concerning future adaptation possibilities were considered in the definition of the perspectives. The individualist coincides with full human adaptation, the egalitarian perspective with a worst case scenario and the hierarchist perspective followed a mid-level relative risk for all effects (details see appendix 4).

For the DALYs, the choice for a future scenario, age weighting and discount rate will influence the results. For the individualist perspective, we applied an optimistic future scenario with age weighting and a discount rate of 3%. The egalitarian perspective followed a worst case scenario that coincides with a pessimistic future scenario, no age weighting and no discount rate. The hierarchist perspective included a baseline future scenario with no age weighting and discount rate at 3%. The choice for age weighting per perspective was based on Hofstetter (1998).

One of the uncertainties in ecosystem damage refers to the ability of dispersal of species (Gitay et al., 2002). For the individualistic and hierarchist perspective we assume nature will be partly able to adapt to the effects of climate change by the ability of species to disperse. While for the egalitarian perspective we assume a precautionary scenario without species dispersal. A second assumption for ecosystem damage is the inclusion of taxa. For the individualist and hierarchist we assume all species are equally important. For the egalitarian perspective the red list species identified by IUCN were included only, giving high importance to species that are already threatened in their existence.

Table 4.1. Combination of value choices for time horizon, influence of ozone depleting chemicals on radiative forcing, Burden of disease (BoD), human and ecosystem adaptation and the species to be included, expressed in three different cultural perspectives.

Value choice	**Individualist**	**Hierarchist**	**Egalitarian**
Time horizon	20 year	100 year	Infinite
Indirect effects of ozone depleting chemicals	Yes	No	No
BoD: - age weighting	Yes	No	No
- discount rate	3%	3%	0%
- Future Scenario	Optimistic	Baseline	Pessimistic
Biological/sociological adaptation	Full	Mean	No
Species dispersal	Yes	Yes	No
Species protection level	All	All	Red list species

4.3 Results

Figures 4.2 and 4.3 show the CFs of respectively human health damage and ecosystem damage for a subset of GHGs for the three cultural perspectives. From the total set of 63 GHG emissions, we show the most important GHGs CO_2, CH_4 and N_2O, and representatives from the substance groups CFCs, HCFCs, Halons, HFCs, HFEs and PFC-14. For the total list of characterization see appendix 4.

The CF of PFC-14 is the largest of the GHGs included and, depending on the cultural perspective chosen, 3.6 to 5.4 orders of magnitude larger compared to the CF of CO_2 (figure 4.2). The individualistic and the hierarchist perspective shows smaller differences between the substances compared to the egalitarian perspective.

Differences between the perspectives are the largest for chemicals with a long residence time in the air and range from 0.1 orders of magnitude for CH_4 up to 5.0 orders of magnitude for PFC-14. For ozone depleting chemicals, such as CFC-11 and Halon-1301, the difference between the perspectives is

amplified by the fact that the individualistic perspective takes into account indirect effects leading to net cooling effects, while it is excluded for other perspectives. A net cooling effect results in negative CFs under the assumption of a global temperature increase as background situation.

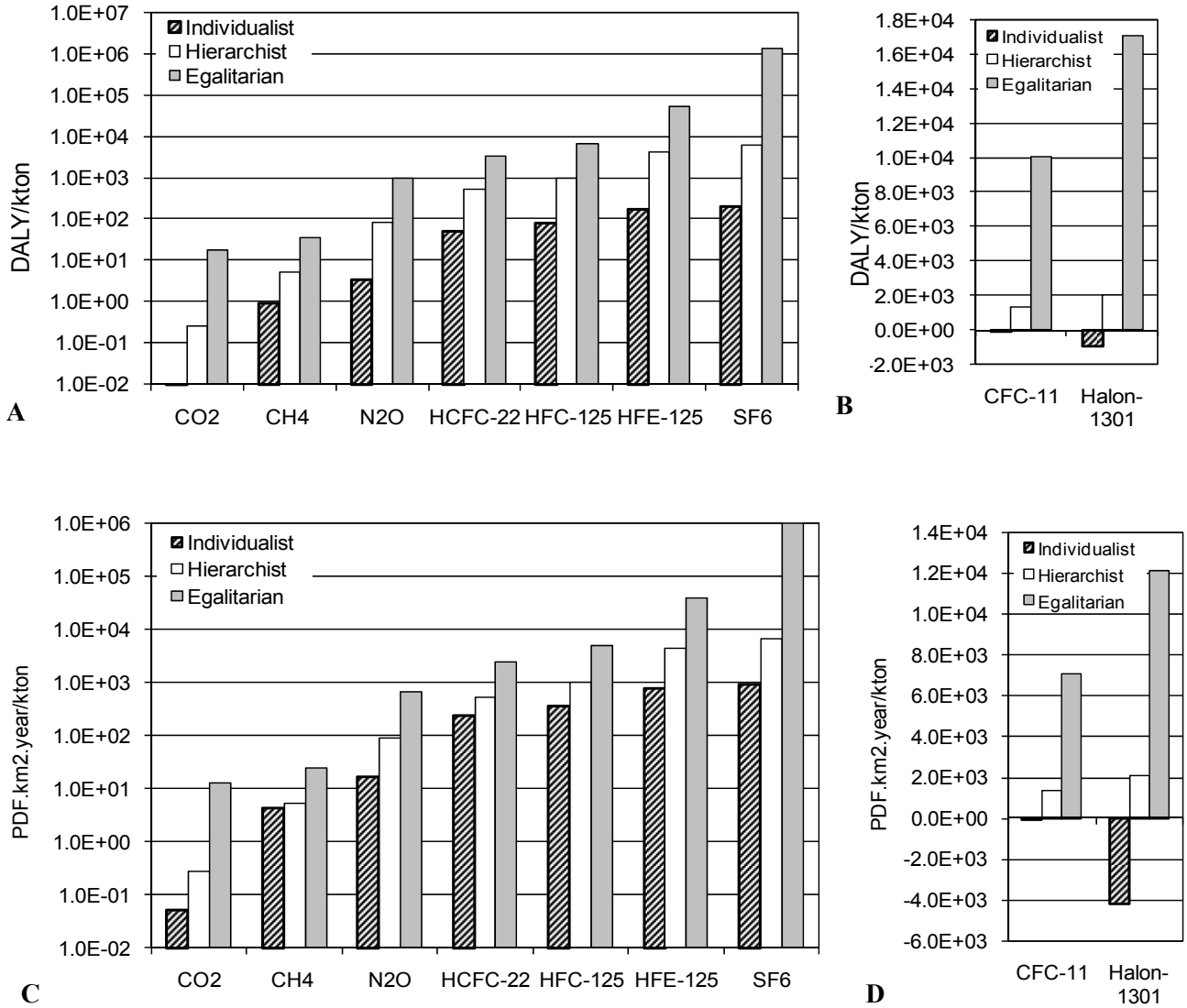

Figure 4.2. CFs for 7 emissions with a net positive impact on global warming (A,C) and 2 ozone depleting emissions with a possible negative impact on global warming (B,D). The factors are related to human health damage (A, B) and the loss of biodiversity (C, D), for an Individualist (I), Hierarchist (H) and an Egalitarian (E) perspective. Note that figure 4.2A and 4.2C is in log-scale, and figure 4.2B and 4.2D in normal scale.

Specifically focusing on the damage part of the CF, figure 4.3 shows that the human damage factor ranges from 1.10^6 to 3.10^7 yearly DALYs per °C temperature increase, depending on the cultural perspective. Climate change mainly influences the incidence of diarrhea, malaria and malnutrition. For the individualistic perspective the effects of diarrhea and malaria play a dominant role. For the hierarchist and egalitarian perspective, malnutrition is most important.

The ecosystem damage factor is 0.06 PDF/°C when dispersal is assumed and all species are taken into account, as employed in the individualistic and hierarchist perspective. When considering no dispersal and using the red list species for the egalitarian perspective, the ecosystem damage factor increases to 0.2 PDF/°C.

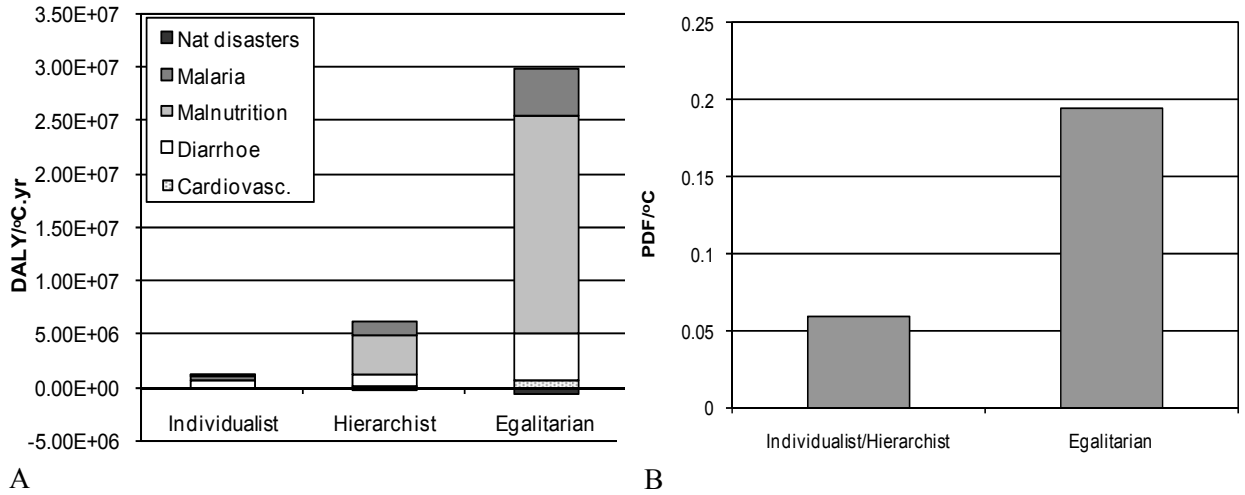

Figure 4.3. The human damage factors (A) and terrestrial ecosystem damage factors (B) related to climate change, following the three cultural perspectives.

4.4 Discussion

Based on a coherent modeling procedure and combining various data sources (Thomas et al., 2004, Mcmichael et al., 2003, Forster et al., 2007, Mathers and Loncar, 2006, Who, 2004), we were able to calculate new CFs for 63 GHGs. The calculated CFs can be used to quantify effects of GHG emissions in LCA case studies, towards human health as DALYs and ecosystem health as disappeared fraction of species. This opens the possibility to aggregate global warming effects with the effect of a wide range of other stressors relevant in life cycle studies, such as ozone depleting gases, radioactive pollutants and priority air pollutants. Compared to previous studies we were able to include a larger number of human health impacts, including malnutrition and diarrhea. For ecosystem damage factors a wide range of species was included, based on the review of Thomas et al. (2004). Value choices in the calculation procedure were assessed by means of establishing three distinct scenarios following the theory of cultural perspectives (Thompson et al., 1990, Hofstetter, 1998).

The analysis showed that scenario-specific differences in the CF of a GHG can be up to 5 orders of magnitude, particularly for gases with a long residence time in air. The choice for a specific time horizon in the step from emissions to GHG air concentrations largely explains the differences between the scenarios. For ozone depleting substances, the choice whether to include indirect cooling effects can also have important consequences, as it can change the direction (positive to negative) of the CFs for climate related impacts. However, the effect on human health of ozone depletion via increase in skin cancer and

cataract, still results in net damage to human health due to emission of ozone depleting substances (Hayashi et al., 2006). Note that a number of priority air pollutants, such as sulfur dioxide, nitrogen oxides and carbon monoxide where not included in our analysis. The inclusion of the highly uncertain indirect effects of these pollutants in a LCIA context certainly needs further attention.

The scenario-specific differences in CFs caused by the choices for human and ecosystem damage were relatively small compared to the choice of a specific time horizon. The difference in the human damage factor is almost fully explained by the choice to what extent malnutrition may play a role in future human health impacts caused by climate change. The different DALYs applied for diarrhea take also a part of the difference. For the ecosystem damage factors, the value choices on including or excluding red list species and the choice for dispersal or non-dispersal of species were equally important.

Data and model uncertainty. Apart from the differences in CFs due to value choices in the modeling procedure, uncertainties in the data employed and the models used may also influence the results (Brakkee et al., 2008). Due to data limitations we were not able to carry out a complete and overall uncertainty analysis. Instead, we describe and quantify the uncertainties of each calculation step separately. Uncertainty in the change in radiative forcing due to a GHG emission change is mainly caused by uncertainty in the atmospheric life time and the radiative forcing efficiency of the GHG under consideration. Forster et al. (2007) estimates a typical uncertainty of ± 35% (90% confidence range) for the direct influence on radiative forcing by non-CO_2 GHG emissions and ±15% for CO_2. Forster et al. (2007) also states that uncertainties for the indirect effects of emission changes on radiative forcing changes are generally much higher than for the direct effects. The indirect effect of chlorides, halons and methane depends on actual background concentrations of the individual substances. We applied concentrations in year 2000 for this purpose. Atmospheric concentrations of halons and chlorides, however, are expected to significantly decline (WMO, 2003). The lower the concentrations the smaller the expected indirect effects. Therefore the indirect effects of ozone depleting chemicals are likely to reduce in the coming century.

The relationship between radiative forcing and temperature change is complex and potentially non-linear. In this paper the calculation step from radiative forcing (RF) to temperature is numerically derived from the atmospheric model of IMAGE(Eickhout et al., 2004). Various factors, such as the choice for a specific background emission scenario and including land and oceanic feedback mechanisms on carbon sinks, may alter the outcomes. However, the influence of these factors on radiative forcing changes due to an emission change is expected to stay within a factor of 1.5 (Brakkee et al., 2008). Fuglestvedt et al. (2003) stated that calculations with different climate models show a range in the conversion factor of RF to temperature of 0.4 to 1.2 $^0C/(Wm^{-2})$. The conversion factor used in this study ranges from 0.34 to 0.67 $^0C/(Wm^{-2})$, depending on the time horizon chosen, which falls mostly in the range reported by Fuglestvedt et al. (2003).

Apart from uncertainties in the atmospheric part of the modeling procedure, uncertainties arise in the calculation of human health damage factors. The effects of climate change on human health are many and complex (Mcmichael and Woodruff, 2006, Patz and Campbell-Lendru, 2005, Confalonieri, 2007). Due to data limitations, not all human health effects were included in the analysis, such as dengue and tick-borne diseases. Additional data on world-wide relative risks of these infectious diseases due to global warming is required to further improve the calculation of human health damage factors. Most of the model uncertainties within the calculations of the Relative Risk factors are reflected in the scenario choices included in the different cultural perspectives (see appendix 4). The uncertainties in the RR for diarrhea are mainly caused by the various pathogens and modes of transmission and the variability in severity of clinical symptoms of pathogens, and are expected to be in the range of 10% uncertainty (Ezzati et al., 2004). For malaria, McMichael et al (2003) applied the MARA model which uses a combination of biological and statistical approaches to quantify the effects of climate on Falciparum malaria. The uncertainty in the RR factors range from no increasing risk to a factor 2 between the midrange and higher range (Ezzati et al., 2004). The effect of climate change on malnutrition is uncertain mainly due to the predicted changes in regional precipitation. The uncertainty is expected to be in the range of no increasing risk to a factor of 2 between midrange and higher range RR estimates (Ezzati et al., 2004).

Uncertainties in DALY calculations, considering different future scenarios, are handled by the different perspectives. Note, however, that future projections in DALY development after 2030 are not taken into account, due to data limitations. This influences the results for the hierarchist and egalitarian perspective, as both have a time frame far beyond 2030, namely 100 and 500 years after the baseline 1990.

Concerning damage towards ecosystems by global warming, IPCC (Fischlin et al., 2007) indicates that, “20 to 30% of plant and animal assessed so far in an unbiased sample are likely to be at increasingly high risk of extinction as global mean temperatures exceed a warming of 2 to 3 °C above pre-industrial levels”. From this information we derive an ecosystem damage factor of typically 0.1 PDF/°C, which lies within the range of our results. The uncertainty range of the average extinction risk is 0.05 -0.2 PDF/°C global mean temperature rise (Fischlin et al., 2007). Note, however, that the temperature effect on individual species groups covers a much larger effect range 0.005-0.43 per °C (Thomas et al., 2004). Furthermore, the damage results on ecosystems do not take into account the possibility of human responses to protect biodiversity. The data provided by Thomas et al. (2004) served as a basis for our results and did not consider this aspect in the analysis.

Comparison with other LCIA methods. When comparing our human health damage CFs for GHG emissions with previous studies (Steen, 1999, M. and Spriensma, 1999), it appears that the results of both other studies fall within our results. However, the differences in CFs between the studies are smaller than the differences caused by values choices within our study. This indicates that value choices generally have a larger influence on the human health damage CFs for GHGs compared to differences caused by applying different modeling concepts that are currently available. A comparison of our ecosystem

damage factors with the EPS method (Steen, 1999) was not possible, because the EPS method does not use the same modeling endpoint as used in our study, which is the time-integrated global disappearance of species.

Concluding remarks. The new CFs are suitable to compare the impacts of GHGs with other types of stressors, such as substances causing acidification and respiratory impacts, for both human health and biodiversity. Particularly for impacts on biodiversity, GHG emissions have not been commonly included in LCA case studies. For human health damage five different health effects were included, but still a number of other diseases related to global warming are missing in the calculations due to data limitations. We also showed that the choice of a specific perspective can substantially alter the CFs for GHGs. This will also alter the relative importance of GHG emissions compared to other stressors. It should also be stressed that by combining global warming damage scores with damage scores from other impact categories, inconsistent modeling assumptions may arise, such as differences in time horizon or assumptions on socio-economic adaptations. The influence of value choices and consistency between impact categories should be carefully assessed in LCA case studies.

4.5 Acknowledgments

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4.7 Appendix 4

Indirect radiative forcing

Methane. Methane causes the formation of the stressors tropospheric ozone and stratospheric water vapor. The effect through the change in formation of tropospheric ozone is given by (Houghton et al., 2001):

$$\frac{d[O_{3\,trop}]}{d[CH_4]} \cdot \frac{dRF}{d[O_{3\,trop}]} = \frac{R_{CH_4}}{[CH_4]} \cdot re_{O_3 trop} \qquad (1)$$

where R_{CH4} is the sensitivity coefficient to determine the change in tropospheric ozone concentration. This value is based on the OxComp experiment (Houghton et al., 2001) and has a value of 6.7. The concentration of tropospheric ozone is given in Dobson Units (DU). The radiative efficiency coefficient of tropospheric ozone (re_{O3trop}) is 0.042 $Wm^{-2}DU^{-1}$.

The effect through the change in formation of water vapor due to the oxidation of CH_4 is given by (Harvey et al., 1997):

$$\frac{d[H_2O_{air}]}{d[CH_4]} \cdot \frac{dRF}{d[H_2O_{air}]} = \frac{0.0009}{\sqrt{[CH_4]}} \qquad (2)$$

$[CH_4]$ in equation 1 and 2 is the actual CH_4 concentration on time t. Since this value is unknown we assume here that with marginal additional emissions of CH_4, $[CH_4]_t$ is $[CH_4]_0$ the concentration in year 2000, which is about 1.83 ppb. The background concentration however also changes. Scenarios (IPCC, 2000) show that the concentration in the 21st century may double. This has consequences for the indirect forcing. For example if the concentration of CH_4 doubles, the $dRF_{O3}/d[CH_4]$ and $dRF_{H2O}/d[CH_4]$ decrease with respectively 50% and 30%. For illustration see figure 4.4.

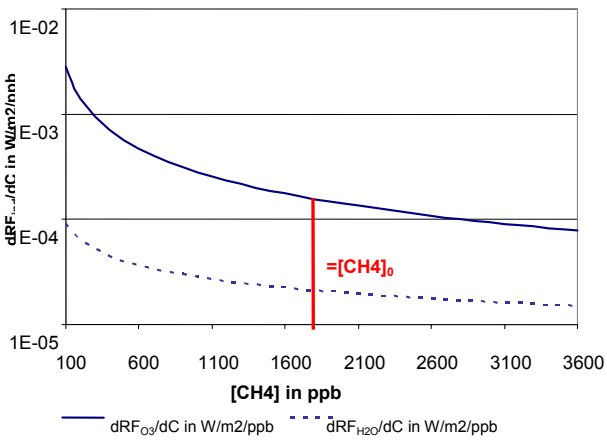

Figure 4.4. The relation between the change in radiative efficiency of respectively the indirect formation of H_2O (dRF_{H2O}/dCH_4) and tropospheric ozone (dRF_{O3}/dCH_4) with the actual CH_4 concentration

Ozone depleting chemicals. CFCs, HCFCs, tetrachloride (CCl_4) and methylchloroform (CH_3CCl_3) cause depletion of stratospheric ozone due to reactive chlorine atoms, which is responsible for a decrease in radiative forcing. The change in this indirect effect is given by:

$$\frac{d[O3_{strat}]}{d[Chloride]} \cdot \frac{dRF}{d[O_{3strat}]} = -0.118 \cdot NCl_C^{1.7} . [Chloride]^{0.7} \tag{3}$$

The same type of effect is also related to the Halons, which is given by:

$$\frac{d[O_{3strat}]}{d[Halon]} \cdot \frac{dRF}{d[O_{3strat}]} = -3.048 \cdot NBr_H \tag{4}$$

NCl_C and NBr_H are the number of reactive atoms per molecule Chloride or Halon. Equations 3 and 4 were derived from Harvey et al. (1997). Like the indirect effect of methane the actual concentration of the Chloride is also of relevance. In figure 4.5 dRF_{O3}/d[Chloride] is shown in relation to concentrations of chlorides with different numbers of Chlorine atoms.

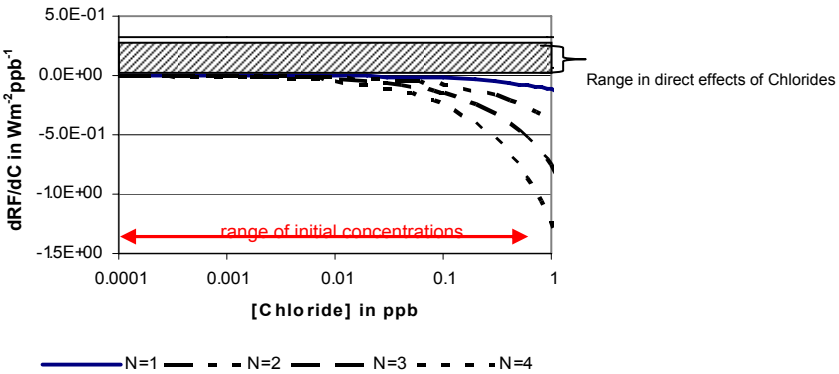

Figure 4.5. The radiative efficiency of the indirect formation of stratospheric ozone is related to the actual Chloride concentration. The shaded area is the range of direct effects of Chlorides. Initial concentrations range from 0 up to 0.55 ppb. For GHGs with an unknown background the indirect effects were set to zero.

Here, we also assume that the typical concentration equals the concentration in year 2000. For ozone depleting chemicals, however, a sharp decline in atmospheric concentration can be expected due to legal restriction of these substances in products, according to the Montreal protocol (UNEP, 2000). Figure 4.5 shows that with declined concentrations the indirect temperature factor will approach zero W.m-2.ppb-1 at concentrations below 0.05 ppb. This implies that the indirect part of dRF/dC of CFCs, HCFCs, tetrachloride and methylchloroform should be considered highly uncertain.

Temperature factors

To calculate the temperature factors (table 4.3), the information on the life times and the radiative efficiency coefficient ($Wm^{-2}ppb^{-1}$) of GHGs were taken from IPCC (Forster et al., 2007) and the substance specific mass to concentration conversion factors were derived with the Law of Boyle (table 4.2). For CO_2 the dC/dE factors for a 20, 100 and infinite time horizon were calculated using the Bern carbon cycle in Foster et al. (2007) and are 1.76E-9, 6.21E-9 and 6.33E-8 ppb.yr/kg respectively. The infinite time horizon is set at 2000 years, when the atmospheric/ocean equilibrium is reached with a 22% of carbon that reside in the atmosphere.

Table 4.2. Substance-specific background information as input in the temperature factor calculations (relevant for Equations 2-5 in the main text).

	Substance specific parameters		
	Mass to conc. factor (ppb/kg)	**Atmos. Lifetime (yr)**	**Rad.eff. Coeff. ($Wm^{-2}ppb^{-1}$)**
CO_2*	1.41E-10		1.4E-05
CH_4	3.87E-10	12	3.7E-04
N_2O	1.41E-10	114	3.03E-03
Substances controlled by the Montreal Protocol			
CFC-11	4.50E-11	45	0.25
CFC-12	5.12E-11	100	0.32
CFC-13	5.93E-11	640	0.25
CFC-113	3.30E-11	85	0.30
CFC-114	3.62E-11	300	0.31
CFC-115	4.01E-11	1700	0.18
Carbon tetrachloride	4.02E-11	26	0.13
Methyl bromide	6.52E-11	0.7	0.01
Methyl chloroform	4.64E-11	5	0.06
HCFC-22	7.16E-11	12	0.20
HCFC-123	4.05E-11	1.3	0.14
HCFC-124	4.54E-11	5.8	0.22
HCFC-141b	5.29E-11	9.3	0.14
HCFC-142b	6.16E-11	17.9	0.20
HCFC-225ca	3.05E-11	1.9	0.20
HCFC-225cb	3.05E-11	5.8	0.32
Halon-1211	3.74E-11	16	0.30
Halon-1301	4.16E-11	65	0.32
Halon-2402	2.38E-11	20	0.33
Hydrofluorocarbons			
HFC-23	8.84E-11	270	0.19
HFC-32	1.19E-10	4.9	0.11
HFC-43-10mee	2.46E-11	15.9	0.4
HFC-125	5.16E-11	29	0.23
HFC-134a	6.07E-11	14	0.16
HFC-143a	7.36E-11	52	0.13
HFC-227ea	3.64E-11	34.2	0.26
HFC-245fa	4.62E-11	7.6	0.28
HFC-152a	9.37E-11	1.4	0.09
HFC-236fa	4.07E-11	240	0.28
HFC-365mfc	4.18E-11	8.6	0.21
Perfluorinated compounds			
Sulphur hexafluoride	4.24E-11	3200	0.52
Nitrogen trifluoride	8.72E-11	740	0.21
PFC-14	7.03E-11	50000	0.1
PFC-116	4.48E-11	10000	0.26

PFC-218	3.29E-11	2600	0.26
PFC-318	3.09E-11	3200	0.32
PFC-3-1-10	2.60E-11	2600	0.33
PFC-4-1-12	2.15E-11	4100	0.41
PFC-5-1-14	1.83E-11	3200	0.49
PFC-9-1-18	1.34E-11	1000	0.56
Trifluoromethyl sulphur pentafluoride	3.16E-11	800	0.57
Fluorinated ethers			
HFE-125	4.55E-11	136	0.44
HFE-134	5.24E-11	26	0.45
HFE-143a	6.19E-11	4.3	0.27
HCFE-235da2	3.35E-11	2.6	0.38
HFE-245cb2	4.69E-11	5.1	0.32
HFE-245fa2	4.12E-11	4.9	0.31
HFE-254cb2	5.16E-11	2.6	0.28
HFE-347mcc3	3.09E-11	5.2	0.34
HFE-347pcf2	3.09E-11	7.1	0.25
HFE-356pcc3	3.40E-11	0.33	0.93
HFE-449sl	2.48E-11	3.8	0.31
HFE-569sf2	2.34E-11	0.77	0.30
HFE-43-10pccc124	2.06E-11	6.3	1.37
HFE-236ca12	3.36E-11	12.1	0.66
HFE-338pcc13	2.64E-11	6.2	0.87
Perfluoropolyethers			
PFPMIE	1.60E-11	800	0.65
Hydrocarbons and other compounds			
Dimethylether	1.34E-10	0.015	0.02
Methylene chloride	7.29E-11	0.38	0.03
Methyl chloride	1.23E-10	1.0	0.01

* The dC/dE factors for a 20, 100 and infinite time horizon derive from Foster et al. (5) and are 1.76E-9, 6.21E-9 and 6.33E-8 ppb.yr/kg respectively.

Table 4.3. Direct, indirect and total temperature factors (°Ckton^{-1}yr).

