Int J Life Cycle Assess (2013) 18:698–706
DOI 10.1007/s11367-012-0504-x

UNCERTAINTIES IN LCA

# The influence of value choices in life cycle impact assessment of stressors causing human health damage

An M. De Schryver · Sebastien Humbert ·
Mark A. J. Huijbregts

Received: 28 March 2011 / Accepted: 6 September 2012 / Published online: 20 September 2012


**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 perspective.
*Results* We found that the average difference in human health damage score goes from 1 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 5 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 a life cycle assessment (LCA) and thus the practical implication of decisions based on the results of an LCA.

**Keywords** Decision making · Human health · Life cycle assessment · Uncertainties · Value choices

Responsible editor: Andreas Ciroth

**Electronic supplementary material** The online version of this article (doi:10.1007/s11367-012-0504-x) contains supplementary material, which is available to authorized users.

A. M. De Schryver (✉)
Institute of Environmental Engineering,
ETH Zurich, ETH Honggerberg,
8093 Zurich, Switzerland
e-mail: an.schryver@gmail.com

S. Humbert
Quantis, Parc Scientifique EPFL Bât D,
1015 Lausanne, Switzerland

M. A. J. Huijbregts
Department of Environmental Science, Institute for Water and Wetland Research, Radboud University Nijmegen,
P.O. Box 9010, 6500 GL Nijmegen, The Netherlands

## 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, 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 coproducts (Schmidt 2008; Luo et al. 2009; Ayer et al. 2007; Werner 2005; European Aluminium Association 2002). The impact assessment methodologies Eco-indicator 99 (Goedkoop and Spriensma 1999) and ReCiPe 2009 (Goedkoop et al. 2009) used the Cultural Theory to define three sets of characterization factors (CFs) reflecting different value choices (considering an individualist, hierarchist, and egalitarian perspective). CFs are used to quantify and aggregate life cycle emissions into damage scores for human health and ecosystem health. The implementation of these three perspectives allows to assess the influence of value choices on LCA results. Each perspective reflects differences in moral beliefs, concerns, and interests that explain one's view on society and nature, and that corresponds to a specific set of values (Schwarz and Thompson 1990; Hofstetter et al. 2000). De Schryver et al. (2011) further investigated how value choices can influence CFs for a number of human health impact categories (expressed in disability-adjusted life years (DALY) per unit of intervention) in a consistent way. The impact categories considered were water scarcity, tropospheric ozone formation, particulate matter formation, human toxicity, ionizing radiation, stratospheric ozone depletion, and climate change. Although CFs can change orders of magnitude from one perspective to the other, it is not clear how these differences influence actual LCA results.

The aim of this paper was to analyze the influence of value choices within human CFs (expressed in DALY per unit intervention) on the LCA outcome of a range of products. We calculated the human health damage score, expressed in DALY, using the three sets of CFs developed by De Schryver et al. (2011). The average relative contribution of each impact category to the human health damage score was 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 were highlighted and explained.

## 2 Methods

### 2.1 Human health impact

The human health damage scores (expressed in DALY) were calculated by applying the CFs from De Schryver et al. (2011). 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, (1) 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; (2) 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 (3) 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 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. (2011).

### 2.2 Life cycle inventory dataset

For all impact categories, except water scarcity (details see Electronic supplementary material (ESM) 1), 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 4,000 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 2). The full list of products included in this analysis is given in ESM 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

**Table 1** Overview of value choices implemented in the CFs developed by De Schryver et al. (2011)

| 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[b] | 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

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 and 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 that results in $9.0 \cdot 10^{-4}$ to $2.7 \cdot 10^{-3}$ $m^3$ of water evaporation per kilowatt hour (Water and Sustainability 2002; Ecoinvent Centre 2007), 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 ESM 1.

**Table 2** Product groups and number of products included (prod. incl.) in the analysis

| Product group | Number of products included | Type of products included |
|---|---|---|
| Agricultural products | 72 | Plant products and by products, 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 |

## 2.3 Alignment data inventory and characterization factors

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

1. The CFs for biogenic carbon dioxide uptake and emissions were set on zero, considering an equal uptake and release balance (Cherubini et al. 2009; Gnansounou et al. 2009).
2. For water scarcity, De Schryver et al. (2011) 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 ESM 1 (see Tables 1, 2, and 3).
3. 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 land, expressed in DALYs, the net heavy metal emissions to agricultural soil are excluded from the analysis.
4. 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 ESM 1, Tables 4 and 5).

