Robustness of Life Cycle Assessment Results

Influence of data variation and modelling choices on results for beverage packaging materials

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Table of content

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1 Introduction

1.1 Life cycle assessment

History and evolution of LCA

Life cycle assessment (LCA) is a well-established method to evaluate the potential environmental impacts and use of resources of a product or service throughout its life cycle (ISO, 2006a). Energy analyses, the predecessors of LCAs, emerged around 1970 and mainly focussed on the inventory of energy use and resources, emissions, and the generation of waste (Baumann and Tillman, 2004; Guinée et al., 2011). The use of LCA in its present format started in the late 1980s. The first impact assessment methods were introduced in the 1980s, and aimed to aggregate inventory data and divide them into classes (Habersatter and Widmer, 1990). Only the next generation impact assessment methods started to focus on understanding the relevance and effect of the inventory data (resources and emissions) on the environment. The so-called characterization factors evolved over time, and are based on scientific models to estimate the impact of substances along their impact pathways (e.g. the CML method from Heijungs et al. (1992)).The shift from the aggregation of inventory data into an environmental scientific based method was a significant change.

LCA went through a standardization and harmonisation of the framework, terminology and methodology during 1990 – 2000 (Guinée et al., 2011). The Society of Environmental Toxicology and Chemistry (SETAC) initially played a key role in harmonizing the framework and methodological part of LCA (Klöpffer, 2006), while the International Organization for Standardization (ISO) performed the standardisation of methods and procedures. The efforts of the SETAC and ISO resulted in two international standards: ISO 14040 and 14044 (ISO, 2006a, b).

The use of LCA has elaborated since the 1990s, and the number of LCA related studies and articles increased exponentially (Chen et al., 2014; Guinée et al., 2011; Hou et al., 2015; Peters, 2009). Meanwhile, the number of LCA approaches increased as well, resulting in a divergence in methods (Guinée et al., 2011). The United Nations Environment Programme (UNEP) and SETAC furthermore aimed, and continue to aim, to incorporate life cycle thinking into practice, and make improvements in the tools, data and impact indicators (Life Cycle

Initiative, 2015). Life cycle thinking continues to be incorporated in European Policy and strategies on the use of resources and waste management.

Standardised ISO LCA method

An ISO standardised LCA consists of four phases. Figure 1.1 presents the general ISO framework and shows the links among the four LCA phases. The first phase, *Goal and scope definition,* defines the purpose of the study and how it will be performed. The *goal definition* describes the objective of the study, the intended use of the results, and the audience. The s*cope definition* describes the applied methodological approach, the definition of the product, and the system boundaries of the studied product system. The scope also sets the methodological framework for the next phases, the life cycle inventory and impact assessment. The *Inventory analysis* examines the processes in the product system and quantifies for each process all the input and output data (i.e. inventory data). Economic inventory data include the amount of resources, materials and energy needed to manufacture a product. Environmental inventory data include all extracted natural resources which are used in the process (inputs), and emissions and waste released to the environment (outputs). *Impact assessment* converts the environmental inventory data into their contribution to the environmental impact in one or more impact categories. This phase first assigns the environmental inventory data to the selected impact categories (classification). The contribution of each environmental input or output to an impact category is next calculated based on characterisation models (characterisation). *Interpretation* evaluates the inventory data and/or impact results from the previous phases according to the defined goal and scope, and draws conclusions and/or recommendations. The interpretation phase also examines the confidence and reliability of the LCA results through sensitivity, scenario and/or uncertainty analyses. The LCA procedure is elaborated in two international standards: ISO 14040 and ISO 14044 (ISO, 2006a, b).

The ISO standards provide detailed guidance on procedures, but not how these procedural steps need to be taken (i.e. the methodology). ISO does not aim to standardise the LCA methods into every detail, which means that there is still room for a range of methodological choices. Methodological guidance is provided by several (national) standards and guidelines, each with their own interpretation of approaches and methods (e.g. AFNOR (2011); Baumann and Tillman (2004); BSI (2011); EC-JRC (2010, 2012); Guinée et al. (2002); Pankaj Bhatia et al. (2011)). The vagueness in the current ISO 14044 procedures is often addressed

(Ekvall and Finnveden, 2001; Guinée et al., 2011; Wardenaar et al., 2012; Weidema, 2014; Zamagni et al., 2008).

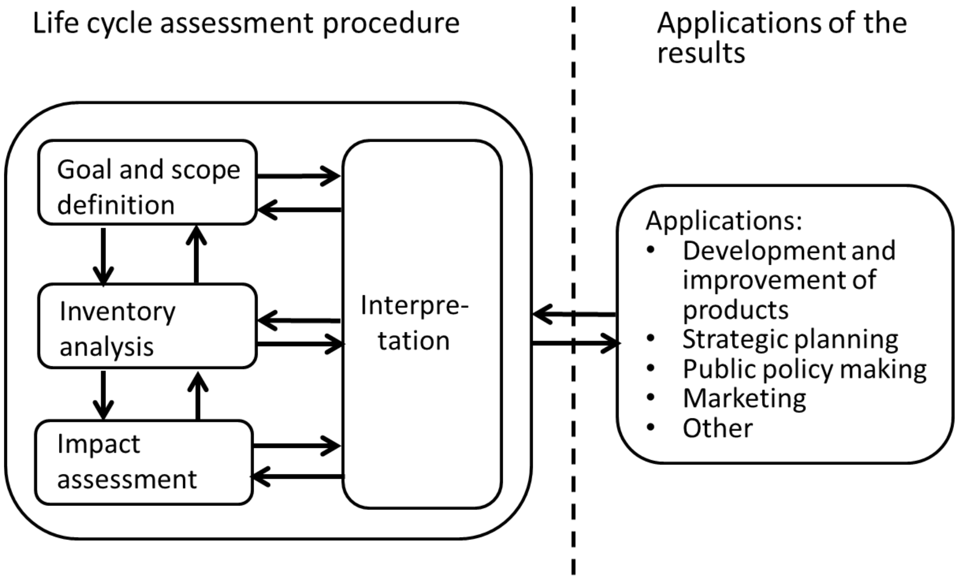

Figure 1.1: The general LCA methodological framework, based on ISO 14040 (2006a). The ISO LCA procedure is displayed on the left side of the dashed line. The application of the results is displayed on the right side of the line, and is not part of the ISO procedure.

Challenges in LCA

Many issues have been debated in the scientific literature since the start of LCA. Methodological debates started already during the early development of the LCA method and were then related to impact assessment, allocation, and the potential to simplify LCA (Baumann and Tillman, 2004). Meanwhile, the number of approaches on the system boundaries, allocation methods, and spatial differentiation increased (Guinée et al., 2011). LCA still contains challenges and suffers from unresolved issues (Finnveden et al., 2009; Reap et al., 2008; Tillman, 2000).

One of the discussions on the impact assessment includes the evaluation and estimation of characterisation factors (e.g. Hauschild et al. (2008) and Huijbregts et al. (2000b) for toxicological impact categories). Other discussions debate the site-dependency of the characterisation factors (e.g. Potting et al. (1998), Huijbregts et al. (2000a), Pennington et al. (2004), Potting and Hauschild (2006)). Impact assessment can be based on the cause-effect relationship between the inventory data and a particular impact category (midpoint indicators), or can be calculated for the effect on areas of protection (endpoint indicators) (Bare et al., 2000). Midpoint indicators are more robust, but endpoint indicators may be more relevant to society's understanding of the final effect. The different impact assessment approaches serve different purposes. Frequently used impact assessment methods include CML2 (Guinée et al., 2002) and its successor CML-IA (2010), Eco-indicator (Goedkoop and Spriensma, 2001) and its successor ReCiPe (Goedkoop et al., 2009), Impact 2002+ (Jolliet et al., 2003), and ILCD Midpoint 2011+ (EC-JRC, 2010). The choice of the impact assessment method influences the results (Dreyer et al., 2003). The number of impact assessment methods continues to increase.

Allocation issues emerge if a process has more than one function, and/or produces more than one product. The allocation problem deals with how to partition the environmental burdens among the multiple processes and products. The allocation problem is one of the most controversial and classical problems in LCA (Reap et al., 2008; Russell et al., 2005). Allocation methods are suggested and discussed by e.g. Ekvall and Tillman (1997), Ekvall and Finnveden (2001), Heijungs and Guinée (2007), Weidema and Schmidt (2010), Suh et al. (2010), and Pelletier et al. (2014). Recycling is a special multifunctional process. The recycling process is on one hand a waste management process, but on the other hand a production process for material. The environmental impacts of the recycling process and the produced recycled material need to be divided between the product system providing recyclable waste, and the product system using recycled material. Different perspectives on sustainability and system boundaries resulted in a range of methods on where and how to assign these impacts (EC-JRC, 2010; Ekvall and Finnveden, 2001; Ekvall and Tillman, 1997; Guinée et al., 2002; Ligthart and Ansems, 2012; Newell and Field, 1998). The application of different recycling modelling methods can result in different LCA outcomes for the same product system (Azapagic and Clift, 1999; Cederstrand et al., 2014; Ekvall and Finnveden, 2001; Liu and Müller, 2012; Wardenaar et al., 2012; Weidema and Schmidt, 2010). The recycling issue is still ongoing and additional methods keep emerging.

Debates on the system boundaries have changed into debates on the use of LCA as an accounting-type of assessment (attributional LCA), or a change-oriented type (consequential LCA). The two types serve different purposes and require a different methodology (Ekvall and Weidema, 2004; Löfgren et al., 2011; Tillman, 2000; Tillman et al., 1994; Weidema et al., 1999).

LCA is nowadays broadly applied and used by academics and industry for research purposes, product development, but also for environmental claims by companies. LCA is an important analytical tool for decision making in business and (public) policy making (Lloyd and Ries, 2007; Tillman, 2000). LCA is used as measurement device for comparisons among products, e.g. for the selection of environmentally friendly products. LCA results require to be robust and trustworthy if LCA is used as a decision-support tool (Finnveden and Ekvall, 1998; Geisler et al., 2005; Guinée et al., 2002; Ingwersen and Stevenson, 2012).

Despite the fact that LCA is standardized, LCA results for the same product provide in practice sometimes different and even conflicting results (e.g. Finnveden and Ekvall (1998), Lazarevic et al. (2010), Michaud et al. (2010), Padey et al. (2012), Price and Kendall (2012), von Falkenstein et al. (2010), Weiss et al. (2012), and Wenzel and Villanueva (2006)). Conflicts in LCA results are not beneficial for the credibility of LCA as a decision-support tool and the usefulness of LCA results. Differences in LCA outcomes for the same product or process are often assigned to uncertainties in data, but methodological choices and assumptions play an important role as well (Brandão et al., 2012).

1.2 Uncertainty in LCA

Uncertainty terminology and classification

Uncertainty is defined as ‘the quality or state of being uncertain; something that is doubtful or unknown; something that is uncertain’ (Merriam-Webster Dictionary, 2015). The terminology and classification for uncertainty is inconsistent in LCA (Finnveden et al., 2009; Heijungs and Huijbregts, 2004). Uncertainty is often characterised by the location where uncertainty occurs, and/or the type of uncertainty. Uncertainty in LCAs can occur according to Huijbregts (1998a) and Zamagni et al. (2008) in parameters (input data), the model (mathematical relationships), or due to choices (scenarios; normative choices). Notten and Petrie (2003) made a division in empirical parameters (input data), model parameters (value parameters and decision variables), and the model structure and form. Data uncertainty occurs in LCA due to data inaccuracy, lack of specific data, and variability (Heijungs and Huijbregts, 2004; Huijbregts et al., 2001). Data inaccuracy involves imprecise measurements, estimations, assumptions, or small number of sites that are investigated. Lack of specific data refers to data gaps or non-representative data. Data variability occurs if more than one value is available. Björklund (2002) additionally added epistemological uncertainty (lack of knowledge of the system behaviour), plain mistakes, and estimation of all types of uncertainties. Variability can be due to spatial or temporal variability, and variability between sources and objects (Huijbregts, 1998a). A detailed explanation of locations and types of uncertainty, as they are used in this thesis, is presented in section 1.4.

Management of uncertainty in LCA

A good environmental model requires the inclusion of uncertainty analysis, especially when the results are used to inform and support decision making (Bennett et al., 2013; Jakeman et al., 2006). Uncertainty is inherent in LCA studies and should be made explicit. Problems concerning uncertainty in LCA results have been recognised from the early start of LCA (Huijbregts, 1998a; Ross et al., 2002). The ISO 14044 procedure (ISO, 2006b) stresses the importance of including sensitivity and uncertainty analyses, but does not provide clear guidelines how to perform these uncertainty analyses.

Uncertainties in data and the model propagate into uncertainties in the LCA results. Tools to handle uncertainties are abundant, and include statistical modelling, sensitivity analysis and scenario analysis (Heijungs and Huijbregts, 2004; Huijbregts, 1998a). Uncertainty analysis was not regularly performed in LCAs in the past (Björklund, 2002; Huijbregts et al., 2001; Ross et al., 2002), despite the availability of uncertainty assessment tools, but the number of LCAs including uncertainty analysis is increasing (Finnveden et al., 2009). Inclusion of uncertainty in LCA results can provide important information when these results are used in decision making, and should then be an integral part of LCA (Ciroth et al., 2004; Finnveden et al., 2009; Geisler et al., 2005; Heijungs and Huijbregts, 2004; Huijbregts, 1998a; Notten and Petrie, 2003; Ross et al., 2013).

Data and parameter uncertainty are the most addressed uncertainties (Finnveden et al., 2009; Heijungs and Huijbregts, 2004; Lloyd and Ries, 2007; Ross et al., 2002). Ways to handle data and parameter uncertainty include quantitative methods (scenarios, stochastic modelling, Monte Carlo analysis, Latin Hypercube sampling, uncertainty propagation, Taylor series, Bayesian analysis, Fuzzy set theory), qualitative methods (data quality indicators), or a combination of both methods (Coulon et al., 1997; Finnveden et al., 2009; Heijungs and Huijbregts, 2004; Huijbregts, 1998b; Lloyd and Ries, 2007; Zamagni et al., 2008). The variability within data sets is often included by means of Monte Carlo analysis (Hung and Ma, 2009; Lo et al., 2005; Sonnemann et al., 2003).

Individual data entries within a date set are often correlated to each other, for instance the relationship between energy sources and their emissions. The variability of individual data entries can be expressed by statistical means, such as minimum and maximum values, average values and confidence intervals, or probability density functions. The use of these statistical values often leads to a loss of the correlation among the separate data entries.

These correlations are furthermore usually poorly known and hardly included (Lloyd and Ries, 2007). Disregarding these correlations may lead to over-or underestimation of the uncertainty in the LCA results (Refsgaard et al., 2007).

The availability of (commercial) data and data sets from companies, organisations and institutions continues to increase. Hence, multiple data sets exist representing a similar process. The influence of the use of different data sets on the LCA results can be significant, and underpins the use of sensitivity analyses on these data sets (Peereboom et al., 1998). The variability among data sets is only sporadically explored, even though its influence can be higher compared to the effect of data variability within data sets (Steinmann et al., 2014). The inclusion of this variability among data sets could thus produce valuable information to the decision makers. A clear-cut approach to handle this variability and translate it into an uncertainty range in the results could be an asset in uncertainty management in LCA.

Uncertainty due to choices is usually handled by scenario analysis. Scenarios are used to address assumptions in the representation of the life cycle of the product (processes, waste treatments, simplifications), or methodological issues (system boundaries, allocation). Scenarios are also used to explore future trends or changes in the life cycle of a product. The results of scenario analyses are normally presented as point values or stack diagrams. Each point value or stack diagram reflects only one specific situation, contrary to the value ranges produced in the statistical methods.

The handling of several types of uncertainties at the same time is only sporadically performed in LCA (e.g. Huijbregts et al. (2003), van Zelm and Huijbregts (2013)). The management of both the variability among data sets and the inclusion of choices is innovative in LCA. There are no ready-made methods in LCA to actually include combinations of uncertainties, and its usefulness is unknown in LCA. In order to find answers to these questions, we take a closer look at integrated assessment, as they experience similar challenges in uncertainty management.

Uncertainty in integrated assessment

Integrated assessment experiences, similar to LCA, uncertainty problems and discussion on uncertainty. The terminology and classification for uncertainties are also inconsistent (Maxim and van der Sluijs, 2011; Walker et al., 2003). Van Asselt and Rotmans (2002) and van Asselt et al. (2001), e.g., distinguish two types of uncertainties: variability and limited knowledge. They furthermore make a distinction in uncertainties in the model quantities (parameters and inputs), the form of the model (structure and relationships), and the completeness and adequacy of the model (system boundaries, representation). Funtowicz and Ravetz (1993), on the other hand, classify uncertainty on the technical, methodological and epistemological level. Walker et al. (2003) added a third dimension of uncertainty. They make a distinction in: the location of uncertainty (where the uncertainty occurs), the level of uncertainty (from determinism to ignorance), and the nature of uncertainty (imperfect knowledge or inherent variability). Van der Sluijs et al. (2003) additionally add the qualification of the knowledge base (from weak to strong) and the value-ladenness of choices (from small to large). Potting et al. (2002) combine classifications of uncertainty types and locations from several authors and apply these to both the socio-economic system and the natural system. They additionally make a distinction between past and future systems.

Framework and guidance on the use of uncertainty in integrated assessments are provided by e.g. Funtowicz and Ravetz (1990), van Asselt et al. (2001), van Aardenne (2002), van Asselt and Rotmans (2002), van der Sluijs et al. (2005), and Gabbert et al. (2010). Tools to manage uncertainty include similar tools as for LCA, but also additional ones on the inclusion of stakeholders and the use of an uncertainty matrix (Gabbert et al., 2010; Refsgaard et al., 2007; van der Sluijs et al., 2005; Walker et al., 2003). Uncertainty problems and the uncertainty management in LCA seem to resemble those in integrated assessment.

The handling of several types of uncertainties at the same time (ensemble modelling) is performed by e.g. van Loon et al. (2007). Results for ensemble modelling in integrated assessment are proven to be more robust compared to results from separate models (Delle Monache and Stull, 2003; van Loon et al., 2007). The simultaneous handling of several types of uncertainties in LCA might thus also provide more robust results, since uncertainty management in LCA has the same features as in integrated assessment. We might learn from the experiences of ensemble modelling in integrated assessment. This thesis explores whether simultaneous handling of variability among data sets and the inclusion of choice provides a meaningful contribution to the robustness of LCA results.

1.3 Research questions and research approach

This thesis focuses on the robustness of LCA results. A robust finding is "one that holds under a variety of approaches, methods, models, and assumptions and one that is expected to be relatively unaffected by uncertainties. Robust findings should be insensitive to most known uncertainties, but may break down in the presence of surprises" (IPCC, 2001). A first step to produce robust LCA results is the identification of uncertainty sources influencing these results, i.e. the locations and/or the types of uncertainties. The next step is the integration of uncertainty management into the performance of an LCA. Inclusion of uncertainty management in LCA results can make the results more robust, and consequently increases the trustworthiness and thereby the usefulness of LCA results in decision making.