	TF20 (°C.yr.kton-1)			**TF100 (°C.yr.kton-1)**			**TFinf (°C.yr.kton-1)**		
	Direct	**Indirect**	**Total**	**Direct**	**Indirect**	**Total**	**Direct**	**Indirect**	**Total**
CO2	8.40E-09		8.40E-09	4.17E-08		4.17E-08	5.94E-07		5.94E-07
CH4	4.74E-07	2.24E-07	6.98E-07	8.24E-07	3.90E-07	1.21E-06	1.15E-06	5.45E-07	1.70E-06
N2O	2.66E-06		2.66E-06	1.36E-05		1.36E-05	3.26E-05		3.26E-05
Substances controlled by the Montreal Protocol									
CFC-11	6.18E-05	-7.66E-05	-1.48E-05	2.17E-04	-2.69E-04	-5.20E-05	3.39E-04	-4.21E-04	-8.14E-05
CFC-12	1.01E-04	-7.95E-05	2.14E-05	4.97E-04	-3.92E-04	1.05E-04	1.10E-03	-8.65E-04	2.33E-04
CFC-13	9.92E-05	0.00E+00	9.92E-05	6.58E-04	0.00E+00	6.58E-04	6.35E-03	0.00E+00	6.35E-03
CFC-113	6.00E-05	-3.04E-05	2.97E-05	2.80E-04	-1.41E-04	1.38E-04	5.64E-04	-2.85E-04	2.79E-04
CFC-114	7.39E-05	-3.28E-06	7.06E-05	4.58E-04	-2.03E-05	4.38E-04	2.26E-03	-1.00E-04	2.16E-03
CFC-115	4.87E-05	-1.00E-06	4.77E-05	3.36E-04	-6.91E-06	3.29E-04	8.21E-03	-1.69E-04	8.04E-03
Carbon tetrachloride	2.48E-05	-5.35E-05	-2.87E-05	6.39E-05	-1.38E-04	-7.40E-05	9.11E-05	-1.97E-04	-1.06E-04
Methyl bromide	1.55E-07	-4.73E-05	-4.71E-05	2.19E-07	-6.68E-05	-6.65E-05	3.06E-07	-9.32E-05	-9.29E-05
Methyl chloroform	4.65E-06	-1.10E-05	-6.33E-06	6.68E-06	-1.58E-05	-9.10E-06	9.33E-06	-2.20E-05	-1.27E-05
HCFC-22	4.74E-05	-9.00E-06	3.84E-05	8.24E-05	-1.57E-05	6.68E-05	1.15E-04	-2.19E-05	9.32E-05
HCFC-123	2.50E-06	-4.67E-08	2.46E-06	3.54E-06	-6.59E-08	3.47E-06	4.94E-06	-9.19E-08	4.84E-06
HCFC-124	1.90E-05	0.00E+00	1.90E-05	2.78E-05	0.00E+00	2.78E-05	3.88E-05	0.00E+00	3.88E-05
HCFC-141b	2.07E-05	-2.87E-06	1.78E-05	3.31E-05	-4.59E-06	2.85E-05	4.62E-05	-6.41E-06	3.98E-05
HCFC-142b	5.04E-05	-1.71E-06	4.87E-05	1.05E-04	-3.58E-06	1.02E-04	1.48E-04	-5.02E-06	1.43E-04
HCFC-225ca	3.94E-06	0.00E+00	3.94E-06	5.56E-06	0.00E+00	5.56E-06	7.76E-06	0.00E+00	7.76E-06
HCFC-225cb	1.86E-05	0.00E+00	1.86E-05	2.72E-05	0.00E+00	2.72E-05	3.79E-05	0.00E+00	3.79E-05
Halon-1211	4.36E-05	-4.43E-04	-3.99E-04	8.61E-05	-8.74E-04	-7.88E-04	1.20E-04	-1.22E-03	-1.10E-03
Halon-1301	7.78E-05	-7.42E-04	-6.64E-04	3.26E-04	-3.10E-03	-2.78E-03	5.79E-04	-5.52E-03	-4.94E-03
Halon-2402	3.38E-05	-6.24E-04	-5.90E-04	7.50E-05	-1.38E-03	-1.31E-03	1.05E-04	-1.95E-03	-1.84E-03
Hydrofluorocarbons									
HFC-23	1.10E-04		1.10E-04	6.74E-04		6.74E-04	3.04E-03		3.04E-03
HFC-32	2.14E-05		2.14E-05	3.08E-05		3.08E-05	4.30E-05		4.30E-05
HFC-43-10mee	3.80E-05		3.80E-05	7.48E-05		7.48E-05	1.05E-04		1.05E-04
HFC-125	5.83E-05		5.83E-05	1.60E-04		1.60E-04	2.30E-04		2.30E-04
HFC-134a	3.51E-05		3.51E-05	6.52E-05		6.52E-05	9.10E-05		9.10E-05

HFC-143a	5.40E-05		5.40E-05	2.04E-04		2.04E-04	3.34E-04		3.34E-04
HFC-227ea	4.87E-05		4.87E-05	1.47E-04		1.47E-04	2.17E-04		2.17E-04
HFC-245fa	3.10E-05		3.10E-05	4.72E-05		4.72E-05	6.59E-05		6.59E-05
HFC-152a	4.01E-06		4.01E-06	5.67E-06		5.67E-06	7.91E-06		7.91E-06
HFC-236fa	7.44E-05		7.44E-05	4.47E-04		4.47E-04	1.83E-03		1.83E-03
HFC-365mfc	2.32E-05		2.32E-05	3.62E-05		3.62E-05	5.06E-05		5.06E-05
Perfluorinated compounds									
Sulphur hexafluoride	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
Nitrogen trifluoride	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
PFC-14	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
PFC-116	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
PFC-218	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
PFC-318	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
PFC-3-1-10	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
PFC-4-1-12	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
PFC-5-1-14	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
PFC-9-1-18	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
Trifluoromethyl sulphur pentafluoride	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03	9.65E-03
Fluorinated ethers									
HFE-125	1.27E-04		1.27E-04	6.80E-04		6.80E-04	1.82E-03		1.82E-03
HFE-134	1.12E-04		1.12E-04	2.88E-04		2.88E-04	4.11E-04		4.11E-04
HFE-143a	2.42E-05		2.42E-05	3.45E-05		3.45E-05	4.81E-05		4.81E-05
HCFE-235da2	1.13E-05		1.13E-05	1.59E-05		1.59E-05	2.22E-05		2.22E-05
HFE-245cb2	2.55E-05		2.55E-05	3.67E-05		3.67E-05	5.12E-05		5.12E-05
HFE-245fa2	2.09E-05		2.09E-05	3.01E-05		3.01E-05	4.20E-05		4.20E-05
HFE-254cb2	1.28E-05		1.28E-05	1.80E-05		1.80E-05	2.51E-05		2.51E-05
HFE-347mcc3	1.82E-05		1.82E-05	2.63E-05		2.63E-05	3.66E-05		3.66E-05
HFE-347pcf2	1.76E-05		1.76E-05	2.64E-05		2.64E-05	3.68E-05		3.68E-05
HFE-356pcc3	3.55E-06		3.55E-06	5.01E-06		5.01E-06	6.99E-06		6.99E-06
HFE-449sl	9.86E-06		9.86E-06	1.40E-05		1.40E-05	1.95E-05		1.95E-05
HFE-569sf2	1.84E-06		1.84E-06	2.60E-06		2.60E-06	3.63E-06		3.63E-06
HFE-43-10pccc124	5.80E-05		5.80E-05	8.55E-05		8.55E-05	1.19E-04		1.19E-04
HFE-236ca12	7.38E-05		7.38E-05	1.29E-04		1.29E-04	1.80E-04		1.80E-04
HFE-338pcc13	4.66E-05		4.66E-05	6.85E-05		6.85E-05	9.56E-05		9.56E-05
Perfluoropolyethers									
PFPMIE	7.00E-05		7.00E-05	4.70E-04		4.70E-04	5.59E-03		5.59E-03
Hydrocarbons and other compounds									
Dimethylether	1.37E-08		1.37E-08	1.93E-08		1.93E-08	2.70E-08		2.70E-08
Methylene chloride	2.82E-07	0.00E+00	2.82E-07	3.99E-07	0.00E+00	3.99E-07	5.57E-07	0.00E+00	5.57E-07
Methyl chloride	4.17E-07	0.00E+00	4.17E-07	5.88E-07	0.00E+00	5.88E-07	8.21E-07	0.00E+00	8.21E-07

Human health damage factors

To calculate the human health damage factors, for cardiovascular diseases, diarrhea, malnutrition and malaria, the region specific relative risk factors for year 2030 were taken from McMichael et al. (2003), while the factors for inland and coastal flooding derived from Ezzati et al. (2004). Table 4.4 gives an overview of the health effects considered and the assumptions taken in the calculations of the relative risk factors. The disease-specific and region-specific relative risk factors were available for the future emission scenario S550 and S750. In scenario S550, a temperature rise of 0.5°C in year 2030 is expected. The S750 scenario predicts a temperature rise of 0.68 °C in year 2030 (McMicael et al., 2003). Table 4.6 and 4.7 present the Relative Risk factors used in the study.

Table 4.4. Health effects considered, related assumptions and burden of disease type.

Causes of health effects	**Assumptions**	**Burden of disease**
Malnutrition	Models of grain cereals and soybean to estimate the effects of change in temperature, rainfall and CO2 on future crop yields were used.	Nutritional deficiencies
Diarrhoea	Effects of increasing temperature on the incidence of all-cause diarrhoea were addressed, while effects of rainfall were excluded.	Diarrhoeal diseases
Heat stress	Temperature attributable deaths were calculated. The burden of disease of all cardiovascular diseases were used.	All cardiovascular diseases
Natural disasters	The increased incidence of coastal and inland flooding were assessed.	Drowning
Vector borne diseases	Malaria was considered, caused by P. falciparum and P. vivax. Increasing incidence within already endemic populations was excluded.	Malaria

Several assumptions are connected to the Relative Risk factors for human health damage as developed by Ezzati (2004) and McMichael et al. (2003). Table 4.5 gives an overview of the assumed adaptations for the different health effects, the scenario range of the relative risk factors and how this is linked to the cultural perspectives. For cardiovascular diseases, malnutrition and natural disasters, the scenario range results partly from adaptation uncertainties considered by the model calculations, what is considered in defining the perspectives. For Diarrhea, the scenario range results only from model uncertainties.

Table 4.5. Assumptions on adaptation and vulnerability in RR factors (McMicael et al., 2003), linked to the different perspectives.

Health effects	Biological adaptation	Socioeconomic adaptation	Uncertainty range	Comment
Cardiovascular diseases	Biological and behavioral adaptation (like the increase use of air-conditioning) are considered in model calculations.	None	Factor 3 – 6 from low to high	Uncertainty range derived from Ezzati et al. (2004) Appendix B, page 1632-1633. For the individualist perspective full adaptation to climate change is assumed, what results in no increasing risk. For the hierarchist and egalitarian perspective the mid and maximum range of the uncertainty is used, what reflects lower adaptation possibilities.
Diarrhea	None	Projections of future changes in GDP are applied. For any country that attains per capita a GDP above US$ 6000/year, a RR=1 is assumed.	Factor 1.2	A 0 to 10% uncertainty range on the mid-range estimate is stated by Ezzati et al. (2004), page 1574. Socioeconomic adaptation is included at the same level for all three perspectives.
Malnutrition	None	An accumulating GDP and a 50% trade liberalization in agriculture is introduced gradually.	0 for low; factor 2 from mid to high	Socioeconomic adaptation is included in all three perspectives. In the absence of formal sensitivity analyses of the complete model, the uncertainty estimates presented by Ezzati et al. (2004) are tentative The lower range covers full adaptation to changes in agricultural output (i.e. no change in risk), and is considered representative for the individualist perspective. The upper range refers to a doubling of the estimate of the most likely relative risk and is used for the egalitarian perspective
Natural disasters	None	Model assumes that protection evolves over time in proportion to projected increases in GDP.	Region dependent. See table 4.7.	Uncertainties relate to the frequency of extreme weather events as modeled by various scenarios, and to evolving protection over time due to projected increases in GNP. Uncertainty factor derived from Ezzati et al. page 1590 (2004). The mid, mean and max range is linked to individualist, hierarchist and egalitarian perspective.
Malaria	None	None	.	

The burden of disease figures for cardiovascular diseases, diarrhea, malnutrition and malaria, for an optimistic, pessimistic and baseline scenario of 2030 were taken from Mathers and Loncar (2006). These DALYs reflect age weighting and a discount rate of 3% [0.03,1] in all scenarios. Using the ratio of the DALY [0,0] and DALY [0.03,1] for year 2002 (WHO, 2004) per disease and world region as scaling factors, the DALYs of the pessimistic future scenario were converted to DALYs with no discount rate and age weighting. The DALYs of the baseline future scenario were converted in the same way, but now using the ratio of the DALY [0.03,0] and DALY [0.03,1] for year 2002 per disease and world region, as scaling factors (table 4.6).

Table 4.6. The DALYs and Relative Risk factors for 2030 (McMicael et al., 2003) used in the calculation of the human health damage factors (Equation 9 in the main text), for cardiovascular diseases, diarrhea, malaria and malnutrition, as applied in the three perspectives.

		Individualist			Hierarchist			Egalitarian		
		$DALY_{r,h}$	RR		$DALY_{r,h}$	RR		$DALY_{r,h}$	RR	
	Scenarios discount rate, age weigthing	Optim. [0.03,1][a]	S550[b]	S750[b]	Baseline [0.03,0][a]	S550[b]	S750[b]	Pessim. [0,0][a]	S550[b]	S750[b]
Card iovascular	African region				2.86E+07	1.004	1.004	4.36E+07	1.007	1.008
	Eastern Mediterranean region				1.57E+07	1.004	1.003	2.40E+07	1.007	1.005
	Latin American and Caribbean region				1.66E+07	1.002	1.003	2.36E+07	1.004	1.005
	South-East Asian region				7.66E+07	1.004	1.005	1.18E+08	1.008	1.009
	Western Pacific region				5.97E+07	1.000	1.000	9.67E+07	1.000	1.000
	Developed countries				5.49E+07	1.000	1.000	7.66E+07	1.000	1.000
Diarrhea	African region	1.11E+07	1.050	1.060	1.86E+07	1.050	1.060	6.67E+07	1.050	1.060
	Eastern Mediterranean region	7.29E+05	1.045	1.045	9.27E+05	1.045	1.045	3.60E+06	1.045	1.045
	Latin American and Caribbean region	8.06E+05	1.000	1.000	6.27E+05	1.000	1.000	2.27E+06	1.000	1.000
	South-East Asian region	4.09E+06	1.055	1.060	5.87E+06	1.055	1.060	2.87E+07	1.055	1.060
	Western Pacific region	2.49E+06	1.000	1.000	1.52E+06	1.000	1.000	8.37E+06	1.000	1.000
	Developed countries	3.50E+05	1.000	1.000	3.74E+05	1.000	1.000	1.22E+06	1.000	1.000
Malnutrition	African region				7.13E+06	1.000	1.045	1.96E+07	1.000	1.090
	Eastern Mediterranean region				9.43E+05	1.030	1.100	2.33E+06	1.060	1.200
	Latin American and Caribbean region				5.04E+05	1.050	1.110	1.26E+06	1.100	1.220
	South-East Asian region				4.26E+06	1.110	1.160	1.27E+07	1.220	1.320
	Western Pacific region				1.62E+06	1.010	1.025	4.71E+06	1.020	1.050
	Developed countries				7.57E+05	1.000	1.000	1.66E+06	1.000	1.000
Malaria	African region	1.13E+07	1.045	1.055	2.04E+07	1.045	1.055	7.48E+07	1.045	1.055
	Eastern Mediterranean region	4.65E+04	1.045	1.135	8.03E+04	1.045	1.135	2.96E+05	1.045	1.135
	Latin American and Caribbean region	2.84E+04	1.075	1.090	4.08E+04	1.075	1.090	9.57E+04	1.075	1.090
	South-East Asian region	1.68E+05	1.005	1.005	3.02E+05	1.005	1.005	1.42E+06	1.005	1.005
	Western Pacific region	1.19E+05	1.215	1.265	2.18E+05	1.215	1.265	9.60E+05	1.215	1.265
	Developed countries	1.27E+05	1.260	1.165	1.28E+05	1.260	1.165	1.62E+05	1.260	1.165

[a][0.03,1] corresponds with 3% discount rate and age weighting; [0.03,0] corresponds with 3% discount rate and no age weighting; [0,0] corresponds with no discount rate and no age weighting. [b]The S550 scenario results in a stabilized temperature increase of about 3 degrees; The S750 scenario results in a stabilized temperature increase of about 4 degrees.

For coastal and inland flooding, no specific DALYs for the year 2030 were available in Mathers and Loncar (2006). Ezzati et al. (2004) published the annual incidence of deaths per 10.000.000 caused by floods for 2030, in the absence of climate change. The underlying database is the EM-DAT database, which records the numbers of deaths and injuries attributed to each natural disaster in the last 100 years (Em-Dat, 2002). Ezzati et al. (2004) scaled the annual incidence of flood death under baseline climate conditions for 1990 to 2030 considering increasing population density and changes in flood defences by rising GDP. To calculate the total number of deaths due to coastal and inland flooding in 2030, the relative incidence estimates were linked to total population estimates in 2030. The corresponding DALYs were calculated with the DALY calculation template provided by the WHO and presented in table 4.7. Table 4.8 shows the region-specific attributable human health burden the five diseases included for the three perspectives.

Table 4.7. The DALYs and Relative Risk factors used for the three perspectives (Ezzati et al., 2004), used to calculate the human health damage factors (Equation 9 in the main text), for inland and coastal flooding.

		Individualist			Hierarchist			Egalitarian		
		$DALY_{r,h}$	RR		$DALY_{r,h}$	RR		$DALY_{r,h}$	RR	
	Scenarios discount rate, age weighting	Optim [0.03,1][a]	S550[b]	S750[b]	Baseline [0.03,0] [a]	S550[b]	S750[b]	Pessim [0,0] [a]	S550[b]	S750[b]
Nat. Disasters (coastal)	Afr-D	0.00E+00	1.22	1.24	0.00E+00	1.44	1.48	0.00E+00	1.89	1.96
	Afr-E	0.00E+00	1.06	1.07	0.00E+00	1.12	1.13	0.00E+00	1.25	1.27
	Amr-A	0.00E+00	1.06	1.07	0.00E+00	1.13	1.14	0.00E+00	1.25	1.27
	AMR-B	1.32E+03	1.45	1.48	1.25E+03	1.90	1.96	2.49E+03	2.81	2.93
	AMR-D	7.80E+01	2.29	2.38	7.30E+01	3.58	3.76	1.46E+02	3.54	6.52
	EMR-B	0.00E+00	1.27	1.28	0.00E+00	1.53	1.57	0.00E+00	8.28	2.13
	EMR-D	0.00E+00	2.01	2.09	0.00E+00	3.01	3.18	0.00E+00	2.50	5.36
	EUR-A	0.00E+00	1.04	1.05	0.00E+00	1.09	1.10	0.00E+00	6.82	1.20
	EUR-B	0.00E+00	2.89	3.01	0.00E+00	4.78	5.02	0.00E+00	1.18	9.05
	EUR-C	7.40E+01	1.01	1.02	7.30E+01	1.03	1.03	1.40E+02	8.55	1.06
	SEAR-B	1.13E+02	1.14	1.15	1.04E+02	1.28	1.30	2.12E+02	1.06	1.59
	SEAR-D	7.90E+03	1.01	1.01	7.29E+03	1.03	1.03	1.48E+04	1.56	1.05
	WPR-A	4.40E+01	1.01	1.02	4.20E+01	1.03	1.03	8.28E+01	1.05	1.06
	WPR-B	4.91E+03	1.02	1.02	4.73E+03	1.04	1.04	9.22E+03	1.06	1.08
Nat. Disasters (inlandl)	Afr-D	1.79E+03	1.00	1.00	1.56E+03	2.30	1.99	3.33E+03	3.13	2.64
	Afr-E	1.02E+04	1.00	1.00	8.96E+03	2.30	1.99	1.94E+04	3.18	2.65
	Amr-A	1.65E+03	1.00	1.00	1.56E+03	11.50	9.66	3.11E+03	18.69	15.61
	AMR-B	2.55E+04	1.00	1.00	2.36E+04	2.60	3.18	4.69E+04	3.67	4.65
	AMR-D	8.54E+03	1.00	1.00	8.05E+03	2.92	2.26	1.61E+04	4.20	3.10
	EMR-B	7.42E+03	1.00	1.00	6.66E+03	3.20	4.04	1.40E+04	4.63	6.03
	EMR-D	3.93E+04	1.00	1.00	3.52E+04	5.29	4.56	7.41E+04	8.17	6.94
	EUR-A	6.86E+02	1.00	1.00	6.74E+02	5.30	5.27	1.29E+03	8.28	8.20
	EUR-B	3.03E+03	1.00	1.00	2.98E+03	2.32	3.16	5.71E+03	3.22	4.65
	EUR-C	4.95E+02	1.00	1.00	4.87E+02	2.45	4.31	9.33E+02	3.42	6.46
	SEAR-B	5.23E+03	1.00	1.00	4.82E+03	2.51	3.57	9.81E+03	3.60	5.37
	SEAR-D	5.33E+04	1.00	1.00	4.91E+04	1.73	1.39	9.99E+04	2.22	1.68
	WPR-A	8.01E+02	1.00	1.00	7.72E+02	2.91	2.04	1.51E+03	4.29	2.80
	WPR-B	2.61E+04	1.00	1.00	2.52E+04	1.88	2.00	4.91E+04	2.50	2.70

[a][0.03,1] corresponds with 3% discount rate and age weighting; [0.03,0] corresponds with 3% discount rate and no age weighting; [0,0] corresponds with no discount rate and no age weighting. [b]The S550 scenario results in a stabilized temperature increase of about 3 degrees; The S750 scenario results in a stabilized temperature increase of about 4 degrees.

Table 4.8. The region-specific attributable human health burden ($DALY_{r,h}$ in yr/yr) of the five diseases included for the three perspectives (outcomes of Equation 9 in the main text).

		Individualist		Hierarchist		Egalitarian	
	Scenario Temp rise (ºC)	S550[a] 0.50	S750[a] 0.68	S550[a] 0.50	S750[a] 0.68	S550[a] 0.50	S750[a] 0.68
	Attributable burden	$DALY_{r,h}$ (Yr/Yr)					
Cardiovascular	African region			9.99E+04	1.14E+05	3.05E+05	3.49E+05
	Eastern Mediterranean region			5.49E+04	3.92E+04	1.68E+05	1.20E+05
	Latin American and Caribbean region			3.32E+04	4.14E+04	9.43E+04	1.18E+05
	South-East Asian region			3.06E+05	3.45E+05	9.41E+05	1.06E+06
	Western Pacific region			0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Developed countries			0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Total ($DALY_h$)			4.94E+05	5.39E+05	1.51E+06	1.65E+06
Diarrhea	African region	5.54E+05	6.64E+05	9.29E+05	1.11E+06	3.34E+06	4.00E+06
	Eastern Mediterranean region	3.28E+04	3.28E+04	4.17E+04	4.17E+04	1.62E+05	1.62E+05
	Latin American and Caribbean region	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	South-East Asian region	2.25E+05	2.45E+05	3.23E+05	3.52E+05	1.58E+06	1.72E+06
	Western Pacific region	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Developed countries	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Total ($DALY_h$)	8.11E+05	9.42E+05	1.29E+06	1.51E+06	5.08E+06	5.89E+06

Malnutrition	African region			0.00E+00	3.21E+05	0.00E+00	1.76E+06
	Eastern Mediterranean region			2.83E+04	9.43E+04	1.40E+05	4.66E+05
	Latin American and Caribbean region			2.52E+04	5.55E+04	1.26E+05	2.78E+05
	South-East Asian region			4.68E+05	6.81E+05	2.79E+06	4.05E+06
	Western Pacific region			1.62E+04	4.04E+04	9.41E+04	2.35E+05
	Developed countries			0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Total ($DALY_h$)			5.38E+05	1.19E+06	3.15E+06	6.80E+06
Malaria	African region	5.08E+05	6.21E+05	9.16E+05	1.12E+06	3.37E+06	4.12E+06
	Eastern Mediterranean region	2.09E+03	6.28E+03	3.61E+03	1.08E+04	1.33E+04	3.99E+04
	Latin American and Caribbean region	2.13E+03	2.56E+03	3.06E+03	3.67E+03	7.18E+03	8.61E+03
	South-East Asian region	8.41E+02	8.41E+02	1.51E+03	1.51E+03	7.08E+03	7.08E+03
	Western Pacific region	2.55E+04	3.14E+04	4.69E+04	5.78E+04	2.06E+05	2.54E+05
	Developed countries	3.29E+04	2.09E+04	3.32E+04	2.11E+04	4.21E+04	2.67E+04
	Total ($DALY_h$)	5.72E+05	6.83E+05	1.00E+06	1.21E+06	3.64E+06	4.45E+06
Nat. Disasters (Inland and coastal flooding)	Afr-D	0.00E+00	0.00E+00	2.03E+03	1.54E+03	7.10E+03	5.47E+03
	Afr-E	0.00E+00	0.00E+00	1.16E+04	8.87E+03	4.22E+04	3.20E+04
	Amr-A	0.00E+00	0.00E+00	1.63E+04	1.35E+04	5.50E+04	4.54E+04
	AMR-B	5.95E+02	6.35E+02	3.88E+04	5.26E+04	1.30E+05	1.76E+05
	AMR-D	1.01E+02	1.08E+02	1.56E+04	1.03E+04	5.18E+04	3.46E+04
	EMR-B	0.00E+00	0.00E+00	1.46E+04	2.02E+04	5.08E+04	7.04E+04
	EMR-D	0.00E+00	0.00E+00	1.51E+05	1.25E+05	5.31E+05	4.40E+05
	EUR-A	0.00E+00	0.00E+00	2.90E+03	2.88E+03	9.40E+03	9.29E+03
	EUR-B	0.00E+00	0.00E+00	3.94E+03	6.44E+03	1.27E+04	2.09E+04
	EUR-C	7.40E-01	1.48E+00	7.08E+02	1.61E+03	3.31E+03	5.10E+03
	SEAR-B	1.58E+01	1.70E+01	7.31E+03	1.24E+04	2.55E+04	4.30E+04
	SEAR-D	7.90E+01	7.90E+01	3.61E+04	1.94E+04	1.30E+05	6.87E+04
	WPR-A	4.40E-01	8.80E-01	1.48E+03	8.04E+02	4.96E+03	2.72E+03
	WPR-B	9.81E+01	9.81E+01	2.24E+04	2.54E+04	7.42E+04	8.42E+04
	Total ($DALY_h$)	8.90E+02	9.39E+02	3.25E+05	3.01E+05	1.13E+06	1.04E+06
	Totaal DALY (Yr/Yr)	1.38E+06	1.63E+06	3.65E+06	4.76E+06	1.45E+07	1.98E+07
	ΔTEMP (ºC)	1.80E-01		1.80E-01		1.80E-01	
	ΔDALY	2.43E+05		1.10E+06		5.32E+06	
	Damage factor (Yr/Yr.ºC)	1.35E+06		6.12E+06		2.95E+07	

[a]The S550 scenario results in stabilized temperature increase of about 3 degrees; The S750 scenario results in stabilized temperature increase of about 4 degrees.

Ecosystem damage factors

The damage factors for ecosystems were calculated using the species extinction projections presented in Thomas et al. (2004). The potentially disappeared percentage of species was calculated for three different interpretations of the species area relationship, using all species and Red list species only (IUCN, 2001) and with and without dispersal (table 4.9). The following steps are taken to calculate the damage factors:

1. Per study, the difference in species disappearance between the highest and lowest global temperature increase was calculated for the different interpretations of the species area relationship, all species or Red list species only, and with or without dispersal. This resulted in 12 species disappearances per study. If only one future scenario within a specific study was available, the current situation was used as the point of departure;
2. Per study, the change of the PDF of the 12 outcomes was divided by the corresponding temperature change to obtain the 12 ecosystem damage factors per study;
3. An average ecosystem damage factor from the different interpretations of the species area relationship was calculated for combinations of the assumption with and without dispersal and

using all species or Red list species only. This resulted in four terrestrial ecosystem damage factors per study;

4. An average terrestrial ecosystem damage factor over all the studies was calculated per scenario, using the number of species per study as a weighting factor.

Table 4.9. Underlying data applied in the calculation of the ecosystem damage factor (DF) with formula (10), using 3 different methods and the red list species (Thomas et al., 2004). The average damage factors are species weighted.

	Number of species	**Temperature change (ºC)**			**Ecosystem damage factor with dispersal (%/ºC)**				**Ecosystem damage factor without dispersal (%/ºC)**				
		low	mid	high	Method 1	Method 2	Method 3	Red list species	Method 1	Method 2	Method 3	Red list species	note
Queensland: Mammals	11	1		3.5	15.2	16.4	26.0	24.4					1
Queensland: Birds	13	1		3.5	16.8	18.0	24.8	29.2					1
Queensland:Frogs	23	1		3.5	12.0	14.0	19.6	22.0					1
Queensland:Reptiles	18	1		3.5	14.4	15.2	12.8	26.8					1
Australia: butterflies	24	0.9	1.8	3	9.4	8.8	11.2	15.3	11.8	12.4	14.1	22.4	2
Mexico: mammals	96	1.35	2		0.0	1.5	3.1	4.6	1.5	1.5	3.1	3.1	4
Mexico, birds	186	1.35	2		1.5	1.5	1.5	1.5	0.0	0.0	0.0	-1.5	4
Mexico:butterflies	41	1.35	2		3.1	1.5	1.5	0.0	4.6	4.6	6.2	9.2	4
Sout Africa: Mammals	5			3	8.0	10.7	15.3	0.0	9.3	12.0	19.7	23.0	3
South Africa: Birds	5			3	9.3	9.7	10.7	0.0	11.0	11.7	13.3	17.0	3
South Africa: Reptiles	26			3	7.0	7.3	9.0	0.0	11.0	12.0	15.0	19.7	3
South Africa: Butterflies	4			3	4.3	2.3	2.7	0.0	11.7	15.0	23.3	26.0	3
Brazil: Cerrado plants	163	1.35	2						15.4	13.8	18.5	48.9	4
Europe: birds	34			3	1.1	1.6	1.6	1.9	3.5	6.8	10.3	13.0	1
South Africa: Proteaceae	243		2		12.0	10.5	13.5	19.0	16.0	15.0	20.0	26.0	3
Europe: plants	192	1.7	1.9	2.3	1.7	1.7	1.7	3.3	6.7	10.0	11.7	18.3	2
Average DF (%/ºC)					5.6	5.4	6.8					19.4	

Note: 1 Difference between high and low local temperature increase. 2 difference between high and low temperature for global temperature increase. 3 only one temperature available, therefore the slope is determined by comparing zero temperature change and damage with high (global) temperature increase. 4 difference between mid and low global temperature increase.