### 2.4 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, 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 (Bland and Altman 1999).

## 3 Results

### 3.1 Relative contribution

Out of an emission list of more than 600 substances, fewer than 50 substances contribute 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 ESM 1, Table 6. Figure 1 shows the average share of each impact category in the human health damage score per product group. ESM 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.

The impact scores from climate change (ESM 2) 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.

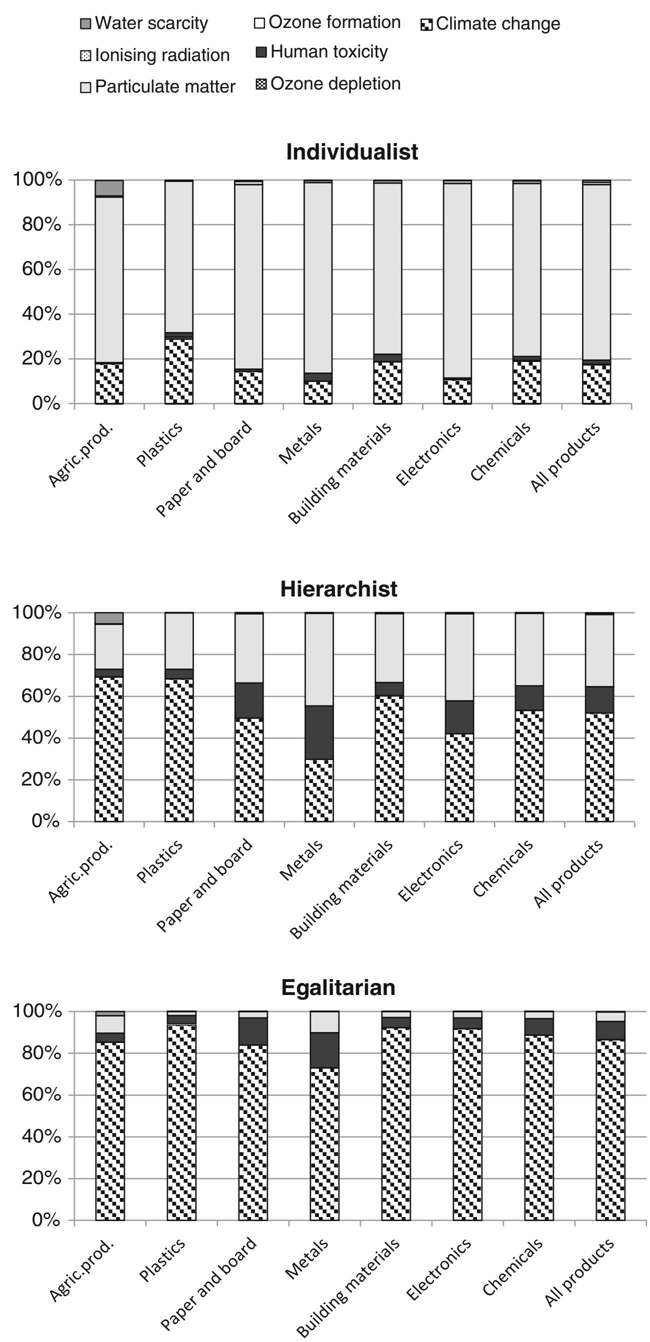


**Fig. 1** The average relative contribution of each impact category to the human health damage score (in percent), per perspective for all products and per product group. *Agric.prod.* agricultural products

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 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 12–26 % of the human health damage score for the hierarchist perspective.

### 3.2 Bland–Altman statistics

Figure 2 presents three Bland–Altman plots showing the difference in the logarithmic human health damage scores across the three perspectives, together with the 95 and 75 % confidence intervals. Note that the inverse logarithmic differences in human health damage scores provide ratios between perspectives (Bland and Altman 1999). 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 ratio in damage scores of 280. The ratio varies from this by up to a factor of 7 with 75 % confidence, or by up to a factor of 30 with 95 % confidence. The egalitarian and hierarchist perspectives show an average ratio in damage scores of 30—the ratio varies from this by up to a factor of 4 with 75 % confidence, or by up to a factor of 10 with 95 % confidence. The hierarchist and individualist perspectives show the lowest average difference in log damage scores, i.e., an average ratio in damage scores of 10. In this case, the ratio varies from this by up to a factor of 5 with 75 % confidence, or by up to a factor to 16 with 95 % confidence. Products showing a ratio in damage scores outside the 75 and 95 % confidence intervals are mainly chemicals and metals. Average differences and confidence intervals per product group are given in ESM 1 (Table 7).