The aim of this thesis is ***to evaluate whether the use of multiple data sets and multiple modelling options can increase the robustness of LCA results.***

The aim of the thesis is addressed through three research questions:

1) What are reasons for differences in LCA results for the same product?
2) Can the use of multiple data sets for a process increase the robustness of LCA results?
3) Can the inclusion of multiple modelling options increase the robustness of LCA results?

These questions will be addressed on the basis of case studies for disposable beverage cups and aluminium beverage cans.

Parameter and data uncertainty are the most considered locations of uncertainty (Finnveden et al., 2009; Heijungs and Huijbregts, 2004; Lloyd and Ries, 2007; Ross et al., 2002). The variability of data ***within*** a certain data set is often considered by means of scenarios or Monte Carlo analysis (Hung and Ma, 2009; Lo et al., 2005; Sonnemann et al., 2003). This thesis, however, addresses the variability ***among*** data sets from different databases. This variability is only sporadically explored, and its influence can be higher compared to the effect of data variability within data sets (Steinmann et al., 2014).

This thesis furthermore aims to combine both variability among data sets and modelling choices simultaneously. The simultaneous handling of several types of uncertainties is only occasionally performed in LCA (e.g. Huijbregts et al. (2003), and van Zelm and Huijbregts (2013)). The simultaneous use of multiple data sets and modelling choices is thus novel within LCA. This thesis explores a new approach to handle this combination of uncertainty modelling in LCA, and explores whether it provides a meaningful contribution to the robustness of LCA results. The research is performed by means of case studies on disposable beverage cups and aluminium cans.

To answer the first research question I evaluated existing (comparative) LCA studies for disposable beverage cups (Chapter 2). I identified sources for differences in LCA results for cups made from the same material. These sources include the properties of the cups itself, but also choices made in the presentation of the life cycle of the cups and methodological choices. To answer the second question I explored a new method in LCA including multiple data sets from different databases, and show the effect of the variability among the data sets on the LCA results (Chapter 3). This method was illustrated using disposable polystyrene (PS) beverage cups (Chapter 3). The multiple data sets method was next applied and evaluated in a comparative LCA of disposable PS, polylactic acid (PLA), and paper beverage cups (Chapter 4). To answer the third question I addressed two modelling choices: the modelling of the product life cycle, and the underlying modelling philosophy. The effect of different waste treatments (a model choice in the product life cycle) on the LCA results for the PS cup is included in Chapter 3, and for the three disposable cups (PS, PLA and paper) in Chapter 4. I touched upon methodological modelling choices (underlying modelling philosophy) in the handling of recycling in Chapters 3 and 4. The method provided in Chapter 3 thus captures the simultaneous incorporating of variability from multiple data sets and the choices of modelling approaches. I examined the effect of different methods to handle recycling in LCA in more detail in Chapter 5. I illustrated this effect by means of two different products, i.e. a disposable PS beverage cup and an aluminium can. Chapter 5 thus also answers part of question three. Finally, I evaluated and discussed the method to include multiple data sets and the modelling choices as an uncertainty analysis tool in LCA and its usefulness in uncertainty management (Chapter 6). Table 1.1 presents the relationship between the research questions and the chapters covering the questions, and the links to the case studies. This thesis thus concentrates on uncertainties in LCA results due to variability of available data sets, choices in the modelling of the product life cycle, and choices in the underlying modelling philosophies.

Table 1.1: Interaction between the research questions, the chapters, and the case studies.

<table>
<tr><th>Research question:</th><th>Polystyrene cups</th><th>Polylactic-acid cups</th><th>Paper cups</th><th>Aluminium cans</th></tr>
<tr><td>1. What are reasons for differences in LCA results for the same product?</td><td colspan="3">Chapter 2</td><td></td></tr>
<tr><td>2. Can the use of multiple data sets for a process increase the robustness of LCA results?</td><td>Chapters 3 and 4</td><td colspan="2">Chapter 4</td><td></td></tr>
<tr><td>3. Can the inclusion of multiple modelling options increase the robustness of LCA results?</td><td colspan="4"></td></tr>
<tr><td>a) Included waste treatments options (choices in representation of the life cycle)</td><td>Chapters 3 and 4</td><td colspan="2">Chapter 4</td><td></td></tr>
<tr><td>b) Options in handling recycling (choice in modelling philosophical)</td><td>Chapter 5</td><td colspan="2"></td><td>Chapter 5</td></tr>
</table>

1.4 Uncertainty aspects covered in this thesis

The previous sections provided a diversity of uncertainty terminology, locations where uncertainty can occur, and types of uncertainty. This section presents types and locations of uncertainties as used in this thesis, and presents examples of locations and their relationship to uncertainty types. It also shows the location and type of uncertainties addressed in this thesis.

This thesis distinguishes three types of uncertainties and two main locations where uncertainties can occur, based on Walker et al. (2003), van der Sluijs et al. (2003), and Potting et al. (2002) (see Table 1.2).

The type of uncertainty refers in this thesis to the nature of the uncertainty. The three types of uncertainties are:

1) variability
2) choices
3) unreliability.

Passive ignorance (i.e. we don't know what we don't know) is not included. Active ignorance (i.e. deliberately ignoring aspects due to limited knowledge or lack of care to understand something) is included in choices and unreliability.

Variability refers in this thesis to observable variation as a result of natural randomness or heterogeneity. Variability is not related to any knowledge, but a reflection of the real world (van Asselt and Rotmans, 2002; van der Sluijs et al., 2003; Walker et al., 2003). Variability in data for processes which produce the same or similar materials or products, for example, occurs due to different production techniques, efficiencies, waste treatments, or legal requirements.

Choices refer in this thesis to the normative choices which are taken by the stakeholders and/or LCA practitioners. This meaning is comparable to the normative choices as part of the 'value-ladenness' in van der Sluijs et al. (2003) and Potting et al. (2002), and includes the 'value diversity' by van Asselt and Rotmans (2002). Choices are unavoidable in LCA, and often there exists more than one correct choice. Choices are subjective decisions based on the goal of the LCA, the stakeholder's interest, knowledge of the subject, etc. As such, choices contain a certain type of interest, ethics, biases, preferences, customs, regulations, ignorance, or assumptions (van der Sluijs et al., 2003). Choices differ from variability and unreliability because stakeholders have an active role and can make normative decisions which reflect their behaviour. Both variability and unreliability of data have a scientific and technical element, opposed to the subjective nature of choices. Stakeholders and LCA practitioner cannot influence the variability of inventory data, but do decide which data or data sets to use in the assessment.

All other types of uncertainties are bundles under the type unreliability. Unreliability is "the level of confidence in the state-of-the-art knowledge that is facilitated by using well-accepted methods or measuring equipment and/or by following well-accepted protocols in applying those methods or equipment" (Funtowicz and Ravetz, 1990; van der Sluijs et al., 2005). Uncertainty refers in this thesis to the inaccurate, inexact, and unrepresentative depiction of data or a model. Unreliability can stem from imprecise measurements, measurement errors, lack of data, lack of knowledge, or ignorance. Unreliability is usually measurable and stems from well-understood processes (van Asselt and Rotmans, 2002).

This thesis distinguishes between uncertainty in the model and in the data. A model is a representation of the real world. An LCA makes use of a model to depict the assumed life cycle of the examined product, i.e. the modelling of the product life cycle. A model is also used to translate, explain, and calculate the causal relationships between substances and their effect on the environment, i.e. the modelling of the environmental impacts. The LCA modelling practice itself follows from a certain methodological approach, hence there is a choice of using a certain modelling concept, i.e. the underlying modelling philosophy. The uncertainty location “data” includes inventory data (i.e. values for process data and parameters), and available databases (i.e. databases with ready to use data sets of processes). The number of available databases has increases through the accessibility of data from organisations, institutions, and via databases in LCA software programs (e.g. ecoinvent (Ecoinvent Centre, 2010), ELCD (2008), USLCI (NREL, 2011)).

Table 1.2 contains a number of identified locations where uncertainty occurs, and presents the relationship between the location and the type of uncertainty. An existing relationship is denoted by a circle (●). This thesis addresses and discusses several uncertainty locations and types (light grey boxes in Table 1.2). The main focus of this thesis covers the choices in the representation of a process (waste treatment), the choices in allocation modelling (in recycling), and the variability among data sets. The dark grey boxes in Table 1.2 represent these research questions.

1.5 Outline of the thesis

The first chapter is an introduction to the thesis. The chapter contains the aim of the thesis and the research questions, and explains the uncertainty concept of LCA.

Chapter 2 presents the analysis of ten existing LCAs for disposable beverage cups. The study compared quantitative results of existing LCAs as to examine the consistency and robustness of the results. Modelling choices and used data sources of each study were evaluated and their influence on the LCA results. This led to the identification of possible sources for discrepancies in LCA results. The influence of selected sources was explored in the next chapters.

Chapter 3 describes a novel method to include the variability among data sets and the uncertainty due to modelling choices in the LCA results. The method is applied on a case study of a disposable polystyrene beverage cup. The study purposely used different data sets from various databases/sources for processes with an influential contribution to the LCA results. The study included two waste treatments (incineration and recycling), and again purposely used different data sources for these waste treatments. This variability in data sets from different databases/sources is presented as a spread in the LCA results.

Chapter 4 presents the results of a comparative LCA for three disposable beverage cups, made from polystyrene, polylactic acid, and paper. The method described in the previous chapter was applied in this comparative study. The study included multiple data sets for influential processes, and multiple waste treatment scenarios. The results identified the main contributing processes in the life cycles of the three cups, and the spread in these results represented the variability in the processes. The overlap in the results among the three cups is valuable information for decision makers. The results show the usefulness of the method in a comparative setting.

Chapter 5 describes the evaluation of six methods which are available and used to model recycling in LCA. The study compared the philosophies and perspectives behind these recycling modelling methods. The six methods were applied and compared on a disposable polystyrene beverage cup and an aluminium beverage can, assuming hypothetical circumstances. Next, two recycling modelling methods were again applied on the polystyrene cup and aluminium can, but now according to actual waste management practices in Europe. The results show the influence of the selected recycling modelling method on the LCA results.

Chapter 6 is the synthesis of this thesis. The chapter discusses the new method from Chapter 3 and its usefulness in uncertainty management. It also debates the implications of the multiple modelling methods for recycling and the consequences for standardisation of the LCA method. The chapter places the context of the previous chapters in a broader perspective, and ends with conclusions for this thesis.

Table 1.2: Types of uncertainties in the data and model used in LCAs. See text for definitions of variability, choices, and unreliability. ● denotes a relationship. The light grey cells refer to the types and locations of uncertainties addressed in this thesis. The dark grey cells refer to the main research questions.

Location of uncertainty	Type of uncertainty: *Variability*	*Choices*	*Unreliability*
Model			
Modelling the product life cycle			
- System boundaries		●	●
- Functional unit		●	
- Reference flows		●	
- Included/omitted life cycle phases		●	
- Representation of product or processes	●	●	●
- Simplification		●	●
Underlying modelling philosophy			
- Sustainability viewpoint	●	●	
- Perspectives	●	●	
- ISO-compliance		●	
- Allocation methods		●	
- Impact categories		●	
- Attributional or consequential		●	
- Cut-off rules		●	
Modelling environmental impacts			
- System boundary		●	●
- Inclusion or emission substances		●	
- Spatial difference	●		
- Temporal difference	●		
- Life time of substances			●
- Linear instead of non-linear modelling			●
- Relative contribution of substances and pollutants to impacts			●
- Characterisation factors		●	
Data			
Inventory data			
- Lack of (representative) data			●
- Inaccurate data			●
- Unreliable data			●
- Measurement errors			●
- Spatial difference	●		
- Temporal difference	●		
- Differences in factories performing the same process	●		
- Value for parameter	●	●	
- Marginal or average data		●	
Available databases			
- Data for similar process in different databases	●	●	
- Site-specific or average data		●	

2 A critical comparison of ten disposable cup LCAs

Abstract

Disposable cups can be made from conventional petro-plastics, bioplastics, or paperboard (coated with petro-plastics or bioplastics). This study compared ten life cycle assessment (LCA) studies of disposable cups with the aim to evaluate the robustness of their results. The selected studies have only one impact category in common, namely climate change with global warming potential (GWP) as its category indicator. Quantitative GWP results of the studies were closer examined. GWPs within and across each study show none of the cup materials to be consistently better than the others. Comparison of the absolute GWPs (after correction for the cup volume) also shows no consistent better or worse cup material. An evaluation of the methodological choices and the data sets used in the studies revealed their influence on the GWP. The differences in GWP can be attributed to a multitude of factors, i.e. cup material and weight, production processes, waste processes, allocation options, and data used. These factors basically represent different types of uncertainty. Sensitivity and scenario analyses provided only the influence of one factor at once. A systematic and simultaneous use of sensitivity and scenario analyses could, in a next research, result in more robust outcomes.

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2.1 Introduction

Reusable cups need rinsing and cleaning. This makes them unpractical in situations where on the spot facilities for rinsing and cleaning are absent, limited or inconvenient (e.g. on-the-go consumption, at once-only events, and during peak consumption). Disposable cups are then a convenient alternative to supply hot and cold drinks. They require no maintenance, and are cheap and easy to use. Disposable cups are typically used by take-away shops and restaurants, at happenings and parties, in vending machines on schools, factories or in offices.

Most disposable cups are made of petro-plastic or paperboard. Petro-plastics are produced from fossil fuels (i.e. oil and natural gas). Paperboard cups mainly consist of paperboard, but they are coated with a thin layer of plastic, usually petro-plastic. This plastic prevents liquid from intruding into the paperboard. Particularly petro-plastic cups are frequently associated with an unnecessary use of limited resources and superfluous production of waste.

As alternatives to petro-plastic cups, there are nowadays biopaper and bioplastic cups available. Bioplastics for disposable cups are produced from renewable resources (i.e. plant material), and are biodegradable. Bioplastics cups do exist already for cold drinks, but they are as yet thermo-instable, which makes them unsuitable for hot drinks. Introduction of these bioplastic cups for hot drinks is expected soon (Wageningen UR, 2012). Biopaper cups, i.e. paperboard cups with a bioplastic coating, are applicable for both cold and hot drinks.

Bioplastic and biopaper cups are often perceived as more environmental sustainable than their petro-plastic alternatives (Butijn et al., 2013). This is not unambiguously confirmed though by comparative studies with life cycle assessment (LCA). LCA is a method to evaluate the environmental performance of products throughout their life cycle, i.e. from resource extraction up to and including waste processing (see Figure 2.1) (ISO, 2006a, b).

The aim of this paper is to evaluate the consistency and robustness of a number of recent LCA studies for disposable cups. The outcomes from these studies are compared qualitatively and quantitatively. The comparison focused on the global warming potential (GWP) as this was the only common category indicator across the studies. Differences between the studies in methodology and data used are identified and evaluated in relation to the differences in GWP results of the LCA studies.

2.2 Methods and means

2.2.1 Life cycle Assessment

The period 1990 to 2000 showed strong activity to harmonize and standardize the LCA framework, terminology and methodology (Guinée et al., 2010). The Society of Environmental Toxicology and Chemistry (SETAC) and International Organization for Standardization (ISO) played a key role in this process. Several international standards for the methods and procedures used in LCA have meanwhile been produced, such as ISO 14040 and 14044 (ISO, 2006a, b). Building on ISO 14044 (2006a, b), the Joint Research Centre of the European Commission (JRC-IES) recently published the ILCD handbook and a Product Environmental Footprint aiming to harmonize LCA methodology and therewith comparability of LCA results in European context (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010, 2012).

LCA is meanwhile a well-established method for assessing the potential environmental impact associated with a product or service system. An LCA consists of four main phases (ISO, 2006a):

1) Goal and scope definition describes the objective of the study (goal), and the methodological approach used (scope).
2) Inventory analysis examines the processes in the product system and quantifies for each process all the input and output data (i.e. inventory data). Economic inventory data include the amount of resources, materials and energy needed to manufacture a product. Environmental inventory data include all extracted natural resources which are used in the process (inputs), and emissions and waste released to the environment (outputs).
3) Impact assessment converts the inventory data into their contribution to environmental impact in one or more impact categories, e.g. global warming potential, resource depletion, eutrophication, toxicity.
4) Interpretation evaluates the results and their robustness from the previous phases and draws conclusions and/or recommendations. This phase also examines the confidence and reliability of the LCA results through sensitivity, scenario and/or uncertainty analyses.

Although the LCA method is standardized, there is still room for a range of methodological choices (ISO, 2006a). The influence of these methodological choices and differences in data sources on the outcomes of an LCA was evaluated for a selection of LCA studies focusing on disposable cups.

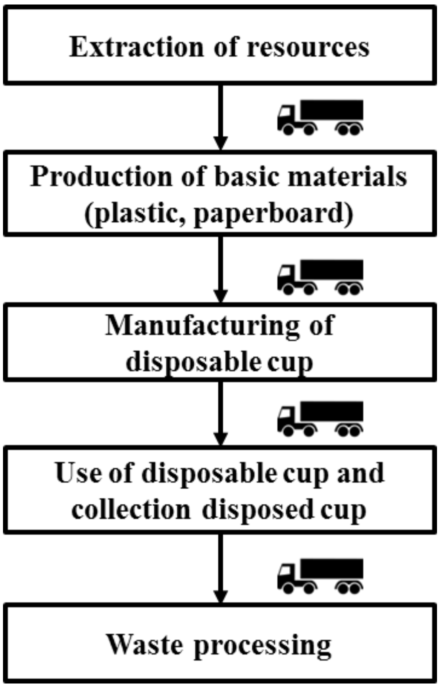

Figure 2.1: Simplified life cycle flow of disposable cups.

2.2.2 Selected studies

Ten LCA studies on disposable cups were collected from scientific journals (three articles) and via internet (seven reports; see Table 2.1). Each report is publicly available from renowned companies. All collected LCA studies are peer-reviewed, published in 2000 or later, include at least one type of disposable cup, encompass the whole life cycle of the cup(s), and cover at least the global warming potential (GWP) as an impact indicator. LCA studies not complying with those criteria have been excluded from our review.

The goal of the selected LCA studies on disposable cups is defined by the involved researchers and stakeholders. All studies included a comparison between two or more disposable cups or between disposable cups and other drinking systems. The goal of the study was diverse. For several studies the goal was to evaluate and compare the environmental impact between cups or drinking systems in order to find the most beneficial cup/system (Franklin Associates, 2006; Garrido and Alvarez del Castillo, 2007; Ligthart and Ansems, 2007; Pladerer et al., 2008; Uihlein et al., 2008; Vercalsteren et al., 2006). The goal of other studies was to benchmark a specific cup against other cups, and to quantify the

Table 2.1: Ordinal and relative ranking of disposable cups according to their global warming potential (GWP) results in comparative LCAs. The cup with the lowest GWP is awarded the number 1 and a relative percentage of 100% (between brackets behind the ranking; 100% is not mentioned in the table). Cups with (almost) the same GWP (difference less than 5%) are considered equal. Several studies included more than one comparison.