Characterization factors

Table 4.10: CFs of human damage (CF HH; dDALY/dE) and ecosystem damage (CF ES; dPDF/dE), due to climate change, for the three perspectives. Ozone depleting chemicals can have negative CFs for global warming in the Individualist perspective due to the inclusion of indirect cooling effects. Temperature factors are included as well.

	Individualist			Hierarchist			Egalitarian		
	TF20	CF HH	CF ES	TF 100	CF HH	CF ES	TF inf	CF HH	CF ES
Unit	°Ckton^{-1}yr	DALY/kton	km^2.yrkton	°Ckton^{-1}yr	DALY/kton	km^{2}yr/kton	°Ckton^{-1}yr	DALY/kton	km^2.yr/kton
CO_2	8.40E-09	1.13E-02	5.35E-02	4.17E-08	2.55E-01	2.66E-01	5.94E-07	1.76E+01	1.24E+01
CH_4	6.98E-07	9.41E-01	4.45E+00	1.21E-06	7.43E+00	7.74E+00	1.15E-06	3.40E+01	2.41E+01
N_2O	2.66E-06	3.58E+00	1.69E+01	1.36E-05	8.33E+01	8.68E+01	3.26E-05	9.61E+02	6.81E+02
Substances controlled by the Montreal Protocol									
CFC-11	-1.48E-05	-2.00E+01	-9.45E+01	2.17E-04	1.33E+03	1.38E+03	3.39E-04	1.00E+04	7.10E+03
CFC-12	2.14E-05	2.89E+01	1.36E+02	4.97E-04	3.04E+03	3.17E+03	1.10E-03	3.24E+04	2.30E+04
CFC-13	9.92E-05	1.34E+02	6.32E+02	6.58E-04	4.03E+03	4.19E+03	6.35E-03	1.88E+05	1.33E+05
CFC-113	2.97E-05	4.00E+01	1.89E+02	2.80E-04	1.71E+03	1.78E+03	5.64E-04	1.67E+04	1.18E+04
CFC-114	7.06E-05	9.52E+01	4.50E+02	4.58E-04	2.80E+03	2.92E+03	2.26E-03	6.66E+04	4.72E+04
CFC-115	4.77E-05	6.44E+01	3.04E+02	3.36E-04	2.06E+03	2.14E+03	8.21E-03	2.43E+05	1.72E+05
Carbon tetrachloride	-2.87E-05	-3.87E+01	-1.83E+02	6.39E-05	3.91E+02	4.07E+02	9.11E-05	2.69E+03	1.91E+03
Methyl bromide	-4.71E-05	-6.36E+01	-3.00E+02	2.19E-07	1.34E+00	1.40E+00	3.06E-07	9.03E+00	6.40E+00
Methyl chloroform	-6.33E-06	-8.53E+00	-4.03E+01	6.68E-06	4.09E+01	4.26E+01	9.33E-06	2.75E+02	1.95E+02
HCFC-22	3.84E-05	5.17E+01	2.44E+02	8.24E-05	5.04E+02	5.25E+02	1.15E-04	3.40E+03	2.41E+03

HCFC-123	2.46E-06	3.31E+00	1.57E+01	3.54E-06	2.16E+01	2.25E+01	4.94E-06	1.46E+02	1.03E+02
HCFC-124	1.90E-05	2.57E+01	1.21E+02	2.78E-05	1.70E+02	1.77E+02	3.88E-05	1.15E+03	8.11E+02
HCFC-141b	1.78E-05	2.40E+01	1.14E+02	3.31E-05	2.02E+02	2.11E+02	4.62E-05	1.36E+03	9.66E+02
HCFC-142b	4.87E-05	6.57E+01	3.10E+02	1.05E-04	6.45E+02	6.72E+02	1.48E-04	4.36E+03	3.09E+03
HCFC-225ca	3.94E-06	5.31E+00	2.51E+01	5.56E-06	3.40E+01	3.54E+01	7.76E-06	2.29E+02	1.62E+02
HCFC-225cb	1.86E-05	2.51E+01	1.19E+02	2.72E-05	1.66E+02	1.73E+02	3.79E-05	1.12E+03	7.93E+02
Halon-1211	-3.99E-04	-5.38E+02	-2.54E+03	8.61E-05	5.26E+02	5.48E+02	1.20E-04	3.56E+03	2.52E+03
Halon-1301	-6.64E-04	-8.95E+02	-4.23E+03	3.26E-04	1.99E+03	2.08E+03	5.79E-04	1.71E+04	1.21E+04
Halon-2402	-5.90E-04	-7.96E+02	-3.76E+03	7.50E-05	4.58E+02	4.78E+02	1.05E-04	3.11E+03	2.20E+03
Hydrofluorocarbons									
HFC-23	1.10E-04	1.48E+02	7.01E+02	6.74E-04	4.12E+03	4.29E+03	3.04E-03	8.97E+04	6.36E+04
HFC-32	2.14E-05	2.89E+01	1.37E+02	3.08E-05	1.88E+02	1.96E+02	4.30E-05	1.27E+03	8.99E+02
HFC-43-10mee	3.80E-05	5.13E+01	2.42E+02	7.48E-05	4.58E+02	4.77E+02	1.05E-04	3.09E+03	2.19E+03
HFC-125	5.83E-05	7.86E+01	3.71E+02	1.60E-04	9.78E+02	1.02E+03	2.30E-04	6.81E+03	4.82E+03
HFC-134a	3.51E-05	4.74E+01	2.24E+02	6.52E-05	3.99E+02	4.15E+02	9.10E-05	2.69E+03	1.90E+03
HFC-143a	5.40E-05	7.29E+01	3.44E+02	2.04E-04	1.25E+03	1.30E+03	3.34E-04	9.85E+03	6.98E+03
HFC-227ea	4.87E-05	6.57E+01	3.10E+02	1.47E-04	8.99E+02	9.37E+02	2.17E-04	6.40E+03	4.54E+03
HFC-245fa	3.10E-05	4.18E+01	1.98E+02	4.72E-05	2.89E+02	3.01E+02	6.59E-05	1.94E+03	1.38E+03
HFC-152a	4.01E-06	5.41E+00	2.56E+01	5.67E-06	3.47E+01	3.61E+01	7.91E-06	2.34E+02	1.65E+02
HFC-236fa	7.44E-05	1.00E+02	4.74E+02	4.47E-04	2.74E+03	2.85E+03	1.83E-03	5.41E+04	3.83E+04
HFC-365mfc	2.32E-05	3.12E+01	1.48E+02	3.62E-05	2.22E+02	2.31E+02	5.06E-05	1.49E+03	1.06E+03
Perfluorinated compounds									
Sulphur hexafluoride	1.49E-04	2.01E+02	9.52E+02	1.04E-03	6.37E+03	6.63E+03	4.72E-02	1.40E+06	9.88E+05
Nitrogen trifluoride	1.23E-04	1.66E+02	7.82E+02	8.22E-04	5.03E+03	5.24E+03	9.08E-03	2.68E+05	1.90E+05
PFC-14	4.78E-05	6.45E+01	3.05E+02	3.37E-04	2.06E+03	2.15E+03	2.36E-01	6.96E+06	4.93E+06
PFC-116	7.92E-05	1.07E+02	5.05E+02	5.57E-04	3.41E+03	3.55E+03	7.81E-02	2.31E+06	1.63E+06
PFC-218	5.80E-05	7.82E+01	3.69E+02	4.03E-04	2.46E+03	2.57E+03	1.49E-02	4.40E+05	3.12E+05
PFC-318	6.71E-05	9.05E+01	4.28E+02	4.68E-04	2.86E+03	2.98E+03	2.12E-02	6.27E+05	4.44E+05
PFC-3-1-10	5.81E-05	7.84E+01	3.70E+02	4.04E-04	2.47E+03	2.57E+03	1.49E-02	4.41E+05	3.13E+05
PFC-4-1-12	5.98E-05	8.06E+01	3.81E+02	4.18E-04	2.55E+03	2.66E+03	2.42E-02	7.15E+05	5.06E+05
PFC-5-1-14	6.08E-05	8.20E+01	3.87E+02	4.24E-04	2.59E+03	2.70E+03	1.92E-02	5.68E+05	4.02E+05
PFC-9-1-18	5.05E-05	6.81E+01	3.22E+02	3.43E-04	2.10E+03	2.18E+03	5.03E-03	1.48E+05	1.05E+05
Trifluoromethyl sulphur pentafluoride	1.21E-04	1.63E+02	7.70E+02	8.12E-04	4.97E+03	5.17E+03	9.65E-03	2.85E+05	2.02E+05
Fluorinated ethers									
HFE-125	1.27E-04	1.71E+02	8.07E+02	6.80E-04	4.16E+03	4.33E+03	1.82E-03	5.39E+04	3.82E+04
HFE-134	1.12E-04	1.51E+02	7.13E+02	2.88E-04	1.76E+03	1.84E+03	4.11E-04	1.21E+04	8.60E+03
HFE-143a	2.42E-05	3.26E+01	1.54E+02	3.45E-05	2.11E+02	2.20E+02	4.81E-05	1.42E+03	1.01E+03
HCFE-235da2	1.13E-05	1.52E+01	7.18E+01	1.59E-05	9.73E+01	1.01E+02	2.22E-05	6.56E+02	4.65E+02
HFE-245cb2	2.55E-05	3.44E+01	1.62E+02	3.67E-05	2.25E+02	2.34E+02	5.12E-05	1.51E+03	1.07E+03
HFE-245fa2	2.09E-05	2.82E+01	1.33E+02	3.01E-05	1.84E+02	1.92E+02	4.20E-05	1.24E+03	8.78E+02
HFE-254cb2	1.28E-05	1.72E+01	8.13E+01	1.80E-05	1.10E+02	1.15E+02	2.51E-05	7.43E+02	5.26E+02
HFE-347mcc3	1.82E-05	2.45E+01	1.16E+02	2.63E-05	1.61E+02	1.67E+02	3.66E-05	1.08E+03	7.67E+02
HFE-347pcf2	1.76E-05	2.37E+01	1.12E+02	2.64E-05	1.61E+02	1.68E+02	3.68E-05	1.09E+03	7.70E+02
HFE-356pcc3	3.55E-06	4.78E+00	2.26E+01	5.01E-06	3.06E+01	3.19E+01	6.99E-06	2.06E+02	1.46E+02
HFE-449sl	9.86E-06	1.33E+01	6.28E+01	1.40E-05	8.56E+01	8.92E+01	1.95E-05	5.77E+02	4.09E+02
HFE-569sf2	1.84E-06	2.48E+00	1.17E+01	2.60E-06	1.59E+01	1.66E+01	3.63E-06	1.07E+02	7.59E+01
HFE-43-10pccc124	5.80E-05	7.82E+01	3.69E+02	8.55E-05	5.23E+02	5.44E+02	1.19E-04	3.52E+03	2.50E+03
HFE-236ca12	7.38E-05	9.95E+01	4.70E+02	1.29E-04	7.88E+02	8.21E+02	1.80E-04	5.31E+03	3.76E+03
HFE-338pcc13	4.66E-05	6.28E+01	2.97E+02	6.85E-05	4.19E+02	4.36E+02	9.56E-05	2.82E+03	2.00E+03
Perfluoropolyethers									
PFPMIE	7.00E-05	9.44E+01	4.46E+02	4.70E-04	2.88E+03	3.00E+03	5.59E-03	1.65E+05	1.17E+05
Hydrocarbons and other compounds									
Dimethylether	1.37E-08	1.85E-02	8.73E-02	1.93E-08	1.18E-01	1.23E-01	2.70E-08	7.98E-01	5.65E-01
Methylene chloride	2.82E-07	3.81E-01	1.80E+00	3.99E-07	2.44E+00	2.54E+00	5.57E-07	1.64E+01	1.16E+01
Methyl chloride	4.17E-07	5.62E-01	2.66E+00	5.88E-07	3.60E+00	3.75E+00	8.21E-07	2.43E+01	1.72E+01

Chapter 5
Value choices in life cycle impact assessment of stressors causing human health damage

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Abstract

This article investigates how value choices in life cycle impact assessment can influence characterization factors (CFs) for human health. The Cultural Theory is used to define sets of value choices in the calculation of CFs, reflecting the individualist, hierarchist and egalitarian perspectives. CFs were calculated for interventions related to the impact categories water scarcity, tropospheric ozone formation, particulate matter formation, human toxicity, ionizing radiation, stratospheric ozone depletion and climate change.

With the Cultural Theory as a framework, we show that individual, hierarchical and egalitarian perspectives can lead to CFs that vary up to six orders of magnitude. For persistent substances, the choice in time horizon explains the differences among perspectives, while for non-persistent substances, the choice in age weighting and discount rate of DALY, and the type of effects or exposure routes, accounts for differences in CFs. The calculated global impact varies by two orders of magnitude, depending on the perspective selected and derives mainly from particulate matter formation and water scarcity for the individualist perspective, and from climate change for the egalitarian perspective.

Our results stress the importance of dealing with value choices in life cycle impact assessment and suggest further research for analyzing the practical consequences for life cycle assessment results.

Keywords uncertainty analysis • value choices • life cycle impact assessment • human health • Cultural Theory

5.1 Introduction

Uncertainties are inevitable in life cycle assessment (LCA), risk assessment, or any other analytical tool to assess environmental impacts (Huijbregts, 1998, Steen, 2006, French and Geldermann, 2005). Different types of uncertainty arise within each step of an assessment — for example, while collecting data, defining system boundaries, or calculating environmental impacts of emissions.

Several typologies are put forward to describe the different types of uncertainty (e.g., Morgan and Henrion, 1990, Wynne, 1992, van Asselt and Rotmans, 2002, Ascough Ii et al., 2008). In general, three types of uncertainties can be distinguished: measurement uncertainty, uncertainty from assumptions and uncertainty from ignorance. In this paper we focus on uncertainties from assumptions in life cycle impact assessment. Uncertainties from assumptions most often involve value choices. Assumptions can derive from lack in knowledge, whereby the choice of one option above another can be influenced by personal values such as, what is commonly accepted or familiarity. Hertwich et al. (2000) describe these value choices as contextual values. On the other hand, assumptions can also be driven by personal believes and values that reflect what we care about, without any science being involved. A typical example is the equity of different age groups or species. These value choices are defined as preference values (Hertwich et al., 2000).

Scenario analyses can be used to investigate the uncertainties related to assumptions or choices that reflect different personal values. Several tools and frameworks exist to cluster different personal values and define model scenarios (e.g., Schwartz and Mark, 1992, Tukker, 2002). Within LCA, the Cultural Theory has been used as a tool, as it both reflects visions on society and views on nature (e.g., Hofstetter, 1998, Frischknecht et al., 2000, Goedkoop et al., 2008). The Cultural Theory distinguishes five different perspectives from which people perceive the world and behave in it. Three of these are generally used within environmental decision making: the individualist, hierarchist and egalitarian perspectives (Hofstetter, 1998, Hofstetter et al., 2000). Each perspective reflects a hypothetical stakeholder or decision maker with a specific set of preferences and contextual values that explains one's view on society and nature (Schwarz and Thompson, 1990, Thompson et al., 1990, van Asselt and Rotmans, 1996). An indication that perspectives correspond to actual human groups can be derived from the study by Tukker et al. (2002). Based on an analysis of the toxicity controversy in Sweden and the Netherlands, they indicate that the individualist perspective corresponds with industry, the hierarchist perspective with the Environmental Protection Agency (in Sweden) or the Dutch environmental ministry and the egalitarian perspective with environmentalists.

Most impact assessment methodologies embed value choices without giving practitioners or decision makers the opportunity to assess the difference in result when applying a distinct world view (e.g., Jolliet et al., 2003, Hauschild and Potting, 2005). Some impact assessment methodologies do handle uncertainties arising from value choices by applying the Cultural Theory, but in a limited and not always

consistent way (e.g., Goedkoop and Spriensma, 1999, Goedkoop et al., 2008). Therefore, we argue for broader implementation of the Cultural Theory in an impact assessment methodology that combines several impact categories.

The goal of this paper is to address uncertainties related to assumptions and value choices in life cycle impact assessment. The goal is to derive three sets of characterization factors (CFs) for human health damage (expressed as disability-adjusted life years or DALYs), by implementing specific value choices for the individualist, hierarchist and egalitarian perspectives in existing impact assessment models. For each perspective, we defined value choices for seven human health impact categories: water scarcity, tropospheric ozone formation, particulate matter formation, human toxicity, ionizing radiation, stratospheric ozone depletion, and climate change. These categories address both local and global effects as well as short- and long-term effects, and are the most widely used environmental impact categories in life cycle assessment of human health (Hauschild et al., 2009). Our work focuses on human health damage, but is equally relevant to analyze impacts for ecosystem quality and resource depletion. To calculate CFs, existing damage models are adapted to the described set of value choices. The value choices recognized as main drivers for differences in CFs among perspectives are outlined and explained. The constructed impact assessment methodology is used to quantify the human health damage from annual global water consumption and outdoor emissions, and to analyze the differences among perspectives. Finally, the limitations of the analysis and future research needs are discussed.

5.2 Methodology

Value choices

The individualist, hierarchist and egalitarian perspectives each have their own contextual and preference values (Schwarz and Thompson, 1990, Hofstetter et al., 2000, Jager et al., 1997, van Asselt and Rotmans, 1996). The individualist perspective is characterized by weak group cohesion and regulations for social relations, and considers nature to be stable and able to recover from any disturbance. This coincides with the view that humans have a high adaptive capacity through technological and economic development. This view considers known damages as the most reliable basis for decisions and emphasizes present effects over future gains or losses. The hierarchist perspective is characterized by strong group cohesion with binding regulations for social relations and considers nature to be in equilibrium. This perspective coincides with the view that impacts can be avoided with proper management and the search for a balance between manageability and the precautionary principle. The egalitarian perspective has strong group cohesion (relationships) coupled with few regulations and considers nature to be fragile and unstable. This vision gives high priority to the precautionary principle and equal importance to present and future effects.

Figure 5.1 presents an overview of the different contextual and preference values, projected along the cause-effect pathway. For seven human health impact categories, existing damage models that calculate CFs were adapted to the three sets of value choices. Table 5.1 is a synopsis of the choices that are used in the calculations. For detailed descriptions see appendix 5.1 (table 5.4).

Preference values reflect what we care about, our moral values and ideas of what is good or bad for society, such as the concern for equity or future generations (Munthe, 1997, Hertwich et al., 2000). The following choices regarding different preferences were considered:

- The temporal vision of life and society is perspective-dependent (Jager et al., 1997). Time perspective can be applied by considering effects within a certain time horizon or by discounting future effects. Different time horizons were applied within the calculation from emission to effect, while discounting was applied to calculate the damage, namely discounting years of life lost in the future (Murray and Lopez, 1996c, Hellweg et al., 2003). Based on Jager et al. (1997) and Janssen and Rotmans (1995), we selected a time horizon of 20 years and a discount rate of 5% for the individualist perspective, emphasizing present and short-term effects. The hierarchist perspective has a more balanced time perspective and follows a 100-year time horizon, which is most frequently used by several organizations (ISO/TR14047, 2003, Steinfeld et al., 2006, PAS 2050, 2008). We propose a 3% discount rate, as this rate is used as default scenario in burden of disease calculations by the World Health Organization (Murray and Lopez, 1996c). The egalitarian perspective gives importance to long-term effects as current and future effects are considered equal. This coincides with an infinite time horizon and 0% discount rate (Jager et al., 1997, Janssen and Rotmans, 1995).
- Assigning value to a year of life at different ages (age weighting) depends on personal preference (Murray and Lopez, 1996c). The individualist perspective gives a higher value to more economically relevant subpopulations. The strong group cohesion of the hierarchist and egalitarian perspectives results in equality and thus no differentiation between individuals of different ages (Hofstetter, 1998).
- Including or excluding positive effects can be considered as a preference value choice (Jager et al., 1997). Examples of positive environmental effects are the cooling effects from chlorofluorocarbons and halons that counter climate change, as well as nitrogen oxides that degrade tropospheric ozone, countering ozone formation. Positive effects were only included for the individualist perspective following their positive attitude towards environmental benefits (Hofstetter, 1998).

Contextual values relate to our idea of how the world works. They reflect the influence of personal and social judgment when choosing one scientific assumption over its alternative, such as familiarity with a certain dataset or common acceptance (Hertwich et al., 2000). The following choices regarding different contextual values were considered:

- Limited knowledge on causalities reflects a different level of risk that is or is not accepted by a certain perspective. According to Thompson et al. (1990) the egalitarian perspective is risk-adverse,

while the individualist is risk taking. The hierarchist accepts a high level of risk, as long as the decision is made by experts (Thompson et al., 1990). Based on this consideration, the egalitarian perspective includes all known effects; the hierarchist perspective, likely effects; and for the individualist perspective, certain (proven) effects.

- Improved health care can reduce the DALYs attributable to a certain impact (Lorenzoni et al., 2005), while the level of legislation, education and research can increase protection and prevention. Differences in assumptions concerning the level of biological and socioeconomic adaptation possibilities, which also can be defined as management style (Ezzati et al., 2004), were considered in the definition of perspectives (Hofstetter et al., 2000). The individualist perspective coincides with an adaptive management style, the egalitarian with a preventive and comprehensive management style, and the hierarchist with a controlling and limited management style (Hofstetter et al., 2000, De Schryver et al., 2009).
- Future projections on demographic developments, population displacements, changes in gross domestic product, years of schooling and technology changes will alter the sensitivity, size and age composition of the population and thus influence the number of incidence cases attributable to a given emission (Mathers and Loncar, 2006). Future optimistic, baseline and pessimistic development scenarios, as defined by Mathers and Loncar (2006), coincide respectively with the individualist, hierarchist and egalitarian perspectives (De Schryver et al., 2009).

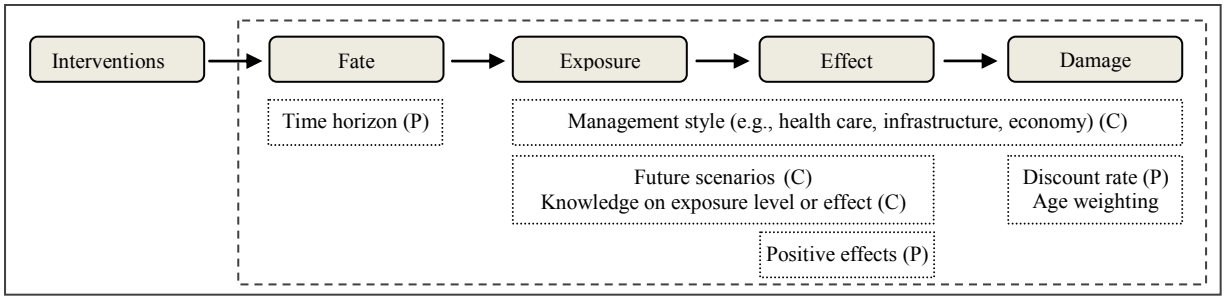

Figure 5.1. Overview of the cause-effect pathway, from intervention to damage. The calculation steps of the characterization factors are presented in the dashed box. Choices deriving from preference values (P) and contextual values (C), considered at each calculation step, are presented in the doted boxes.

Global damage of water consumption and outdoor emissions

The damage from outdoor emissions on the global scale was based on inventory data for the year 2000 from Sleeswijk et al. (2008). For water scarcity, country-specific water consumption data for the year 1995 were derived from the Watergap2 global model (Alcamo et al., 2003). For each impact category, the substances contributing most to the global impact per capita (DALY/(capita.yr)) were identified and presented. A population of 5.7 billion people in 1995 and 6.1 billion people in 2000 was considered (Undesa, 2008). To evaluate the global damage for the year 2000, the per capita global impact was multiplied with the 2000 population. Therefore, for water scarcity, it was assumed that the water impact per capita did not change between 1995 and 2000.

5 Value choices in life cycle impact assessment

Table 5.1. Combination of value choices deriving from preference values (P) and contextual values (C) for the CFs, expressed for three different cultural perspectives.

Impact category	Original choices[a]	Value choices	P/C	Individualist	Hierarchist	Egalitarian
All impact categories		Time horizon	P	20 years	100 years	Infinite
		Discount rate	P	5%	3%	0%
		Age weighting	P	Yes	No	No
Water scarcity Pfister et al. (2009)	Age weighting: yes Discount rate: 3% Regulation of flow: standard Food water requirement: 1350m^3/yr.capita	Regulation of flow (management style)	C	High	Standard	Standard
		Food water requirement (management style)	C	1000m^3/yr.capita (i.e., efficient management)	1350m^3/yr.capita (i.e., standard management)	1350m^3/yr.capita (i.e., standard management)
Ozone formation Van Zelm et al. (2008)	Age weighting: no Discount rate: 0% Morbidity effects: not included Positive effects from NO_x are included and excluded.	Morbidity effects[b]	C	No	No	Yes
		Positive effects from tropospheric ozone degradation from NO_x	P	Yes	No	No
Particulate matter Van Zelm et al. (2008)	Age weighting: no Discount rate: 0% Type of PM: primary PM_{10} and secondary PM from SO_2, NO_x and NH_3	Effects from primary PM_{10} and secondary PM from SO_2, NO_x and NH_3	C	Primary PM_{10}	Primary PM_{10} + Secondary PM from SO_2	Primary PM_{10} + Secondary PM from SO_2, NO_x and NH_3
Human toxicity Huijbregts et al. (2005)	Age weighting: no Discount rate: 0% Bioaccumulation essential metals: yes Carcinogenity: all substances Noncarcinogenic effects: included Time horizon: infinite	Bioaccumulation for essential metals	C	No	Yes	Yes
		Included substances on basis of carcinogenity		IARC classification: 1	IARC classification: 1, 2A, 2B	All
		Noncarcinogenic effects	C	No	Yes	Yes
Ionizing radiation Frischknecht et al. (2000)	No difference in cancer types Discount rate: 0% time horizon: individualist,100yr; hierarchist and egalitarian, 100,000yr Age weighting: individualist, yes; hierarchist and egalitarian, no	Cancer types[c]	C	Definite cancers	Definite and probable cancers	Definite, probable, possible and remainder cancers without information
Ozone depletion Hayashi et al. (2006)	Age weighting: no Discount rate: 0% Cataract: included Time horizon: infinite	Cataract	C	No	No	Yes
Climate change De Schryver et al. (2009)	Individualist: same as presented here except discount rate is 3% Hierarchist: same as presented here Egalitarian: same as presented here	Positive effects from ozone depletion	P	Yes	No	No
		Management style (Ezzati et al., 2004)	C	Adaptive management style	Controlling management style	Comprehensive management style
		Future developments (Mathers and Loncar, 2006)	C	Optimistic	Baseline	Pessimistic

Note: Detailed descriptions can be found in the appendix 5.1, table 5.4. m3/(yr.capita)= cubic meter per year per capita; yr= year; IARC= International Agency for Research on Cancer; PM= particulate matter.

[a]For all impact categories, except climate change and ionizing radiation, the original method developers presented one set of CFs embedding a certain set of value choices. For ionizing radiation and climate change, the original method developers presented CFs for the individualist, hierarchist and egalitarian perspectives.

[b]Morbidity effects included are asthma attacks, minor restricted activity days, respiratory hospital admissions, symptom days.

[c]Definite cancers are thyroid, bone marrow, lung, breast cancer; probable cancers are bladder, colon, ovary, liver, oesophagus, skin and stomach cancer; cancers without information are bone surface and all other cancers.

5.3 Results

Figure 5.2 presents the CFs for a subset of interventions (in DALY/kg, DALY/kBq or DALY/m^3) for the three cultural perspectives. The full list of CFs for 1239 substances and water consumption can be found in appendix 5.2. The number of substances included for the impact categories particulate matter and human toxicity depends on the level of knowledge about effects or exposure assumed for each perspective. For particulate matter, effects of secondary particulates from SO_2, NH_3 and NO_x are excluded for the individualist perspective, while effects from NH_3 and NO_x are excluded for the hierarchist perspective. For human toxicity, the availability in knowledge about carcinogenity and including or excluding non-carcinogenic effects results in CFs for 25 substances when applying the individualist perspective, 620 substances for the hierarchist perspective, and 1002 substances for the egalitarian perspective. For the individualist perspective positive effects are included and therefore the CF of some substances turns negative, such as nitrogen oxides for ozone formation and chlorofluorocarbons and halons for climate change.

Table 5.2 lists the differences in CFs for each impact category and the relevant choices that lead to differences among perspectives. Table 5.2 does not cover, however, the differences in case a CF becomes zero due to specific choices concerning the certainty of effects. It also does not address the fact that for ozone formation and climate change, some CFs range from negative (i.e., reducing the impacts) to positive (i.e., causing impacts) values, depending on whether positive effects are included for the perspectives. The type of DALY refers to the combination of age weighting and discount rate, which both influence the number of DALYs calculated per case (see tables 5.5 and 5.6 in appendix 5.1). The difference in CFs among perspectives is the largest for substances with a relatively long residence time in the environment (> 100 years). This is particularly the case for a number of emissions connected to the impact categories human toxicity (metals), ionizing radiation, ozone depletion and climate change. For example, the difference in CF between the individualist and egalitarian perspectives is five orders of magnitude for PFC-14 (for climate change) and four orders of magnitude for I-129 (for ionizing radiation). For toxicity of metals, the difference in CFs among perspectives can be as much as six orders of magnitude, due to the long lifetime and inclusion or exclusion of bioaccumulation of metals. For ozone depletion, the difference in CFs among perspectives is smaller, with two to three orders of magnitude between the individualist and egalitarian perspectives. Impact categories that cover substances with a shorter residence time in the environment, i.e., ozone formation and particulate matter, each show smaller differences among perspectives (up to 1.2 orders of magnitude). However, combining the effects of particulate matter and ozone formation for NO_x gives a difference of three orders of magnitude between the hierarchist and egalitarian perspectives. This is due to the exclusion of highly uncertain effects for the hierarchist perspective.