## 4 Discussion

### 4.1 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. The lack in alignment is due to both missing inventory data (such as regional specific CFs for water consumption) and missing CFs (such as missing CFs for heavy metal uptake from agricultural products).

Within this study, impacts are calculated by combining total emission data with average (nonregion 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 and Sustainability 2002). Overall, the water scarcity impact results should be interpreted as a first approximation.

For heavy metal emissions to agricultural soils, the inventory data and characterization factors for human toxicity do not match. While the inventory data provides the uptake of heavy metals by agricultural products, no CFs exist that quantify the human toxicity impact from this step on. Therefore, 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

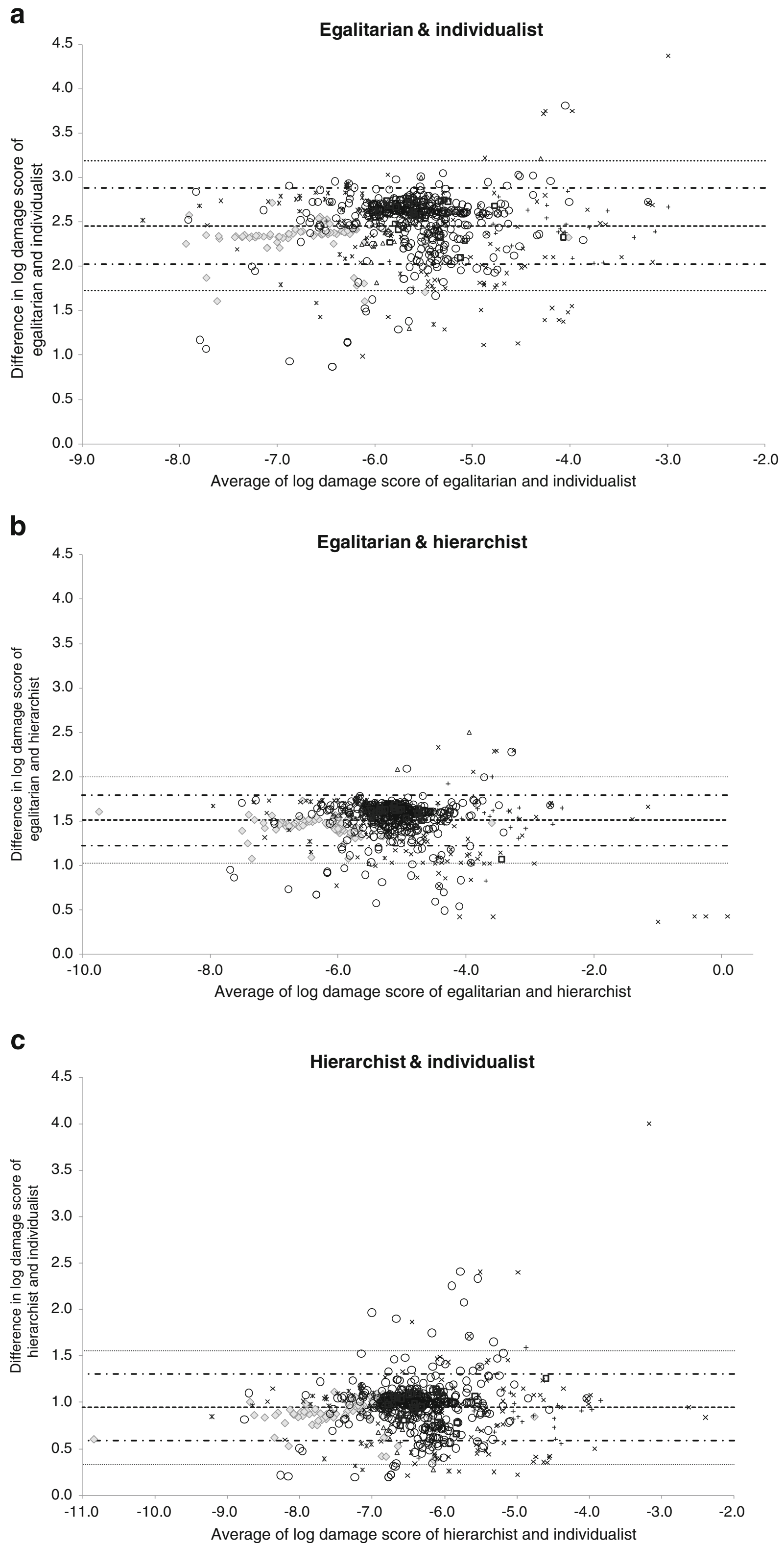


**Fig. 2** Bland–Altman plots showing the difference damage scores (*y*-axes) plotted against their average (*x*-axes) after log transformation of the data; for the egalitarian and individualist perspectives (**a**), for the egalitarian and hierarchist perspectives (**b**) and for the hierarchist and individualist perspectives (**c**). Note that the inverse logarithmic values on the *x*- and *y*-axes results in geometric means on the *x*-axis and the ratio between perspectives on the *y*-axis. The *dashed line* presents the average difference; the *dotted lines* indicate the 2.5 and 97.5 confidence intervals; the *dash–dot* lines indicate the 12.5 and 87.5 confidence intervals. Each marker type represents a product group: ◇ Agricultural products - Building materials ○ Chemicals + Electronics × Metals △ Paper and board □ Plastics

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 life cycle inventory data and CFs. Impacts, such as those from noise and indoor air emissions, should be included in order to improve the human health damage calculations.

### 4.2 Interpretation

Bearing in mind the aforesaid uncertainties and 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 products 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 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 1 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. 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 ratio difference in human health damage score among perspectives above the 95 % confidence interval. 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. 2011). 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 results for the egalitarian perspective.

Should the comparative damage scores of two products differ more than a factor of 30, we can be 95 % confident that such ranking will not be influenced by the choice in perspective. If we accept 75 % confidence, the relevant ratio of damage scores is reduced to 7. Furthermore, product ranking is minimally 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 differ largely between production processes. An example is the comparison between corrugated kraftliner board and chipboard. Corrugated kraftliner board has relatively high PM10 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 CO2 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.