No.	Study	Petro-plastic[a]				Bio[a] plastic	Paperboard lined with[a]			Comments [b]
		(HI)PS	EPS	PP	PET & RPET	PLA	PE	PLA	wax	
1	Franklin Associates (2006)	3 (167)		1	4 (208)	2 (148)				
2	Franklin Associates (2009a)		1 (102)		1		1 (104)			RPET burden excludes virgin material processing
			1		2 (110)		1 (102)			RPET burden based on open-loop principle
3	Franklin Associates (2011)		1				3 (137)	2 (127)		Maximum decomposition PB in landfill
			3 (790)				2 (161)	1		No decomposition PB in landfill
			1			1 (104-108)	2 (119)		3 (214)	Maximum decomposition PB in landfill; Two weights for PLA cups
			3 (915)			4 (919-992)	1		2 (129)	No decomposition PB in landfill; Two weights for PLA cups
4	Garrido and Alvarez del Castillo (2007)									No comparison, PP is only disposable cups
5	Häkkinen and Vares (2010)				1			1 (103)		Landfilling of cups
					2 (715)			1		Incineration of cups
6	PE Americas (2009) [c]			2 (133)	3 (225)	1				Two weights for PP cup and PLA cup; Two cup manufacturing processes PLA cup
7	Ligthart and Ansems (2007)	2 (339)					1			
8	Pladerer et al. (2008) [c]	4 (380)			3 (220)	2 (155)	1			Incineration PET and PLA cups
		4 (380)			2 (145)	3 (195)	1			Incineration and recycling of PET cups, composting of PLA cups
9	Uihlein et al. (2008) [d]	2				1				
10	Vercalsteren et al. (2006)			3 (171)		2 (153)	1			Small events, incineration of PP cups
				2 (192)		3 (220)	1			Large events , incineration and fuel substitution PP cups

[a] HIPS = high impact polystyrene, EPS = expanded polystyrene, PP = poly propylene, PET = poly ethylene terephthalate, RPET = recycled PET, PLA = polylactic acid, PE = poly ethylene, PB = paperboard.
[b] The numbers before the brackets refer to the ordinal ranking and the numbers between the brackets to the relative percentages.
[c] Percentages estimated from histograms.
[d] Percentages could not be calculated, no absolute data available.

environmental impact of a specific cup with new, improved or different technology or waste options (Franklin Associates, 2009a, 2011; Häkkinen and Vares, 2010; PE Americas, 2009). The goal of an LCA influences among others the methodological choices, system boundaries and used data (e.g. company specific data versus generic or secondary data), and thus influences the GWP results.

2.2.3 Review approach

Figure 2.1 depicts a simplified life cycle flow or product system for disposable cups. The cycle starts with the extraction of natural resources for the production of basic materials (petro-plastic, bioplastic, or paperboard). These materials are used in the manufacturing of the disposable cups. Next, disposable cups are transported, often via a distributor, to the user of the cup. The user disposes the cup in a waste recipient after consumption of a drink. Several waste processing options are available for the cups, e.g. landfilling, incineration, recycling, and composting (depending on the properties of the cups).

The scope definition in the studies specified the used methodological approach. It includes the definition of the functional unit (the function of the studied product system), the reference flow, system boundaries, cut-off criteria, inventory data collection, allocation principles, and impact categories. Choices, such as alternative waste processing options or selected impact categories, are also incorporated in the scope definition. The last LCA phase, interpretation, can include a confidence or reliability check of the results.

We analysed the LCA studies on their characteristics in each of the four LCA phases:

- Goal and scope definition: cup properties (hot/cold beverage cups, cup size, cup material), included life cycle processes, waste processing option, cut-off rules, allocation principles
- Inventory analysis: geography of the data sources, reported inventory data
- Impact assessment: reported impact indicators
- Interpretation: reliability of the LCA results based on inclusion of multiple cup systems, sensitivity or scenario analysis, statistical uncertainty analysis.

See Table 2.2 for all included characteristics (and how they were included in the studies).

Table 2.2: Characteristics of LCA studies on disposable cups.

Study number	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Source	**Franklin Associates (2006)**	**Franklin Associates (2009a)**	**Franklin Associates (2011)**	**Garrido and Alvarez del Castillo (2007)**	**Häkkinen and Vares (2010)**	**PE Americas (2009)**	**Ligthart and Ansems (2007)**	**Pladerer et al. (2008)**	**Uihlein et al. (2007)**	**Vercalsteren et al. (2006)**
Investigated cup										
Hot beverage cup		X	X				X			
Cold beverage cup	X		X	X	X	X		X	X	X
Cup size										
• 180 ml - 200 ml - 330 ml				X			X		X	X
• 16-oz - 0.5 l	X	X	X			X		X		
• 32-oz			X							
• no size mentioned					X					
Cup material										
• polystyrene (PS, EPS, HIPS)	L/I	L/I	L/I				R	I	I	
• polypropylene (PP)	L/I			L/I		L				I; I/F
• PET and recycled PET (RPET)	L/I	R			L; I	L		I; I/R		
• polylactic acid (PLA)	L/I		L/I			L		I; C	I	I/C
• coated paperboard		L/I	L/I		L; I		I	I		I; I/F
Processes included										
(Raw) material extraction and production	X	X	X	X	X	X	X	X	X	X
Manufacturing of cup	X	X	X	X	X	X	X	X	X	X
Use and disposal of cup				X			X	X	X	X
Waste processing	X	X	X	X	X	X	X	X	X	X
Transport	X	X	X	X	X	X	X	X	X	X
Cut-off rules										
Weight	X	X			X	X				
Capital goods/infrastructure	X	X			X	X	X			X
Simplification processes	X	X	X			X	X			

Study number	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Source	**Franklin Associates (2006)**	**Franklin Associates (2009a)**	**Franklin Associates (2011)**	**Garrido and Alvarez del Castillo (2007)**	**Häkkinen and Vares (2010)**	**PE Americas (2009)**	**Ligthart and Ansems (2007)**	**Pladerer et al. (2008)**	**Uihlein et al. (2007)**	**Vercalsteren et al. (2006)**
Allocation										
Mass	X	X	X				X			
Economic							X			
System expansion	X	X	X	X	X		X	X	X	X
Geography data sources										
Europe	X			X	X		X	X	X	X
United States	X	X	X			X		X		X
LCI indicators										
Energy use	X	X	X		X				X	
GHG gasses (CO_2, CH_4, N_2O)	X				X					
GHG gasses (CO_2, CH_4, N_2O, CFC's)					X					
Solid waste	X	X	X							
Extended air- and water emissions	X									
Water use			X			X				
LCIA indicators										
Global warming potential/climate change	X	X	X	X	X	X	X	X	X	X
Acidification				X		X	X	X	X	X
Eutrophication				X		X	X	X	X	X
Ozone depletion				X			X	X	X	X
Human toxicity (incl. respiratory effects, carcinogens, heavy metals)				X			X		X	X
Eco toxicity (including pesticides)							X		X	X
Land use									X	
Photo-oxidant formation				X		X	X			
Energy				X		X		X		
Abiotic resource depletion/ fossil fuels							X	X	X	X
EI99 (total)								X	X	

Study number	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Source	**Franklin Associates (2006)**	**Franklin Associates (2009a)**	**Franklin Associates (2011)**	**Garrido and Alvarez del Castillo (2007)**	**Häkkinen and Vares (2010)**	**PE Americas (2009)**	**Ligthart and Ansems (2007)**	**Pladerer et al. (2008)**	**Uihlein et al. (2007)**	**Vercalsteren et al. (2006)**
Reliability LCA results										
Different cup systems included										
• Material properties or production process		X				X				
• Waste process option or conditions			X		X			X		X
• Weight of cup			X			X				
Sensitivity or scenario analysis										
• Energy use, material use, or fabrication process		X				X		X	X	X
• Waste process option or conditions		X	X			X	X	X		X
• Weight of cup							X			X
• Transport distances										X
Statistical uncertainty analysis								X		

X = included in study, x1/x2 = combination of two waste processes in one scenario, x1; x2 = two scenarios with different waste processes
Waste processes: L= landfilling, I = incineration, R = recycling, C = composting, F = fuel substitute

The quantitative comparison focused out of necessity on GWP alone. GWP is namely the only indicator covered by all included studies (see Table 2.2). First, we ranked ordinal, within each study, GWP results for the different disposable cup systems from the lowest to highest result. We estimated GWPs from histograms whenever exact numbers were not given. Next, to enable a quantitative comparison between the cup systems across studies, we resized the GWPs for all cups to the volume of the most used cup, i.e. a volume of 16 ounce (473 ml). Ligthart and Ansems (2007) and Franklin Associates (2011) report a proportional increase in GWP to the weight of the cups. For simplification purposes, we assumed a proportional increase in GWP to the volume of the cups. This simplification is justified by the relation between the volume and weight of cups. While it limits the accuracy of the GWP results, it does provide an adequate indication of the GWP. Then, we compared the GWPs of all cup systems from all studies to each other.

Next, we evaluated the potential relation between differences in GWP results and the listed characteristics for each study. Several studies included multiple scenarios for the same cup, which gave the opportunity to analyse the influence of specific changes or assumptions to the GWP.

2.3 Results

2.3.1 Cup systems in selected articles

The ten selected LCA articles and reports include cold and hot cups (see Tables 2.1 and 2.2). Two studies examined only hot cups (Franklin Associates, 2009a; Ligthart and Ansems, 2007), seven studies examined only cold cups (Franklin Associates, 2006; Garrido and Alvarez del Castillo, 2007; Häkkinen and Vares, 2010; PE Americas, 2009; Pladerer et al., 2008; Uihlein et al., 2008; Vercalsteren et al., 2006), and one study included both hot and cold cups (Franklin Associates, 2011).

Raw materials for the petro-plastic cups were polystyrene (PS), high impact PS (HIPS), expanded PS (EPS), polyethylene terephthalate (PET), recycled PET (RPET), and polypropylene (PP). The studies include only one bioplastic, polylactic acid (PLA). Paperboard cups are lined with either (low density) polyethylene ((LD)PE), PLA or wax.

All studies examined at least one cradle-to-grave cup system (i.e. one cup material and one waste processing option for one cup weight; reference flow). Most studies incorporated additional cup systems with either alternative cup materials or production properties

(Franklin Associates, 2009a; PE Americas, 2009), alternative waste processes (Franklin Associates, 2011; Häkkinen and Vares, 2010; Pladerer et al., 2008; Vercalsteren et al., 2006), or various weights of the cups (Franklin Associates, 2011; PE Americas, 2009) (see also Tables 2.1 and 2.2). The results from these additional cup systems were also included in our study as cup systems on their own. This means that several studies provided more than one result per cup material, but each with different settings or conditions for the cup systems. These studies, moreover, provided valuable information on the influence of these differences on the GWP results.

The GWP results of all cup systems from all studies were used in the ranking of the cups and the comparison of their absolute values, as described in the following paragraphs.

2.3.2 Ordinal ranking on GWP of cup systems within LCA studies

The cup systems were ranked in ordinal order according to the GWP results within each of the ten studies (see Table 2.1). Based on GWP only, no cup material ranks consistently better than other cup materials in all studies (see Table 2.1). No cup material can be labelled as the most environmentally friendly one. The studies nevertheless report several consistent outcomes in the ordinal ranking of the different cup materials.

Cups made from paperboard rank on GWP at least equally compared to R(PET) cup systems (Franklin Associates, 2009a; Häkkinen and Vares, 2010; Pladerer et al., 2008). The paperboard cup has a lower GWP impact compared to the PS cup (Ligthart and Ansems, 2007; Pladerer et al., 2008). The PLA cup shows a lower GWP compared to the (HI)PS cup (Franklin Associates, 2006; Pladerer et al., 2008; Uihlein et al., 2008).

The studies provide also contradictory GWP ranking for the materials used in the cup systems. These are most evident in the comparison between RPET and EPS cups (Franklin Associates, 2009a), PET and (HI)PS cups (Franklin Associates, 2006; Pladerer et al., 2008), PLA and PET cups (Pladerer et al., 2008), PLA and PP cups (Vercalsteren et al., 2006), paperboard and EPS cups (Franklin Associates, 2011), paperboard and RPET cups (Franklin Associates, 2009a), and paperboard and PLA cups (Franklin Associates, 2011).

The GWPs of the cups were also ranked within their own material group, i.e. paperboard and petro-plastic. The bioplastic group contains only one material, PLA, hence no evaluation within this material group is possible.

Paperboard cups can be lined with (LD)PE, PLA or wax. Only Franklin Associates (2011) covered more than one paperboard cups system. They rated the GWP of hot paperboard cups with PLA lining above the PE lined ones, and the cold paperboard cups with PE lining above the wax lined cups.

The ranking on materials of the cup systems within the petro-plastic group is not consistent across studies. Franklin Associates (2006) reported a lower GWP for PS cup systems compared to PET ones, but Pladerer (2008) on the other hand judged PET cups to be environmentally friendlier than PS cups. According to Franklin Associates (2009a), EPS cup systems have an equal or slightly lower GWP compared to RPET cup systems, depending on the used allocation method for the recycled material in two different life cycles for RPET cups (Franklin Associates, 2009a). Both Franklin Associates (2006) and PE Americas (2009) showed a lower GWP for PP cups compared to PET cups.

In summary, no cup material or cup system ranks consistently as most environmental friendly one. The ranking of the cups between studies is not consistent. Reasons for these differences could be due to differences between one or more characteristics as mentioned in the comments in Table 2.2.

2.3.3 Comparison of absolute GWP results across all studies

The GWP results from all cup systems were recalculated to a cup size of 16 oz (473 ml), the most used cup size in the reviewed studies, in order to enable comparison. Garrido and Alvarez del Castillo (2007) and Uihlein et al. (2008) provided no absolute GWP data, and were excluded from this comparison. Häkkinen and Vares (2010) did not mention a cup size, we assumed the most used size of 16 oz. A total 42 GWP results were in this way derived: 18 petro-plastic cup systems, 9 PLA cup systems, and 15 paperboard cup systems.

Figure 2.2 shows the GWP results for all cup systems in all studies (from the lowest to the highest value). Figure 2.2 does not show a distinct area for cup systems within each material group, i.e. neither paperboard nor bioplastic or petro-plastic scores consistently best or worst. GWP results for the cup systems within each of the three materials are scattered over the graph and prevent a deduction on the preferred cup material.

The variation in GWP results among all cup systems is great. The ratio between the highest and the lowest GWP score for all cup systems is 30. The ratio is 3.4 within the petro-plastic group, 1.7 within PLA, and nearly 20 within the paperboard group. Especially the variation in the GWP of the paperboard cups is very high.

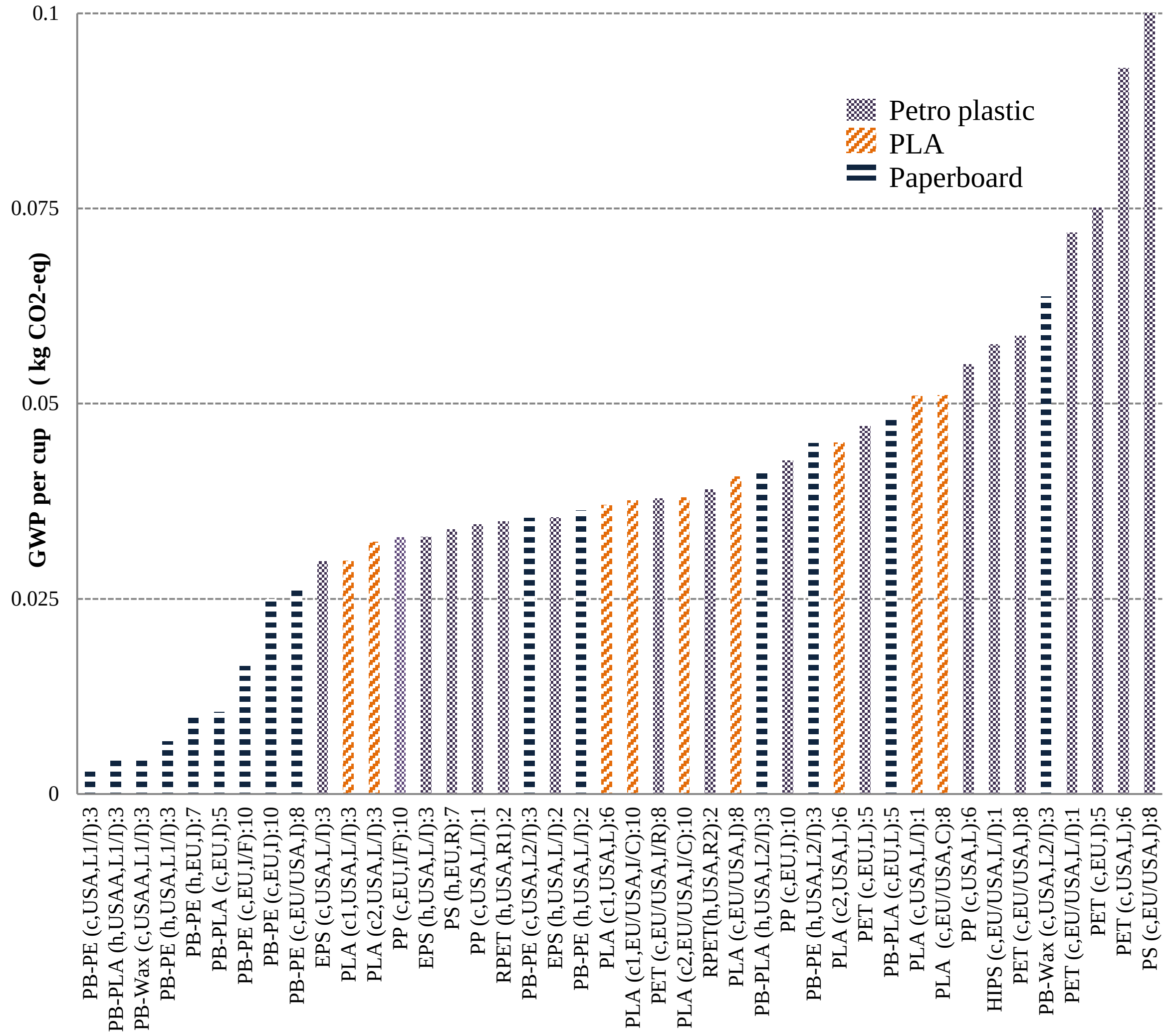

Figure 2.2: GWP (in kg CO_2-eq) of disposable cups. The GWP is recalculated to a reference size of 16 oz (473 ml). Each cup has the following notation: Material type (cold or hot cup, location of data sources, waste processing option): study number.

c = cold cup; h = hot cup; EU = European Union; USA = United States of America; L = landfilling; I = incineration (with energy recovery); F = fuel substitute; R = recycling; C = composting. The number 1 or 2 behind the c (cold cup) or waste processing options (R or L) refers to different assumptions used in the study. Study numbers refer to the number of the source as mentioned in Table 2.1.