For water scarcity, the CFs show relatively small differences among perspectives attributable to value choices.

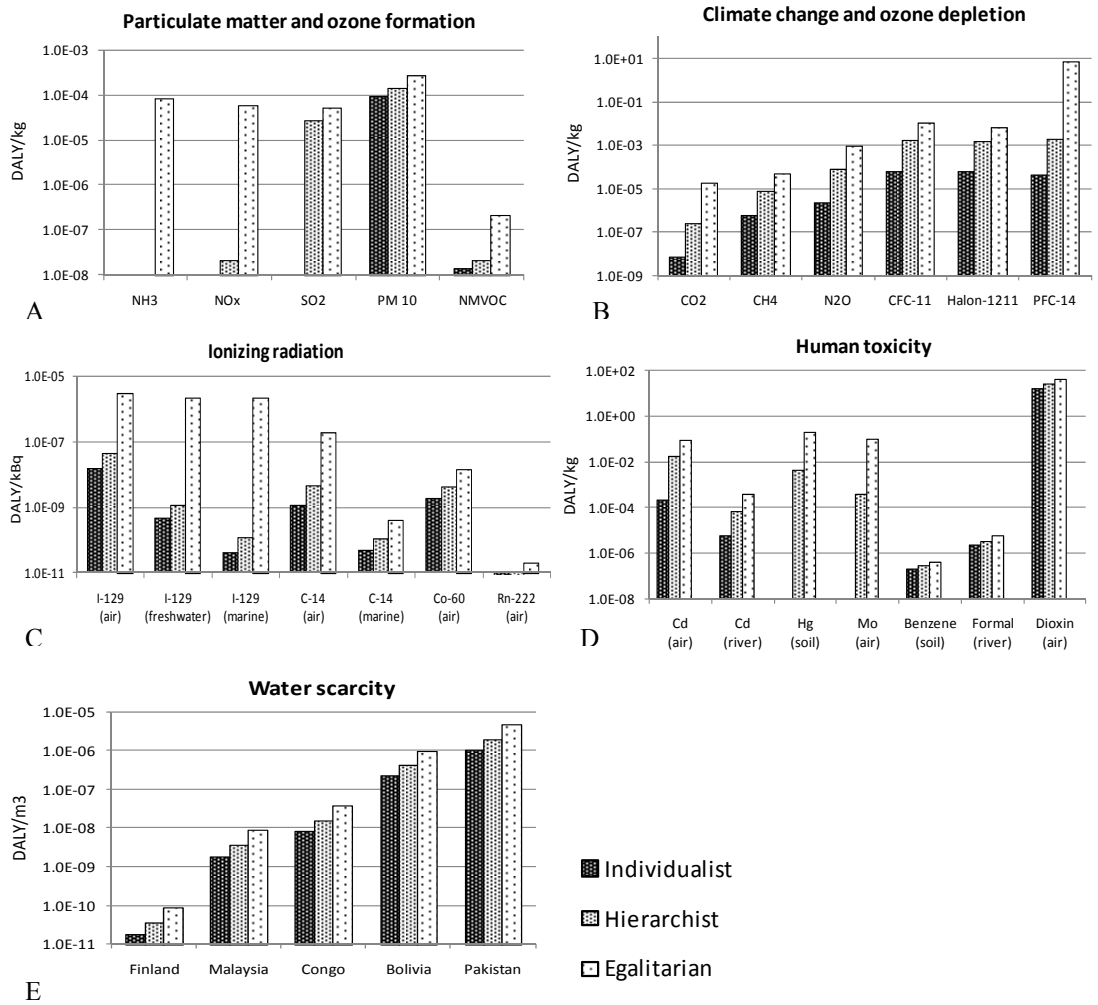

Figure 5.2. CFs for a range of selected substances following an individualist, hierarchist and egalitarian perspective. Graph A: combined CFs for particulate matter and ozone formation. Note that the negative CF for NO_x for the individualist perspective, $-4.2.10^{-8}$ DALY/kg, is not presented; Graph B: combined CFs for climate change and ozone depletion; Graph C: CFs for ionizing radiation; Graph D: CFs for human toxicity. Graph E: CFs for water scarcity. Note: CFs are expressed in DALY/kg, DALY/kBq or DALY/m^3 using a log-scale. DALY/kg= DALY per kilogram; DALY/kBq= DALY per kilobecquerel; DALY/m^3= DALY per cubic meter; Formal= Formaldehyde; Mo= Molybdenum; Cd= Cadmium;Hg=mercury; Dioxin= 2,3,7,8-tetrachlorodibenzo-p-dioxin.

Table 5.2. Difference in CFs among perspectives, per impact category (IC). For impact categories for which the time horizon is important (human toxicity, ionizing radiation, ozone depletion and climate change) a difference is made between long-lived (LL, i.e., > 100 years) and short-lived (SL, i.e., < 100 years) substances.

IC	egalitarian/ individualist	egalitarian/hierarchist	hierarchist/ individualist
Water scarcity	Max. difference is a factor of 18	Max. difference is a factor of 2.3	Max. difference is a factor of 8
	The regulation of flow is the most important choice, followed by the choice in type of DALY and water requirement	The choice in type of DALY is the only choice responsible for the difference	The regulation of flow, the choice in type of DALY and water requirement are all equally important
Ozone formation[a]	Max. difference is a factor of 15	Max. difference is a factor of 10	Max. difference is a factor of 1.5
	The choice in including effects with low amount of knowledge is 1.8x more important than the choice in type of DALY	The choice in including effects with low amount of knowledge is 2.5x more important than the choice in type of DALY	The choice in type of DALY is the only choice responsible for the difference
Particulate matter	Max. difference is a factor of 2.8	Max. difference is a factor of 1.9	Max. difference is a factor of 1.5
	The choice in type of DALY is the only choice responsible for the difference	The choice in type of DALY is the only choice responsible for the difference	The choice in type of DALY is the only choice responsible for the difference
Human toxicity[a]	Metals: max. 6 orders of magnitude Non metals: max. factor of 10	Metals: max. 4 orders of magnitude Non metals: max. factor of 22	Metals: max. 5 orders of magnitude Non metals: max. factor of 7
	Metals: the choice in time horizon and bioaccumulation determines the difference in perspective Non metals: the choice in including noncarcinogenic effects is 2x more important than the choice in type of DALY	Metals: the choice in time horizon determines the difference in perspective Non metals: the choice in including carcinogenic effects (IARC classification) is 12x more important than the choice in type of DALY	Metals: the choice in time horizon and bioaccumulation mainly determines the difference in perspective Non metals: the choice in including noncarcinogenic effects is 2x more important than the choice in type of DALY
Ionizing radiation	LL: max. 4.4 orders of magnitude SL: max. factor of 11	LL: max. 4 orders of magnitude SL: max. factor of 2.2	LL: max factor of 11 SL: max factor of 5.2
	LL: the choice in time horizon is more than 2000x more important than the choice in type of DALY or the knowledge about effects SL: the choice in type of DALY, the knowledge about effects and time horizon all contribute with the same importance to the difference between scenarios	LL: the choice in time horizon is more than 2000x more important than the choice in type of DALY or the knowledge about effects SL: the choice in type of DALY, the knowledge about effects and time horizon all contribute with the same importance to the difference between scenarios	LL: the choice in time horizon is 2x more important than the choice in type of DALY or the knowledge about effects SL: the choice in type of DALY, the knowledge about effects and time horizon all contribute with the same importance to the difference between scenarios
Ozone depletion	LL: max. 2.5 orders of magnitude SL: max. factor of 30	LL: max. 1.5 orders of magnitude SL: max. factor of 20	LL: max. factor of 9 SL: max. factor of 6
	LL: choice in time horizon can rise up to 60x more important than choice in type of DALY and up to 7x than the inclusion of cataract SL: the choice in including cataract is 8x more important and twice as important than choice in type of DALY and time horizon	LL: the choice in time horizon and including cataract are main important for the difference in scenario, being respectively 15x and 11x more important than the choice in type of DALY SL: the choice in including cataract is 9x and 11x more important than the choice in time horizon and type of DALY	LL: the choice in time horizon is 4 to 5x more important than the type of DALY or including cataract SL: the choice in time horizon is up to 2x more important than the type of DALY or including cataract
Climate change[b]	LL: max. 5 orders of magnitude SL: max. 2.5 orders of magnitude	LL: max. 3 orders of magnitude SL: max. factor of 10	LL: max. 2 orders of magnitude SL: max. 1.5 orders of magnitude
	LL: the choice in time horizon is max. 400x more important than choice in management style and future scenarios, that on its turn is 4x more important than the choice in type of DALY SL: the choice in management style and future scenarios is 4x more important than choice in type of DALY, and 1.3x more important than the choice in time horizon	LL: the choice in time horizon is max. 300x more important than choice in type of DALY or future scenarios and management style SL: the choice in time horizon, management style, future scenarios and type of DALY is equally important	Independent of the lifetime of the substance, the choice in time horizon is equally important than the choice in management style and future scenarios; The choice in DALY is less important

Note: The numerical values represent the maximum ratios of the egalitarian/individualist, egalitarian/hierarchist and hierarchist/individualist scenarios. The type of DALY refers to age weighting and discount rate.
[a]Only the max. differences for substances included in both perspectives are presented. For the egalitarian/individualist perspective vinylchloride shows a maximum difference of a factor of ten. For the egalitarian/hierarchist scenario the same substance shows a max. difference of a factor of 1.5.
[b]Both positive and negative CFs are reported (see appendix 5.2). Only positive CFs are considered in the ratio calculations. The negative and zero values are further discussed in the text.

Figure 5.3 illustrates the share of each impact category to the global damage to human health (in %) caused by water consumption and outdoor emissions in 2000, for the three perspectives. Depending on the perspective, the damage is driven mainly by three impact categories: water scarcity, particulate matter and climate change. All other impact categories contribute less than 2% of the damage. Of the total damage score of the individualist perspective, 67% is attributable to particulate matter formation and 21% to water scarcity. For the egalitarian perspective, 95% of the total damage score is related to climate change (mainly CO_2 emissions). For the hierarchist perspective, the total damage score is distributed almost equally between climate change (~50%) and particulate matter (~40%), together with a ±10% contribution by the impact of water scarcity. Independent of perspective, six substances and water consumption are responsible for more than 95% of the total damage (see table 5.3). Substance contributions per impact category can be found in the appendix 5.1 (table 5.7).

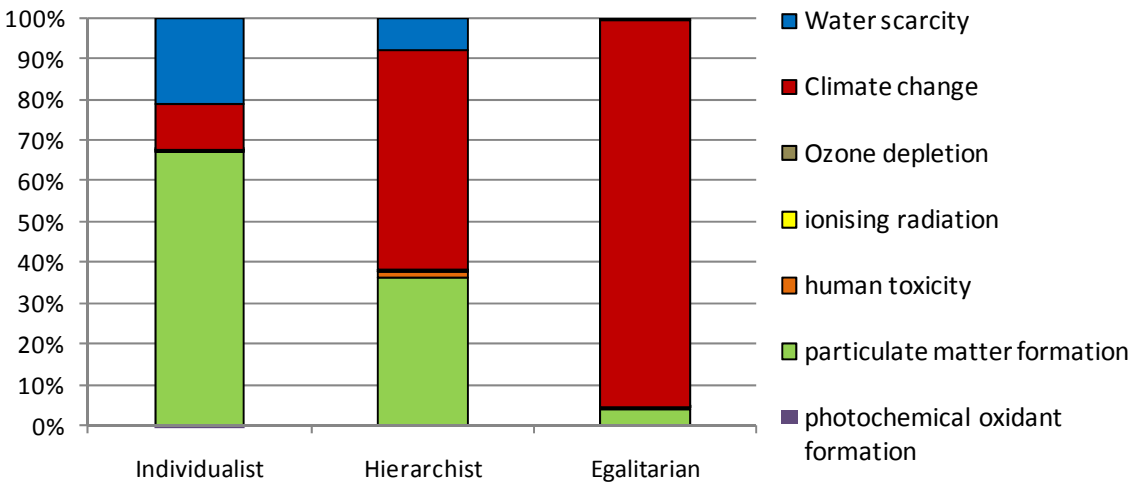

Figure 5.3. Global human health damage (share of each impact category, expressed in %) caused by emissions and water consumption in 2000, following the individualist, hierarchist and egalitarian perspectives.

The cumulated global environmental damage to human health is 4 million DALY/yr for the individualist, 21 million DALY/yr for the hierarchist perspective, and 570 million DALY/yr for the egalitarian perspective (table 5.3). The resulting yearly disability loss is 0.2 to 34 days per capita, and a total loss of 19 days to 7.5 years within a total human lifespan of 80 years, which corresponds to 0.1% to 10% of life lost.

Table 5.3. Substances responsible for 95% of the human health damage in the world, of emissions in 2000. The damage scores (in DALY/(capita.yr)) and percentage damage contribution (in %) are presented.

Substance/source	Compartment of emission	Individualist		Hierarchist		Egalitarian		Driving choices
		DALY/ (capita.yr)	%	DALY/ (capita.yr)	%	DALY/ (capita.yr)	%	
Water consumed		1.4E-4	21.1%	2.8E-4	8.3%	6.7E-4	0.7%	Management style (regulation of flow) + type of DALY
Particulates, < 10 μm	Air	4.4E-4	66.9%	6.5E-4	19.1%	1.2E-3	1.3%	Type of DALY
Carbon dioxide, fossil	Air	3.5E-5	5.3%	1.2E-3	35.2%	8.2E-2	87.8%	Time horizon
Methane[a]	Air	3.0E-5	4.6%	3.6E-4	10.7%	2.5E-3	2.6%	Management style-future scenarios
Sulfur dioxide	Air	2.4E-8	0.0%	5.7E-4	16.8%	1.1E-3	1.2%	Type of effects included
Dinitrogen monoxide	Air	4.4E-6	0.7%	1.6E-4	4.6%	1.8E-3	1.9%	Management style-future scenarios + time horizon
CFC-12	Air	9.9E-7	0.2%	5.7E-5	1.7%	5.6E-4	0.6%	Management style-future scenarios + time horizon
Others		7.8E-6	1.2%	1.2E-4	3.6%	3.6E-3	3.8%	
Total damage (in DALY/(capita.yr))		6.5E-4	100%	3.4E-3	100%	9.3E-2	100%	
Total damage (in disability-adjusted life days/(capita.yr))		0.2		1.2		34		
Total damage for the world (in DALY/yr) using a population of 6.12 billion people for year 2000		4.0E+6		2.1E+7		5.7E+8		

Note: The column "driving choices" presents the value choices responsible for the difference among perspectives. The type of DALY refers to the choice in age weighting and discount rate.
[a]Sum of methane from biogenic and fossil origin. No degradation products are included in the inventory dataset or CFs, therefore there is an underestimation of the damage from methane (more details can be found in the appendix 5.1).

5.4 Discussion

In this paper, existing models to calculate CFs for water consumption and 1239 substances (Pfister et al., 2009, Van Zelm et al., 2008, Huijbregts et al., 2005, Frischknecht et al., 2000, Hayashi et al., 2006, De Schryver et al., 2009) were adapted following the Cultural Theory, using different value choices for individualist, hierarchist and egalitarian perspectives (Schwarz and Thompson, 1990, Thompson et al., 1990, van Asselt and Rotmans, 1996, Hofstetter, 1998, Hertwich et al., 2000, Hofstetter et al., 2000, French and Geldermann, 2005, Steen, 2006). This application allows for a transparent way of handling value choices for environmental assessments and decision making. While we illustrate this work by focusing on damage to human health, the technique can also be applied to assess impacts on other damage categories such as, ecosystem quality and resource depletion.

The calculations show that scenario-specific differences in CFs depend on the persistence of the substance in the environment. For persistent substances, such as long-lived greenhouse gases or metals, the difference in CFs can reach up to six orders of magnitude. The chosen time horizon used for the fate and exposure factor is mainly responsible for this difference (preference value). For short-lived substances, the difference in CFs among perspectives mainly derives from choices regarding the effect and damage factor, namely the exposure or effects included based on the amount of knowledge (contextual value) and the choice in type of DALY (preference value). For water scarcity, the difference

among perspectives is driven mainly by the management style that decides on how the water flow is regulated (contextual value).

Limitations in defined choices

Despite the substantial effort to create a coherent implementation of all value choices identified in the models employed, it was not possible to include all choices in the calculation of the CFs.

First, due to data limitations, perspectives on future scenarios (demographic changes, migration patterns, gross domestic product projections, education) and management style (socioeconomic adaptations) were not included, except for climate change. For climate change, the choices for future scenarios and socioeconomic adaptation made the CFs from the egalitarian and individualist perspectives differentiate by one order of magnitude. An improved health care system, better education or research can lower the damage factors, while demographic changes can influence the number of cases affected. The inclusion of future scenarios and management style can be an important contributor to the difference among perspectives (1.5 order of magnitude) for substances with long response times, such as a number of ozone-depleting chemicals. Further research on including future scenarios for all impact categories is therefore needed.

For ionizing radiation, limited data sources constrained the available exposure factors at the corresponding time horizon (see appendix 5.2). In total, CFs were derived for 52 ionizing substances. For the hierarchist perspective, CFs for three substances are missing (Pu-238 and Ra-226 emitted to air; Ra-229 emitted to freshwater) and for the individualist perspective, CFs for four substances are missing (Pu-238, Pb-210 and Ra-226 emitted to air; Ra-229 emitted to freshwater). This lack of data can lead to an underestimate of the damage score for the individualist and hierarchist perspectives when analyzing emission data that cover these substances (such as nuclear waste and electricity from nuclear and coal power plants). Note that for the global damage calculations, the missing substances were not present in the inventory dataset and thus no underestimate attributable to the lacking CFs arises.

Information on cause-effect relationships was not always available and therefore not all health effects could be included in the different scenarios (see appendix 5.1, table 5.4). Examples are diarrhea incidence from water scarcity (Banda et al., 2007), premature deaths from particulate matter (Reiss et al., 2007), solar keratosis from ozone depletion (Lucas et al., 2008) and dengue and tick-borne encephalitis from climate change (Haines et al., 2006). This omission results in an underestimation of CFs, particularly for the egalitarian perspective, as this perspective considers all possible effects, including those with limited knowledge.

Subjective assumptions

The ultimate goal for developing different scenarios is to provide tools to evaluate possible outcomes. Exploring various trajectories and considering alternative plausible states of the world widens stakeholder's or decision maker's perspective and highlights issues that otherwise would be missed (Mahmoud et al., 2009). In this study, we applied the Cultural Theory for exploring plausible states of the world (Thompson et al., 1990). However, subjective assumptions were inevitable in the construction of the three scenarios:

- To account for different temporal visions of life and society, various time horizons were considered for the fate and exposure factor, while the damage factors included specific discount rates. The egalitarian perspective has no discount rate, which is consistent with an infinite time horizon. A 5% discount rate for the individualist perspective results in a maximum of 20 life years lost at birth, and the same maximum DALYs result when applying a 20-year time horizon. A 3% discount rate for the hierarchist perspective results in a maximum of 30 life years lost at birth, which is lower than the maximum DALY obtained when applying a 100-year time horizon. In general, applying a 5% or 3% discount rate for the individualist or hierarchist perspectives respectively gives a lower damage factor than applying a time horizon of 20 or 100 years for the damage factor. The combined use of time horizon and discount rate is common practice in life cycle assessment (e.g., Hauschild and Potting, 2005, Goedkoop et al., 2008). Consistently applying a time horizon or a discount rate throughout the calculation steps of CFs is recommendable and warrants further research.
- Which exact time horizon to select is difficult to underpin. Other time horizons than those applied in the three scenarios could be selected. For example, for the egalitarian perspective one can argue that an infinite time horizon is unrealistic for some emissions (residence time of > 100,000 years) and a more appropriate time horizon could be selected, such as a 500-year time frame used within the IPCC calculations (IPCC, 2000).
- Positive effects, such as NOx emissions regarding tropospheric ozone degradation, were only included for the individualist perspective following their positive attitude towards environmental benefits (van Asselt and Rotmans, 1996). However, in most cases positive effects are also uncertain. This is contradicting with the individualist perspective which only includes proven effects (Thompson et al., 1990). The high level of uncertainty argues for excluding positive effects for the individualist and hierarchist perspectives and including them for the egalitarian perspective. Here, positive effects are essentially assessed on basis of their positive environmental impacts and not their level of uncertainty.
- Causalities with limited knowledge are manifold, such as uncertainty in morbidity effects from ozone formation, and the effects from secondary aerosols. Effects or substances with limited scientific proof are excluded from the individualist perspective, while included for the egalitarian perspective. However, for the hierarchist perspective, the required level of knowledge is more difficult to define.

For instance, some researchers (Gloria et al., 2006, Ligthart, 2004) argue that the fate and exposure models used to address the human toxicity of metals, such as USES-LCA (Van Zelm et al., 2009), are highly uncertain. For essential metals (i.e., cobalt, copper, manganese, molybdenum and zinc), bioaccumulation is expected to be overestimated. Excluding bioaccumulation in the hierarchist perspective would decrease CFs for essential metals up to four orders of magnitude.

- In this paper only the effects of equal weights and unequal weights as provided by the WHO are assessed by age weighting (WHO, 2008b). However, not all studies agree in assigning different weights to a year of life lost at different ages, nor in the relative magnitude of the weights (Lopez et al., 2006).

The Cultural Theory is recognized as not being able to account for the full variety of world visions and perspectives (van Asselt and Rotmans, 1996). Therefore, the constructed scenarios can be seen as default scenarios and, depending on the questions to be answered, we recommend the development of a flexible system that allow users to adapt and construct their own scenarios.

Global damage of water consumption and outdoor emissions

Depending on the perspective chosen, the global damage caused by water consumption and outdoor emissions in 2000 is mainly caused by the impacts from climate change or particulate matter. The absolute yearly damage score (in DALY/(capita.yr)) for the individualist perspective is two orders of magnitude lower than for the egalitarian perspective and derives mainly from the difference in time horizon chosen for climate change. The chosen time horizon determines how much damage caused by persistent substances, in this case carbon dioxide, is included in the damage score. For climate change, taking only part of the damage into account causes a difference of three orders of magnitude between the egalitarian and individualist perspectives. This makes the time horizon the most important value choice for the difference in global damage among perspectives. The damage from particulate matter dominates the global damage outcome for the individualist perspective. As a matter of comparison, our result for particulate matter represents approximately half of the burden of disease due to outdoor air pollution in 2000 calculated by Cohen et al. (2005).

The global damage calculations have several limitations:

- Not all human health impacts were considered in this assessment, mainly due to lack of data. Impacts, such as from noise and indoor air emissions, should be included in order to improve the calculation of the global damage.
- Except for water scarcity, all impacts are calculated by combining global total emission data with average CFs. Further research is needed to evaluate both the inventory and the CFs of impacts at regional and local levels (e.g., urban versus rural versus remote emissions). Within this study, regionalization is especially required for ozone formation and particulate matter;

- For particulate matter, the CF of PM10 (from Van Zelm et al., 2008) is used to calculate the global damage from global PM10 emissions (from Sleeswijk et al. 2008). Because PM10 is an important contributor to the total damage, better assessment of the size distribution below PM10, i.e. PM2.5 and PM0.1, would be necessary to reduce uncertainty in the results (Dockery et al., 1993);
- We considered total water consumption (industry, households and irrigation) to evaluate the damage from water scarcity. The result is the potential number of DALYs/yr that can be avoided if all water consumed today was saved under current water scarcity conditions. It is however virtually impossible to save all water used for irrigation. This implies that the global damage from water consumption is probably overestimated in our current calculations, particularly because water from irrigation accounts for 85% of the global water consumption (Shiklomanov, 1999);
- To evaluate the damage caused by water consumption in 2000, water consumption per capita for 1995 is multiplied with the population in 2000. Therefore, pressure per capita is assumed not to change between 1995 and 2000, which is a source of uncertainty that needs further refinement.

5.5 Conclusion

Value choices in impact assessment modeling were implemented by applying the Cultural Theory. CFs for 1239 substances and water consumption are provided, covering the human health impact categories of water scarcity, ozone formation, particulate matter, human toxicity, ionizing radiation, ozone depletion and climate change. Depending on the chosen perspective, CFs can range from negative to positive values and differ up to six orders of magnitude. The most important value choice for long-lived substances is the choice in time horizon (fate factor), followed by the effects included and the choice in age weighting and discount rate of the DALY calculation (damage factor). For substances with a relative short life time, the most important choices are the effects included and choice in age weighting and discount rate.

When applying the three sets of CFs to assess the global emissions and water consumption, the damage to human health differs by 2 orders of magnitude among the chosen perspectives and is mainly driven by the time horizon chosen for climate change. The global impact comes mainly from particulate matter when considering an individualist perspective, climate change when considering an egalitarian perspective, and particulate matter and climate change when considering a hierarchist perspective. Water scarcity should also not be neglected, as it contributes considerably to the global impact for the individualist and hierarchist perspectives. All other impact categories contribute less than 2% to the total global damage.

The results of this study clearly indicate that value choices within impact assessment modeling influence the absolute values of CFs and the overall damage calculation. Further research is required to evaluate whether cultural perspectives can also change the ranking among products and services, and conclusions of life cycle assessment studies.

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5.8 Appendix 5.1

Methodology

Value choices

Table 5.4. Suggested combination of value choices on the level of concern (C) and the level of uncertainty (U), applied to the three perspectives for human health. Choices on the level of fate factor, exposure factor, effect factor and damage factor are separated for each impact category (IC).