## 5 Conclusions

We can conclude that value choices in impact modeling have direct implications for LCA outcomes. Human health damage scores can vary by up to 4 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 (1) the human health damage scores of two products differ less than a factor of 7 (75 % confidence interval) whatever the perspective chosen and (2) the comparing products are based on largely different underlying processes and corresponding emissions (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, it is not only the different system boundaries and applied assumptions that are important, but also 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.

**Acknowledgments** The research was partly funded by the European Commission under the 7th Framework Program on Environment; ENV.2008.3.3.2.1: PROSUITE—Sustainability Assessment of Technologies, grant agreement number 227078. We would like to thank two anonymous reviewers for their useful and constructive comments.

## References

ABS (2004) Water Account Australia 1996–97. Australian Bureau of Statistics, Canberra

Ayer N, Tyedmers P, Pelletier N, Sonesson U, Scholz A (2007) Co-product allocation in life cycle assessments of seafood production systems: review of problems and strategies. Int J Life Cycle Assess 12(7):480–487

Bauer C, Bolliger R, Tuchschmid M, Faist-Emmeneger M (2007) Wasserkraft. In: Dones R (ed) Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Öokobilanzen für die Schweiz. Paul Scherrer Institut Villingen, Swiss Centre for Life Cycle Inventories, Dübendorf, p 104

Bland J, Altman D (1986) Statistical methods for assessing agreement between two methods of clinical measurement. Lancet 1:307–310

Bland JM, Altman DG (1999) Measuring agreement in method comparison studies. Stat Methods Med Res 8(2):135–160

Cherubini F, Bird ND, Cowie A, Jungmeier G, Schlamadinger B, Woess-Gallasch S (2009) Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: key issues, ranges and recommendations. Resour Conserv Recycl 53(8):434–447

De Schryver AM, van Zelm R, Humbert S, Pfister S, McKone TE, Huijbregts MAJ (2011) Value choices in life cycle impact assessment of stressors causing human health damage. J Ind Ecol 15 (5):796–815

EC (2001) Green paper on Integrated Product Policy. COM (2001) 68 final. European Commission, Brussels

EC (2005) Taking sustainable use of resources forward: a thematic strategy on the prevention and recycling of waste. COM(2005) 666 final. European Commission, Brussels, Belgium

Ecoinvent Centre (2007) ecoinvent data v2.01, ecoinvent reports No. 1–25. CD-ROM. Duebendorf, Switzerland

Environment Canada (2004) Threats to water availability in Canada. National Water Research Institute, Burlington

European Aluminium Association (2002) LCA and legislation. http://www.eaa.net/en/environment-health-safety/lca/lca-legislation/. 11–2009