Based on Figure 2.2, the paperboard cup systems with waste incineration or a combination of landfilling/incineration tend to have the lowest GWP. The paperboard cup systems including landfilling/incineration, but with a different assumption in the degradation rate of the paperboard, also appear though in the middle and the high end of the graph. Similarly, PET cups are situated at the high end of the graph when PET is incinerated or landfilled, but are located in the middle range when PET is recycled or a combination of recycling/incineration is used.

The GWP results in Figure 2.2 do not show a clear distinct section between the different petro-plastic materials. Similarly, different lining materials for the paperboard cups do not show up in distinct areas in Figure 2.2. Other factors than cup material only must play a role in the GWP results. The discrepancies and variations between the outcomes of the studies may be based on differences in data used and methodological choices. This is evaluated in the next section.

2.3.4 Analysis of data used and methodological choices

The ten studies were analysed for differences on several methodological aspects and the influence of those differences on GWP. Table 2.2 summarizes the results for some characteristics of the LCA studies. The functional unit, the included life processes, the cut-off rules, the used data sources, and the reported inventory and impact data are discussed below.

Several studies included multiple cup systems with alternative choices in material properties, production processes, weight of the cups, or waste treatment options. These additional results provided an opportunity to find a relationship between methodological choices and data used on the one hand, and GWP results on the other hand.

Functional unit (FU)

The FU in the studies is either based on the number of included cups (1, 1000, 10,000 or 100,000 cups), or on the amount of cups needed to serve an amount of liquid (100 or 1000 l). The cup size across studies varied considerably from 180 ml up to 946 ml (32 oz), but cup volumes were the same within most studies.

The volume has a major influence on the weight of the cups. The weight of the cups is an influential factor (PE Americas, 2009). The GWP outcomes within studies increased nearly proportional to the weight of the cups across cup systems (Franklin Associates, 2011; Vercalsteren et al., 2006). The weight of the cups also partly depends on the cup material and other properties of the cups. Different properties of the cups, e.g., lining material or waste processing options, therefore also can lead to some spread in the GWP results (see section on Waste processing).

The cup materials have different specific weights, and particularly paperboard cups can be considerably heavier than plastic ones of the same volume. Materials for petro-plastic are divers (PS, HIPS, EPS, PP, PET and RPET) and inhibit different properties, such as density, heating values, clarity (crystalline or amorphous), and insulating capacity. The examined petro-plastic and paperboard cups can be suitable for hot and/or for cold drinks. This differs across studies. The PLA cups can only be used for cold drinks, since thermostable PLA is not yet on the market.

Cups intended for the use of hot drinks can have different properties compared to cups intended for cold drinks. Paperboard cups for cold drinks need both an inner and outer lining, for example, while paperboard cups for hot drinks only need an inner lining (International Paper, 2012).

The weight of the cup turns out to be an influential factor in the GWP results. The weight depends on the type of material(s) used, the volume of the cup and the intended purpose (cold or hot drinks).

Life cycle processes included in the LCAs

The life cycle of the cup systems consists of a sequence of stages. A number of studies provided GWP information for the individual life cycle stages. This made it possible to identify the major and minor contributing phases. The main contributors to GWP are the production of cup materials, the manufacturing of cups, and the waste treatment of cups. Table 2.3 shows the relative contribution of these main life cycle phases to the GWP results. Some of the processes are highlighted below.

The contribution to GWP from transport is small in most studies. Only Vercalsteren et al. (2006) showed up to 15% contribution in the PLA cup, mainly due to the distance of 850 km from the distributor to the event site.

PE Americas (2009) modelled the manufacturing processes of the PLA cup after two different processes: according to PET cups and according to PP cups. This was done due to lack of information on the manufacturing process of PLA cups from PLA granulates. The PLA cup which was modelled according to the PET cup showed a 15% increase in GWP compared to the PLA cup modelled according to the PP cup manufacturing process (PE Americas, 2009).

Most studies included all life-cycle process stages (see Figure 2.1). Some studies omitted the use and disposal phase of the cup (Franklin Associates, 2006, 2009a, 2011; Häkkinen and Vares, 2010; PE Americas, 2009). The other studies included the collection medium (bag, container or box) and/or the waste treatment of the packaging material and/or the transport of the disposed cups to the waste facility. Franklin Associates (2006, 2009a, 2011) considered the use phase to be the same for all cups, and excluded this phase. Ligthart and Ansems (2007) reported a 10-15% contribution to GWP of the use and disposal phase, but included the transport of the cups to the waste processor. Other studies reported a minor contribution to GWP of the use and disposal phase.

All studies identified the production of the cup material and the manufacturing of the cup as important contributors to the GWP. The influence of the waste process varies and is discussed in the next subsection.

Waste processing

The disposed cups can enter various waste treatment routes, partly depending on the properties of the cups (notably their biodegradability). Included waste treatment processes for the cups are landfilling, incineration (with energy recovery), their use as a fuel substitute, recycling, composting, and combinations of several waste options. Most studies applied energy credits for avoided production of conventional energy thanks to the recovered energy during incineration. Garrido et al. (2007) did not include credits for the incineration of the cups (15% of the disposed cups were sent to the incinerator; the other 85% were landfilled). Four studies incorporated different waste treatment options for the cups and modelled these as separate cup systems (Franklin Associates, 2011; Häkkinen and Vares, 2010; Pladerer et al., 2008; Vercalsteren et al., 2006). This provided a valuable insight on the influence of the waste processing options on the GWP.

Incineration of PET cups instead of landfilling increased the GWP by 60% (Häkkinen and Vares, 2010). Incineration of PP cups in a cement kiln provided additional credits for avoided GWP, due to avoided use of fossil fuels oil and coal, compared to incineration in a municipal solid waste incinerator (Vercalsteren et al., 2006). Recycling PET cups instead of incineration lowered the GWP by 30% (Pladerer et al., 2008). These results are supported by the relative contribution to GWP of the waste process in the total life-cycle of the cup in Table 2.3. Landfilling of petro-plastics contributes very little to GWP (less than 2%), because petro-plastics do not decompose in landfills. Incineration of petro-plastics contributes 30-40% to GWP due to the net release of fossil carbon dioxide after correction for energy recovery from combustion heat (see Table 2.3). The relative contribution of recycling is not unanimous and depends on the allocation principle of the used material (i.e. the credits for the recycled material). Overall, incineration of petro-plastic cups increases the GWP compared to landfilling, and recycling can decrease the GWP.

The waste options for PLA cups included landfilling, incineration and composting. PLA can be composted in industrial composting plants under the right conditions (temperature 60^{o}C, high humidity, and needs to be mixed with other organic materials) (Greene, 2007; Nielsen and Weidema, 2002). Similar to most petro-plastics, the studies considered PLA an inert material during landfilling (probably due to the short aerobic phase). PLA degrades under anaerobic circumstances only when specific conditions are met (temperature 55^{o}C, high humidity, balanced mixture with other organic materials) (Merrild and Hedal Kløverpris, 2010; Yagi et al., 2009). The carbon uptake from the original biomass is sequestered in PLA and not released back to the environment during landfilling. The studies applied various allocation approaches on the uptake of carbon dioxide in biomass and these are further discussed under the allocation section (Accounting for biogenic carbon dioxide). Incineration of PLA provides GWP credit for the recovered energy, but also releases the carbon dioxide. Franklin Associates (2011) reported a higher credit, and thus a lower GWP, for carbon sequestration in landfilling compared to credits from recovered energy from combusting PLA. Composting PLA instead of incineration led to different results. The GWP of composting PLA was relatively high according to Pladerer et al. (2008), because incineration received credits for the recovered energy and composting did not. Composting, on the other hand, also led to a reduction of GWP in Vercalsteren et al. (2006).

Table 2.3: Relative contribution (%) of the main life cycle phases to the GWP of disposable cups.

Material[a]	MP [b]	CM [b]	MP + CM[b]	Waste treatment[c]		CO_2 mod[d]	Study
PP	60	36	96	<2	Landfill		PE Americas (2009)
PET			100	0	Landfill		Häkkinen and Vares (2010)
PET	68	29	97	<2	Landfill		PE Americas (2009)
EPS			94	6	80% landfill and 20% incineration		Franklin Associates (2009a)
EPS			94	6	80% landfill and 20% incineration		Franklin Associates (2011)
PS			67	33	Incineration		Pladerer et al. (2008)
PP	33	10	43	40	Incineration		Vercalsteren et al. (2006)
PET			63	37	Incineration		Häkkinen and Vares (2010)
PET			70	30	Incineration		Pladerer et al. (2008)
PET			105	-20	Incineration/ recycling		Pladerer et al. (2008)
PS	92	17	109	-33	Recycling		Ligthart and Ansems (2007)
RPET			94-98	6	Recycling		Franklin Associates (2009a)
PLA	32-38	60-68	98	<2	Landfill	A	PE Americas (2009)
PLA			186	-86	80% landfill and 20% incineration	N	Franklin Associates (2011)
PLA			115	-20	Incineration	N	Pladerer et al. (2008)
PLA	45	15	60	20	50% incineration and 50% composting	A	Vercalsteren et al. (2006)
PLA			90	7	composting	N	Pladerer et al. (2008)
PB			22	77	Landfill	N	Häkkinen and Vares (2010)
PB			86	14	80% landfill and 20% incineration	N	Franklin Associates (2009a)
PB			47-53	47-53	80% landfill and 20% incineration; 100% decomposition PB in landfill	N	Franklin Associates (2011)
PB			87-97	3-13	80% landfill and 20% incineration; 50% decomposition PB in landfill	N	Franklin Associates (2011)
PB			340-700	-240 to -600	80% landfill and 20% incineration; no decomposition PB in landfill	N	Franklin Associates (2011)
PB			104	-4	Incineration	N	Häkkinen and Vares (2010)
PB	110	10	120	-30	Incineration	N	Ligthart and Ansems (2007)
PB			107	-18	Incineration	N	Pladerer et al. (2008)
PB	-45	45	0	60	Incineration	A	Vercalsteren et al. (2006)

[a] See Table 2.1 for explanation of abbreviations
[b] MP = material production, CM = cup manufacturing
[c] Incineration with energy recovery
[d] Modelling approach for biogenic carbon dioxide: N = carbon-neutral, A = accounted for similar as fossil carbon dioxide

Vercalsteren et al. (2006) included sequestered carbon in compost and credited compost for the displacement of plant growing media and soil conditioners. GWP is measured over a timeframe of 100 years. The sequestered carbon in compost will be respired into carbon dioxide within this timeframe, which means that counting for sequestered carbon is actually not correct.

The paperboard cups in the studies were landfilled and/or incinerated. Here, similar to PLA, different approaches are used in the uptake of carbon in wood. These allocation differences are discussed in the allocation section (Accounting for biogenic carbon dioxide).

Methane is formed during the decomposition of paperboard in landfills (Franklin Associates, 2011; Häkkinen and Vares, 2010). The GWP of the paperboard cup greatly depends on the assumptions on the decomposition grade, forming of methane, and the management of the landfill gasses. The GWP of the cup is at maximum decomposition rate ten times higher compared to no decomposition (Franklin Associates, 2011). Table 2.3 illustrates the change in relative GWP impact of the different life-cycle phases for the landfilling option.

Incineration of paperboard cups instead of landfilling lowers the GWP due to credits of recovered energy. The reduction can be up to of 80% (Häkkinen and Vares, 2010). Incineration of paperboard cups in a cement kiln or in a subcoal route (as alternative fuel) instead of a waste incinerator further reduced the GWP impact (Vercalsteren et al., 2006). The energy recovery efficiency in the waste incinerator was taken as 20%, but in the kiln as 100% since the waste was used as secondary fuel (Vercalsteren et al., 2006).

The choice of the waste treatment option has a major influence on the GWP. Incineration instead of landfilling can drastically increase the GWP of the petro-plastics cups, but on the other hand decreases the GWP of paper cups (due to avoided release of methane and recovered energy). The studies showed no consensus on the effect of recycling PET, and neither on composting PLA. The impact of the waste treatment is furthermore influenced by the amount of applied credits (incineration, composting), the allocation of material (in recycling) and degradation assumptions (landfilling and composting).

Cut-off rules and allocation

Not all studies collected inventory data for all life-cycle phases or all materials involved. Seven LCA studies used cut-off criteria, i.e. a point where no further data was collected. Six studies performed a cut-off if the weight of material inputs to processes was small (less than 1%) and/or omitted capital goods/infrastructure. Five studies used simplified processes. Omission of materials or goods, or the simplification of processes can all influence the GWP, but its magnitude is not clear from the studies.

Allocation is necessary in multiple-output processes to assign upstream input and output data among the multiple products of a process within a product system. Franklin Associates (2006, 2009a, 2011) used mass or enthalpy allocation, based on a case-by-case consideration of the processes and products. Ligthart and Ansems (2007) applied both mass and economic allocation. System expansion, i.e. avoided allocation, was also used by all studies which used a credit for recovered energy or recovered materials.

Franklin Associates (2009a) used two ways of allocating credits for recycled PET (RPET) as cup material. The first way treated RPET as PET made from recycled material, and excluded any burden from producing virgin material. The second way considered RPET as (partly) replacing virgin PET, and assigned part of the burden of virgin PET to RPET. This approach increased the GWP by 10%.

Accounting for biogenic carbon dioxide

Carbon dioxide is the main contributor to GWP. Biogenic carbon dioxide can be accounted for in different manners. Christensen et al. (2009) described two consistent approaches in which they paid special attention to the waste treatment options and the system boundary of the waste treatment.

In the first approach biogenic carbon dioxide is not accounted for and has no contribution (neutral) towards GWP, i.e. the carbon-neutral approach. The uptake of atmospheric carbon dioxide and the sequestration of carbon in biomass are not accounted for. Biogenic carbon stays sequestered if the biomass is landfilled (assuming no carbon is released). This means that a credit must be applied for landfilling in the carbon-neutral approach, since the carbon is not released back to the environment. Carbon dioxide is released back to the atmosphere during incineration of the biomass, but the carbon-neutral approach does not account for this release.

In the second approach the carbon dioxide is accounted for and contributes to GWP in the same manner as fossil carbon dioxide does (Christensen et al., 2009). Here, the biomass is credited for the uptake of atmospheric carbon dioxide. Landfilling of the biomass does not receive credit, since the biogenic carbon sequestration was already accounted for in the production of the biomass. The release of carbon dioxide during incineration of the biomass should in this approach be included since it is treated equally as fossil carbon dioxide.

Both biogenic carbon dioxide approaches are used in the reviewed studies (see Table 2.3). PE Americas (2009), Uihlein et al. (2008) and Vercalsteren et al. (2006) credit PLA and paperboard for the uptake of carbon dioxide in the biomass. The other studies treated PLA and paperboard as carbon-neutral.

The different approaches led to dissimilar contributions of the GWP in the life cycle phases of PLA cups (see Table 2.3). Landfilling hardly changed the GWP according to PE Americas (2009), because the carbon uptake was already included in the production of the biomass. Franklin Associates (2011), on the other hand, applied a carbon-neutral approach and credited landfilling for the sequestered carbon in PLA. Incineration of PLA releases the sequestered carbon back to the environment. Pladerer et al. (2008) took a carbon-neutral approach and applied GWP credits for recovered energy during incineration (see Table 2.3). Vercalsteren et al. (2006) accounted for the uptake from carbon dioxide in the PLA. They applied GWP credits for the recovered energy from PLA incineration, but also included carbon dioxide emission from combusting PLA.

Only Vercalsteren et al. (2006) credited paperboard for the uptake of carbon dioxide in the biomass, the other studies took a carbon-neutral approach. This means that incineration contributed to GWP according to Vercalsteren et al. (2006), but provided a GWP credit according to the others (Häkkinen and Vares, 2010; Ligthart and Ansems, 2007; Pladerer et al., 2008), as can be seen in Table 2.3.

The allocation choice of carbon dioxide uptake in biomass should in principle not affect the GWP of the complete life cycle. It just shows carbon uptake and/or releases in different processes in the life cycle. This can create confusion in communication to decision makers or stakeholders on the credits for possible waste treatment options.

Data sources

Inventory data for the studies were collected from several sources. Three studies included data from the USA, four from Europe, and three used data from both the USA and Europe. Data for raw material production is frequently taken from a database (often average data), and complemented with company specific data on the manufacture of the cups. Differences in inventory data can lead to different GWP results. The influence from different data sources on the GWP can only be evaluated if the same data is applied in modelling the rest of the life cycle. Especially the same waste process is relevant, since this process plays an important role.

Oil and natural gas are the resources for petro-plastics. Data from PlasticsEurope, the US GaBi database, the US LCI database, and the ecoinvent database represent the used data sources for petro-plastics. Two Franklin Associates studies (2009a, 2011) used the same data source for production of EPS, and used a combination of landfilling and incineration as waste treatment. The GWP results for both EPS cups systems are close to each other.

The GWP of PET cups with landfilling according to PE Americas (2009) is double as high as landfilled PET cups according to Häkkinen and Vares (2010). The GWP of PET with incineration is almost 30% higher for Häkkinen and Vares (2010) compared to Pladerer et al. (2008). PE Americas (2009), Häkkinen and Vares (2010), and Pladerer et al. (2008) all used different databases for production and/or waste processing of PET (i.e. GaBi data, Boustead data, and data from other published studies), which could be the reason for the variation in GWP results.

Polylactic acid (PLA), based on corn, was the only examined bioplastic. All but one study refer to PLA made by NatureWorks in the USA. Only Uihlein et al. (2008) calculated and estimated PLA data based on European information. The GWP of the PLA cup based on PLA data from 2005 (Franklin Associates, 2006) was 60% higher than the one based on PLA data from 2010 (Franklin Associates, 2011).

Paperboard cups were sometimes called paper, carton, or carton board cups. Several studies state that it is hard to collect data on the production of paperboard. The studies used solid bleach board, liquid packaging board, or company specific information as material data. Paperboard production plants can use a variety of fuels in the paperboard production process (e.g. fossil and/or renewable fuels). The GWP of paperboard can nearly double depending on the fuel mix used for the production of paperboard (i.e. wood residues instead of fossil fuels) (Franklin Associates, 2009a).