IC	Step	Value choices	Individualist	Hierarchist	Egalitarian	Note
All impact categories (generic value choices)	Fate + exposure	The change in concentration due to a change in emission is substance lifetime specific and depends on the time horizon after which the change in concentration is measured (Huijbregts et al., 2005). As temporal vision on life and society is perspective dependent, different timeframes are applied for each perspective (Jager et al., 1997, De Schryver et al., 2009).	20years	100years	Infinite	C
		Future projections on demographical developments, population displacements, changes in GDP, years of schooling and technology changes will alter the sensitivity, size and age composition of the population and thus influence the number of cases (incidence) per emitted substance (Mathers and Loncar, 2006). Future optimistic, baseline and pessimistic scenarios (Ipcc, 2000, Murray and Lopez, 1997), can be linked to the individualist, hierarchist and egalitarian perspectives.	Optimistic development	Baseline development	Pessimistic development	U[a]
	Damage	Discounting years of life lost in the future is perspective dependent. Janssen et al. (1995) propose a 0% time discount rate for the egalitarian perspective, a 2% discount for the hierarchist perspective and a 5% discount for the individualist perspective. We follow this vision, except for the hierarchist perspective where a 3% discount rate is chosen, as this is used as default scenario by the World Health Organization (Murray and Lopez, 1996c, Who, 2008b).	5%	3%	0%	C
		Age weighting allocates a higher importance to a year of life at young age than at old age or infants (Murray and Lopez, 1996c). A higher value for economically more relevant subpopulations corresponds with the individualist perspective, while the group bounded hierarchists and egalitarian perspectives do not differentiate between individuals with different ages (Gold et al., 1996, Murray and Lopez, 1996c).	Yes	No	No	C
		For the impact category particulate matter and ozone formation, part of the damage (chronic diseases) takes place in the future. For other impact categories, the lifetime of the substances is important regarding effects that take place in the future. Future effects are affected by the level of manageability/ adaptation. Better health care system, education and legislation can reduce the disability-adjusted life years (DALYs) per case in the future (Hofstetter, 1998). The type of management is perspective dependent (Thompson et al., 1990, Van Asselt et al., 1996).	Adaptive management style	Controlling management style	Comprehensive management style	U[a]
Water scarcity	Fate + exposure	Water availability depends on variability in precipitation. Variability in precipitation gives a certain water stress that depends on the water storage capacities. The correction factor (Corr) for water stress due to variability in precipitation (VF) depends on the level of flow regulation by providing sufficient storage structures(Pfister et al., 2009). According to van Asselt and Rotmans (1996), the individualist perspective coincides with an adaptive management style. Therefore a lower variability factor is suggested. For the hierarchist and egalitarian perspectives the original variability factor, as presented by Pfister et al.(2009), is maintained.	Strong regulated flows: Corr=1 Weak regulated flows: Corr=√VF	Strong regulated flows: Corr=√VF Weak regulated flows: Corr=VF	Strong regulated flows: Corr=√VF Weak regulated flows: Corr=VF	U
	Effect	A food water requirement of 1350m^3 per capita per year is the minimum direct human dietary requirement and is used to derive malnutrition cases per amount of water deprivation (Pfister et al., 2009). Good management can drop the food water requirement to 1000m^3 per person by 2050 (Rockstrom2006). i.e., 74% of the water requirement and consequently of the expected health effect. This results in 1 case per 1823m^3 per year water deprived. A water requirement of 1350m^3 per year is applied for the hierarchist and egalitarian perspectives. A good management level, and thus 1	0.74 case per 1350m^3/yr = 1 case per 1823m^3/yr	1 case per 1350m^3/yr	1 case per 1350m^3/yr	U

		case per 1823m^3 per year is assumed for the individualist perspective.				
	Damage	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
(Tropospheric) ozone formation	Fate + exposure	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
	Effect	The effect factor is calculated for an average 24h concentration and the daily highest 8h concentration. The average 24h concentration gives a negative total damage from NO_x due to more ozone degradation than formation (Van Zelm et al., 2008). Including or excluding the positive effects of ozone degradation (applying the 24h or 8h scenario) is considered to be a value choice on the level of concern. Positive effects (24h scenario) are only included for the individualist perspective as they consider nature as being stable with assured recovery (Hofstetter, 1998). For the egalitarian and hierarchist perspectives we apply the 8 highest hours of concentration to calculate the effects.	24 hours	8 hours	8 hours	C
		The amount of knowledge about ozone-related morbidity is limited (Vonk and Schouten, 2002, Anderson et al., 2004). Therefore, morbidity from asthma, minor restricted activity days, respiratory hospital admissions and symptom days is only included in the egalitarian perspective (Hofstetter, 1998).	Excluded	Excluded	Included	U
	Damage	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
Particulate matter	Fate + exposure	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
	Effect	Evidence for effects from primary PM_{10} is available (Pope et al., 2009) and therefore included for all perspectives. Evidence concerning human health risks at ambient concentrations of secondary PM from SO_2, NO_x and NH_3 is available (Reiss et al., 2007, Usepa, 2009). However, the level of effect is still being debated (Hofstetter, 1998, Torfs et al., 2007) and therefore excluded for the individualist perspective. Reiss et al. (2007) shows that there are more studies indicating health effects from secondary PM from SO_2 than from NO_x or NH_3. Therefore, in the hierarchist perspective, we decided to include effects from secondary PM from SO_2 only.	Primary PM_{10}	Primary PM_{10} + secondary PM from SO_2	Primary PM_{10} + secondary PM from SO_2, NO_x and NH_3	U
		The amount of knowledge about asthma, chronic obstructive pulmonary disease, croup in preschool children and ischaemic heart disease is limited. Therefore, these effects are only included in the egalitarian perspective (Hofstetter, 1998).	*Excluded*	*Excluded*	*Included*	*U*
	Damage	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
Human toxicity	Fate + exposure	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
		The bioconcentration factor for metals is less than proportional with the environmental concentration (Hendriks et al., 2001). Therefore, oral intake of metals via food (bioaccumulation) is excluded for the individualist perspective, but included for the hierarchist and egalitarian perspectives.	Intake through drinking water, air	Intake through all routes	Intake through all routes	U
	Effect	The International Agency for Research on Cancer evaluates the carcinogenic risk of chemical substances to humans and group substances according to the level of prove on human and animal carcinogenity (Iarc, 2004). According to Hofstetter (1998) the egalitarian perspective is risk adverse and includes all substances with insufficient evidence of carcinogenity (IARC categories 1, 2A, 2B and 3), the hierarchist perspective reflects a balance between evidence and probability and includes substances with sufficient evidence (IARC categories 1, 2A and 2B), and the individualist perspective includes substances with strong evidence only (IARC category 1). We follow this vision, except for the egalitarian perspective where all substances with a TD50 are included.	IARC classification: 1	IARC classificatio n: 1, 2A and 2B	All substances	U
		The type and level of response for noncarcinogenic effects is uncertain (Huijbregts et al., 2005) and therefore excluded from the individualistic perspectives.	Excluded	Included	Included	U

	Damage	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
Ionizing radiation	Fate + exposure	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
	Effect	The 'dose and dose-rate effectiveness factor' (DDREF) describes the ratio between the risk increase per Man.Sv observed at high doses and the assed risks at low doses. The DDREF is found to be between 2 and 10. A value of 2 is recognized as being conservative and therefore assumed for the egalitarian perspective (Icrp, 1990). A DDREF factor of 6 is preliminary proposed for the hierarchist perspective and value of 10 for the individualist perspective.	10	6	2	U
		Based on the amount of knowledge not all effects are included in the different perspectives. Bladder, colon, ovary, skin, liver, oesophagus and stomach cancer are possibly or probably connected with ionizing radiation and thus only included in the egalitarian and hierarchist perspectives. For bone surface and remainder cancer no information about the level of proof is available and therefore are only included in the egalitarian perspective. Thyroid, bone marrow, lung and breast cancer are definitely associated to ionizing radiation and thus considered for all perspectives (Frischknecht et al., 2000).	Thyroid, bone marrow, lung and breast cancer	Thyroid, bone marrow, lung, breast, bladder, colon, ovary, skin, liver oesophagus, and stomach cancer	Thyroid, bone marrow, lung, breast, bladder, colon, ovary, skin, liver, oesophagus, stomach, bone surface, and remaining cancer	U
	Damage	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
(Stratospheric) ozone depletion	Fate + exposure	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
	Effect	Not all effects of ozone have the same level of evidence. Effects of skin cancer (malignant malenoma, basel cell carcinoma and squamous cell carcinoma) solar keratoses and photoaging are certain and can be included for all perspectives. The evidence of increased incidence of cataract, pterygium herpes and sunburn due to increased UV-B radiation is weak (Lucas et al., 2008) and therefore included for the egalitarian perspective only.	Malignant malenoma, basel cell carcinoma, squamous cell carcinoma, solar keratoses and photoaging	Malignant malenoma, basel cell carcinoma, squamous cell carcinoma, solar keratoses and photoaging	Malignant malenoma, basel cell carcinoma, squamous cell carcinoma, solar keratoses, photo aging, cataract, herpes, sunburn and pterygium	U[b]
		The inclusion of positive effects, like vitamin D efficiency is considered a value choice on the level of concern (Jager et al., 1997) and only included for the individualist perspective as they consider nature as being stable with assured recovery (Hofstetter, 1998).	*Included*	*Excluded*	*Excluded*	*C*
	Damage	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
Climate change	Fate + exposure	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/
	Effect	The inclusion of positive effects from ozone depletion substances is a value choice on the level of concern (Jager et al., 1997) and only included for the individualist perspective as they consider nature as being stable with assured recovery (Hofstetter, 1998).	Included	Excluded	Excluded	C
	Damage	Generic value choices only (presented for all ICs at start of the table).	/	/	/	/

Note: The value choices printed in grey italic are not implemented in our CFs. C= value choice on the level of concern; U= value choice on the level of uncertainty; m^3/yr = cubic meter per year.
[a]Value choice only considered within the CFs of the impact category climate change.
[b]Only the effects of skin cancer and cataract are included in the CFs.

Time horizon

For the impact categories human toxicity, ionizing radiation, stratospheric ozone depletion and climate change, time horizon specific calculations were required. For human toxicity, USES-LCA readily provides fate and exposure results for a 100 year and infinite time horizon Van Zelm et al. (2009). USES-LCA was adapted to calculate fate and exposure factors for a time horizon of 20 years as well. For ionizing radiation, time horizon-specific exposure factors of most radio-active substances were given by IAEA (1985) and Frischknecht et al. (2000). For a limited number of substances, exposure factors for a 20years time horizon were derived by linear extrapolation between a 10 and 50 years time horizon (emitted to freshwater and marine water: I-129; emitted to air: C-14, I-229, C-137) or by using a 50years time horizon as first approximation (emitted to freshwater: C-137; emitted to air: H-3; emitted marine water: Am-241, C-14, Cs-137, H3, Ru-106) (see appendix 5.2). For stratospheric ozone depletion, CFs were provided for an infinite time horizon only (Hayashi et al., 2006). Fate and exposure factors for the time horizons of 20 years and 100 years were derived by calculating the fraction of exposure via:

$$F_t = 1 - e^{(-t-ts)-k}$$

where F_t is the fraction of exposure for time horizon t, k the degradation rate of the substance in the atmosphere ($year^{-1}$) and ts the time needed for the substance to reach the atmosphere (year). A transport time (ts) of 3 years was assumed (WMO, 1995). For climate change, the fate and exposure factors were readily available for all three time horizons considered (De Schryver et al., 2009).

Disability-adjusted life years

Each perspective has different visions on age weighting and discount rate, both affecting the disability-adjusted life year (DALY) values. For the individualist perspective, 5% discount rate and age weighting was assumed, presented as [0.05,1], for the hierarchist perspective 3% discount rate and no age weighting was assumed, presented as [0.03,0], and for the egalitarian perspective no age weighting or discounting was assumed, presented as [0,0].

The DALY values were calculated by implementing the necessary information into the world health organization burden of disease template (Who, 2008a). For the impact categories human toxicity, climate change and ionizing radiation the age specific duration values, incidence rates, age at onset and number of deaths were taken from the report Human Health Statistics 1990 (Murray and Lopez, 1996b), and the disability weights were derived from the Global Burden of Disease 1990 (Murray and Lopez, 1996a). For climate change (De Schryver et al., 2009) the DALYs [0.03,1] of the optimistic 2030 scenario (Mathers and Loncar, 2006) were converted to DALY with 5% discount rate and age weighting. Therefore, the ratio of the DALY [0.03,1] and DALY [0.05,1] for year 1990 per disease and world region was used as scaling factor (Murray and Lopez, 1996a, Murray and Lopez, 1996b). The CFs for ozone depletion

(Hayashi et al., 2004) and water scarcity (Pfister et al. 2009) were adapted to the corresponding age weighting and discount rate by applying the ratio of the required DALY and the original DALY. For the impact categories particulate matter and photochemical ozone formation the age specific population numbers and number of deaths were taken from the Global Burden of Disease 2004 update (Who, 2008b). Age specific duration times, disability weights and incidence rates were derived from van Zelm et al. (2008).For each impact category the DALYs per incidence case are presented in table 5.5. For climate change, the DALYs per degree Celsius increase are presented in table 5.6.

Table 5.5. The disability-adjusted life year per incidence case calculated for the different impact categories, following three different perspectives.

	Individualist	Hierarchist	Egalitarian
Impact category	[0.05,1]	[0.03,0]	[0,0]
Water scarcity			
Nutritional deficiencies	1.5E+1	2.0E+1	4.1E+1
Ozone formation			
Acute mortality	8.8E-2	1.3E-1	2.5E-1
Asthma attacks	*3.1E-4*	*2.7E-4*	2.7E-4
Minor restricted activity days	*1.0E-4*	*8.5E-5*	8.5E-5
Respiratory hospital admissions	*1.2E-2*	*1.1E-2*	1.1E-2
Symptom days	*1.6E-4*	*1.3E-4*	1.4E-4
ERV for asthma	*9.4E-4*	*8.2E-4*	8.2E-4
Particulate matter			
Chronic mortality	3.5E+0	5.2E+0	1.0E+1
Acute respiratory morbidity	3.0E-2	2.6E-2	2.6E-2
Acute cardiovascular morbidity	3.3E-2	2.8E-2	2.8E-2
Human toxicity			
Cancer average	4.8E+0	7.9E+0	1.1E+1
Noncancer average	*1.4E+0*	1.9E+0	2.7E+0
Ionizing radiation			
Thyroid cancer	4.8E+0	7.9E+	1.1E+1
Bone marrow	5.7E+0	8.4E+0	1.4E+1
Lung cancer	6.8E+0	1.2E+01	1.6E+1
Breast cancer	3.2E+0	5.1E+0	7.6E+0
Bladder cancer	*2.1E+0*	3.8E+0	5.0E+0
Colon cancer	*3.8E+0*	6.5E+0	8.8E+0
Ovary cancer	*5.5E+0*	8.6E+0	1.3E+1
Skin cancer	*2.7E+0*	4.3E+0	6.3E+0
Liver cancer	*9.7E+0*	1.5E+1	2.2E+1
Oesophagus cancer	*7.5E+0*	1.3E+1	1.8E+1
Stomach cancer	*5.8E+0*	9.9E+0	1.4E+1
Bone surface	*4.8E+0*	*7.9E+0*	1.1E+1
Remainder	*4.8E+0*	*7.9E+0*	1.1E+1
Hereditary[a]	1.4E+1	2.1E+1	5.7E+1
Ozone depletion			
Melanoma and other skin cancer	2.7E+0	4.3E+0	6.3E+0
Cataract	*8.2E-1*	*1.0E+0*	1.1E+0

Note: The figures printed in grey italic are not included for the corresponding perspective. [0.05,1]= 5% discount rate and age weighting; [0.03,0]= 3% discount rate and no age weighting; [0,0]= no age weighting or discounting; ERV= emergency room visits.
[a]No future generation discounting is considered.

Table 5.6. The disability-adjusted life year per degree Celsius increase for climate change (based on expected DALYs for the year 2030), following three different perspectives.

	Individualist	Hierarchist	Egalitarian
Impact category	[0.05,1]	[0.03,0]	[0,0]
Climate change			
Cardiovascular	0.0E+0	2.5E+5	7.6E+5
Diarrhoe	4.8E+5	1.2E+6	4.5E+6
Malnutrition	0.0E+0	3.6E+6	2.0E+7
Malaria	4.1E+5	1.2E+6	4.5E+6
Nat disastersa	2.0E+2	-1.3E+5	-5.0E+5

Note: [0.05,1]= 5% discount rate and age weighting; [0.03,0]= 3% discount rate and no age weighting; [0,0]= no age weighting or discounting.
[a] The model assumes that protection evolves over time in proportion to projected increases in GDP, this results in negative burdens for the hierarchist and egalitarian perspective.

Results

CFs of water scarcity

For water use CFs were calculated on a country level. This is defined as regionalized impact assessment. Data on annual freshwater availability and water withdrawals were derived from the Watergap2 global model (Alcamo et al., 2003), while data on flow regulation were derived from Pfister et al. (2009). Geographic information system allows data processing on different spatial resolutions (ESRI, 2004) and was used to calculate the new water scarcity index (WSI) per country (see appendix 5.2). Using the ratios of our calculations, the water requirement, the WSI and the damage factor of Pfister et al. (2009), the CFs from Pfister et al. (2009) were extrapolated for the three perspectives. CFs were calculated for 165 countries and are presented in appendix 5.2.

CFs of all impact categories

The CFs for each substance and each impact category can be found in appendix 5.2.

Global damage

The link between inventory data and impact assessment CFs is not always achieved. All carbon containing substances degrade partly to carbon dioxide. For example, over 90% of atmospheric methane degrades to carbon dioxide while the rest is absorbed by micro-organisms in the soil (Badr et al., 1992). In this analysis, no degradation products are included in the inventory dataset or the calculated CFs. For fossil emissions this results in a slight underestimation of the calculated global damage. For biogenic emissions (emissions from the product originally derived from absorbed carbon dioxide from air; such as biogenic methane released by plant products) both the uptake of carbon dioxide as the degradation of the emitted carbon containing substance is excluded, what compensates each other and results in an relative zero effect. The global inventory dataset presents ‘methane’ emissions as a combination of biogenic and fossil methane. We applied the CF of fossil methane what generates a slight underestimation of the damage.

Table 5.7. Percentage damage contribution (%) of substances that contribute for more than 5% to the global damage of emissions and water consumption of the year 2000.

IC	Country	Water consumption (m^3/capita)		CFs (DALY/m^3)			Global damage result (% contribution)		
				Individualist	Hierarchist	Egalitarian	Individualist	Hierarchist	Egalitarian
Water scarcity	India	5.1E+02		9.5E-7	2.0E-6	4.7E-6	59.0	59.8	59.8
	Pakistan	6.8E+02		9.9E-7	1.9E-6	4.5E-6	11.1	10.6	10.6
	China	2.4E+02		1.4E-7	2.7E-7	6.5E-7	5.2	4.9	4.9
	Afganistan	2.0E+03		7.4E-7	1.4E-6	3.4E-6	3.5	3.3	3.3
	All other countries						21.3	21.4	21.4
	Total damage for this impact category (in DALY/capita)						1.4E-4	2.8E-4	6.7E-4
	Substance	Emiss. Comp.	Emission (kg/capita)	CFs (DALY/kg or DALY/kBq)			Global damage result (% contribution)		
Ozone formation	Non-methane volatile organic compounds (NMVOC), unspecified	Air	2.7E+1	1.4E-8	2.0E-8	2.0E-7	98.1	55.1	55.1
	Nitrogen oxides	Air	1.9E+1	-4.2E-8	2.0E-8	2.0E-7	-210.0[a]	38.3	38.3
	Sulfur dioxide	Air	2.1E+1	1.1E-9	1.7E-9	1.6E-8	6.2	3.5	3.5
	Remaining substances						0.7	0.3	0.3
	Total damage for this impact category (in DALY/capita)						-3.8E-7	1.0E-6	1.0E-5
Particulate matter	Particulates, < 10 µm	Air	4.8E+	9.2E-5	1.4E-4	2.6E-4	100.0	53.2	34.3
	Sulfur dioxide	Air	2.1E+1	-	2.7E-5	5.1E-5	-	46.8	30.1
	Nitrogen oxides	Air	1.9E+1	-	-	5.7E-5	-	-	30.0
	Ammonia	Air	2.5E+	-	-	8.2E-5	-	-	5.7
	Remaining substances						0.0	0.0	0.0
	Total damage for this impact category (in DALY/capita)						4.4E-4	1.2E-3	3.6E-3
Human toxicity	Benzene	Air	2.4E+0	3.5E-7	8.4E-7	1.1E-6	40.9	3.9	0.5
	Formaldehyde	Air	1.2E-2	4.9E-5	6.7E-5	1.3E-4	27.3	1.5	0.3
	Chromium	Air	2.6E-4	1.3E-3	3.3E-3	1.4E-7	16.4	1.6	0.0
	Dioxins	Air	3.0E-9	5.5E+1	8.6E+1	1.3E+2	8.0	0.5	0.1
	Nickel	Air	5.7E-4	2.2E-4	4.0E-4	6.4E-4	6.0	0.4	0.1
	Mercury	Air	4.9E-5	-	4.2E-1	9.5E-1	-	39.2	8.5
	Chlorine	Water	1.1E-2	-	5.2E-5	7.0E-5	-	14.2	1.8
	Lead	Air	1.5E-3	-	4.8E-3	9.6E-3	-	13.4	2.5
	Arsenic	Air	1.8E-4	4.2E-6	1.5E-2	2.3E-1	0.0	5.3	7.3
	Selenium	Air	1.3E-4	-	1.8E-3	2.2E+0	-	0.5	51.5
	Selenium	Water	1.3E-4	-	2.3E-3	2.8E+0	-	0.1	5.1
	Barium	Soil	5.0E-4	-	5.1E-5	2.5E-2	-	1.2	3.6
	Zinc	Soil	3.3E-3	-	2.0E-6	1.3E-4	-	1.6	2.3
	Lead	Soil	1.5E-3	-	9.2E-7	6.8E-4	0.0	13.4	2.3
	Barium	Air	5.0E-4	-	2.1E-4	2.3E-2	-	0.2	2.1
	Manganese	Water	7.7E-4	-	2.1E-4	5.3E-3	-	0.3	1.9
	Manganese	Air	7.7E-4	-	9.8E-4	1.1E-2	-	0.3	1.5
	Barium	Water	5.0E-4	-	1.3E-4	2.7E-2	-	0.3	1.4
	Arsenic	Soil	1.8E-4	3.0E-8	8.3E-5	1.4E-1	0.0	0.2	1.2
	Cadmium	Air	6.8E-5	2.0E-4	1.9E-2	9.1E-2	0.7	0.2	1.1
	Vanadium	Air	6.0E-4	-	1.1E-3	8.4E-3	-	0.2	0.9
	Remaining substances						0.6	1.7	4.1
	Total damage for this impact category (in DALY/capita)						2.1E-6	1.2E-2	4.9E-5
Ionizing radiation	Cesium-137	Water	3.9E+1	2.4E-8	2.4E-8	2.4E-8	78.9	71.8	19.1
	Carbon-14	Air	9.5E+1	1.6E-9	1.6E-9	1.6E-9	12.9	19.9	59.4
	Cobalt-60	Water	6.2E+	7.0E-9	7.0E-9	7.0E-9	3.6	3.0	0.8
	Cesium-134	Water	1.2E+	2.3E-8	2.3E-8	2.3E-8	2.2	1.9	0.5
	Technetium-99	Water	2.2E+2	2.0E-11	2.0E-11	2.0E-11	0.4	1.3	0.3
	Iodine-129	Water	2.4E+	6.7E-10	6.7E-10	6.7E-10	0.1	0.1	18.1
	Remaining substances						1.9	2.1	1.8
	Total damage for this impact category (in DALY/capita)						1.2E-6	3.9E-6	2.8E-5
Ozone depletion	CFC-12	Air	1.7E-2	4.1E-5	2.6E-4	1.4E-3	25.4	41.5	50.5
	CFC-11	Air	6.6E-3	7.9E-5	3.5E-4	1.3E-3	19.6	22.6	19.4

	HCFC-141b	Air	2.7E-2	1.9E-5	3.6E-5	1.2E-4	19.0	9.3	7.1
	HCFC-22	Air	4.9E-2	7.7E-6	1.6E-5	5.4E-5	14.0	7.6	5.8
	Halon 1211	Air	7.9E-4	4.1E-4	1.0E-3	3.4E-3	12.2	7.7	5.8
	Halon 1301	Air	1.5E-4	8.5E-4	4.6E-3	2.0E-2	4.8	6.7	6.5
	Remaining substances						5.0		5.0
	Total damage for this impact category (in DALY/capita)						2.7E-6	1.0E-5	4.6E-5
Climate change	Carbon dioxide, fossil	Air	4.7E+3	7.4E-9	2.6E-7	1.8E-5	47.7	65.3	92.7
	Methane[b]	Air	4.9E+1	6.2E-7	7.4E-6	5.0E-5	41.7	19.9	2.8
	Dinitrogen monoxide	Air	1.9E+	2.4E-6	8.3E-5	9.6E-4	6.1	8.6	2.0
	HCFC-22	Air	4.9E-2	3.4E-5	5.0E-4	3.4E-3	2.3	1.4	0.2
	Remaining substances						2.2	4.8	2.3
	Total damage for this impact category (in DALY/capita						7.3E-5	1.8E-3	8.8E-2

Note: For water scarcity, the four most contributing countries are presented. Global emissions (in kg or kBq) and water consumption data per capita (in m^3) is presented, together with the corresponding CFs (in DALY/m^3, DALY/kg or DALY/kBq). Emiss. Comp= the compartment of emissions. m^3= cubic meter; kg= kilogram; kBq= kilobecquerel.

[a]For the individualist perspective positive effects from nitrogen oxides are included and therefore the CF turns negative. This results in a negative damage for the emission of nitrogen oxides and a total negative damage for ozone formation (-100%).

[b]Sum of methane from biogenic and fossil origin. We applied the CF of fossil methane, generating therefore a slight underestimation of the damage.

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5.9 Appendix 5.2

CFs for water consumption and 1239 substances, covering seven human health impact categories, are presented as an excel document and referred to as appendix 5.2. For each impact category the new recalculated CFs are presented together with the original CFs (defined as "original CFs"). For human toxicity the original CFs are not presented, as the figures directly derive from the model USES-LCA.

Chapter 6
Practical consequences of value choices in life cycle impact assessment

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Abstract

Purpose. This study analyzes the influence of value choices in impact assessment models for human health, such as the choice of time horizon, on life cycle assessment outcomes.

Methods. For 756 products the human health damage score is calculated using three sets of characterization factors (CFs). The CFs represent seven human health impact assessment categories: water scarcity, tropospheric ozone formation, particulate matter formation, human toxicity, ionizing radiation, stratospheric ozone depletion and climate change. Each set of CFs embeds a combination of value choices following the Cultural Theory, and reflects the individualist, hierarchist or egalitarian perspectives.

Results. We found that the average difference in human health damage score goes from one order of magnitude between the individualist and hierarchist perspectives, to 2.5 orders of magnitude between the individualist and egalitarian perspectives. The difference in damage score of individual materials among perspectives depends on the combination of emissions driving the impact of both perspectives and can rise up to five orders of magnitude.

Conclusions. The value choices mainly responsible for the differences in results among perspectives are the choice of time horizon and inclusion of highly uncertain effects. A product comparison can be affected when the human health damage score of two products differ less than a factor of 5, or the comparing products largely differ in their emitted substances. Overall, our study implies that value choices in impact assessment modeling can modify the outcomes of an LCA and thus the practical implication of decisions based on the results of an LCA.

Keywords Uncertainties • Value choices • Life cycle assessment • Human health • Decision making

6.1 Introduction

Value choices within life cycle assessment (LCA), such as the choice of time horizon, are unavoidable. Transparency in value choices and caution about the choices included in the outcome of an environmental assessment is important for decision making, such as policy making and legislation actions (EC, 2001, EC, 2005). A consistent pattern of value choices throughout the whole decision analysis is required to analyze environmental problems in an accurate way. Moreover, a broader modeling framework that allows both scientifically valid impact assessment modeling and the representation of the decision maker or the human actor's vision would provide an extended decision support basis (French and Geldermann, 2005).

Several studies provide guidelines on how to deal with value choices within data collection, such as where to set the system boundaries and how to allocate the inventory data of co-products (Schmidt, 2008, Luo et al., 2009, Ayer et al., 2007, Werner, 2005, European Aluminium Association, 2002). Janssen and Rotmans (1995) presented the consequences of adopting different perspectives on allocation, discount rate and time horizon for policy making concerning the allocation of carbon dioxide emission rights. Other researchers approached the influence of perspectives on weighting different environmental impacts (Finnveden, 1997, Schmidt and Sullivan, 2002, Eldh and Johansson, 2006). Unfortunately, the practical consequences of value choices made within impact assessment modeling are rarely analyzed and understood. A few studies showed that decision variables, such as the choice of time horizon and the choice of discount rate or disability weights within disability-adjusted life years (DALY) calculations, can have a large influence on the outcome of a study (Arnesen and Kapiriri, 2004, Frischknecht et al., 2000, Watkiss and Downing, 2008, De Schryver et al., 2010). However, most studies focus on one specific part of impact assessment models, a few specific environmental impacts or the effect of on characterization factors. More research is required, to better understand the influence of different perspectives in impact assessment models on life cycle assessment outcomes, when applying an impact assessment methodology that combines several environmental impacts.

The aim of this paper is to analyze the influence of value choices within human impact assessment modeling on the LCA outcome of a range of products, covering different product groups. We calculate the human health damage score, expressed in DALY, following three perspectives. Each perspective reflects a type of person, hypothetical stakeholder or decision maker with differences in moral beliefs, concerns and interests that explains one's view on society and nature, and that corresponds to a specific set of values (Schwarz and Thompson, 1990, Hofstetter et al., 2000). The average relative contribution of each impact category to the human health damage score is calculated per product group and the differences in human health damage scores are presented. Finally, within the discussion the main choices responsible for different outcomes among perspectives are highlighted and explained.

6.2 Methodology

Human health impact

The human health damage scores (expressed in DALY) were calculated by applying the characterization factors (CFs) from De Schryver et al. (2010). They used the Cultural Theory to define coherent sets of value choices in the calculation of CFs, reflecting the individualist, hierarchist and egalitarian perspectives (Thompson et al., 1990, Hofstetter, 1998). By implementing these value choices in existing impact assessment models, they recalculated CFs for interventions related to the impact categories water scarcity, tropospheric ozone formation, particulate matter formation, human toxicity, ionizing radiation, stratospheric ozone depletion and climate change. Following the Cultural Theory, (i) the individualist perspective coincides with the view that mankind has a high adaptive capacity through technological and economic development, that known damages are the most reliable basis for decisions and that present effects are more important than future gains or losses; (ii) the hierarchist perspective coincides with the view that impacts can be avoided with proper management, that the choice of what to include in the model is based on the level of (scientific) consensus and that time perspective is balanced; and (iii) the egalitarian perspective coincides with the view that nature is fragile and unstable, that a worst case scenario is needed (the precautionary principle) and that a long time perspective is justified. Table 6.1 presents a summary of the different choices taken by each perspective. Further information regarding the methodological choices reflecting these perspectives can be found in De Schryver et al. (2010).

Table 6.1. Overview of value choices implemented in the CFs developed by De Schryver et al. (2010).

Value choices	Individualist	Hierarchist	Egalitarian
Time horizon	20 years	100 years	Infinite
Discount rate	5%	3%	0%
Age weighting	Yes	No	No
Including positive effects[a]	Yes	No	No
Level of knowledge	Only considers certain (proven) effects	Considers likely effects	Considers all known effects
Biological/sociological adaptation	Full	Mean	No
Future projections[c]	Future optimistic development scenarios	Baseline development scenarios	Pessimistic development scenarios

[a]Examples are cooling effects from chlorofluorocarbons and halons that counter climate change, and nitrogen oxides that degrade tropospheric ozone, countering tropospheric ozone formation

[b]The level of biological and socioeconomic adaptation possibilities (also defined as management style; Ezzati et al., 2004), such as improved health care which can reduce the DALYs attributable to a certain impact (Lorenzoni et al., 2005), or the level of legislation, education and research which can increase protection and prevention

[c]Demographic developments, population displacements, changes in gross domestic product, years of schooling and technology changes alter the sensitivity, size and age composition of the population

Life cycle inventory dataset

For all impact categories, except water scarcity, the inventory data were directly taken from the ecoinvent 2.01 database (Ecoinvent Centre, 2007). This database contains consistent and well documented life cycle inventory data for over 4000 life cycle inventory datasets, covering activities and products which are mostly interlinked with each other. To reduce data interdependency, the products selected for the analysis

are those at the start of the production chain, such as 'at farm' for agricultural products or 'at plant' for building materials, and those with a wide geographical preference, such as global or European location. In total, 756 products, from cradle to gate, were included in our analysis and sorted in seven product groups: agricultural products, building materials, chemicals, electronics, metals, paper and board, and plastics (table 6.2). The full list of products included in this analysis is given in appendix 6.2.

Ecoinvent includes inventory data for water withdrawal but not for water consumption. The human health damage calculation for water scarcity is based on the amount of water consumed, i.e., the amount of water withdrawal that is evaporated, integrated into the product or displaced to another watershed or the sea, and therefore does not go back to the same watershed. Within the inventory data of the 756 products, default consumption fractions were calculated, reflecting the amount of water consumed from the total amount of water withdrawn. For irrigation of agricultural products, we calculated that 70% of the water withdrawal is consumed (Shiklomanov, 1999, Unesco, 2001, ABS, 2004), for industrial cooling we selected a 'once-through' cooling system with 1% water consumption (Water & Sustainability, 2002, Yang and Dziegielewski, 2007) and for industrial processing we assumed 10% of water withdrawal to be consumed (Environment Canada, 2004, Solley et al., 1998, Unesco, 2001). For electricity production, water consumption values (l/kWh) are used for cooling water what results in $9.0 \cdot 10^{-4}$ to $2.7 \cdot 10^{-3}$ m^3 of water evaporation per kWh (Water & Sustainability, 2002, Ecoinvent Centre, 2006), depending on the type of power plant. For turbine water use the water consumption is $3.5 \cdot 10^{-3}$ m^3/kWh for alpine dams and 10 times higher for non alpine dams because of the lower water drop (Stewart and Howell, 2003, Bauer et al., 2007). Details on the calculations are given in appendix 6.1.