Euser AM, Dekker FW, le Cessie S (2008) A practical approach to Bland–Altman plots and variation coefficients for log transformed variables. J Clin Epidemiol 61(10):978–982

Ezzati M, Lopez AD, Rodgers A, Murray CJL (eds) (2004) Comparative quantification of health risks: global and regional burden of diseases attributable to selected major risk factors. World Health Organization, Geneva

French S, Geldermann J (2005) The varied contexts of environmental decision problems and their implications for decision support. Environ Sci Policy 8(4):378–391

Gnansounou E, Dauriat A, Villegas J, Panichelli L (2009) Life cycle assessment of biofuels: energy and greenhouse gas balances. Bioresour Technol 100(21):4919–4930

Goedkoop M, Spriensma RT (1999) The Eco-indicator 99: a damage oriented method for life cycle impact assessment methodology. PRé Consultants, Amersfoort

Goedkoop M, Heijungs R, Huijbregts M, De Schryver AM, Struijs J, Van Zelm R (2009) ReCiPe 2009. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level; First edition Report I. Characterisation. VROM, Den Haag

Hofstetter P (ed) (1998) Perspectives in life cycle impact assessment. A structured approach to combine models of the technosphere, ecosphere and valuesphere. Kluwer, Dordrecht

Hofstetter P, Baumgartner T, Scholz RW (2000) Modelling the valuesphere and the ecosphere: integrating the decision makers' perspectives into LCA. Int J Life Cycle Assess 5(3):161–175

Lorenzoni I, Lowe T, Pidgeon N (2005) A strategic assessment of scientific and behavioural perspectives on 'dangerous' climate change. Technical report 28. Tyndall Centre for Climate Change Research, Norwich

Luo L, van der Voet E, Huppes G, Udo de Haes H (2009) Allocation issues in LCA methodology: a case study of corn stover-based fuel ethanol. Int J Life Cycle Assess 14(6):529–539

Nemecek T, Kägi T, Blaser S (2007) Life cycle inventories of agricultural production systems. Ecoinvent report version 2.0, vol 15. Swiss Centre for LCI, ART, Duebendorf and Zurich

Reiss R, Anderson EL, Cross CE, Hidy G, Hoel D, McClellan R, Moolgavkar S (2007) Evidence of health impacts of sulfate-and nitrate-containing particles in ambient air. Inhal Toxicol 19 (5):419–449

Renehan AG, Jones J, O'Dwyer ST, Shalet SM (2003) Determination of IGF-I, IGF-II, IGFBP-2, and IGFBP-3 levels in serum and plasma: comparisons using the Bland–Altman method. Growth Horm IGF Res 13(6):341–346

Schmidt J (2008) System delimitation in agricultural consequential LCA. Int J Life Cycle Assess 13(4):350–364

Schwarz M, Thompson M (eds) (1990) Divided we stand: re-defining politics, technology and social choice. University of Pennsylvania Press, Pennsylvania

Shiklomanov AI (1999) World water resources at the beginning of the 21st century in International Hydrological Programme. State Hydrological Institute (SHI)/UNESCO, St. Petersburg

Solley WB, Pierce RR, Perlman HA (1998) Estimated use of water in the United States in 1995. United States Geological Survey, Denver

Stewart BA, Howell TA (eds) (2003) Encyclopedia of water science. Dekker Encyclopedias Series. Marcel Dekker, New York

Water and Sustainability (2002) (Volume 3) U.S. Water Consumption for Power Production—The Next Half Century. EPRI, Palo Alto, US

Thompson M, Ellis R, Wildavsky A (1990) Cultural theory. Westview Press, Boulder

Unesco (2001) Securing the food supply. United Nations. World Water Assessment Programme. United Nations Education Scientific and Cultural Organization, Paris

Van Zelm R, Huijbregts MAJ, Den Hollander HA, Van Jaarsveld HA, Sauter FJ, Struijs J, Van Wijnen HJ, Van de Meent D (2008) European characterization factors for human health damage of PM10 and ozone in life cycle impact assessment. Atmos Environ 42:441–453

Werner F (ed) (2005) Ambiguities in decision-oriented life cycle inventories. The role of mental models and values, vol 17. Eco-Efficiency in Industry and Science. Springer, Amsterdam

Yang X, Dziegielewski B (2007) Water use by thermoelectric power plants in the United States. J Am Water Works Assoc 43(1):160–169