Vercalsteren et al. (2006) used the solid bleached board data from the ecoinvent database (version 2000). These data, as they turned out, included a mistake in the reported fossil and biogenic carbon dioxide, which led to a threefold account for carbon dioxide emissions. The (incorrect) results of this study were used in the study of Pladerer (2008). The GWPs of the paperboard cup system with incineration as waste process from Vercalsteren et al. (2006) and Pladerer (2008) are more than double compared to the GWPs from Ligthart and Ansems (2007) and Häkkinen and Vares (2010).

The influence of different data sets on the GWP cannot be clearly established from the studies since also the differences in other characteristics influence the GWP results. Different data sets can cause various GWP results and this could be the source of discrepancies between studies.

Life cycle inventory (LCI) data

The collected inventory data were sometimes included in the study reports. Five studies presented LCI data, merely on energy use and greenhouse gasses, but solid waste and water use were also reported. The inventory data were used to calculate the life cycle inventory assessment data of the cups. The omission of substances in the inventory data collection can have an effect on the GWP results.

Impact categories

All studies contained data on GWP as category indicator for climate change. This was a selection criterion for the studies. Five studies included additionally impact results on acidification, eutrophication, ozone depletion and human toxicity/health. Comparison of these LCIA results (except GWP) was hard since the studies used different characterization methods. Uihlein et al. (2008) and Vercalsteren et al. (2006) used the EI99 indicator, Garrido and Alvarez del Castillo (2007) provided only relative impact percentages, and Pladerer et al. (2008) used a mix of impact categories from diverse methods (IPPC, CML 2001, and Impact 2002+). Since GWP was the only common impact indicator, the comparison of the LCA results between the studies was therefore limited to GWP results.

2.3.5 Reliability check

The life cycle of the cups can be modelled according to various system configurations (e.g. different cup materials, various waste processing options). Seven studies incorporated multiple cup systems and thus increased the reliability of the GWP results (see also Table 2.1 and 2.2). Franklin Associates (2009a) and PE Americas (2009) included alternative materials within a material group or covered production properties. Franklin Associates (2011), Häkkinen and Vares (2010), Pladerer et al. (2008), and Vercalsteren et al. (2006) evaluated alternative waste processes. Franklin Associates (2011) and PE Americas (2009) evaluated various weights of the cups. The influences of these alternatives on the GWP results, as visible from outcomes within and across studies, were discussed above.

Several studies performed additional sensitivity or scenario analyses to evaluate the reliability of the GWP results. The studies evaluated the influence of the material or energy use (Franklin Associates, 2009a; PE Americas, 2009; Pladerer et al., 2008; Uihlein et al., 2008; Vercalsteren et al., 2006), the waste processes (Franklin Associates, 2009a, 2011; Ligthart and Ansems, 2007; PE Americas, 2009; Pladerer et al., 2008; Vercalsteren et al., 2006), the weight of the cup (Ligthart and Ansems, 2007; Vercalsteren et al., 2006), and transport (Vercalsteren et al., 2006). Pladerer et al. (2008) were the only ones to include an uncertainty analysis of the input data. They reported the environmental results for the base scenarios including error ranges.

The above sensitivity and scenario analyses support the results from the studies which included multiple cup systems. The inclusion of multiple cup systems per cup material aims to examine alternative circumstances purposely and is part of the goal of these studies. Sensitivity or scenario analyses suggest, on the other hand, that alternatives have been examined as part of the interpretation phase.

2.4 Discussion

The ten peer-reviewed LCA studies for disposable beverage cups were analysed on the differences in methodology and their influence on the quantified GWPs of the cups. The GWPs were recalculated to a cup size of 16 oz (473 ml) to enable a comparison between the different cup systems. Based on a proportional relationship between the GWP and the weight of the cups (Franklin Associates, 2011; Vercalsteren et al., 2006), this recalculation assumed a proportional relationship between the GWP and the cup volume. A more precise recalculation could have resulted in more accurate GWP results, but the simplification seemed adequate in offering a good indication.

The ten studies showed transparency in their reporting. This made it possible to compare the data sources and methodological choices in the studies and to evaluate their influence on the GWP. The analysis of these data sources and methodological choices revealed multiple influential factors for the GWP of the cups (see Table 2.4). We could not single out the most influential factor, but few lessons can be taken. Improved PLA production decreased the GWP of PLA cups by 60% (Franklin Associates, 2006, 2011; Vercalsteren et al., 2006) and the used fuel mix has a big influence in paper production (Franklin Associates, 2009a). This also implies that the used data set is of major importance. A switch in waste treatment can reduce the GWP up to 80% (Häkkinen and Vares, 2010). Assumptions on the decomposition of paper in landfills and the formation of methane can lead to a tenfold increase of the GWP (Franklin Associates, 2011). No study examined all of our identified influential factors, which makes it nearly impossible to identify the most important factor.

LCA in its present form originates from the 1990s. Also then, different LCA studies for the same products happened to lead to varying and sometimes conflicting results that could often be traced back to methodological discrepancies and/or data used. Ekvall (1999) identified the choice in material (recycled or virgin), waste treatment options, type of energy used, type of recovered energy during incineration, allocation options, and the used data as key methodological issues in the life cycle of paper recycling. Merrild et al. (2008) showed the importance of the type of technology used in the production of virgin paper, the reprocessing of paper and the incineration of paper, and the choice of the system boundary in the comparison between paper incineration and recycling. This is consistent with our main influencing factors of GWP, and suggests little progress from 20 years of standardization of methods and procedures in LCA (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010; ISO, 2006a, b). Especially system boundaries and the choice of used data are hard to standardize. However, harmonization of the LCA contributed in fact to transparency in the LCA procedures. The transparency made it possible to compare the methodological choices in the disposal cup studies. This is a great step forwards. Furthermore, methodological choices are often based on actual product systems. Variations in products and processes between companies are unavoidable and a fact of life. Also, waste treatment options depend on national and cultural customs and habits. Note that the goal of the LCA to a large extent determines the methodological choices.

Different LCA studies for the same product can thus still show different outcomes. This inconsistency is shown in this paper and other studies (Padey et al., 2012; Weiss et al., 2012). Such discrepancies are not beneficial for the credibility and reliability of LCA studies and its use as a decision-support tool. Decision makers need robust and trustworthy information if they want to integrate the environmental impact of products in their judgement.

Above influencing factors basically all represent sources for uncertainty in the GWP outcome. Uncertainty is inherent in LCA results. Problems concerning uncertainty in LCA results have been recognized from the early start (Huijbregts, 1998b; Ross et al., 2002). Several classifications of uncertainty have been proposed as a starting point to address uncertainty in LCA (Finnveden et al., 2009; Heijungs and Huijbregts, 2004; Huijbregts, 1998b; Huijbregts et al., 2001). These proposals take their basis in roughly two ways of classifying.

Uncertainty can be classified by its location in the modelling framework, e.g. parameter uncertainty (input data), model uncertainty (limitations of the modelling process), and scenario uncertainty (normative choices) (Zamagni et al., 2008). Uncertainty can also be classified according to the type of uncertainty, e.g. absence of values (completeness), inappropriate values (reliability), or more than one value available (variability) (Heijungs and Huijbregts, 2004). Potting et al. (2002) have combined these two ways of classification in one consistent scheme (and also combined these with additional sources of uncertainty, i.e. whether modelling the past or future, and the social or physical reality). The classification of Potting et al. (2002) was developed for uncertainties in integrated assessment modelling, but is also well applicable to classify uncertainties in life cycle assessment. Table 2.4 shows the classification of the uncertainties from this paper in an adapted version of their scheme.

A classification of uncertainty is useful as it also structures the way how these uncertainties can be addressed. A number of tools exist to deal with the uncertainties, e.g. sensitivity analysis, scenario analysis and statistical approaches (Heijungs and Huijbregts, 2004; van der Sluijs et al., 2004). Most of these uncertainty tools address selected sources of uncertainty only (van der Sluijs et al., 2004). This was also visible in the LCA studies including multiple cup systems. These alternative systems, as well as the sensitivity and scenario analyses, always showed the influence of just one uncertainty parameter at a time. This led to a series of outcomes, where each outcome represents a specific case.

Similar to what this study shows for the GWP of the cup, the LCA results for products in general are usually influenced by multiple factors (i.e. multiple sources of uncertainty). The combined effect of several influential factors is usually not simultaneously presented in the selected cup LCAs for this study and is neither usually covered in other LCA uncertainty assessments. Systematic and simultaneous inclusion of all influential factors would portray the influence of these several factors at once. This so-called ensemble modelling provided in climate studies more robust results than results from separate models (van Loon et al., 2007). Ensemble modelling of multiple data sources and modelling choices could result in a more robust outcome in the comparison of disposable cups. Further research is needed to show whether ensemble modelling provides additional value to the outcome.

Table 2.4: Location and type of uncertainty in the LCA studies of disposable cups. The main influencing factors on the GWP are printed in bold.

Location of uncertainty	**Type of uncertainty**			
	Variability	Reliability	Ignorance	Choices and assumptions
Model completeness:				
- System boundaries			X	X
- Cut-off rules			X	X
- Impact indicators			X	X
- **Allocation methods**			X	X
- Included life-cycle phases			X	X
Model structure:				
- Simplification of model			X	X
- Functional unit				X
- **Cup material**				X
- **Cup weight**				X
- **Production of cup material**	X	X		
- **Manufacturing of cup**	X	X		
- Use and disposal cups	X	X	X	
- **Waste treatment option**	X	X		X
- Transport	X	X		
Input data:				
- Inventory data	X	X		
- **Data sources**	X			X
Possibility of dealing with uncertainty	X	X		X

2.5 Conclusion

Ten peer-reviewed LCA studies for disposable cups were qualitatively and quantitatively compared to each other in order to evaluate the consistency and robustness of their results. The disposable cups were made from petro-plastics, bioplastic and paperboard. The quantitative comparison focused on the global warming potential (GWP) as this was the only common category indicator across the studies. The ordinal ranking of the GWPs of the cup materials within each study was not consistent across these studies (Table 2.1). No cup material ranks consistently better than other cup materials in all studies, and neither can one cup material be labelled as the most environmentally friendly one. Ranking of absolute GWPs of all cup systems from all studies did also not show one cup material as consistently the best or worst (Figure 2.2). The GWP of the cups was for this purpose recalculated to a cup size of 16 oz (473 ml) to enable a quantitative comparison between the various cup sizes. The variation in GWP results among all cup systems is great. The ratio between the highest and the lowest GWP score for all cup systems is 30.

The methodological choices and data used in the studies were evaluated on their relationship to the GWP results. Several studies included additional cup systems which provided insight on the influence of these choices and data. Sensitivity and scenario analyses in some studies furthermore provided a valuable insight, particularly on the influence of the waste processing options on the GWP. The evaluation shows that GWP is influenced by multiple factors: the properties of the cups (e.g. cup material and weight), the production of the cup material, the manufacturing of the cup, the waste processing option of the disposed cups, allocation choices, and used data. The GWP is proportionally related to the weight of the cups.

The influence of the waste processes on GWP varies among the cup materials and its influence is not always consistent. Incineration of petro-plastic cups increases the GWP compared to landfilling. Recycling of petro-plastics instead of landfilling can decrease the GWP. Landfilling of PLA cups includes a credit for sequestration of carbon in PLA, while incineration includes a credit for recovered energy from combustion. Landfilling PLA cups show a lower GWP compared to incineration. There was no consensus on the comparison between composting and landfilling PLA cups. The GWP of the paperboard cup in a landfill significantly depends on the assumptions on its decomposition grade, formation of methane, and the landfill management. The GWP of the paperboard cup is ten times higher if a maximum decomposition is assumed compared to no decomposition in the landfill. Incineration of paperboard cups instead of landfilling can decrease the GWP.

Landfills can capture the emitted methane and flare it into less harmful carbon dioxide, or use it as fuel and thus receive a credit for avoided fuel production. Methane is released to the environment if the landfill does not capture the methane. The landfill management options have thus an important influence on the GWP. The credits received during the incineration process depend on the lower heating value (LHV) of the cup material. Petro-plastics have an LHV from 24 MJ/kg (PET) to 44 MJ/kg (PP), while the LHV of PLA is 18 MJ/kg and 16 MJ/kg for paper. Incineration of petro-plastics receives thus a higher credit compared to PLA and paper.

Harmonization and standardization of the LCA methodology has led to transparency in LCA studies. This transparency facilitates a comparison between the studies. The standardized LCA method nevertheless still leaves room for methodological choices, which can lead to inconsistency in outcomes between studies. For example, biogenic carbon dioxide can be treated as carbon-neutral, or accounted for similarly as fossil carbon dioxide. The choice should be stated explicitly in the study and special attention should be paid to the system boundary and waste treatment to prevent double counting (or omission). Including multiple modelling approaches and data sets simultaneously in an LCA might be a sensible method to increase the robustness of outcomes.

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3 Variation in LCA results for disposable polystyrene beverage cups due to multiple data sets and modelling choices

Abstract

Life Cycle Assessments (LCAs) of the same products often result in different, sometimes even contradictory outcomes. Reasons for these differences include using different data sets and deviating modelling choices. This paper purposely used different data sets and modelling choices to identify how these differences propagated in LCA results. Vehicle for this methodological exploration was an LCA case study of a typical polystyrene (PS) disposable cup. An initial LCA of PS cups was made using only one data set per process. Contribution and sensitivity analysis identified those processes with influential contribution to the overall environmental impact. Next additional data sets were acquired for all influential processes. The spread in impact results for each life cycle process was calculated *after* impact assessment for each individual inventory data set as to preserve the correlation between inventory data within each individual data set. The spread in impact results reflects uncertainty existing between different data sets for the same process and due to modelling choices. The influence on overall LCA results was quantified by systematically applying all combinations of data sets and modelling choices.

Results from the different data sets and modelling choices systematically point to the same processes as main contributors to all impact categories (PS production, cup manufacturing, PS incineration and PS recycling). The spread in toxicity indicators exceeds the energy-related impact categories. Causes of spread are resources and energy used (type, amount, date and origin), reported emissions, and applied allocation procedures. Average LCA results show slight preference for recycling PS compared to incineration in most impact categories. Overlapping spread in results of the two waste treatments, however, does not support the preference for recycling. The approach in this paper showed how variation in data sets and modelling choices propagates in LCA outcomes. This is especially useful for generic LCAs as systematic use of multiple data sets and multiple modelling choices increases the insight in relative contributions of processes to, and uncertainty in the overall LCA. These results might be less easy to perceive, but they provide decision makers with more robust information.

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3.1 Introduction

Life Cycle Assessment (LCA) evaluates the interactions of a product with the environment through its whole life cycle (i.e. product system) (ISO, 2006a). A full life cycle starts with the extraction of resources, continues with processing these resources into materials, follows with the manufacturing of a product from these materials, proceeds with the use of the product, and stops after waste processing of the disposed product. An LCA can evaluate the environmental impact of a single product, e.g., for product system optimisation. LCA is also used to compare the environmental performance of several product alternatives. Different LCA studies of the same product, or covering the same product alternatives, should theoretically result in the same outcome. In practice, however, LCAs frequently show dissimilar and sometimes even conflicting results (Padey et al., 2012; Weiss et al., 2012). This discrepancy is not beneficial for the credibility and reliability of LCA studies and its use as a decision-support tool.

A recent review by van der Harst and Potting (2013a) illustrates the discrepancy between a number of LCA studies on disposable beverage cups. These studies compared disposable beverage cups made from different types of material such as paper, petro-plastics, and bioplastics. Petro-plastics are usually produced from fossil fuels as crude oil or natural gas, while bioplastics are made from renewable resources. The most widely used bioplastic for disposable beverage cups is poly lactic acid (PLA) manufactured from corn (NatureWorks LLC, 2011). PLA is also made from sugarcane, tapioca or sugar beets (Corbion Purac, 2013b; Galactic, 2011). Petro-plastics for disposable cups include polystyrene (PS), polypropylene (PP), polyethylene terephthalate (PET) and recycled PET (RPET).

The LCAs in the review by van der Harst and Potting (2013a) all quantified climate change, but did not have any other impact categories in common. None of the cups consistently performed worst or best on climate change in all included studies. A closer look into the climate impact across studies showed a ratio between the highest and lowest climate impact value of 3.4 for petro-plastics, 1.7 for PLA, and 20 for paper.

Van der Harst and Potting (2013a) also looked closer into possible sources of discrepancies between the LCAs. They identified amongst others differences in the properties of the cups (material choice and weight), the production processes covered, the energy sources used (fossil or renewable), and the waste processing options considered (i.e. landfilling, incineration, recycling, composting). Especially the choice of the waste processing option seemed to influence the outcomes. Pladerer et al. (2008), for instance, found a lower climate change impact for recycling of PET cups, compared to their incineration. Häkkinen and Vares (2010) showed a lower climate change impact for landfilling of PET cups compared to their incineration. Other assumptions and choices for waste processing also caused variation in LCA outcomes. Crediting recycled petro-plastics, for example, played an important role in the outcome of an LCA (Franklin Associates, 2009a; Ligthart and Ansems, 2007). Alternative assumptions in the degradation extent of paper cups during landfilling were responsible for most of the spread in the paper cup results (Franklin Associates, 2011). Furthermore, the use of different data sets led to a spread in results.

The findings of van der Harst and Potting (2013a) are supported by other studies. See Text box 3.1 for an overview of these studies.

The above studies all point to several important sources for variation in LCA results for the same product. The Joint Research Centre of the European Commission (JRC-IES) aims to increase the reproducibility and consistency in LCAs through on-going initiatives in the form of recommendations in the ILCD handbook and the Product Environmental Footprint (PEF) (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010, 2012). The PEF includes data quality requirements and data collection approaches at product level. Additionally, product category rules (PCR) for materials include specific guidance on issues such as boundaries, allocation, data collection, calculation, etc. (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010; The International EPD System, 2013). PEF and PCR may harmonize LCA procedures and data collection at specific production sites, but this does not necessarily lead to more robust LCA's for average or typical products. These guidelines namely, however, cannot solve the variation which exists between different production sites, and thus cannot reduce this uncertainty in LCA results.

Variation in LCA outcomes results from uncertainty in LCA input data, parameters or modelling choices (Huijbregts, 1998a; Huijbregts et al., 2003). A good modelling process should include performance tests such as sensitivity and uncertainty analysis (Bennett et al., 2013; Jakeman et al., 2006). The importance of uncertainty and sensitivity analysis in LCA has been recognized from the beginning (Huijbregts, 1998a; Ross et al., 2002), and is also recommended by the ISO standard (ISO, 2006b). LCA studies often explore uncertainties by means of sensitivity analysis, scenario analysis, and/or with help of Monte Carlo analysis for uncertainty in input data (Huijbregts et al., 2003; Huijbregts et al., 2001). Monte Carlo analysis in LCA typically relies on probability density functions for separate inventory items as based on given inventory item values across different inventory data sets and sources (Huijbregts et al., 2001). This analysis is very useful in assessing the uncertainty in inventory data of a specific process or specific material at a specific production site, for which multiple data sets of the same specific process/material are gathered. Monte Carlo analysis is also used for uncertainty analysis of inventory data of average or typical products, where it takes its basis in multiple inventory data sets for various production sites. This ignores, however, the correlation between the inventory items values as existing within one data set from a particular production site. Variation between different data sources is, as a matter of fact, frequently ignored in LCA studies for typical or average products by only using one data set per material or process.