Table 6.2. Product groups and number of products included (prod. incl.) in the analysis

Product group	**Number of prod. incl.**	**Type of products included**
Agricultural products	72	Plant products and byproducts, animal feed, organic fertilizers
Building materials	46	Bricks, insulation, concrete, construction glass
Chemicals	445	Acids, inorganic fertilizers, pesticides, washing agents, silicones, inks, paints, elements in gaseous or liquid state
Electronics	49	Cables, inductors, plugs, printing wiring boards, batteries, screens, printers, computers, toners
Metals	90	Alloys, ferro- and non-ferro metals
Paper and board	30	Pulp, packaging paper, corrugated board, graphic paper
Plastics	24	Biopolymers, rubbers, thermoplasts and thermosets

Alignment data inventory and CFs

To ensure an appropriate link between data inventory and CFs, the following calculation rules were adopted:

- The CFs for biogenic carbon dioxide uptake and emissions were put on zero, considering an equal uptake and release balance (Cherubini et al., 2009, Gnansounou et al., 2009).
- For water scarcity, De Schryver et al. (2010) present region specific CFs. By using the country specific water consumption as a weighting factor, a European average CF is calculated and applied to all water consumption values in the inventory dataset, except electricity production and

agricultural products. For the latter two, region specific CFs were used. More information on the CFs for water scarcity is given in appendix 6.1 (tables 6.3-6.5).

- For heavy metal emissions to agricultural soil, the inventory data includes the removals through uptake by harvested products, leaching and erosion (Nemecek et al., 2007). As no appropriate impact assessment method exists which characterizes human toxicity impacts for metal uptake from agricultural grounds, expressed in DALYs, the net heavy metal emissions to agricultural soil are excluded from the analysis.
- For each group of substances (defined as sum parameters; e.g., aldehydes unspecified and hydrocarbons chlorinated), a weighted average CF needed to be calculated. This was done by using as weighting factor the global emission level of the year 2000 of the individual substances covered by the group (see appendix 6.1, table 6.6 and 6.7).

Data analysis

The calculated damage scores for each perspective were analyzed in two ways. First, the substance contribution and average relative contribution of each impact category to the human health damage score was calculated and presented per perspective and product group. Second, the Bland-Altman statistical approach was used to define systematic differences between the scenarios (Bland and Altman, 1986, Bland and Altman, 1999). In a Bland-Altman plot, for each product the difference between the damage scores of two scenarios is plotted against the average damage score. This type of statistical approach is commonly used in clinical studies (e.g., Euser et al., 2008, Renehan et al., 2003) and provides direct information about the absolute difference between calculation methods. It also allows investigating whether the difference between scenarios is randomly distributed within the dataset. Because the data extent over several orders of magnitude, both differences and average damage scores are calculated after log transformation of the data (Dewitte et al., 2002).

6.3 Results

Relative contribution

Out of an emission list of more than 600 substances, less than 50 substances contribute for more than 5% to the total human health damage score for each of the products. The list of substances contributing with more than 5% is presented in appendix 6.1, table 6.8. Figure 6.1 shows the average share of each impact category in the human health damage score per product group. Appendix 6.2 presents the CFs calculated per product, for each impact category and perspective. Depending on the perspective, the damage is mainly driven by the impact categories particulate matter formation and/or climate change. For a number of product groups following the hierarchist and egalitarian perspectives, human toxicity also plays a role. Specifically for agricultural products, the share of water scarcity in the human health damage score is

relevant within the hierarchist and individualist perspectives (5-7% on average), particularly for irrigated products in water stressed countries (e.g., water scarcity accounts for 97% of the damage score of jute production in China). All other impact categories contribute, on average, to less than 2% of the human health damage scores.

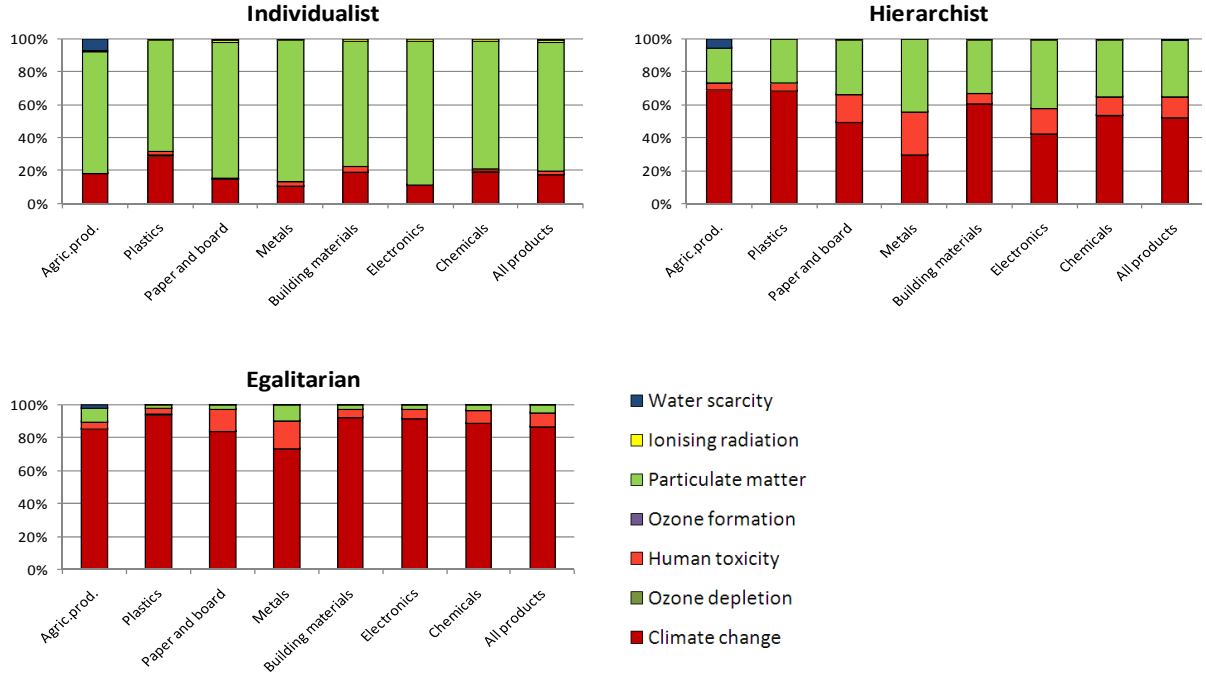


Figure 6.1. Relative contribution of each impact category to the total human health damage score (in %), per perspective for all products and per product group. Agric.prod.= agricultural products.

The impact scores from climate change show the largest difference between perspectives, namely up to 3 orders of magnitude between the egalitarian and individualist perspectives. Carbon dioxide is for more than 90% of the products the dominating greenhouse gas for all three perspectives (impact>70%), except for agricultural products. For agricultural products, climate change is mainly driven by dinitrogen monoxide when following an individualist or hierarchist perspectives, and carbon dioxide when using an egalitarian perspective.

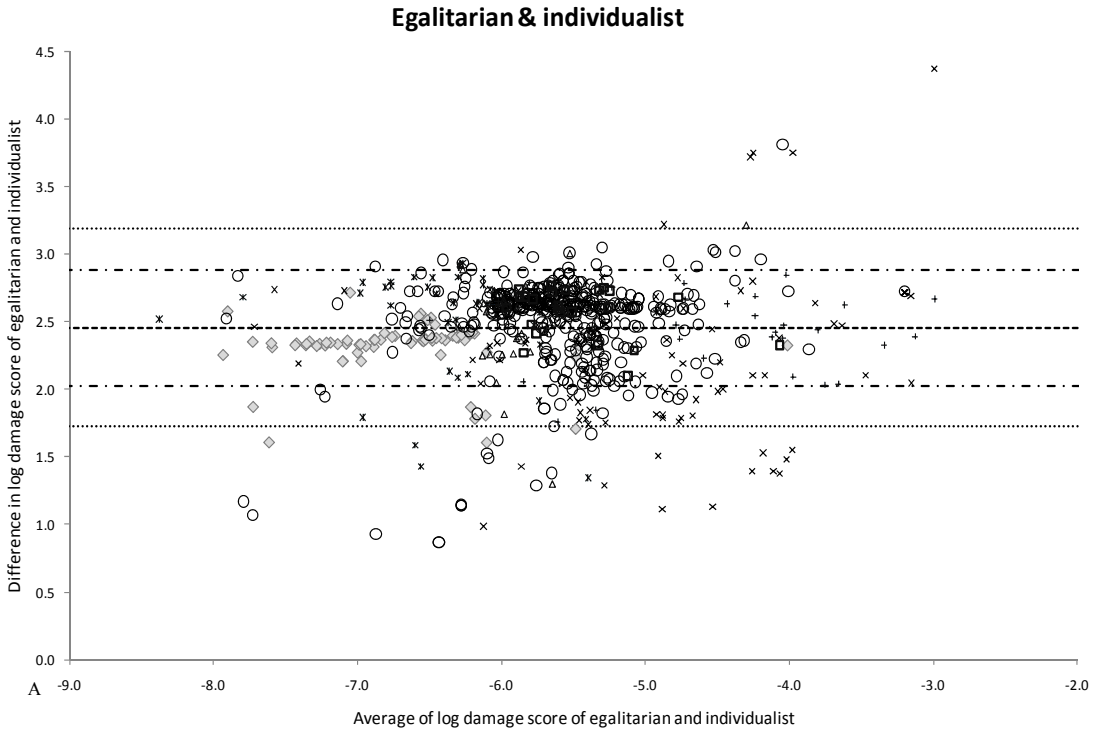
The impact scores for particulate matter formation show a difference of maximum 1.8 orders of magnitude among perspectives. For the individualist perspective, the impact mainly originates from primary fine particulate matter (particulates smaller than 2.5 µm, $PM_{2.5}$ and particulates between 2.5 µm and < 10 µm, $PM_{10-2.5}$). For the hierarchist perspective, the impact mainly derives from primary fine particulate matter and sulfur dioxide which are considered to be relevant for this perspective. For the egalitarian perspective, all types of particulates are considered to be relevant and the main impact derives from sulfur dioxide and nitrogen oxide emissions.

For human toxicity, the human health damage score is mainly driven by metal emissions, independent of the perspective chosen. For the product groups ‘chemicals’, ‘electronics’, ‘metals’ and ‘paper and board’

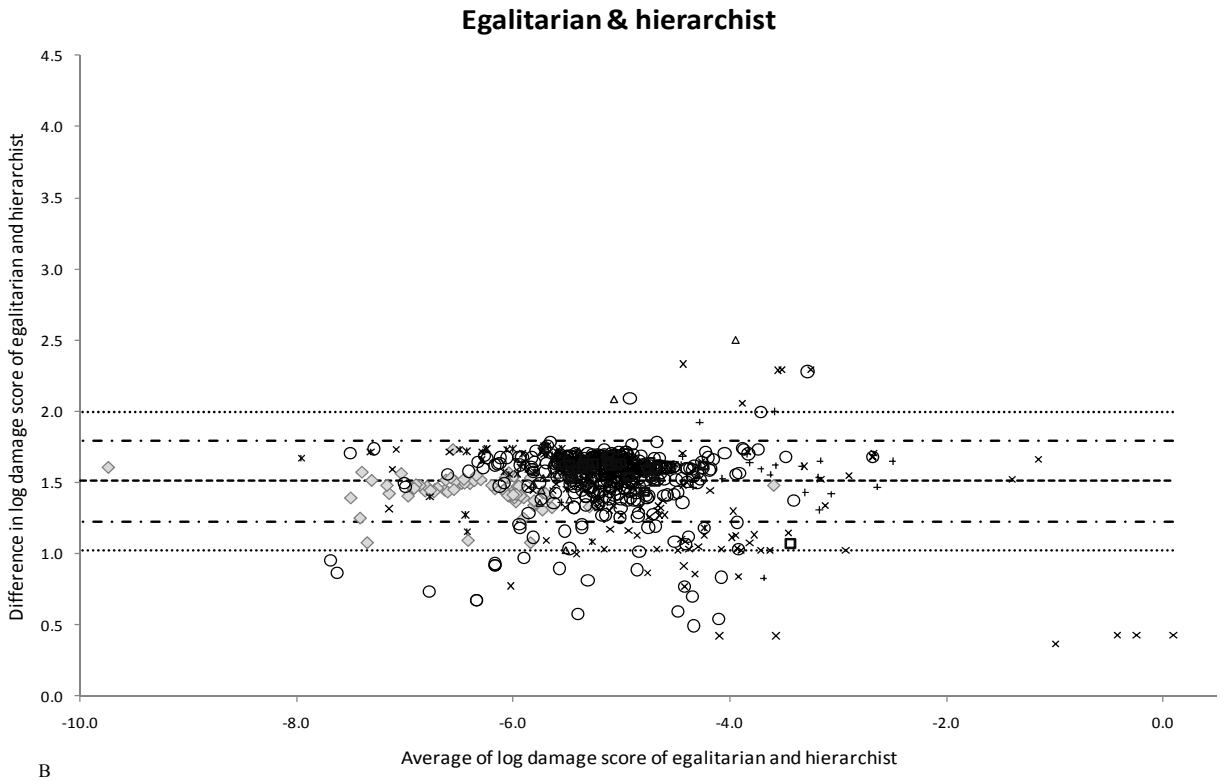
human toxicity shows the highest contribution, on average, with 12% to 26% to the human health damage score for the hierarchist perspective.

Bland-Altman statistics

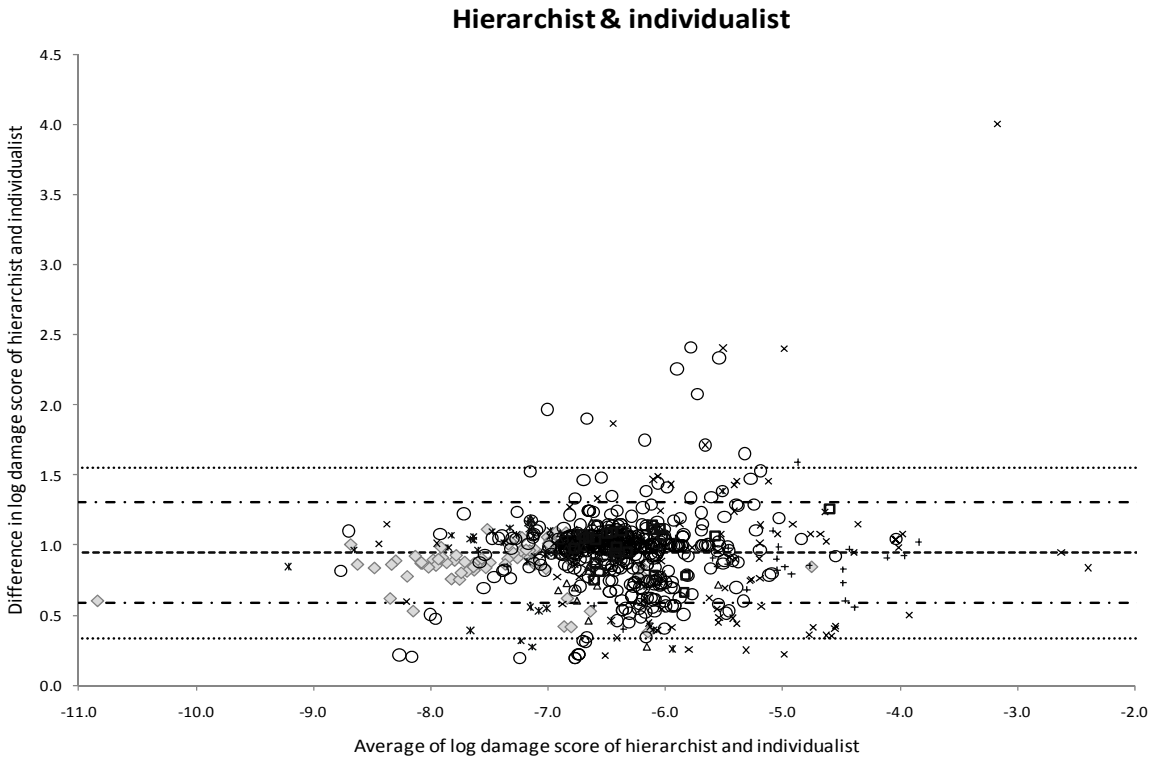
Figure 6.2 presents three Bland-Altman plots showing the difference in log transformed human health damage scores among the three perspectives, together with the 95% and 75% confidence interval. Note that anti-logging the difference in human health damage scores results in the ratio among perspectives. For all plots the difference among perspectives is randomly distributed within the dataset. The egalitarian and individualist perspectives represent the two most distinct perspectives, with an average difference in damage scores of a ratio of 280. The difference can vary up to a factor of 7 using a confidence interval of 75% or a factor of 30 using a 95% confidence interval. The egalitarian and hierarchist perspectives show an average difference of a ratio of 30, which can change up to a factor of 4 using a confidence interval of 75% or a factor of 10 using a confidence interval 95%. The hierarchist and individualist perspectives show the lowest average difference in log damage scores, i.e., a ratio of 10 between the damage scores of the perspectives. In this case, the difference can vary up to a factor of 5 using a confidence interval of 75% or a factor of 16 using a 95% confidence interval. Products showing a difference between perspectives outside the 75% and 95% confidence intervals are mainly chemicals and metals. Average differences and confidence intervals per product group are given in appendix 6.1 (table 6.9).



A



B



C

Figure 6.2. Bland-Altman plots showing the difference in damage scores (y-axes) plotted against their average (x-axes) after log transformation of the data; for the egalitarian and individualist perspectives (A), for the egalitarian and hierarchist perspectives (B) and for the hierarchist and individualist perspectives (C). Note that anti-logging the values on the x-axis and y-axis results in geometric means on the x-axis and the ratio of measurements on the y-axis. The dashed line presents the average difference; the dotted lines indicate the 2.5 and 97.5 confidence interval; the dash-dot lines indicate the 12.5 and 87.5 confidence interval. Each marker type represents a product group:

◇ Agricultural products × Building materials ○ Chemicals + Electronics × Metals △ Paper and board □ Plastics

6.4 Discussion

Uncertainties

Despite the efforts to make a complete and overall analysis, uncertainties are present within the life cycle inventory dataset and in the alignment of inventory data and CFs.

Within this study, impacts are calculated by combining total emission data with average (non region specific) CFs, except for water scarcity. For water scarcity, regional specific CFs are applied for agricultural and electricity water consumption. Further regionalization would improve the impact assessment, particularly for water consumption in all life cycle inventory datasets as well as for emissions for particulate matter and ozone formation.

An additional source of uncertainty is the water consumption values applied for industrial cooling and processing. Within industry, processing and cooling water consumption is reported as a single value with variations ranging from 2% for the primary textile sector to 29% for the transport equipment sector (Environment Canada, 2004). Based on this, for industrial processing a default consumption fraction of 10% is assumed. For industrial cooling, no specific data was found and thus a default consumption fraction of 1% is used assuming a 'once-through' cooling system as within electricity production (Water & Sustainability, 2002). Overall, the water scarcity impact results should be interpreted as a first approximation.

For heavy metal emissions to agricultural soils the alignment of inventory data and characterization is incomplete. The characterization of heavy metal impacts on agricultural soil was not considered in the analyses. This results in an underestimation of the human toxicity impact for the product group 'agricultural products', in particularly for the egalitarian and hierarchist perspectives as these perspectives have high CFs for metal emissions.

Within the impact category 'particulate matter formation', the CF of PM_{10} is based on Van Zelm et al. (2008) and applied to both $PM_{2.5}$ and $PM_{10-2.5}$. Particularly for the individualist perspective PM_{10} is an important contributor to the human health damage score, where the damage from $PM_{2.5}$ and $PM_{10-2.5}$ are equally presented. The inclusion of human health effects of different sizes of PM_{10} would be necessary to refine the results (Reiss et al., 2007).

Furthermore, degradation products (e.g., the degradation from methane to carbon dioxide) are not included in the applied CFs not in the inventory dataset. For fossil emissions this results in a slight underestimation in the calculated human health damage scores.

Finally, not all human health impacts were considered in this assessment, due to lack of CFs. Impacts, such as those from noise and indoor air emissions, should be included in order to improve the human health damage calculations.

Interpretation

Bearing in mind the aforesaid limitations in the application of the methodology, for 756 products the calculated human health damage scores (using three sets of CFs) are interpreted and discussed below.

The magnitude of the difference among perspectives is determined by the combination of interventions driving the impact of both perspectives. For most of the product included, the value choice mainly responsible for the differences among perspectives is connected to the characterization of climate change, i.e., the choice of time horizon for carbon dioxide. For products driven by human toxicity or particulate matter formation, however, an important value choice is the accepted level of knowledge (see table 6.1). For particulate matter formation, evidence concerning human health risks at ambient concentrations of secondary PM from sulfur dioxide, nitrogen oxides and ammonia is available, although the level of effect is still being debated and therefore excluded for the individualist perspective. For human toxicity, exposure routes of metals through bioaccumulation are highly uncertain and therefore only included for the hierarchist and egalitarian perspectives, and excluded for the individualist perspective.

Particularly large absolute differences between the perspectives are caused by emissions of long living greenhouse gasses, such as sulfur hexafluoride (life time of $3.2 \cdot 10^3$ years) and tetrafluoromethane (life time of $5.0 \cdot 10^4$ years). These are responsible for large differences in human health damage scores, up to one order of magnitude above the average difference among the egalitarian and the other two perspectives. Again this is mainly due to the choice of time horizon. For example, sulfur hexafluoride or tetrafluoromethane emissions related to magnesium or aluminum production contribute with more than 60% to the human health damage score of the egalitarian perspective and are responsible for a difference in human health damage score among perspectives above the 95% confidence interval. Apart from this, for very specific emissions, such as mercury emissions during the production of liquid mercury, large differences between the individualist and hierarchist or egalitarian perspectives appear due to the choice of including or excluding bioaccumulation. The same holds for products emitting substances with highly uncertain effects, such as secondary effects of SO_2 for particulate matter. In this case the large difference between the individualist and hierarchist perspectives is caused by the choice of including or excluding uncertain effects.

On the contrary, human health damage scores show minimal differences among perspectives when the impact is driven by rather short-lived substances with certain effects such as particulate matter emissions ($PM_{2.5}$ or $PM_{10-2.5}$). For particulate matter, the difference among perspectives derives from the choice of discounting years of life lost in the future (discount rate) and in allocating a higher importance to a year of life at economically more relevant ages (age weighting) (De Schryver et al. 2010). The difference between the egalitarian and other two perspectives is the smallest when the impact of particulate matter contributes with more than 10% to the egalitarian perspective.

The ranking of two products within a product comparison will not be influenced by the choice in perspective, when the damage scores of the compared products differs more than a factor of 30 considering a 95% confidence interval (whatever the perspective chosen). When a confidence interval of 75% is considered, this difference is reduced to a factor of seven. Furthermore, a product ranking is minimal influenced when the products are based on common underlying processes, such as the electricity mix. The chosen perspective can be influential, however, if material inputs with their corresponding emissions largely differ between production processes. An example is the comparison between corrugated kraftliner board and chipboard. Corrugated kraftliner board has relatively high PM_{10} emissions, mainly from direct emissions (65%) and electricity (11%). This results in a higher human health damage score compared to chipboard when an individualist perspective is applied. In contrast, chipboard has relatively high CO_2 emissions, mainly from direct fossil emissions (33%), electricity use (24%) and disposal of plastic (11%). This results in a higher human health damage score compared to corrugated kraftliner when an egalitarian perspective is applied.

6.5 Conclusions

We can conclude that value choices in impact modeling have direct implications for LCA outcomes. The difference in human health damage scores can range up to four orders of magnitude between the individualist and egalitarian perspectives; and the value choices responsible for the large differences in results are the choice of time horizon and including or excluding highly uncertain effects.

The choice in perspective can alter the ranking of a product comparison when (i) the human health damage score of two products differ less than a factor of seven (75% confidence interval) whatever the perspective chosen and (ii) the comparing products are based on largely different underlying processes and corresponding emissions (e.g., long living versus short living substances). The most important contradicting substances are carbon dioxide (or other long living substances) versus particulate matter ($PM_{2.5}$ or $PM_{10-2.5}$). Therefore, when comparing the results from different studies, caution should be given to not only the different system boundaries and applied assumptions, but also to the perspective used within the applied methodology. Overall, our study implies that value choices in impact assessment modeling can modify the outcomes of an LCA and thus the practical implication of decisions based on the results of an LCA.

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6.8 Appendix 6.1

Methodology

Life cycle inventory dataset

The human health damage calculation for water scarcity is based on the amount of water consumed. Therefore, default consumption fractions were needed, reflecting the amount of water consumed from the total amount of water withdrawn. Default consumption fractions were calculated for water used by the underlying electricity production processes and the products included in the analysis. Default consumption fractions used for the products included in the analysis are: 0.7 for irrigation of agricultural products, 0.01 for industrial cooling and 0.1 for industrial processing. These values represent the fraction of water withdrawal that is evaporated or incorporated into the product; also defined as blue water by the water footprint network (Hoekstra et al., 2009). Within electricity production processes, the amount of water consumed for cooling water and turbine water use depends on the type of power plant.

- Irrigation: According to Shiklomanov (1999) agricultural industry uses 66% of global water withdrawal being 3790 km^3, and consumes 85% of global water consumption being 2070 km^3. This results in a 70% of agricultural withdrawals being consumed and used for irrigation (also confirmed by Unesco, 2001, ABS, 2004).

- Industrial cooling: we assume that the amount of water evaporated during industrial cooling processes is the same as 'once through cooling systems', namely 1% of the total water consumed for cooling (Yang and Dziegielewski, 2007, Water & Sustainability, 2002).
- Industrial processing: water used for industrial purposes such as fabrication, processing or washing, results in water evaporation and incorporation into the product. Depending on the industrial sector, 1% to 30% of used water is consumed, see table 6.3 (Environment Canada, 2004, Solley et al., 1998, Unesco, 2001). Note that the water consumption values in table 4 include water used for cooling and escaped steam. Here, an average water consumption of 10% is assumed.
- Electrical cooling: Thermal electricity power plants use water to discharge 'heat' waste, resulting in evaporation of water. The amount of evaporated water is a function of the efficiency of the power plant which depends on the technology (cooling system) and mode of operation (type of fuel), which is on average 35% (e.g., 30-40% efficiency for coal power plants and 30-50% efficiency for natural gas power plants; Dones et al., 2007). Table 3 presents the amount of water evaporated (consumed) for heat discharge producing of 1 kWh for different fuel types and cooling systems (Yang and Dziegielewski, 2007, Freedman and Wolfe, 2007, Torcellini et al., 2003, Water & Sustainability, 2002).
- Turbine water use: Ecoinvent defines two types of hydropower: reservoir and run-of-river power plants. For run-of-river power plants the water continuously runs in a horizontal way through turbines, without storage. Therefore, no extra evaporation from stagnant water arises and thus the water consumption is zero. Reservoir power plants store large amounts of water, thereby increasing the surface area of the reservoir compared to the free flowing stream. This results in additional evaporative water loss from the surface. The amount of water evaporated from stagnant waters depends on parameters such as the wind speed, the surface area and the air temperature (Lakshman, 1972). The annual evaporation from lakes ranges between 0.5 to 2.2 meters per year (Stewart and Howell, 2003). To define the amount of water evaporated per kWh (m^3/kWh), an annual evaporation rate of 1 meter (m/yr) is combined with the area occupied by the reservoir per kWh (m^2.yr/kWh), see table 3. Note that for alpine reservoirs the amount of water used per kWh is a factor of ten smaller than non alpine reservoirs due to their higher water drop. Therefore the size of reservoir and the amount of water consumed per kWh is also smaller.

Table 6.3. Water withdrawal and consumption values (in m^3/yr or %) for different sectors. Note that the presented values of water consumption include water use for both industrial cooling and processing. Data is extracted from Environment Canada (2004).

Sector	**Water consumption (%)**
Food	10.9
Beverages	23.1
Rubber products	7.8
Plastics	9.4
Primary textiles	2.4
Textile products	14.1

Wood	26.9
Paper	8.9
Primary metals	8.4
Fabricated metals	5.6
Transportation equipment	29.0
Non-metalic mineral products	18.7
Petroleum and coal products	6.1
Chemicals	8.1

Table 6.4. Water consumption values (l/kWh) for heat discharge in electricity production.

	Water consumption (l/kWh)[a]		**Technology used by ecoinvent (fraction)[b]**		**Water consumed (l/kWh)**
Fuel type	**Once through system (river cooling)**	**Cooling Tower**	**Once through system (river cooling)**	**Cooling Tower**	**Applied value**
Oil	1.1	1.8	1	/	1.1
Peat			0.25	0.75	1.7
Lignite			0.25	0.75	1.7
Nuclear	1.5	2.7	/	1	2.7
Coal	1.1	0.8	0.25	0.75	0.9

[a]Water and Sustainability (2002)
[b]Ecoinvent Centre (2007)

Table 6.5. Water consumption values (l/kWh) for turbine use by hydropower stations.

	Area occupied (m^2.yr/kWh)[b]	**Water evaporation (m/year)[a]**	**Water consumed (l/kWh)**
Non-alpine reservoir, European average	3.5·10-2	1	35
Alpine reservoir, European average	3.5·10-3	1	3.5
Reservoir power, Switzerland	3.5·10-3	1	3.5
Reservoir power, Brazil	3.5·10-2	1	35
Reservoir power, Finland	3.5·10-2	1	35
Run-of-River power plant	0	0	0

[a]Steward and Howell (2003)
[b]Bauer et al. (2007)

Alignment data inventory and CFs

The CFs for biogenic carbon dioxide uptake and emissions were put on zero, considering a net uptake and release balance (Cherubini et al., 2009, Gnansounou et al., 2009). Biogenic methane is characterized in the same way as methane from fossil sources.

For water scarcity, De Schryver et al. (2010) provides country specific characterization factors (CFs). As most products within our inventory dataset (75%) are based on European conditions, an average European CF is constructed, using country specific water consumption data as a weighting factor. The European geographic area is applied, including 48 countries and excluding Russia. Country-specific water consumption for the year 1995 was calculated by combining water use data from the Watergap2 global model (Alcamo et al., 2003) with regional consumption fractions from UNESCO (Shiklomanov, 1999). This results in an average European CF of $5.3 \cdot 10^{-8}$ DALY/$m^3_{consumed}$ for the individualist perspective, $1.1 \cdot 10^{-7}$ DALY/$m^3_{consumed}$ for the hierarchist perspective and $2.6 \cdot 10^{-7}$ DALY/$m^3_{consumed}$ for the egalitarian perspective. For all agricultural products (77 from 79 products represent a specific country), and energy specific water flows (cooling water and turbine water) country specific CFs were applied.

For each group of substances (defined as sum parameters) a weighted average CF was calculated (table 6.7). As weighting factor for each substance, the European annual emissions of the year 2000 were taken from Sleeswijk et al. (2008), see table 6.6.

Table 6.6. List of substances and corresponding yearly European air and water emissions used to calculate the weighted average CF of several sum parameters.

	Annual European emission		**Note**
Substances	To air	To water	
Aldehydes, unspecified (kg)			
2-Butenal	6.34·10+3	9.49·10+3	
Acetaldehyde	7.48·10+6	1.98·10+6	
Benzaldehyde	1.55·10+5	4.51·10+5	
Formaldehyde	1.97·10+7	2.48·10+6	
PAH, polycyclic aromatic hydrocarbons (kg)			
Anthracene	1.03·10+4	1.03·10+4	
Benzo(a)pyrene	4.05·10+5	4.05·10+5	
Fluoranthene	5.51·10+4	5.51·10+4	
Naphthalene	6.61·10+5	6.61·10+5	
Phenanthrene	1.52·10+5	1.52·10+5	**a**
Pyrene	1.52·10+5	1.52·10+5	
Actinides, radioactive, unspecified (kBq)			
Americium-241	5.10·10+5	3.40·10+8	
Uranium-234	7.05·10+6	3.35·10+8	**a**
Uranium-235	3.06·10+5	1.47·10+7	**a**
Uranium-238	6.60·10+6	4.20·10+8	**a**
Plutionium-241	3.06·10+6	3.63·10+10	**a**
Carboxylic acids, unspecified (kg)			
Formic acid		1.98·10+5	**a**
Acrylic acid		2.68·10+5	
Hydrocarbons, chlorinated (kg)			
Ethane, 1,1,1-trichloro-, HCFC-140	9.77·10+4		
Methane, tetrachloro-, CFC-10	1.14·10+5		
Ethane, 1,1,1,2-tetrachloro-	1.32·10+3		**b**
Ethane, 1,1,2,2-tetrachloro-	1.26·10+3		**b**
Ethane, 1,1,2-trichloro-	1.58·10+5		**b**
Benzene, 1,2,4-trichloro-	4.09·10+4		**b**
Ethane, 1,2-dichloro-	0.00·10+0		
Ethene, 1,2-dichloro-	2.56·10+3		**a**
Propane, 1,2-dichloro-	1.30·10+6		**b**
Benzene, 1,3-dichloro-	1.57·10+3		**a**
Propene, 1,3-dichloro-	4.73·10+3		**b**
Allyl chloride	1.60·10+5		**b**
Benzotrichloride	4.75·10+2		**b**
Benzyl chloride	5.33·10+3		**b**
Benzene, chloro-	4.71·10+5		**b**
Ethane, chloro-	1.45·10+6		**b**
Chloroform	3.32·10+5		**b**
Propene, 1-chloro-1-	2.51·10+5		**b**
Methane, dichloro-, HCC-30	7.15·10+6		
Butadiene, hexachloro-	1.10·10+3		**b**
Benzene, hexachloro-	1.59·10+4		**b**
Cyclopentadiene, hexachloro-	3.37·10+2		**b**
Ethane, hexachloro-	8.54·10+3		**b**
Methane, monochloro-, R-40	3.28·10+6		
Toluene, 2-chloro-	1.30·10+4		**b**
Benzene, 1,2-dichloro-	1.18·10+5		**b**

Benzene, 1,4-dichloro-	5.70·10+6		**a**
Benzene, pentachloro-	4.20·10+1		**b**
Ethane, pentachloro-	3.54·10+2		**b**
Ethene, tetrachloro-	1.38·10+6		**b**
Ethene, dichloro- (trans)	1.13·10+4		**b**
Butene, 1,4-dichloro-2- (trans)	6.57·10+1		**b**
Ethene, trichloro-	4.47·10+6		**b**
Ethene, chloro-	6.66·10+5		**b**
Hydrocarbons, aliphatic, alkanes, cyclic (kg)			
Cyclohexane	2.12·10+6		
Cyclohexanol	3.79·10+4		
Cyclohexylamine	1.56·10+4		
Dicyclopentadiene	7.74·10+4		
Hydrocarbons, aromatic (kg)			
Benzene, 1,2,4-trimethyl-	2.34·10+6	2.16·10+3	**a**
Benzene, 1,3,5-trimethyl-	2.26·10+6	2.17·10+2	**a**
Benzene	4.25·10+9	7.81·10+4	
Benzene, ethyl-	1.36·10+7	1.17·10+4	
Toluene	1.49·10+8	1.95·10+5	
Noble gases, radioactive, unspecified (kBq)			
Krypton-85	8.38·10+14		**a**
Argon-41	6.78·10+13		
Radon-222	2.49·10+10		

a: No CF calculated by De Schryver et al. (2009)
b: No CF calculated for the impact category climate change (De Schryver et al., 2009)

Table 6.7. CFs (in DALY/kg or DALY/kBq) of sum parameters for the impact categories human toxicity, ionizing radiation and climate change.

	Individualist				**Hierarchist**				**Egalitarian**			
Emission compartment	**air**		**water**		**Air**		**water**		**air**		**water**	
	urban	**rural**	**fresh**	**sea**	**urban**	**rural**	**fresh**	**sea**	**urban**	**rural**	**fresh**	**sea**
Human toxicity (DALY/kg)												
Aldehydes	4.9·10-5	9.6·10-6	2.3·10-6	6.1·10-9	4.9·10-5	9.5·10-6	1.6·10-6	5.7·10-9	9.3·10-5	1.8·10-5	3.1·10-6	1.0·10-8
Carboxiylic acid	a/b	a/b	b	b	a	a	4.4·10-9	1.1·10-12	a	a	5.9·10-9	1.4·10-12
Hydrocarbons, aliphatic, alkanes, cyclic	a/b	a/b	b	b	3.7·10-8	3.1·10-9	a	a	5.0·10-8	4.1·10-9	a	a
Hydrocarbons aromatic	3.5·10-7	9.4·10-8	9.4·10-8	1.8·10-8	8.2·10-7	2.2·10-7	7.5·10-8	1.4·10-8	1.1·10-6	2.9·10-7	1.0·10-7	1.9·10-8
Hydrocarbons, chlorinated	3.3·10-6	3.1·10-7	a	a	1.1·10-5	5.7·10-6	a	a	1.5·10-5	7.8·10-6	a	a
PAH, polycyclic aromatic hydrocarbons	b	b	b	b	1.5·10-5	2.9·10-5	6.4·10-7	4.2·10-9	2.2·10-5	4.1·10-5	1.5·10-6	1.2·10-8
Ionizing radiation (DALY/kBq)												
Actinides	1.4·10-7		8.9·10-12	c	3.9·10-7		2.5·10-11	c	7.6·10-7		5.0·10-11	c
Noble gases	1.6·10-14		d		6.1·10-14		d		1.2·10-13		d	
Climate change (DALY/kg)												
Hydrocarbons, chlorinated	-4.3·10-8		d		7.3·10-6		d		5.0·10-5		d	

a:Emission not considered by ecoinvent
b:No CF for this perspective
c:No European emission levels defined by Sleeswijk et al. (2008)
d:Emission does not exist

Results

Relative contribution

Table 6.8 gives an overview of the substances contributing with more than 5% to the human health damage score.

Table 6.8. List of substances contributing with more than 5% to the human health damage score. The figures present per perspective the number of products for which the substance contribute with more than 5% to the damage score. HFC-134a=Ethane, 1,1,1,2-tetrafluoro-; CFC-14=Methane, tetrafluoro-; Dioxins=measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin; HCFC-22=Methane, chlorodifluoro-; HFC-23=Methane, trifluoro-; CFC-10=Methane, tetrachloro-; CFC-113=Ethane, 1,1,2-trichloro-1,2,2-trifluoro-; HCFC-124=Ethane, 2-chloro-1,1,1,2-tetrafluoro-; CFC-12=Methane, dichlorodifluoro-; HFC-152a =Ethane, 1,1-difluoro- .

Substance	**Compartment**	**Individualist**	**Hierarchist**	**Egalitarian**
2-Methyl-2-butene	Water	0	3	0
Acrylic acid	Air	0	4	0
Aldehydes, unspecified	Air	1	0	0
Arsenic	Air	0	22	13
Arsenic	Water	0	5	13
Barium	Water	0	0	21
Cadmium	Air	3	15	7
Carbon dioxide, fossil	Air	600	676	746
Carbon dioxide, land transformation	Air	0	16	17
CFC-10	Air	4	5	0
CFC-113	Air	1	1	1
CFC-12	Air	1	4	2
CFC-14	Air	0	1	148
Chlorine	Air	0	6	0
Chlorine	Water	0	8	5
Chromium	Air	10	1	0
Chromium	Air	10	1	0
Dinitrogen monoxide	Air	70	96	75
Dioxins	Air	8	0	0
Ethane, 1,2-dichloro-Air	Air	0	2	1
Ethene, chloro-	Air	1	0	0
Ethyl cellulose	Air	3	3	3
Formaldehyde	Air	12	2	0
HCFC-124	Air	1	0	0
HCFC-22	Air	1	1	1
HFC-134a	Air	2	1	1
HFC-152a	Air	3	3	1
HFC-23	Air	2	1	2
Lead	Air	0	33	3
Manganese	Water	0	0	11
Mercury	Air	0	133	6
Methane, biogenic	Air	12	11	5
Methane, fossil	Air	161	183	0
Nitrogen oxides	Air	0	0	1
Particulates, < 2.5 um	Air	754	534	8
Particulates, > 2.5 um, and < 10um	Air	753	337	26
Potassium-40	Air	0	0	1
Radon-222	Air	1	0	0
Selenium	Air	0	0	17
Selenium	Water	0	0	10
Sodium chlorate	Air	6	4	0
Sodium formate	Air	3	3	0
Sulfur dioxide	Air	0	530	31
Sulfur hexafluoride	Air	4	3	6
Vanadium	Air	0	1	0
Vanadium, ion	Water	0	0	1
Zinc	Air	0	3	2

Table 6.9 presents the average differences and confidence intervals of the Bland-Altman plots, per product group.

Table 6.9. Average difference and limits of agreement (95% confidence interval; CI) of the Bland-Altman statistics between the egalitarian and individualist perspectives, the egalitarian and hierarchist perspectives, and the hierarchist and individualist perspectives.

Scenarios	egalitarian-individualist			egalitarian-hierarchist			hierarchist-individualist		
Product group	Av. diff.	75% CI	95% CI	Av. diff.	75% CI	95% CI	Av. diff.	75% CI	95% CI
All product groups	285	106-765	53-1535	32	17-62	11-99	9	4-20	2-36
Agricultural products	204	119-349	81-515	27	20-36	16-45	8	5-11	4-15
Building materials	288	99-838	46-1819	40	26-61	19-83	7	4-14	2-24
Chemicals	319	130-783	69-1474	35	20-61	13-92	9	4-19	3-31
Electronics	282	152-523	98-815	35	19-64	12-98	8	4-15	3-22
Metals	200	40-1013	12-3225	21	7-65	3-146	10	2-47	1-146
Paper and board	295	116-751	58-1505	39	25-61	18-85	8	4-13	3-20
Plastics	395	247-633	173-903	39	28-54	21-70	10	7-15	5-19

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6.9 Appendix 6.2

For each product the human health damage factor per impact category and perspective is presented in appendix 6.2 (excel document).

Chapter 7
Synthesis and conclusions

7.1 Introduction

Life cycle assessment (LCA) has emerged as a valuable decision-support tool in assessing the impacts of a product or service, for both policy makers and industry. All over the world, programs and action plans are being developed with the aim to support life cycle thinking in policy making (e.g., UNEP, 2004, EC, 2005, USEPA, 2006). Examples where LCA results have already been used as a guidance for law enforcement are The 30 January 2008 Amendment to Switzerland's Mineral Oil Tax Law (Steenblik et al., 2008) and The independence and security act of 2007 (H.R. 6) of the United States (Koplow and Track, 2007). Nevertheless, several studies have raised the question whether LCA is the best tool to support policy making (e.g., Hertwich, 2008, CIELAP, 2009). First of all, LCA is recognized as being resource and time intensive. Second, the results of an LCA study are known to vary, depending on the geography and technology considered, the type of inventory database, and the applied impact assessment model. Third, uncertainties are often not considered and unknown effects are intentionally or unintentionally left out.

In recent years, international LCA guidelines, standards and databases are being developed which substantially improves the credibility of LCA results. For example, the Join Research Center published guiding documents for inventory data collection and methodology development (JRC, 2010b, 2010d), and a recommendation for available LCA methodologies (JRC, 2010a). Examples of reference life cycle databases are those for Europe (JRC, 2009) and Japan (JAMAI, 2003). While examples of LCA guidelines are those provided by the United Nations Environment Program (UNEP, 2007) and Joint Research Center (JRC, 2010c). In these guidelines, practitioners are warned about uncertainties in LCA results and are recommended to address them. Detailed recommendations on how to address the full range of uncertainties has, however, not been provided by these guidelines. Moreover, the use of standardized approaches such as proposed by JRC (JRC, 2009, 2010a) does not accommodate different world visions. Inevitable assumptions driven by a specific perspective or world vision are incorporated in the underlying data, life cycle model and impact assessment methodology. In this context, life cycle impact assessment (LCIA) methodologies do not always provide a clear and consistent overview about the assumptions and underlying value choices that are part of the assessment (e.g., Jolliet et al., 2003, Hauschild and Potting, 2005). This is particularly the case for impact indicators that quantify the effect at the end of the cause-effect pathway (Bare, 2009, Reap et al., 2008, Hauschild et al., 2009). Several studies show that differences in world view influence the way impact assessment models are structured (e.g., Arnesen and Kapiriri, 2004, Watkiss and Downing, 2008).

In this PhD thesis, uncertainties were analyzed in LCIA models of three ecosystem impact categories with relatively high uncertainty, namely climate change, land use and ecotoxicity. Chapter 2 describes the uncertainties in ecotoxicity and land use impact assessment modeling, while Chapters 3 and 4

analyze the uncertainties arising from value choices within land use and climate change. Additionally, the uncertainties from value choices made within human health LCIA models for water scarcity, tropospheric ozone formation, particulate matter formation, human toxicity, ionizing radiation, stratospheric ozone depletion and climate change were addressed using the Cultural Theory as a framework (Chapter 4, 5 and 6). Three cultural perspectives are used, i.e., the individualist, the hierarchist and the egalitarian perspectives. The individualist coincides with the view that mankind has a high adaptive capacity through technological and economic development and that present effects need emphasis. The hierarchical perspective coincides with the view that impacts can be avoided with proper management, and that the choice on what to include in the model is based on the level of (scientific) consensus. The egalitarian coincides with the view that nature is strictly accountable, that future effects are important, and a worst-case scenario is needed (the precautionary principle).

Here, the overall findings of this PhD thesis are presented. First, an overview the different uncertainties in LCIA of ecosystem health and value choices in LCIA of human health are discussed. Then, an outline of the practical implications of value choices in LCIA modeling is given. Finally, the chapter ends with recommendations on how to handle uncertainties from value choices in LCA.

7.2 Ecosystem health damage

Ecosystem damage was quantified using the damage indicator ‘potential disappeared fraction of species’ (PDF; Muller-Wenk, 1998), which measures the relative change in species richness. Table 7.1 presents an overview of the value choices made within ecotoxicity, land use and climate change LCIA modeling. Note that this list of value choices is not complete. Other value choices could be implemented, such as for land use the choice in reference land use type and the choice in species to calculate the PDF. At the end of the section, the uncertainties from value choices are placed in a broader context by capturing both measurement uncertainties and ignorance.

Table 7.1. Proposal of value choices deriving from preference values (P) and contextual values (C) within the impact categories land use, ecotoxicity and climate change.

Impact category	Value choice	C/P	Individualist	Hierarchist	Egalitarian
All impact categories	Time horizon	P	20 years	100 years	Infinite
	Included species	C/P	All	All	Focal species[d]
Land use Extracted from Chapter 3 (De Schryver et al., 2010a)	Species accumulation factor[a]	C	Constant z value	Variable z value	Variable z value
	Inclusion regional effect I and II[b]	C/P	Not applicable	Regional effect I and II	Regional effect II
Ecotoxicity Extracted from Chapter 2 (De Schryver et al., 2010c)	Bioaccumulation for essential metals	C	No	Yes	Yes
	Toxic effects of essential metals in oceans[c]	C	No	Yes	Yes
	Effect model	C	Linear	Linear	Non-linear
Climate change Extracted from Chapter 4 (De Schryver et al., 2009)	Species dispersal	C	Yes	Yes	No

[a]A constant z-value can be applied as a simplification of the model because it makes it more robust and independent of the life cycle inventory data (Koellner and Scholz, 2008). However, a range of land use type and spatial scale specific z-values are published (e.g., Crawley and Harral, 2001).
[b]Regional effect I: the reduced species richness in the region due to reduced area size; Regional effect II: the increase in species richness due to the enlargement of land use type i when the area occupied with land use type i gets connected with already existing land of the same type. The regional effects are zero when using a constant z value and therefore not applicable for the individualist perspective.
[c]The potential toxic impact in the marine environment may strongly depend on the statement that additional inputs of essential metals to oceans lead to toxic effects (Ligthart, 2004). Therefore this effect is considered highly uncertain. Essential metals are Cobalt, Copper, Manganese, Molybdenum and Zinc.
[d]For climate change, the red list species identified by IUCN were used for the egalitarian perspective (De Schryver et al., 2009).

Depending on the preferred time horizon, short-term or long-term potential effects are emphasized. This is particularly relevant for the impact categories 'ecotoxicity' and 'climate change', as metals and a number of greenhouse gases have a relatively long residence time in the environment (> 100 years). Based on Jager et al. (1997), a time perspective of 20 years, 100 years and infinite is assumed for an individualist, hierarchist and egalitarian perspective respectively. However, which exact time horizon to select is difficult to underpin. A short time horizon for the individualist perspective emphasizes present effects. Though, other time horizons than 20 years could be selected as well, such as 10 or 50 years. The hierarchist perspective is connected to a 100-year time horizon, as used by several international organizations (ISO/TR14047, 2003, Steinfeld et al., 2006). However, solid scientific arguments for a choice of a 100-year time horizon are not available. An infinite time horizon for the egalitarian perspective corresponds with the emphasis for long-term effects, following the precautionary principle. Although, one can argue that an infinite time horizon is unrealistic for some emissions (residence time of > 100,000 years) and a more appropriate time horizon could be selected, such as a 500-year time frame used within the IPCC calculations (IPCC, 2000).

Limited knowledge on causalities results in different levels of risk considered by each perspective. Two types of causalities with limited knowledge can be defined: (i) uncertain if effects occur, i.e. effects of essential metals in oceans, bioaccumulation of essential metals, and regional effects of land use; and (ii) uncertain mechanistic understanding of effects, i.e. the selection of respectively species to be protected, the species accumulation factor, the toxic effect model, and the potential of species dispersal.

Effects with limited scientific proof are excluded from the individualist perspective, while included in the egalitarian perspective. For the hierarchist perspective, a common accepted limit of proof has to be defined. This is, however, not always straightforward. For ecotoxicity for example, the modeling assumptions of the USEtox™ method can be used as a first guidance for the hierarchist perspective. The USEtox™ method results from a consensus building effort amongst modeling experts (Rosenbaum et al., 2008). Based on USEtox™, bioaccumulation of essential metals is suggested for the hierarchist perspective. On the other hand, there is also sufficient scientific evidence that internal concentrations of essential metals appear to be rather constant for a broad concentration range (e.g. Loos et al., 2009), implying no or limited bioaccumulation. In fact, this high uncertainty argues for excluding bioaccumulation for the hierarchist perspective. The same discussion holds for toxic effects of essential metals in oceans, which are also considered very uncertain (Ligthart, 2004).

The lack of mechanistic understanding of effects can have consequences for the perspectives as well. One such value choice relates to the judgment whether we need to aim for protection of a few 'umbrella' species or 'all' species. The focal species approach selects a few species as 'umbrellas', because of their habitat requirements or sensitivity to a particular pressure (Lambeck, 1997). It assumes that if the most sensitive and relevant species are protected, the majority of other species are protected as well. Based on this argument, the use of focal species is applicable for the egalitarian perspective which follows the precautionary principle. A multi-species' method incorporates all species as indicators, useful for reflecting general trends across all species (Curran et al., 2010). The individualist and hierarchist perspectives correspond with a multi-species approach assuming that ecosystems do not largely depend on the presence or absence of specific species. On the other hand, the argument that the individualist gives higher value to more relevant species and the egalitarian perspective emphasizes equality (preference value; based on Hofstetter, 1998) promotes the use of focal species for the individualist and multi-species for the egalitarian perspective. For now, the precautionary principle is arbitrarily set as the most important principle (contextual value, table 1.1). Another approach is to weight the different species included according to the different tropic levels. For example, Garnier-Laplace et al. (2006) constructed weighted species sensitivity distributions by applying weights of 0.64, 0.26, and 0.1 for primary producers, invertebrates and vertebrates, respectively. A second example of lack of mechanical understanding is the choice of exposure/effect model, such as the use of a linear or non-linear effect model in ecotoxicity. Based on USEtox™, a linear effect model is proposed for the hierarchist perspective. Linear models give in most cases conservative results suggesting linearity for the egalitarian perspective, following the precautionary principle, and non linearity for the individualist perspective. On the other hand, relatively simple and data extensive models can be related to the individualist perspective, which argues for linearity. The egalitarian and hierarchist perspectives are assumed to apply more complex models with a structure that approaches reality, but mostly are uncertain and data intensive, such as a non-linear effect model.

The same arguments hold for the choice in variable or constant species accumulation factor. Finally, there is a lack of understanding concerning the level of biological adaptation that can be expected. Biological adaptation refers to the ability of species to response and adapt to a changing ecosystem. An example is the ability of species to disperse with changing biomes caused by climate change. Past research demonstrated that species can adapt to changing conditions (e.g., Pitelka, 1988). The consequences can be both negative, such as plant invasions (e.g., Pauchard et al., 2009), and positive, such as adaptation to soil erosion and aridity (e.g., Jiao et al., 2009). Biological adaptation can appear as a response to all type of stressors. In this thesis, only the effects of biological adaptation to moving biomes from changing climate is implemented in the perspectives.

In the introduction of this PhD thesis three types of uncertainty are presented, namely measurement uncertainties (on parameters), assumptions (assessed mainly by value choices) and ignorance. For land use, the uncertainty in characterization factors (CFs) (around one order of magnitude) derived from a combination value choices and parameter uncertainties (Chapter 3). The uncertainty in the species accumulation factor z was driving both the parameter uncertainty and the uncertainty from value choices. For climate change, uncertainties in parameters (within one order of magnitude) were much lower than the uncertainties from value choices (up to four orders of magnitude). For ecotoxicity, uncertainties from measurements or assumptions were not quantified within this thesis. Chapter 2 indicates that ecotoxicity models involve several sources of uncertainty, in both the fate/exposue factor and effect factor of the cause-effect pathway. The main uncertainty in the fate/exposure factor (up to several orders of magnitude, Chapter 5) is mainly related to the choice in time horizon when metals are present (Ligthart, 2004). For the effect factor, the uncertainty can range over more than ten orders of magnitude and derives main mainly from the availability of toxicity data and the choice in effect model (van Zelm et al., 2009, 2010).

Finally, by use of the Cultural Theory or probabilistic approaches, uncertainties from ignorance are not taken into account. For instance, for all three ecosystem impact categories the applied endpoint indicators cover only part of the cause-effect pathway. Several effects are not covered so far, like loss of unique landscapes, pests or coral bleaching (see Chapter 2 and 3). Furthermore, regionalization would reduce the overall uncertainty in the CFs and make the methods more globally applicable.

7.3 Human health damage

The damages to human health are quantified by changes in both mortality and morbidity, using the damage indicator disability-adjusted life years (DALY; Hofstetter, 1998). Table 7.2 gives an overview of the value choices made within seven human health impact assessment models (corresponding to seven impact categories).

Table 7.2. Combination of value choices deriving from preference values (P) and contextual values (C) for the CFs of human health, expressed for three different cultural perspectives.

Impact category	Value choices	P/C	Individualist	Hierarchist	Egalitarian
All impact categories	Time horizon	P	20 years	100 years	Infinite
	Discount rate	P	5%	3%	0%
	Age weighting	P	Yes	No	No
Water scarcity Pfister et al. (2009)	Regulation of flow (management style)	C	High	Standard	Standard
	Food water requirement (management style)	C	1000m^3/yr.capita (efficient management)	1350m^3/yr.capita (standard management)	1350m^3/yr.capita (standard management)
Ozone formation Van Zelm et al. (2008)	Morbidity effects[a]	C	No	No	Yes
	Positive effects from tropospheric ozone degradation from NO_x	P	Yes	No	No
Particulate matter Van Zelm et al. (2008)	Effects from primary PM_{10} and secondary PM from SO_2, NO_x and NH_3	C	Primary PM_{10}	Primary PM_{10} + Secondary PM from SO_2	Primary PM_{10} + Secondary PM from SO_2, NO_x and NH_3
Human toxicity Huijbregts et al. (2005a)	Bioaccumulation for essential metals	C	No	Yes	Yes
	Included substances on basis of carcinogenity	C	IARC classification: 1	IARC classification: 1, 2A, 2B	All
	Noncarcinogenic effects	C	No	Yes	Yes
Ionizing radiation Frischknecht et al. (2000)	Cancer types[b]	C	Definite cancers	Definite and probable cancers	All cancers
Ozone depletion Hayashi et al. (2006)	Cataract	C	No	No	Yes
Climate change De Schryver et al. (2009)	Positive effects from ozone depletion	P	Yes	No	No
	Management style (Ezzati et al., 2004) and future development	C	Adaptive and optimistic	Controlling and baseline	Preventive and pesimistic

Note: m^3/(yr.capita)= cubic meter per year per capita; yr= year; IARC= International Agency for Research on Cancer; PM= particulate matter
[a]Morbidity effects included are asthma attacks, minor restricted activity days, respiratory hospital admissions, symptom days.
[b]Definite cancers are thyroid, bone marrow, lung, breast cancer; probable cancers are bladder, colon, ovary, liver, oesophagus, skin and stomach cancer; cancers without information are bone surface and all other cancers.

In the modeling of human health damage a number of choices are influenced by preference values. First, to account for different time perspectives, specific time horizons are considered for the fate and exposure factors, while the damage factors include specific discount rates (e.g., Jolliet et al., 2003, Goedkoop et al., 2008). The combined use of time horizon and discount rate is common practice in LCA (Jolliet et al., 2003, Hauschild and Potting, 2005, Goedkoop et al., 2008). It may be more consistent, however, to select one of the two concepts (time horizon or discount rate) for the calculation of fate, exposure, effect and damage factors. Applying a 3% discount rate for the hierarchist perspective (based on WHO, 2008) results in a lower damage factor than when applying a time horizon of 100 years (following e.g., ISO/TR14047, 2003) within the damage calculation. Furthermore, both the choice in discount rate and time horizon is subject to uncertainty, particularly for the choices within the individualist and hierarchist perspectives.

Second, not all studies agree in assigning different weights to a year of life lost at different ages (defined as age weighting by Murray and Lopez, 1996), nor in the relative magnitude of the weights. Lopez et al. (2006) sums several arguments against age weighting, such as, every year of life is of equal value, age weights have not been validated for large populations, and age weights add an extra

level of complexity to burden of disease analysis that obscures the method. Within this thesis only the effects of equal weights

and unequal weights as provided by the WHO are assessed (WHO, 2008). Another value choice in the DALY calculations is the use of disability weights. Disability weighting means that life years are assigned different value according to health state, on a scale from one (death) to zero (perfect health). According to Arnesen and Kapiriri (2004) disability weights reflect the values of wealthy experts and tend to underestimate the diseases typical for poor populations. This value choice is not addressed in this thesis as it is directly incorporated in the DALY values presented by the WHO and therefore difficult to adapt. However, it could be argued to exclude disability weighting and only consider years of life lost for the individualist perspective based on its preference for proven and certain effects.