The studies listed in the beginning of the introduction list causes for uncertainty and variability in LCA results. This may qualitatively explain the variation in LCA results, but does not yet tell how changes in sources quantitatively propagate in LCA results. The consequences of changing a combination of sources are particularly difficult to oversee. This paper explores a relative new way of uncertainty analysis in LCA by using multiple data sets and multiple modelling choices in LCA of non-specific products (i.e. average or typical products within a certain region or time frame). A similar approach of using multiple data sets and modelling choices is often used in the field of spatial explicit modelling of transboundary environmental problems (Lundie and Huppes, 1999; Nakícenovíc et al., 2000; Potting and Bakkes, 2004). The results for this type of modelling, also called ensemble modelling, show in the field of transboundary modelling to be more robust compared to results from separate models (Delle Monache and Stull, 2003; van Loon et al., 2007).

The objective of the paper is to systematically quantify the influence of the outcomes of LCAs for PS cups for changes in data sets and modelling choices. The vehicle for this methodological exploration is an LCA case study of a typical white disposable polystyrene (PS) beverage cups for hot drinks (180 ml). We focussed on a whole range of impact categories in our analysis. The results of the analysis will be discussed in the light of the broader LCA discussion about uncertainties.

Note that a short version of this manuscript, which outlines the method and qualitative results, was included in the proceeding of the LCM 2013 conference (van der Harst and Potting, 2013b).

Text box 3.1: Studies which identify key issues for variation in LCA results.

Several studies on paper production and waste processing also identified issues that are responsible for variation in results. Ekvall (1999), for example, examined differences in life cycle inventory data results for paper recycling and paper incineration and pointed to material choice, waste processing options, type of energy used, and allocation methods as key methodological issues. Also, the choice between marginal or average electricity production can lead to large differences in inventory results in the Scandinavian countries (Weidema et al., 1999). Merrild et al. (2008) compared the influence of various technologies for virgin paper production, paper recycling, and paper incineration on climate change. Merrild et al. (2008) concluded that a preference for recycling or incineration is influenced by the data sets of virgin pulp production and recycling technologies, the energy recovery rate of the incineration plant, and the system boundary. A review of nine LCA studies on paper and cardboard waste treatment options by Wenzel and Villanueva (2006) shows a preference for recycling above landfilling, but the comparison between recycling and incineration was not unanimous. Wenzel and Villanueva (2006) related this inconsistency mainly to differences between system boundaries and the type of used energy. Similarly, an extensive study for Waste & Resource Action Programme (WRAP) (Michaud et al., 2010) compared numerous LCA studies of different waste treatments and shows landfilling as the least preferred waste treatment for paper and cardboard, but the preference between recycling and incineration is inconclusive. The main influencing issues were identified as the electricity mix, the used technologies, and the treatment of carbon sequestration.

The WRAP study also looked into waste treatment of petro-plastics. Most of the reviewed studies in the WRAP study agreed on the preference for petro-plastic recycling compared to petro-plastic incineration, but some preferred incineration. The choice of the credited material and the ratio of substituted material plays an important role in the recycling option, and the efficiency of the incinerator and the substituted energy mix in incineration (Michaud et al., 2010).

Weiss et al. (2012) presented the results of a meta-analysis on 44 LCA studies on biobased materials, and observed differences in LCA studies concerning assumptions and choices in system boundaries, functional units, scenarios and allocation approaches. Variation in LCA results was traced back to the type of biomass used and the production technologies of the biobased materials.

3.2 Methods and means

3.2.1 Life cycle assessment

The procedure for an LCA, as laid down in ISO 14040 and 14044 (ISO, 2006a, b), consist of four phases. The first phase, goal and scope definition, specifies the why (goal) and how (scope) of an LCA. The second phase, inventory analysis, quantifies all the environmental inputs and outputs for all processes in the whole product system at stake. Environmental inputs are natural resources, for example ores and fossil energy carriers, entering the processes. Environmental outputs are emissions and final waste which may leave these processes. The third phase, impact assessment, translates the environmental inputs and outputs into their contribution to a number of environmental impact categories (e.g., resource depletion, climate change, acidification, toxicity). The fourth phase, interpretation, evaluates the results from the previous phases in order to draw conclusions related to the initially defined goal.

The goal or objective of this study is already described in the introduction. Scope definition lays the basis for all later phases in an LCA by specifying its methodological approach, including the assumptions and choices involved. A number of subjects need to be clarified in the scope definition, e.g., the functional unit, the flow of resources and materials from process to process (product system flow), inventory data collection and processing (including allocation), and impact assessment. These subjects are explained in more detail below.

3.2.2 Functional unit

The environmental impact of a product is expressed in the so-called functional unit. The functional unit describes the function(s) of the studied product (or product *system* to be more precise). A well-defined functional unit includes both quantitative and qualitative information about the product's functionality. This may also include technical qualities (e.g., strength, durability, maintenance), appearance, colour, or capacity. A well-defined functional unit constrains the relevant product alternatives, and provides an equal reference to which all inputs and outputs of the system are related and for which the environmental impacts are calculated.

The functional unit in this study is to provide a disposable beverage cup fit for serving hot beverages from a vending machine as frequently used in the Netherlands in big organizations (e.g., offices, companies, and schools). This cup typically has a volume of 180 ml, is made of polystyrene (PS), is white of colour and weights between 3.8 and 4.4 g, as turned out from consultation among several cup sellers (Dispo International, 2012; Huhtamaki, 2012a; Krings & Schuh OHG, 2012; Papstar, 2012b). We used a weight of 4.2 g as representative for the typical disposable PS beverage cup.

3.2.3 Product system

Figure 3.1 shows a simplified flow chart of the life cycle of the disposable PS beverage cups. The life cycle starts with the extraction of oil and natural gas as the basic resources for PS production. Different types of PS exist. A mixture of two PS types is used for disposable cup production, i.e. general purpose PS (GPPS) and high impact PS (HIPS). A small amount of titanium dioxide (1-2%) is added as a colouring agent for whitening PS granulate (Ligthart and Ansems, 2007). The PS granulate is in several steps extruded and thermoformed into cups.

The ready disposable PS cups are packed first as a stack in plastic foil. Next several stacks are put in a cardboard box. A common box contains 30 stacks of 100 beverage cups each, resulting in 3000 cups per box (Depa Disposables and Packaging, 2012; Huhtamaki, 2012a; Moonen, 2012a; RPC Group, 2012). The cardboard boxes with cups are transported via distributors to the customers. The customer is usually not an individual hot beverage drinker, but rather an organization using vending machines for hot beverage supply. The vending machines are replenished with cups (and coffee, tea, etc.). Replenishing can be done by the organization itself or by a distributor. In the latter case, additional transport might be needed.

We assumed that (hot) beverage drinkers use the cups and dispose them in a (waste) bin. Most organizations employ bins for collecting commingled waste, i.e. used cups and other waste. This waste is in the Netherlands generally sent to a municipal solid waste incinerator (MSWI). Most Dutch MSWIs recover energy from the waste incineration (Otten and Bergsma, 2010). Alternatively, an organization can collect the discarded cups separately and send them to a recycler to produce recycled PS. We included both waste incineration with energy recovery and recycling in this study. We assumed either 100% incineration of the cups, or 100% recycling. This facilitated comparison of both waste processing methods, and also allows follow-up scenarios for situations where one part of the cups is incinerated and another part is recycled. In practice, a 100% recycling rate is unlikely since not all cups will be collected in the recycling bin and contaminated cups might be rejected and sent to the waste incinerator after all. We did not consider (possible) contamination in the cups and how this may influence the waste processing of the contaminated cups. We did not consider landfilling as a possible waste option since landfilling of municipal waste is not allowed anymore in the Netherlands since 1996 (De Boer, 1995). We also ignored plastic littering in the environment assuming that in large buildings virtually all cups are disposed of in waste bins.

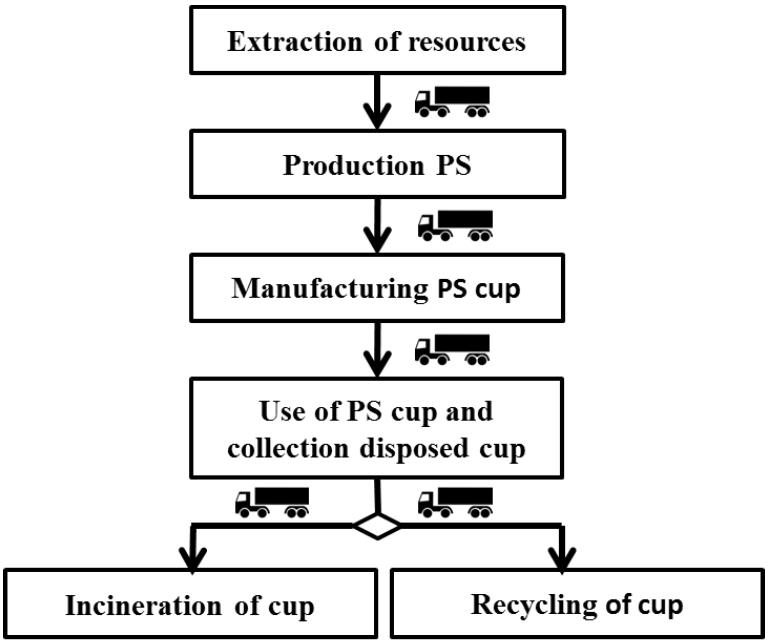

Figure 3.1. Simplified life cycle flow chart of the disposable polystyrene beverage cup

3.2.4 Inventory data collection

We considered a disposable PS cup in this study as typically used for vending machine in the Netherlands in big organizations. The collected data sets should preferably reflect the variability which exists between different producers or manufacturers in the life cycle of a typical disposable PS cup.

Data for the PS cup should reflect data from PS cups which are presently available on the Dutch market. We limited the study to PS cups manufactured in Europe. The use and the disposal of the cups take place in the Netherlands, and data collection for these processes focussed on Dutch situations. The collected data for all processes should represent current technologies. We used cut-off criteria of 1% (mass) for data inclusion and excluded infrastructure from the data collection.

In order to keep inventory data collection feasible, we first collected one data set for each of the processes in the life cycle of the disposable PS cup (see Figure 3.1). These initial data sets were used to make an initial LCA. The initial LCA enabled contribution and sensitivity analysis to identify processes with a minor contribution to the environmental impact. Next, multiple data sets were only collected additionally for the processes with an influential contribution to the environmental impact results. We consider a contribution influential in this study if a process contributes at least 15% to the environmental impact in five or more impact categories.

We only used secondary data sets for the background processes, such as transport, resource extraction, and upstream processes. Data for the background processes were taken from the ecoinvent database (Ecoinvent Centre, 2010).

The data sets for the foreground processes were collected mainly from publicly and commercially available databases, reports, and articles. We gathered additional primary data, however, for the manufacturing and use of the cup, and recycling of the cup. Qualitative and quantitative information was acquired from companies. This information is property of the companies and will not be recognizable published as to guard the confidentiality of data and company specific information.

The number of available data sets for PS production in Europe was limited. Europe also imports a (very) small amount of PS produced in the United States of America (USA). We therefore also included data sets from the USA. Manufacturing of the cups can be based on an inline thermoforming procedure where the extruded PS sheet is directly thermoformed into cups. Alternatively, the extruded PS sheet is cooled down and stacked, and later reheated to form cups in an off-line thermoforming process. The inline thermoforming process is more energy efficient, because the PS sheet does not require reheating. Additionally, the plastic waste created during the inline cup forming process is recycled internally in a closed-loop system. Two data sets for cup manufacturers are based on an inline procedure, but it was not clear which technique was used in the other data sets. Data sets typical for waste treatment in the Netherlands were also sparse, hence we included data sets for the European situation which were considered representative for the Netherlands.

Table 3.1 shows all data sets that are used in the foreground processes in this study. The LCA-software SimaPro 7.3 (PRé Consultants, 2011) is used to express all inventory data per functional unit, i.e. per PS cup of 4.2 g. Inventory data collection and processing for the initial situation is further elucidated here.

Table 3.1: Data sets used in the different life cycle processes of the PS cup. The data sets in the first row are used in the initial LCAs.

PS Production	Cup manufacturing	Incineration cup	PS recycling process	Credits for recycled PS
Ecoinvent database, GPPS and HIPS (Ecoinvent Centre, 2010)	Ecoinvent database, Thermoforming (Ecoinvent Centre, 2010)	Ecoinvent database, MSWI of polystyrene [c] (Ecoinvent Centre, 2010)	Bergsma et al. (2011)	60% based on economic value (PlasticNews and plastic recyclers)
PlasticsEurope, Eco-profiles GPPS and HIPS (PlasticsEurope, 2012)	Ligthart and Ansems (2007) [b]	ELCD database, Waste incineration of plastics [c] (ELCD, 2008)	Ligthart and Ansems (2004) and Plastic recycler 1 (confidential)	Ligthart and Ansems (2004)
ELCD database, GPPS and HIPS (ELCD, 2008)	Garrido and Alvarez del Castillo (2007)	Eggels et al. (2001)	Shen et al. (2011)	Ligthart and Ansems (2007)
Franklin Associates, GPPS and HIPS (2010)	Plastic Cup producer 1 (confidential)	Croezen and Bergsma (2000)	Plastic granulator and Plastic recycler 1 (confidential)	Bergsma et al. (2011)
US LCI database, GPPS and HIPS(NREL) (NREL, 2011) [a]	Plastic Cup producer 2 [b] (confidential)		Plastic recycler 2 (confidential)	

[a] Data are based on Franklin Associates (2010).
[b] Packaging material added.
[c] Credits for recovered energy added.

We used easy available generic data sets in the initial situation. Data for the production of PS and its extrusion and thermoforming into a cup were taken from the ecoinvent database (Ecoinvent Centre, 2010) as included in SimaPro 7.3 (PRé Consultants, 2011). We assumed that the composition of the cup exists of a 50/50 mixture of GPPS and HIPS, and that the cup consists 1% of TiO_2. This is within the range provided by Ligthart and Ansems (2007). The ecoinvent thermoforming data for the manufacturing of the cups include the use and production of packaging materials (including their transport to the cup manufacturer).

On average, 30 stacks of 100 cups need 202 g polyethylene (PE) or polypropylene (PP) foil and 1197 g of cardboard box as packaging material according to confidential information, Garrido (2007) and a cup seller Papstar (2012a). PE foil is used more frequently than PP, hence we use PE as foil material in this study. Data on the manufacturing of these packaging materials were derived from the ecoinvent database. Data for the incineration of the PE foil were taken from the ecoinvent database and adjusted for recovered energy (see incineration PS below). Data for the recycling of the cardboard box originate from (Bergsma et al., 2010). The waste processing of the disposed packaging material was included in the use stage of the cup.

The beverage drinker deposits the used cup in the designated bin, typically one for commingled waste. Next, cleaners collect the disposed cups for handing over to a waste processer. We assumed that the impact of the disposal of the cups is negligible.

The initial incineration situation considered incineration with energy recovery. Data for the incineration of PS were taken from the ecoinvent database. The ecoinvent incineration process specifies recovered energy amounts, but does not credit the incineration process for the recovered energy. Recovered energy is in the Netherlands typically used for electricity and often also for heat production (Otten and Bergsma, 2010). The ecoinvent data for PS incineration were, therefore, supplemented with credits from their specified recovered energy amounts, which are based on an efficiency of 12% electricity and 23% thermal heat recovery (Ecoinvent Centre, 2010). The avoided conventional Dutch electricity and heat from natural gas production, both from the ecoinvent database, were used as credit for the incinerated cups (i.e. avoided allocation by system expansion) in all incineration data sets.

We included PS recycling as an alternative waste processing method in our analysis. Here, similarly to incineration, one data set was used to establish an initial situation for this process. Data on energy and material use for the recycling process of PS were taken from Bergsma et al. (2011). Between 3 and 5% of the PS material gets lost during the recycling process (Arena et al., 2003; Shen et al., 2010). Here a worst case situation of 5% was assumed. The PS cup was credited for the recycled PS based on the avoided production of virgin PS (avoided allocation), but was corrected for the reduced quality of the recycled PS as expressed by its economic value (economic allocation). Prices of PS and recycled PS are very volatile and strongly related to crude oil prices. We used a replacement of 60% (i.e. 1 kg recycled PS replaces 0.6 kg virgin PS) for the initial data set, based on PS prices in February

2012 from industry (PlasticsNews, 2012) and plastic recyclers. Data for the replaced virgin PS are from general purpose PS (GPPS) (Ecoinvent Centre, 2010). Recycled PS can also be credited according to different allocation principles, which are for reasons of readability further elaborated in the results section (see 3.3.2)

Transport distances were based on locations of known production sites. PS producers are located throughout Europe (Eni S.p.A., 2012; Styrolution, 2012; Styron, 2012; Synthos, 2012; Total Petrochemicals, 2012). Manufacturers of disposable PS cups are also distributed among multiple locations in Europe (Huhtamaki, 2012b; Paccor, 2012; RPC Group, 2012). Table 3.2 specifies the average transport distances and transport means from the production sites of PS to all further downstream processes. Other transports, i.e. upstream from the PS producers are already included in the data of PS production. Data for transport means were taken from the ecoinvent database (Ecoinvent Centre, 2010).

Table 3.2: Distances from and to locations and the transport mode used in the LCA of disposable PS cups.

From	To	Distance (km)	Transport means
Producers of PS	Manufacturer of PS cups	500	Lorry 32 t
Packaging material	Manufacturer of PS cups	100	Lorry 16 t
Manufacturer of cups	Distributor cups	500	Lorry 16 t
Distributor cups	Customer	100	Lorry 16 t
Customer	MSWI	150	Lorry 16 t
Customer	Recycler	300	Lorry 16 t

3.2.5 Impact assessment

An impact assessment translates the inventory data into their potential contributions to a range of environmental impact categories. We used the CML Baseline 2001 methodology (Guinée et al., 2002), that quantifies ten impact categories, and supplemented these with the cumulated energy demand (CED) from Frischknecht et al. (2003). The ten impact categories in the CML Baseline 2001 methodology include abiotic depletion (ADP), global warming (GWP 100) (GWP), acidification (AP), eutrophication (EP), ozone layer depletion (ODP), photochemical oxidation (POCP), human toxicity (HTP), fresh water aquatic ecotoxicity (FAETP), marine aquatic ecotoxicity (MAETP), and terrestrial ecotoxicity (TETP). SimaPro 7.3 was used for the calculation of the impact assessment results (PRé Consultants, 2011).

First we assessed the environmental impact for each individual data set and methodological choice. Then, for each process in the life cycle, we calculated the average impact result for that process based on the multiple data sets related to that process. Information about the (market) share of specific data sets in the processes was not available. Advanced statistical data processing with realistic probability density functions was therefore not relevant. Therefore, each data set has equal weight in the calculation of the average results. Next, we determined the spread in result for that process by taking the highest (maximum) and lowest (minimum) impact result from the multiple data sets for that process. Finally, we combined the results from the individual processes into the LCA results for the PS cup. In our approach, the spread was estimated *after* the impact assessment was performed for each individual data sets and modelling choice. This deviates from the conventional approach where uncertainty analysis is performed in the inventory phase.

3.3 Results

3.3.1 Initial LCA

Figure 3.2 shows the contribution to the environmental impact from each process in the life cycle of the disposable PS cup in the initial situation. The PS production and the manufacturing of the cup both contribute considerably in all impact categories. Transport contributes little to most categories and is only clearly visible in ODP (due to Halon emissions during the crude oil extraction for the production of diesel). Also waste processing of the packaging materials of the new cup has minor importance. Incineration of the used PS cup on the other hand does show relevant. A number of impact categories have negative contributions due to the avoided emissions related to the recovered energy. The credit from PS incineration in ODP (primarily due to avoided Halon emissions from natural gas transport) even exceeds the environmental impacts from all other processes.

Figure 3.2 indicates that the production of PS, manufacturing of the cup, and incineration of PS are relevant processes for collecting multiple data sets, since they all contribute more than 15% in at least five impact categories. Transport and the production and waste processing of packaging materials seemed less important. However, we performed a sensitivity analysis to check whether these processes indeed are of minor importance in the total life cycle given the possible uncertainties in our assumptions in the initial LCA.

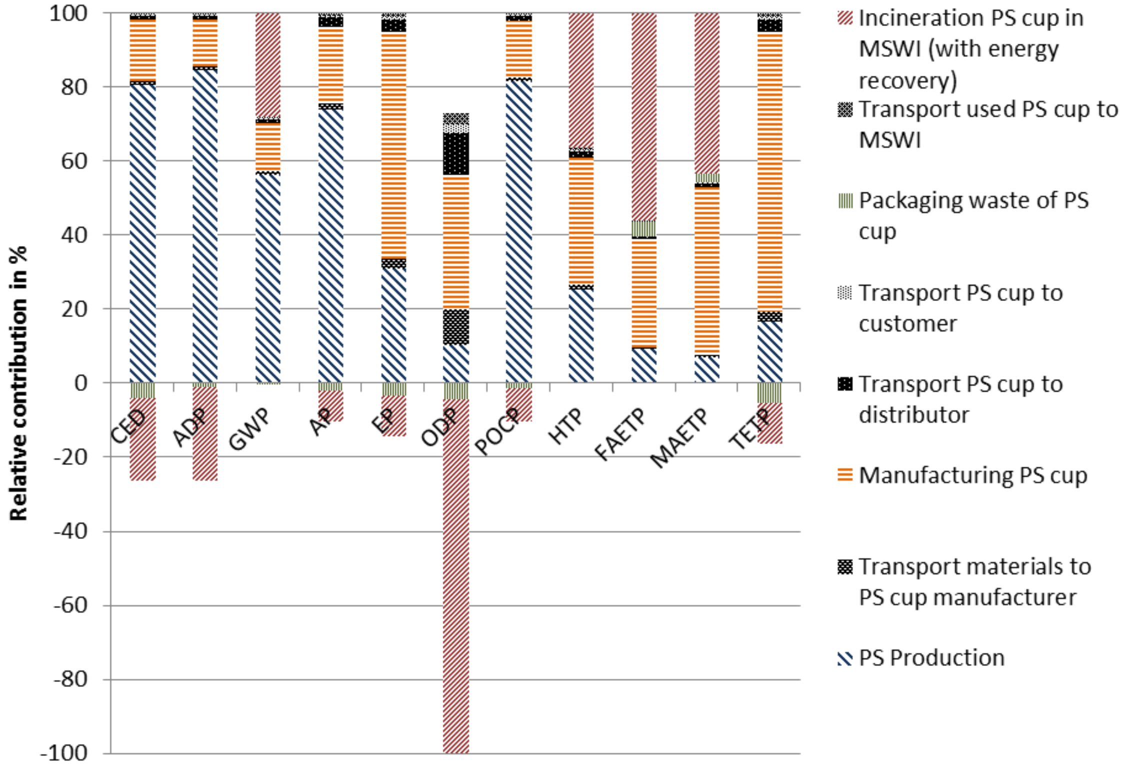

Figure 3.2. Relative contribution of the separate processes to the environmental impact of PS disposable beverage cup (see footnote below Table 3.3 for abbreviations).

Increasing all transport distances in Table 3.2 by 50% raised the results in most impact categories by at maximum 5%. The overall (negative) result in ODP decreased by 50%, but this number was distorted due to the high negative contribution from incineration (see Figure 3.2). Hence transport was not selected for collection of additional multiple data sets.

Production of packaging materials, i.e. PE foil and cardboard box, is included in the cup manufacturing process. Sensitivity analysis for the production and waste processing of the packaging material was performed using the lowest packaging data we acquired (135 g PE and 1084 g cardboard per box of 3000 cups) and the highest ones (240 g PE and 1215 g cardboard). The analysis showed a maximum difference of 7% in the overall results for most impact categories. The results in ODP differ by 17%, but the results in this impact category were again clouded by the high credits of incineration (see Figure 3.2). Production of packaging material was not selected, because the sensitivity analysis did not show its production and waste processing to gain importance in any impact category (except in ODP).

Also the waste processing of the disposed packaging material contributes very little. Sensitivity analysis with the lowest and highest packaging data showed a difference of 0-2% in most impact categories. In ODP the difference was 4%, however the actual increase of the impact was much less because ODP has a big credit from incineration. The waste processing of the packaging has thus minor influence and was not selected for further inquiries.

We also performed sensitivity analysis on two properties of the disposable PS cup, i.e. the composition of the PS mixture and the weight of the cup. A change in the PS mixture from the assumed 50-50% ratio to a ratio of 60% GPPS and 40% HIPS (Ligthart and Ansems, 2007) hardly influenced the overall environmental impacts since the impacts of GPPS and HIPS are very close. An increase or decrease in the weight of the cup by 5% resulted, as to be expected, in a proportional increase respectively decrease in all impact categories. The weight of the cup is thus very important in the absolute quantitative outcome of the PS cups, but does not influence the shares of processes in the absolute outcome. Since this paper is not a comparison between cups of different weight, the weight of the cup was not selected.

In addition to the sensitivity analyses on minor contributors we also evaluated a scenario with recycling as an alternative waste processing option. The initial recycling LCA reveals that both the recycling process and the credits of the recycled PS have a large influence in the impact results.

We selected the following processes for additional data collection based on the results of the contribution and sensitivity analysis for the initial incineration and initial recycling LCA:

- production of PS
- manufacturing of PS cups
- incineration of PS
- recycling process of PS.

The multiple data sets for these processes represent the variability in inventory data.

We also selected two modelling choices for additional data collection:

- waste treatment options (incineration and recycling)
- credit allocation for recycled PS.

3.3.2 Spread per process based on multiple data sets

Next, we collected alternative data sets for each of the selected processes and modelling choices (see Table 3.1). The impact results for the multiple data sets were used to calculate the average results and the spread (maximum and minimum) in the results per process.

Figure 3.3 shows per impact category, for each process, the impact result for the initial data set, the average impact as calculated for all data sets, and the spread from the multiple data sets and modelling choices as used for the selected processes. The process with the highest average contribution in each impact category was set at 100%. All other results were calculated relative to this highest average contribution. The results in Figure 3.3 confirm our contribution and sensitivity analysis. PS production, cup manufacturing and PS incineration remain the major contributors to the environmental impact. Even though their contribution to the impact results is high, their spread in impact results is small in some impact categories. This applies for instance to PS production in the cumulated energy demand (CED) and the abiotic depletion (ADP). The impact categories often contain only one or few processes with high variations in the results. The sources of this high variation in processes (i.e. high compared to the spread in the other processes) are summarized in Table 3.3.

Five data sets were collected for the production of virgin HIPS and GPPS (see Table 3.1). Three data sets are based on European situations and two on USA situations. Differences in impact results for the production of PS were traced back to the origin of the data. USA data contains higher sulphur (di) oxides and barium emissions compared to the European data. These higher emissions in the USA are apparently released during the production of gas, refining of petroleum and pyrolysis of gasoline. It is not clear whether these differences in emissions stem from factual deviations between Europe and the USA (e.g., gas extraction techniques (barium emissions), sulphur content of the gas, flue gas cleaning techniques for sulphur dioxide emissions), or from uncertainties in emission factors.

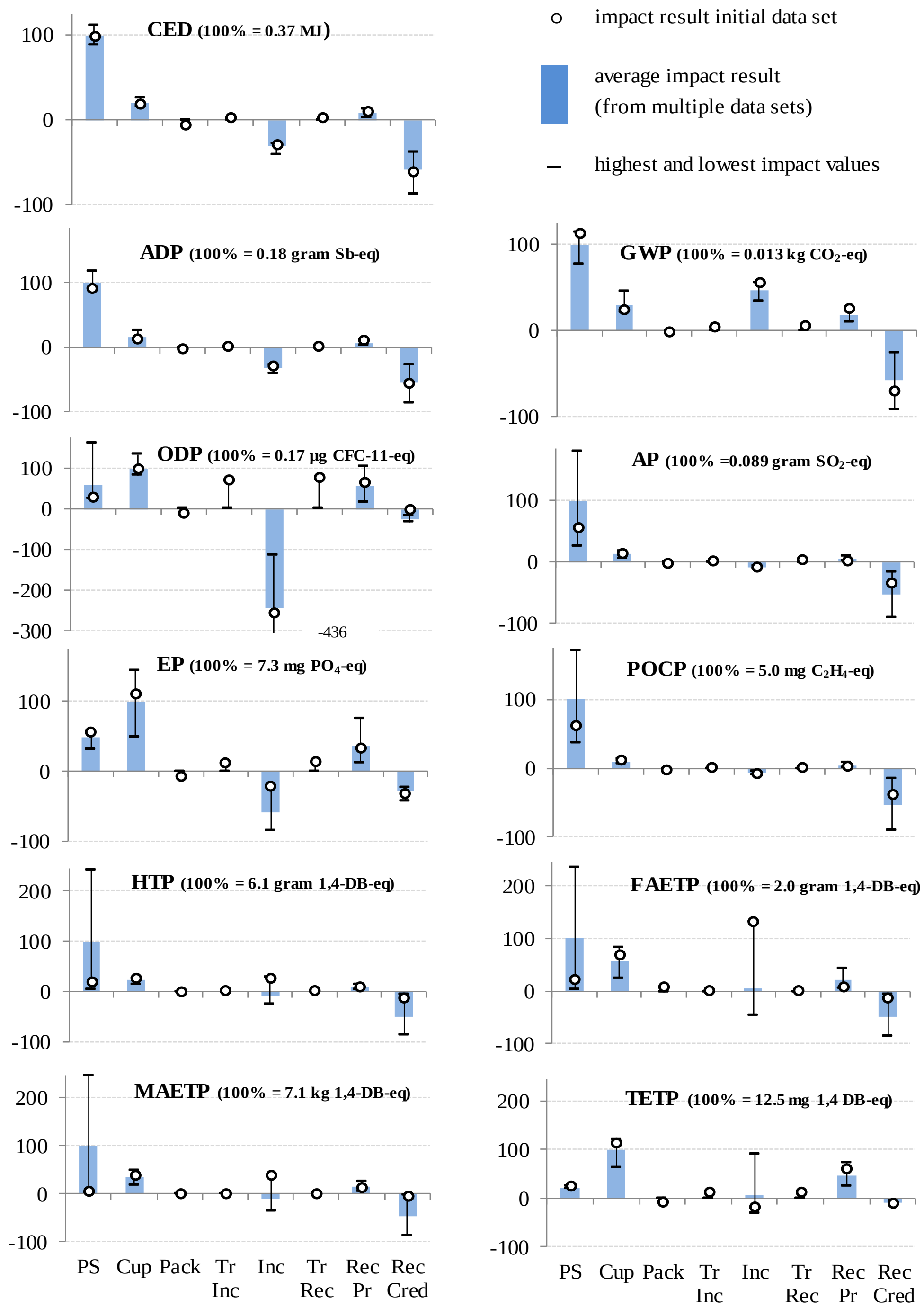

Figure 3.3. Environmental impact of the processes of the life cycle of a disposable PS beverage cup. The graph shows the impact as calculated in the initial LCA, and for the multiple data sets as indicated in Table 3.1. For the multiple data sets, the impacts are shown as average of all data sets, and as a range where the minimum and maximum are the lowest and highest values, respectively, as calculated for the set of data sets. Abbreviations graph: PS = production of PS, Cup = cup manufacturing, Pack = waste processing packaging material, Tr Inc = transport in incineration LCA, Inc = incineration PS in MSWI, Tr Rec = transport in recycling LCA, Rec Pr = recycling process, Rec Cred = credits for recycled PS (See footnote below Table 3.3 for abbreviations of impact categories).

Besides the differences in PS production location, the time frame of the PS production also appeared influential. Recently updated data (from 2012) compared to data from 2002 on PS production in Europe showed a great improvement. The environmental impact in GWP, EP and POCP decreased by more than 30% and in AP more than 50% (PlasticsEurope, 2012). The decrease in impacts is the result of improved production processes, changes in the type of energy used, improved control of emissions, and new data for benzene production. PS production data are thus highly influenced by the origin of the production site and the time of data collection. The used data sets (see Table 3.1) comply with the time horizon set in the scope definition (see section 3.2.4). The geographical scope was also met, since PS used in Europe is mainly produced in Europe but a small amount is imported from the USA.

The range in impact results for cup manufacturing follows from different quantities and types of energy used (some used only electricity, others also heat), and could be related to the type of process used (inline or off-line). Similarly, credits for recovered energy from PS incineration differ in amount and type of energy credited (only electricity or also heat). Incinerators report different amounts of heavy metals emissions, and this explains the variation in the toxicity impact categories.

PS incineration may be replaced by PS recycling as a waste processing method. Data sets for the crediting of PS recycling use different allocation procedures that we also included in this study. Some allocation procedures start from avoided allocation by replacing recycled PS with virgin GPPS (based on the average of the five data sets in Table 3.1). To correct for the drop in quality of recycled compared to virgin PS, the initial data set allocated 60% of the recycled PS to the cups, based on the present economic value of virgin GPPS as obtained from industry (PlasticsNews, 2012) and plastic recycling companies. We included three alternative ways of crediting for the recycled PS, which were used in existing studies. Ligthart and Ansems (2007) used an economic allocation of 50%, whereas Ligthart and Ansems (2004) used an allocation of 90%. The allocation approach can also be based on avoided allocation from other materials that can be substituted by recycled PS. Bergsma et al. (2011) substituted recycled PS for concrete, wood (azobe) and virgin polypropylene. Data for these materials were taken from the ecoinvent database.

Table 3.3: Sources of variation in impact results in the life cycle stages of the PS disposable cup.

Life cycle phase	PS Production	Cup manufacturing	Incineration cup	PS recycling process	Credits for recycled PS
Impact category:					
CED[a]					Crediting allocation
GWP[a]					Crediting allocation
ADP[a]					Crediting allocation
AP[a]	SO_2 and SO_X emissions				Crediting allocation
EP[a]		Energy use and energy source	Credited energy and COD emission	Energy use and energy source	
POCP[a]	SO_X and CO emissions				Crediting allocation
HTP[a]	Ba emission				Crediting allocation
FAETP[a]	Ba emission		V, Cu, Co, Ba emissions and credited energy		
MAETP[a]	Ba emission		V, Ba, Co, Be emissions and credited energy		Crediting allocation
TETP[a]		Energy use and energy source	Hg emissions	Energy use and energy source	
ODP[a]	HCFC-22 emissions		Amount of avoided heat production	Amount of heat used	

[a] CED = cumulated energy demand, GWP = global warming potential, ADP = abiotic depletion potential, AP = acidification potential, EP = eutrophication potential, ODP = ozone layer depletion potential, POCP = photochemical oxidation potential, HTP = human toxicity potential, FAETP = fresh water aquatic ecotoxicity potential, MAETP = marine aquatic ecotoxicity potential, TETP = terrestrial ecotoxicity potential.

PS recycling also shows to be of importance. Figure 3.3 separately presents the environmental impacts of the recycling process and of the credits for the cup from the recycled PS. The variation in impact results of the recycling process stems from various amounts and types of energy used (electricity and/or heat). Variation in PS crediting is related to different crediting allocation procedures. The allocation of 90% PS (Ligthart and Ansems, 2004) provides more credit than 60% (initial situation) and almost twice as much as 50% (Ligthart and Ansems, 2007). The substitution by concrete, wood and PP (Bergsma et al., 2011) shows often the least credits, because the environmental impact of these individual products are often less than PS. If PS production contributes substantially in an impact category, then consequently the spread in recycled PS results will be clearly visible.

The above results demonstrate that variability in data sets, choices in waste processing options and allocation methods all have an important role in the considerable spread in the results for the relevant processes. The spread in the PS production is often considerably higher compared to the spread in the other processes, and relates to the origin of the production site and the time of data collection.

3.3.3 Total LCA results and the spread in total LCA results

We analysed the total impact of PS cups by impact category (Figure 3.4) by combining the average contributions from the separate processes to calculate the overall average contribution in each impact category accumulated over all processes. The spread in each impact category for the LCA results is the difference between the highest and lowest contribution. In Figure 3.4, the average contribution in each impact category was set at 100% for a PS cup with incineration as waste processing option (incineration LCA). The results of the LCA with PS recycling were next expressed relative to the results of the incineration LCA.

The calculated average environmental impacts of PS cups are generally lower when assuming recycling than when assuming incineration (for CED, GWP, ADP, AP, POCP, HTP, FAETP, and MAETP). The majority of these impact categories, on the other hand, show a smaller spread for the incineration LCAs than for the recycling LCAs. Their spreads moreover overlap. The average incineration LCAs on the other hand perform better than the average recycling LCAs in EP, TETP and ODP. Also the spread in these impacts categories is large and overlaps for the incineration and recycling LCAs (the spread for ODP is again less meaning full).