Positive effects were only included for the individualist perspective following their positive attitude towards environmental benefits (van Asselt and Rotmans, 1996). However, in most cases positive effects are also uncertain. This is contradicting with the individualist perspective which only includes proven effects (Thompson et al., 1990). The high level of uncertainty argues for excluding positive effects for the individualist and hierarchist perspectives and including them for the egalitarian perspective. Here, positive effects are essentially assessed on basis of their positive environmental impacts and not their level of uncertainty.

As discussed in Section 7.2, limited knowledge on causalities is dealt with in a different way by each perspective. In the human health LCIA models, causalities with limited knowledge are manifold, such as uncertainty in morbidity effects from ozone formation, noncarcinogenic effects and the effects from secondary aerosols. Effects or substances with limited scientific proof are excluded for the individualist perspective, while included for the egalitarian perspective. For the hierarchist perspective, the required level of knowledge is difficult to define. For example, the impact of secondary aerosols from NO_x and NH_3 is excluded for the hierarchist perspective, as the amount of evidence is limited (Reiss et al., 2007). Though, NO_x and NH_3 have recognized effects (USEPA, 2009) what can be an argument to include these effects.

Uncertainties regarding future projections derive from assumptions in demographic developments, population displacements, changes in gross domestic product, and technology changes. An assumption concerning the level of socioeconomic adaptation possibilities (defined as management style by Ezzati et al., 2004) is for example the increase use of air-conditioning against climate change. These assumptions will alter the sensitivity, size and age composition of the population and thus influence the number of incidence cases attributable to a given emission. This uncertainty is complex to grasp and due to data limitations difficult to include. Only for climate change, different development scenarios (as defined by Mathers and Loncar, 2006) and adaptation scenarios (Hofstetter, 1998) are included in the perspectives (De Schryver et al., 2009). For all other impact categories with long-term

effects, i.e., human toxicity, ozone depletion and ionizing radiation, uncertainties regarding future developments and adaptation are not addressed in this thesis. More research is needed to include future developments and adaptations in the calculation of CFs and assess the related uncertainties.

7.4 Practical implications

As described in the introduction of this PhD thesis, uncertainties from assumptions (choices) can be influenced by both preference values and/or contextual values (Hertwich et al., 2000). Sections 7.2 and 7.3 indicate that most choices are mainly driven by contextual values. The uncertainties deriving from these choices can be reduced by more research and data inventory. Three value choices were indicated to be mainly driven by preference values, namely time perspective, equality of humans or species (defined as age weighting and included species) and inclusion of positive effects. These value choices are strongly debatable and can change when the perspective of the people involved changes. Therefore the uncertainty introduced by these type of value choices is difficult to reduced.

It is important to note that not all value choices described above result in the same level of uncertainty (Chapter 5). The level of uncertainty is shown to depend mainly on the life time of the substance. For impact categories containing substances with a relatively long environmental residence time (i.e. > 100 years), namely human toxicity, ionizing radiation, ozone depletion and climate change, the uncertainties in CFs can rise up to several orders of magnitude. The value choices mainly responsible for this difference are the choice in time perspective (preference value choice) and the inclusion of bioaccumulation of essentials metals (contextual value choice). On the contrary, for impact categories driven by substances with a relatively short environmental residence time (i.e. < 100 years), namely water depletion, particulate matter and ozone formation, the uncertainties in CFs rise up to one order of magnitude. For these impact categories, the difference among perspectives mainly derives from the choice in discounting and age weighting (preference values), and in including uncertain effects or exposure (based on limited knowledge; contextual values). The inclusion of positive effects makes the CFs change from negative to positive values. This is the case for the impact categories climate change, ozone formation and possibly the regional effects of land use. For these impact categories, including positive effects may result in a net benefit, what rewards the human activity and is debatable.

Note that for long living substances, the two value choices responsible for the highest uncertainties (namely choice in time perspective and the inclusion of bioaccumulation) are also valid on a midpoint level. For substances with a relatively short environmental residence time, the value choices valid on midpoint level are the choices of including or excluding positive or non-proven effects. Although it has commonly been assumed that midpoint indicators are less uncertain and more scientifically robust than endpoint indicators (Bare, 2009, UNEP-SETAC, 2003, Reap et al., 2008, Hauschild et al., 2009),

the results of this thesis indicate that for impact categories with long living substances or uncertain effects, midpoint CFs can have a relatively high uncertainty as well.

Next to their influence on CFs, value choices can also affect the final outcome of an LCA. In Chapter 6 of this thesis, for over 700 products the human health damage score was calculated using three sets of CFs. The results imply that the choice for a specific perspective can substantially modify the absolute outcome of an LCA. The average difference in damage score goes from one order of magnitude to 2.5 orders of magnitude. The magnitude of the difference in damage scores among perspectives is determined by the combination of emissions driving the impact of both perspectives. When long living substances are emitted, the difference in damage score among perspectives can rise up to five orders of magnitude for specific materials. On the contrary, damage scores show small differences among perspectives (one order of magnitude) when the impact of the egalitarian or hierarchist (if the latter is compared to the individualist) is driven by short living substances with relatively certain effects, such as particulate matter emissions ($PM_{2.5}$ or $PM_{10-2.5}$) and water consumption.

It can be concluded that the choice in perspective can alter the ranking of a product comparison when (i) the human health damage score of two products differ less than a factor of seven (75% confidence interval) whatever the perspective chosen, and (ii) the comparing products are based on largely different underlying processes and corresponding emissions (long living versus short living substances). Therefore, when comparing the results from different studies, caution should be given to not only the different system boundaries and applied assumptions, but also to the perspective used within the applied methodology (see Chapter 6). Overall, the results of this study imply that value choices within impact assessment modeling can modify the outcome of an LCA and thus the practical implication of decisions based on the results of an LCA.

7.5 Recommendations

Within current LCA studies, uncertainties (especially from value choices) are often analyzed in a simplified way if not entirely left out of the analysis (Strang, 2009, Reap et al., 2008). Based on the findings of this thesis, recommendations on how to handle uncertainties in LCA are presented.

First, for most value choices, the correspondence to a certain perspective is not always straightforward. For example, defining the accepted level of risk for the hierarchist perspective is a difficult task. Within this PhD thesis, the different scenarios were defined based on literature review and personal communication with model developers. An overall consensus about the accepted level of risk can be derived, for example, by the use of expert workshops or surveys. This was, however, outside the scope of this thesis. The expertise of social sciences could also help to define scenarios. In

this respect, there is a need to integrate both social and natural sciences into environmental management (Strang, 2009). Strang (2009) discusses the obstacles to interdisciplinary collaboration and proposes guidelines for collaboration between social and natural science. Among all guidelines, the most important was to specifically allocate people to assist the collaborative process and the communication of the findings (Strang, 2009). The work of Brewer et al. (2005) can be used to find appropriate tools to, for example, clarify decision participants' values and preferences concerning alternatives and to understand the disagreements about the implications of choice options. Furthermore, they provide an overview of characteristics for a good public decision process, such as (i) appropriately representing the knowledge and perspectives of the spectrum of interested and affected parties to the decision, (ii) explicitly addressing scientific disagreements and scientific ignorance and (iii) allowing for reconsideration of choices in response to new information or changing values.

Second, the Cultural Theory is recognized as being imperfect and rather rigid, and not being able to account for the full variety of world visions and perspectives (van Asselt and Rotmans, 1996, De Schryver et al., 2010b). A common misunderstanding of the Cultural Theory is that every individual fully fits into a certain perspective. Most people would at best fit somewhere between two perspectives and switch perspective depending on their task, and role in society (Janssen and Rotmans, 1995, Thompson et al., 1990). Therefore, specific sets of value choices, others than presented in this PhD thesis, can be developed. Regarding the question which or who's perspective to implement, one can distinguish between LCAs applied on governmental level and on industry level. For governmental bodies, it is recommended to develop a set of value choices reflecting the vision of the government on the environment and society. Within the European Union, this can be done in line with the recommendations for methods (JRC, 2010a) and inventory database (JRC, 2009). The set of value choice can then be used as a default when using LCA results in environmental decision making. However, the complex mixture of science and value judgments (reflecting moral and esthetic values) within policy and environmental decision making requires the participation of all actors (French and Geldermann, 2005, Kloprogge and Sluijs, 2006, Mahmoud et al., 2009). This suggests the development of a set of value choices reflecting the vision of the parties involved. Within industry, LCA results can be used strictly internally with the goal to improve the environmental performance of the product or used externally to communicate to consumers and stakeholders. In this case, the set of value choices applied throughout the whole LCA can reflect the vision of the company, the consumers or stakeholders. Depending on the focus group, different perspectives can be defined and implemented throughout the LCA. As pointed out before, especially the choice in time perspective and accepted level of risk are responsible for high uncertainties. Therefore, focusing on these two value choices as a start allows to capture, in most cases, the main uncertainties from value choices.

Third, an analysis of the value choices made within environmental impact assessment models covering other impact categories, such as characterization models for resource depletion, is recommendable. Next to this, more research is required to reduce uncertainties from contextual value choices which largely influence LCA results or are insufficiently analyzed. Examples are including or excluding bioaccumulation of chemicals, uncertain effects from secondary particulate matter and defining future scenarios for impact categories with future effects. Furthermore, as indicated for ecosystem health, part of the uncertainty derives from 'ignorance', which is difficult, if not impossible, to quantify. Therefore, there is a need for further research to develop impact indicators which cover a broader part of the cause-effect pathway and consider region specific effects.

Fourth, a practical LCA framework needs to be developed that is able to accommodate frames and perceptions corresponding to the actors involved and allow a discussion to be organized among the perceptions (Bras-Klapwijk, 1998). Mahmoud et al. (2009) proposed a formal approach for scenario development in environmental impact assessment studies. Within their approach five different steps are described (scenario definition, construction, analysis, assessment and risk management), each step involving interaction and cooperation of stakeholders and/or scientists. These steps can be used in parallel with the LCA stages (goal and scope definition, inventory, impact assessment and interpretation), as described in ISO 14040 (ISO, 2006). Within the 'goal and scope definition', different scenarios can be defined and constructed, influencing for example the selected impact assessment methodology. This requires the development of a flexible impact assessment methodology that allows users to implement their defined scenarios (see Chapter 5). This corresponds to the suggestions of Bras-Klapwijk et al. (1998), who proposed an extra step in the LCA procedure, structuring the concept of the environmental burden and in this way creating the openness and transparency of the methodology used. Within the 'inventory' and 'impact assessment', the influence of the various scenarios on the LCA outcomes can be quantified (analysis and assessment). Within the 'interpretation' stage the results can be further interpreted and management options can be formulated. This practical framework can indeed stimulate the explanation of values and perspectives and the evaluation of alternatives in LCAs.

7.6 References

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Definition table

Definition table

Definitions derive from the Joint Research Center (JRC, 2010b, JRC, 2010a) and International Standard Organization (ISO, 2006b, ISO, 2006a).

Area of protection: an entity that we want to protect, such as 'human health', 'ecosystem health' or 'resource depletion'.

Category indicator (impact category indicator result): a quantifiable representation of an impact category. This quantification can be done on midpoint and endpoint level, see definition of 'midpoint indicator' and 'endpoint indicator'.

Cause-effect pathway (environmental mechanism): an outline of the impact mechanism that includes all the relevant pathways (physical, chemical and biological processes), linking the life cycle inventory analysis results to category indicators and category endpoints.

Characterization: quantitatively modeling the impact from each emission according to the underlying environmental mechanism.

Characterization method: a model or combination of models used to calculate characterization factors.

Characterization factor (CF): a factor derived from a characterization model or method which is applied to convert an assigned life cycle inventory analysis result to the common unit of the category indicator.

Contextual value: a value reflecting social and moral judgment of the choice for an assumption, dataset or estimation method, because of limited scientific knowledge. For example the definition of the products life cycle system boundaries, the inclusion of uncertain effects and how to divide emissions over different co-products. Contextual values appear on both model level and parameter level. In this thesis we only define and consider them on model level.

Damage factor: damage part of the cause-effect pathway that calculates for example the amount of Disability Adjusted Life Years (DALYs) per disease case.

Endpoint (damage) indicator: a quantifiable representation of an impact category, identifying an environmental issue giving cause for concern.

Functional unit: the quantified performance of a product system for use as a reference unit.

Impact assessment methodology: a methodology combines several impact categories.

Impact category: a class representing an environmental issues of concern to which life cycle inventory analysis results may be assigned.

Definition table

Impact score (using endpoint indicators this is also called damage score): outcome of an assessment, namely the conversion of LCI results to common units and the aggregation of the converted results within the same impact category (e.g., DALY per functional unit). This conversion uses characterization factors.

Inventory data: a collection of inputs and outputs (outgoing emissions and ingoing raw materials or land use) with regard to the system being studied, also called interventions.

Life cycle assessment (LCA): a compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle.

Life cycle impact assessment (LCIA): a phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product.

Midpoint indicator: a quantifiable representation of an impact category, identifying an environmental issue based on an indicator chosen somewehere along the cause-effect pathway. The midpoint indicator is commonly selected at the point in the cause-effect chain where individual stressor-specific differences within an impact category are still evident, e.g. the influence of various greenhouse gases on radiative forcing in the atmosphere.

Preference value: a value that embodies moral beliefs without any science being involved, such as the concern for equity or future generations.

Reference flow: a measure of the outputs from processes in a given product system required to fulfill the function expressed by the functional unit.

Value choice: a choice that includes the use of values and subjectivity (from differences in moral beliefs, interests and concerns).

Summary

Summary

Life Cycle Assessment (LCA) is the compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle. The product's life cycle includes all processes which can be related to the production, use and disposal of the product. Modeling the life cycle of a product and assessing its environmental impacts is known to be a challenging task due to the complexity, incomplete and uncertain knowledge. The presence of uncertainties in the life cycle impact assessment (LCIA) models for LCA are recognized, though seldom quantified. Especially when applying an impact indicator that quantifies the ultimate damage effect (such as loss of species or human life years), using endpoint characterization factors (CFs), the uncertainties are high. Some studies provide guidelines and rules of thumb to quantify uncertainties in CFs, however, uncertainties from value choices introduced by the method developer are mostly disregarded.

The goal of this PhD thesis is to assess uncertainties in LCIA models. The focus is on (i) a number of impact categories assessing ecosystem health with relatively high uncertainty, namely climate change, land use and ecotoxicity, and (ii) uncertainties from value choices in impact categories addressing human health.

Within these topics the following three research questions are tackled:

1. What are the sources of uncertainty in LCIA models for ecosystem health, in particular for ecotoxicity, land use and climate change?
2. What are the uncertainties deriving from value choices made in LCIA models for human health and how can the Cultural Theory be applied to quantify these uncertainties?
3. What are the practical implications of value choices within LCIA modeling?

Chapter 2 of this thesis provides an overview of the cause-effect pathways related to the release of toxic chemicals and physical land use practices caused by food production practices. It also discusses the background and application of several LCIA models that produce so-called CFs to quantify the environmental effects of the agricultural activity occurring along the cause-effect pathways. Particular attention is paid to advances in the data and modeling of ecotoxicological and land use impacts that resulted in the development of a consensus model to calculate CFs for aquatic ecotoxicity and several models to calculate CFs for land use. Finally, for both ecotoxicity and land use modeling, a number of uncertainties are discussed and several requirements for improvement are proposed

In Chapter 3, a model framework is developed to analyze various key assumptions and uncertainties within the development of CFs for land use, using the Cultural Theory as framework. The CFs are expressed as potential disappeared fraction (PDF) of vascular plant species based on species area relationships. It is found that the absolute values of the CFs can change from negative to positive scores with an average difference of 0.8 PDF between the two extreme perspectives, i.e., individualistic and egalitarian. The difference between these scenarios is for 40% explained by the choice in the species accumulation factor z and for 60% by the choice in including regional effects.

Within the egalitarian and hierarchist perspective the species accumulation factor z is for more than 80% responsible for the parameter uncertainty. Modeling choices and uncertainties within the species area relationship hardly change the ranking of the different land practices but largely influence the absolute value of the CFs for land use. The absolute change in the land use CFs can change the interpretation of land use impacts compared to other stressors such as climate change.

In Chapter 4, the Cultural Theory is used to handle uncertainties from value choices within the calculations of CFs for greenhouse gasses. New CFs are derived for 63 greenhouse gasses that quantify the impact of an emission change on human and ecosystem health damage. For human health damage, the disability-adjusted life years (DALYs) per unit emission related to malaria, diarrhea, malnutrition, drowning, and cardio-vascular diseases were quantified. For ecosystem health damage, the PDF over space and time of various species groups, including plants, butterflies, birds and mammals, per unit emission was calculated. The use of PDF as unit makes the CFs suitable for comparison with other types of stressors, such as substances causing acidification and human toxicity. The study shows that the CF of a GHG can change up to four orders of magnitude, depending on the defined value choices. The value choice mainly responsible for this difference in results is the choice for a specific time horizon. This indicates that by combining global warming damage scores with damage scores from other impact categories, inconsistent modeling assumptions may arise, such as differences in time horizon or assumptions on socio-economic adaptations. Therefore, there is a need to assess the consistency between impact categories characterizing CFs with the same unit, such as DALY, what is done in Chapter 5.

Chapter 5 explores a broader implementation of the Cultural Theory by combining seven human health impact categories: water scarcity, tropospheric ozone formation, particulate matter formation, human toxicity, ionizing radiation, stratospheric ozone depletion and climate change. Existing LCIA models are adapted to reflect a consistent set of value choices and new CFs are calculated. The study shows that individual, hierarchical and egalitarian perspectives can lead to CFs that vary up to six orders of magnitude. For substances with a relative long residence time in the environment (i.e., >100 years), the choice in time horizon and inclusion of bioaccumulation of essential metals mainly explains the differences among perspectives. For substances with a shorter residence time (i.e., <100 years), the difference in CFs among perspectives is smaller. For these substances the difference in CFs mainly derives from the choice in discounting and age weighting, and including uncertain effects or exposure. The results stress the importance of dealing with value choices in LCIA and suggest further research for analyzing the practical consequences for LCA results (done in Chapter 6).

Chapter 6 indicates the consequences of value choices within impact assessment modeling for human health on a range of products. In this study, the three sets of CFs developed and presented in Chapter 5 are used to calculate the human health damage score for 756 products. The results indicate that the

average discrepancy in damage score goes from one order of magnitude between the individualist and hierarchist perspectives to 2.5 orders of magnitude between the individualist and egalitarian perspectives. The difference in damage score among perspectives for individual products depends on the combination of emissions driving the impact of both perspectives and can raise up to five orders of magnitude. Also here, the value choices mainly responsible for the differences in damage score among perspectives are the choice of time horizon and the inclusion or exclusion of highly uncertain effects or exposure routes. The choice in perspective can alter the ranking of a product comparison when (i) the human health damage score of two products differ less than a factor of seven (75% confidence interval) whatever the perspective chosen, and (ii) the comparing products are based on largely different underlying processes and corresponding emissions (long living versus short living substances). The results imply that value choices within impact assessment modeling can modify the outcome of an LCA and thus the practical implication of decisions based on the results of an LCA.

Chapter 7 gives an overview of the different uncertainties in LCIA methods for ecosystem health and the value choices in LCIA methods for human health. A broader insight is provided in the uncertainties and discussions related to the value choices made to reflect the different perspectives. The practical implications of value choices in LCIA modeling are discussed and four recommendations are made on how to handle value choices in LCA. First of all, the expertise of social sciences is suggested to help defining proper scenarios. Second, regarding the level of complexity it is suggested to develop a set of value choices reflecting the vision of the parties involved, or the government specific vision. Third, more research and analysis is recommended to quantify uncertainties within impact categories other than addressed here and to reduce the major uncertainties from value choices. Second, At last, a practical LCA framework is recommended by applying the different steps of the scenario approach of Mahmoud et al. (2009) in parallel with the LCA stages described in ISO 14044.

Samenvatting

De milieugerichte levenscyclus analyse (LCA) is een methode die de potentiële milieueffecten van een product of dienst kwantificeert. Hierbij worden alle processen die gerelateerd kunnen worden aan productie, gebruik en afval van het product meegenomen. Het berekenen van milieueffecten gebeurt met complexe milieumodellen die impactfactoren berekenen. De impactfactoren zijn een maat voor het optreden van milieueffecten. De factoren varieren vanwege de onzekerheid in gegevens en beperkte kennis over de werkelijkheid. Hoewel de onzekerheden in impactfactoren algemeen worden erkend, zijn deze slechts zelden expliciet zichtbaar gemaakt. Een aantal studies geeft richtlijnen en vuistregels over hoe onzekerheden in de effectbeoordeling gekwantificeerd kunnen worden; echter de onzekerheden te wijten aan keuzes in de modellen zelf worden nauwelijks in beschouwing genomen.

Het doel van dit proefschrift is het inventariseren en analyseren van keuzes in milieumodellen en het kwantificeren van hun invloed op impactfactoren die gebruikt worden in LCA. Binnen dit hoofddoel zijn de volgende drie onderzoeksvragen beantwoord:

1. Wat zijn de bronnen van onzekerheden in milieumodellen die effecten op ecosystemen kwantificeren in LCAs, specifiek voor ecotoxiciteit, landgebruik en klimaatverandering?

2. Wat is de onzekerheid in impactfactoren door keuzes in milieumodellen die menselijke gezondheid quantificeren en hoe kan de Culturele Theorie gebruikt worden om deze onzekerheden te kwantificeren?

3. Wat zijn de praktische gevolgen van keuzes in milieumodellen die gebruikt worden in LCA?

Hoofdstuk 2 van deze thesis geeft een overzicht van de oorzaak-effectketens van respectievelijk de uitstoot van giftige stoffen en landgebruik gerelateerd aan de productie van voedsel. Dit hoofdstuk bespreekt de achtergrond en toepassing van diverse milieumodellen die milieueffecten van landbouwactiviteiten kwantificeren. Voor zowel ecotoxiciteit als landgebruik is een aantal onzekerheden besproken en een aantal mogelijke verbeteringen in de modellering voorgesteld. De modellen kunnen verbeterd worden door onder andere landgebruiktypes beter te specificeren en meer milieueffecten te quantificeren en meer pesticiden mee te nemen.

In hoofdstuk 3 is een model opgesteld, gebruik makend van de Culturele Theorie, om de belangrijkste keuzes en onzekerheden in de effectbeoordeling van landgebruik te analyseren. Volgens de Culturele Theorie kunnen verschillen in keuzes gegroepeerd worden op basis van sociaal-maatschappelijke structuren en perspectieven. Drie belangrijke perspectieven worden onderscheiden, het individualistisch, hiërarchisch en egalitair perspectief, waarbij elk perspectief een bepaalde visie op natuur en maatschappij reflecteert. De effectbeoordeling bij landgebruik drukt de fractie van verlies aan plantensoorten uit, gebruik makend van de soorten-oppervlakterelatie. De studie toont aan dat de de impactfactoren, die gebruikt worden in de effectbeoordeling van landgebruik, van negatief naar

positief kunnen gaan afhankelijk van welk perspectief gekozen wordt. Het verschil tussen de perspectieven wordt deels verklaard door de keuze in het meenemen van soortenverlies te wijten aan het verkleinen van omliggend land en de keuze van de exponent in de soorten-oppervlakterelatie. Deze keuzes en onzekerheden blijken echter nauwelijks invloed te hebben op de ranking van verschillende landgebruik types, maar wel op de absolute waarde van de impact factoren. Deze absolute verandering kan de interpretatie van landgebruik beïnvloeden ten opzichte van andere effecten, zoals klimaat verandering.

In hoofdstuk 4 wordt de Culturele Theorie gebruikt om de onzekerheden, te wijten aan modelkeuzes, in de effectbeoordeling van broeikasgas emissies te analyseren. Voor 63 broeikasgassen zijn nieuwe impactfactoren berekend om de effecten van een emissieverandering op mensen en ecosystemen te kwantificeren. Voor mensen is de kans op malaria, diarree, ondervoeding, verdrinking en hart- en vaatziekten en daaraan gekoppeld het aantal verloren levensjaren per eenheid emissie berekend. Voor ecosystemen is voor verschillende soorten, inclusief planten, vlinders, vogels en zoogdieren, het relatief verlies aan soorten over tijd en ruimte per eenheid emissie berekend. Dit maakt de impactfactoren geschikt voor vergelijking met andere milieueffecten, zoals verzuring van ecosystemen of het effect van giftige stoffen op mensen. Deze studie toont aan dat, afhankelijk van de gemaakte keuzes, de impactfactoren van broeikasgassen met vier orde grootes kunnen veranderen (een factor 10,000). De keuze voor een specifieke tijdshorizon bepaalt tot hoeveel jaar na de emissie de milieueffecten meegenomen worden in de effectbeoordeling en is hoofdverantwoordelijk voor de grote verschillen. Dit resultaat geeft aan dat het combineren van klimaateffecten met andere effecten (bijvoorbeeld humane toxiciteit) gevoelig is voor inconsistente keuzes in de modellering, zoals verschillen in tijdshorizon. Daarom is het verstandig de consistentie tussen effectbeoordelings-methoden te analyseren. Dit wordt gedaan in hoofdstuk 5.

Hoofdstuk 5 beschrijft de implementatie van de Culturele Theorie voor zeven stressoren die invloed hebben op de menselijke gezondheid, namelijk water schaarste, vorming van troposferische ozon, vorming van fijn stof, giftige stoffen, ionizerende straling, afname van stratosferische ozon en klimaatverandering. Bestaande modellen voor effectbeoordeling zijn op een consistente manier aangepast en drie nieuwe sets van impactfactoren zijn berekend. De studie laat zien dat de keuze in een individualistisch, hiërarchisch of egalitair perspectief kan zorgen voor een verandering in impactfactoren die tot zes orde grootes reikt (een factor 1,000,000). Voor stoffen met een relatief lange levensduur (> 100 jaar), wordt het verschil tussen de perspectieven verklaard door de keuze in tijdshorizon. Voor stoffen met een relatief korte levensduur (< 100 jaar) is het verschil in impactfactoren tussen de perspectieven kleiner. In dit geval worden de verschillen verklaard door de keuzes gemaakt in het bepalen van de verloren levensjaren per gezondheidseffect en het al dan niet meenemen van (onzekere) effecten en blootstellingsroutes. De resultaten benadrukken het belang van

het kwantificeren van de invloed van keuzes in de effectbeoordeling. Verder onderzoek naar de praktische gevolgen in LCA-resultaten is beschreven in hoofdstuk 6.

Hoofdstuk 6 kwantificeert de gevolgen van keuzes in de beoordeling van humane gezondheidseffecten voor een reeks van producten. In deze studie worden de drie sets van impactfactoren, beschreven in hoofdstuk 5, gebruikt om de gezondheidsschade voor 756 producten uit te rekenen. De resultaten geven aan dat de gemiddelde afwijking varieert van één orde grootte (factor 10) tussen het individualistisch en hiërarchisch perspectief, tot 2.5 orde groottes (factor 300) tussen het individualistisch en egalitair perspectief. Voor sommige producten kan het verschil in berekende gezondheidsschade echter oplopen tot vijf orde groottes (factor 100,000). Ook hier zijn de keuzes in tijdshorizon en het al dan niet meenemen van onzekere effecten en blootstellingsroutes verantwoordelijk voor het verschil. De keuze in perspectief kan bij het vergelijken van producten de ranking van twee producten beinvloeden wanneer (i) het verschil in gezondheidsschade van twee producten kleiner is als een factor zeven (75% confidence interval), and (ii) de producten op verschillende onderliggende processen en daarbij verschillende type emissies berusten (emissies met een relatief lange levensduur ten opzichte van emissies met een relatief korte levensduur). De resultaten geven aan dat keuzes in de methoden voor effectbeoordeling de uitkomst van een analyse kan beinvloeden.

Hoofdstuk 7 reflecteert op de studies beschreven in de eerdere hoofdstukken en bediscussieert de verschillende onzekerheden in de methoden voor effectbeoordeling. Verder zijn aanbevelingen gedaan hoe gebruikers systematisch om kunnnen gaan met onzekerheden in de effectbeoordeling. Ten slotte is een praktisch raamwerk geformuleerd die de verschillende stappen in het omgaan met keuze-onzekerheden combineert met de practische LCA stappen.

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Dankwoord

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"To my family"

Curriculum vitae

Curriculum vitae

An De Schryver was born in Jette, Belgium, on November the 10th 1982. In 2000 she started to study environmental biology at the Free University of Brussels in Belgium. In august 2003 she started her master thesis as an invited scientist at the Carmabi research institute in Curaçao. In collaboration with the Department of Animal Ecology and Ecophysiology from the Radboud University of Nijmegen she worked on the structural complexity of mangrove habitats and the attraction of juvenile reef fish species. Next to this she analysed mangrove forest degradation on Curaçao using geographical information systems. In July 2004 she got her MSc degree in Environmental Biology at the Free University of Brussels. As a follow up of her studies she did a one year teaching education at the Radboud University of Nijmegen. After getting her teaching degree in July 2005 she worked for a few months as researcher for the Department of Environmental Sciences at the Radboud University of Nijmegen. She entered PRé Consultants in January 2006, as a junior life cycle assessment consultant. At PRé Consultants she participated in a number of projects ranging from life cycle assessments for branch associations to impact assessment research for the European Union. Within the ReCiPe 2008 project she was able to concentrate her research activities on the development of climate change, land use and resource depletion impact assessment models. In June 2008 she took the opportunity to expand her research activities in the field of life cycle impact assessment and started to work part time for the Department of Environmental Sciences at the Radboud University of Nijmegen. In June 2009 she left PRé Consultants for a fulltime PhD position at the Department of Environmental Sciences at the Radboud University of Nijmegen. In August 2010 she started her PostDoc at the Swiss Federal Institute of Technology in Zurich.

List of publications

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