The recycling LCAs tend to make smaller average contributions than the incineration LCAs, but the spread in all impact categories is large and overlap for both waste management options. The smallest spread, calculated as difference between highest and lowest percentages, occurs in the incineration LCA for CED (53%), GWP (45%) and ADP (63%). The larger spreads in CED (104%), GWP (78%) and ADP (126%) in the recycling LCA are mainly higher due to the crediting alternatives of the PS. The largest spread is calculated for ODP. The height of this spread is unjust, as mentioned before, because it is compared to the very small absolute average value of ODP in the incineration LCA. Except for TETP, the difference between the highest and lowest percentages in the toxicity categories exceeds 200%. This is too large to support any meaningful conclusions.

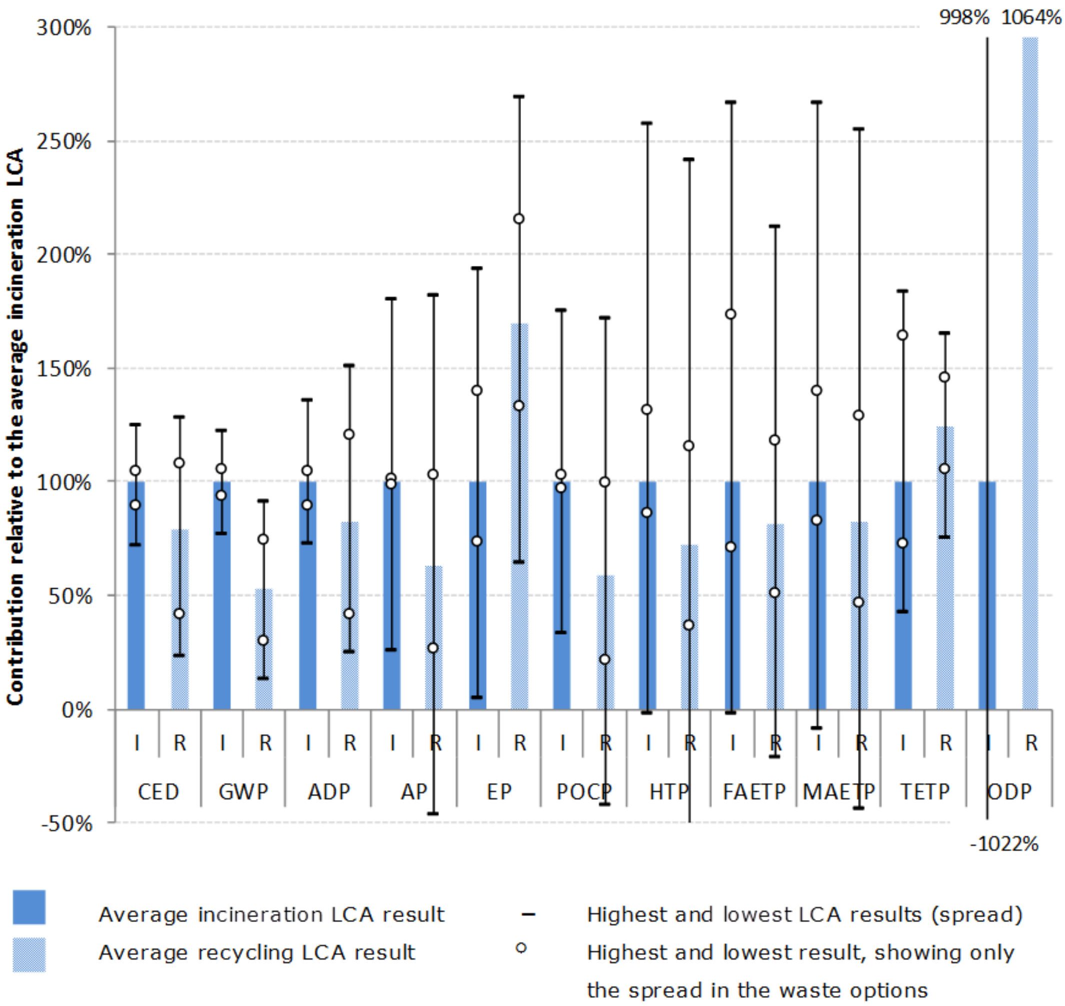

Figure 3.4. Total impact of a PS cup over the entire life cycle. Bars represent the average impact results of the LCA with disposal option incineration (I) or recycling (R). All impact results are calculated relative to the average impact of the LCA with incineration. The dashes show the spread (highest and lowest value) in the total LCA results. The circles show the spread in LCA only due to the spread in the waste processes. Note that the range for ODP exceeds the Y axis scale: for the incineration case the highest and lowest values are 998% and -1022%, respectively; for recycling the average value is 1064%, and the highest and lowest values are 1783% and 684%, respectively. For explanation of the impact categories see the footnote below Table 3.3.

This study considered the two waste processing options incineration and recycling, that customers to some extent can choose between. Customers normally have little or no influence, however, on the preceding processes leading to the formation of the cup. Figure 3.4 shows also the spread within the waste processes only. This spread is smaller compared to the spread from the total life cycle, and shows a preference for recycling above incineration in GWP.

Simple mathematical combinatorics was applied additionally to create every possible LCA combination from the multiple data sets. Five data sets for PS production, five data sets for cup manufacturing and four incineration data sets produced 100 possible combinations for the incineration LCA. We created a histogram of these combinations to see what type of distribution the 100 possible LCA results represent. The distribution of the impact results displays a normal distribution in several impact categories such as CED (see Figure 3.5a), but a rather polarized distribution for other impacts such as AP (see Figure 3.5b) or HTP (Figure 3.5c). The segmentation in Figure 3.5b and 3.5c can be traced back to the origin of the PS (USA or Europe). The clearly different types of distribution across impact categories also means that presenting the uncertainty interval using statistical average and standard deviation would not be appropriate in this situation.

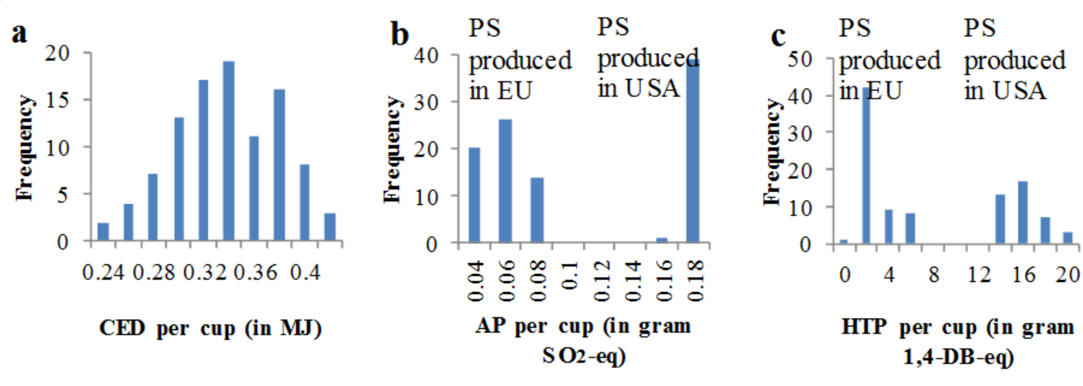

Figure 3.5. Distribution in the 100 possible LCA results of PS cups for the impact categories: a) cumulated energy demand (CED), b) acidification potential (AP), and c) human toxicity potential (HTP). The waste processing option is incineration.

3.4 Discussion

This paper evaluated the spread in LCA results for a typical disposable PS cup due to the use of multiple inventory data sets for processes, alternative waste treatment options (modelling option), and different allocation procedures to credit for PS recycling (modelling option). Before discussing the methodological benefits and drawbacks of the followed approach, we discuss the robustness of the case study that served as a vehicle to explore this in LCA relative new methodological approach for dealing with variability across data sets, and uncertainty in modelling choices.

3.4.1 Discussion of the case study of PS cups

A considerable uncertainty can exist in the characterisation factors which are used for the calculation of the environmental impact, especially in the human- and ecotoxicity impact categories. The fate-effect modelling of toxicity of substances is based on simplified environmental models, which include uncertainties itself. Use of site-generic factors for impact categories as acidification and eutrophication also may lead to considerable uncertainty in impact assessment results (Potting and Hauschild, 2006). The uncertainty in the characterisation factors can be very high (Huijbregts, 1998a), but are not addressed in this study.

Both the use of multiple data sets and modelling choices show to lead to a considerable spread in impact assessment results. The results from these different data sets and modelling choices, however, systematically point to the same processes as the main contributors to all impact categories (PS production, cup manufacturing, PS incineration and PS recycling).

Inventory data from the different sets in this study (see Table 3.1) may vary considerably in the number of included inventory items. Several data sets moreover provided inventory data aggregated to “resources” and “emissions”, and also regularly accumulated the data in cradle to gate processes. This makes it nearly impossible to know which (sub) processes and emission data are included. These results show the importance of the data quality of the inventory data. The introduction of product category rules (PCRs), which would provide guidance on the collection of data, could partly enhance the quality of inventory data at specific sites. Results from multiple data set approaches, as followed in this study, may furthermore function as a benchmark to evaluate the completeness and believability of data from specific sites.

The smallest variation in results of this study was found in the energy related impact categories CED and ADP. Weidema et al. (2003) similarly found that the smallest differences in inventory data were located in the energy consumption (CED) and material use, and the biggest variation were seen in the emissions. Weidema et al. (2003) pointed out that monitoring of emission data differs between companies. Some emissions might be measured more precisely in certain facilities compared to others. Differences in inventory data and background data for plastic materials lead to dissimilar impact results in especially the toxicity categories (European Bioplastics, 2012). This could explain the large spread in the toxicity categories.

The discrepancy in the toxicity impacts of PS production can be traced back to differences in the use of resources in the USA and EU. Steam cracking is one of the main processes in the PS production. Europe uses naphtha from crude oil as the main feedstock for steam cracking. The USA mainly uses ethane and propane (both by-products from oil and gas production) in the steam cracking process (Neelis et al., 2008). Consequently, this means the use of different amounts of resources for the production of the same product.

PS production, a main contributor in the LCA results, shows a high spread in several impact categories, caused by differences in the country origin of the PS production and the time period of the PS data collection. The inclusion of the USA data reveals the importance of the geographical coverage of the data. Europe imports only a very small percentage of PS from the USA, according to European statistical data (Eurostat, 2013). This implies that the use of USA PS production data may not be appropriate in the manufacturing of the PS cup in Europe. Omission of the USA data sets will lead to a decrease in the spread in the PS production results, which will then show only the variation in the data collecting period and technological difference between the European data sets.

Variation in impact results is not limited to inventory data alone. The crediting of PS recycling, i.e. the followed allocation procedure, turns out to have an important effect. This study used avoided allocation by system expansion in combination with allocation based on economic value. Avoided allocation is used to credit the PS cup for substitution of other materials (PS, PP, wood and concrete). Crediting for the recycled PS can follow very different allocation procedures based on very different considerations. Some allocation procedures for example also credit the next product in the PS cascade, or split credits between several cascade levels (Ekvall and Tillman, 1997). These and other allocation principles, such as the cut-off approach, can lead to entirely different results (Frischknecht, 2010). The crediting allocation choice is thus a very influential factor and should be carefully selected in accordance with the objective of the study. ISO guidelines (2006a) include several allocation principle, and express a preference but do not prescribe which one to use. The differences in crediting allocation principles are thus a topic for further study.

PS incineration received credit for the recovered energy (electricity and/or heat). Electricity production in the Netherlands consists mainly of fossil fuel based power plants. As such, the credits for the incinerated PS were based on the avoided use of fossil fuels. The credits will decrease if the Dutch electricity would be generated by renewable energy (i.e. wind energy, solar energy, or biomass). Michaud et al. (2010) confirm the dependency of the credits on the mixture of energy sources. A decrease in credits for incineration, and thus an increase in impact results, affects the comparison between incineration and recycling. The evaluation and preference between the waste treatment options therefore depends significantly on the electricity production of the country.

The environmental impact from PS production continuous to improve (PlasticsEurope, 2012). Enhanced PS production leads to fewer credits for recycled PS. Although it sounds counter-intuitive, improved PS production could point to incineration as a preferred waste treatment instead of recycling.

Recycling is included as an alternate waste treatment option. The two reviewed LCAs in the WRAP study (Michaud et al., 2010) both report a preference for PS recycling above incineration for the impact categories CED, GWP, ADP, AP and POCP . These findings are consistent with our average LCA results. The two studies do not agree on the preferred waste treatment in the impact category EP. It should be noted, however, that recycling is often less feasible for practical reasons. Recycling of PS requires a clean and homogeneous waste stream. Contamination with other material (such as other plastics, paper, metal cans, coffee or thee, food remnants, cigarette butts etc.) can interfere with the recycling process and deteriorate the quality of the recycled PS. A waste stream which contains too many impurities will be sent to the incineration. If the customer chooses recycling as waste treatment, the customer needs to provide separate designated bins for the collection of the PS cups and needs to arrange transport of the cups to a recycler.

3.4.2 Comparison of our approach to other uncertainty analysis methods

The spread caused by multiple data sets basically represents uncertainty in LCA results due to variability in inventory data. The spread from different allocation procedures or waste treatment options basically represents uncertainty due to modelling choices. LCA typically relies on scenario analysis to evaluate uncertainty from modelling choices and on Monte Carlo type of uncertainty analysis to statistically evaluate uncertainty from variability in inventory data. An important added value of the approach in this paper is the combined approach of handling both uncertainty from variability in inventory data sets and due to modelling choices.

Monte Carlo analysis uses stochastic information to determine confidence intervals (Heijungs and Huijbregts, 2004). This means that inventory data for a process require average values, distribution and confidence intervals. For a specific process on a specific production site, these values represent the uncertainty in measurements at that location. The stochastic information for an average or typical process usually relies on a survey from different production sites of the same process from which then per inventory item an average and distribution is calculated. Averaging inventory data from multiple data sets can be challenging or even infeasible if the format of the inventory data is not the same. For instance, detailed aggregated inventory data on resources and emissions, versus a list of used materials and energy sources (as is common in environmental product declarations (EPD)). This problem can arise if, as in our case for a typical product, multiple data sets from different sources are used. Monte Carlo analysis would not be very useful in this situation.

A more important problem with the use of Monte Carlo analysis, which is based on averaging data from multiple data sets, is that it also ignores the correlation between data inventory items within one data set for an individual operator, e.g., the relationship between type of energy and its emissions. These correlations are conserved, as done in this study, by calculating the spread *after* impact assessment has been carried out.

3.4.3 Added value of the approach

Multiple data sets were gathered for the main contributing processes. Guinée et al. (2002) emphasized that inventory data should be representative for the studied system and should be complete, accurate, appropriate and not obsolete. This applies to the inventory data used in this study aiming at a typical PS cup as presently used in vending machines in the Netherlands. This LCA shows the possible mistake from using specific data, i.e. anecdotic data, to represent an average or typical product by evaluating the spread in LCA results for a typical disposable PS cup for which no specific suppliers are known.

If the LCA here was aiming at a specific cup from a specific manufacturing facility, it would have been more appropriate to use specific data from single processes. The multiple data set and modelling approach can also then be useful, namely for serving as a benchmark for specific LCAs by hinting to possible mistakes in specific data obtained for specific sites. This may include possible discrepancies in results which might be related to differences in production methods, assumptions, choices, but also to missing inventory items. The multiple data sets and modelling approach can also serve as a benchmark for comparison purposes of specific product LCA with other products.

The multiple data set approach in this study uses impact results instead of the inventory results for calculating averages and spread, and is thus independent of the format in which inventory items are supplied. Though this was not at stake in this LCA, this also enables inclusion of incidental sets for which no inventory data are available, but only impact data.

The review of the disposable cup LCA studies by van der Harst and Potting (2013a) shows the inconsistency in outcomes between the studies and identified sources for this discrepancy. The use of multiple data sets and modelling choices incorporates some of these sources, which is reflected as spread in the results. The outcome can be less clear, but correlations within inventory data sets are respected, and the certainty of the outcome is higher.

3.5 Conclusion

This paper systematically explored the influence of multiple data sets and multiple choices on the environmental impact. The systematically use of different data sets and modelling choices revealed their influence on the Life Cycle Assessment (LCA) results. The life cycle of disposable polystyrene (PS) cups was examined as a case study. Two disposal options were considered: incineration and recycling.

The applied approach started with an initial LCA of the PS cup with incineration as waste treatment and one data set for each life cycle phase. An additional LCA was made with recycling as waste treatment option. The major impact contributors in the life cycle of disposable PS cups are PS production, cup manufacturing, PS incineration, PS recycling and crediting recycled PS. Additional data sets were gathered for these processes. Average results for the separate processes were calculated based on the multiple data sets. The spread in results (i.e. the difference between the highest and lowest values) was also established for each process. Finally, the average results and their spread of the separate processes were combined into the total LCA results.

The results show that variability in PS production, cup manufacturing, incineration and PS recycling all lead to a considerable spread in impact results. Causes of the spread were traced back to differences in resources, amount of used energy, energy sources, and release of emissions. The difference between PS produced in the USA and Europe especially create a large variation in results, but also the difference between older and newer European data contribute to the spread. Choices in crediting options of recycled PS also create variation in impact results. More research is needed to investigate additional crediting approaches. The spread in the separate life cycle processes propagated into the uncertainty of the total LCA results. The impact categories cumulative energy demand and the abiotic depletion potential show the least spread, while the toxicity categories exhibit the greatest variation.

Comparison of the two waste treatment alternatives (incineration and recycling) shows on average a better performance for recycling in eight impact categories (cumulated energy demand, climate change, abiotic depletion, acidification, photochemical oxidation, human toxicity, fresh water aquatic ecotoxicity, and marine aquatic ecotoxicity). Incineration performs better in eutrophication, terrestrial ecotoxicity and ozone layer depletion. For all impact categories, however, there is an overlap in spread between the recycling and incineration LCA results. This overlap prevents a decisive conclusion on the most preferred waste treatment.

Calculation of the spread was performed on the impact results and not, as is common in Monte Carlo analysis, on the inventory data items. This preserves any relation which exists within the different inventory items in the data sets. It also allows to include data sets with different aggregation levels of information, or even impact data if inventory data are unavailable.

The presented study shows the contribution of separate processes, the spread in their contribution, and their influence on the spread in the total LCA results. The approach enables combining uncertainty due to variability and uncertainty due to choices in a systematic way. The calculation of the variability between data sets is particularly useful in

LCAs of typical or generic products, but can also be applied in the comparison of specific product LCAs with other product. The approach can effectively be used to provide more unambiguous and robust LCA results, specifically in typical LCAs. The results may be less easy to perceive, but the outcome is more certain. This provides decision makers with more trustworthy information.

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4 Multiple data sets and modelling choices in a comparative LCA of disposable beverage cups

Abstract
This study used multiple data sets and modelling choices in an environmental life cycle assessment (LCA) to compare typical disposable beverage cups made from polystyrene (PS), polylactic acid (PLA; bioplastic) and paper lined with bioplastic (biopaper). Incineration and recycling were considered as waste processing options, and for the PLA and biopaper cup also composting and anaerobic digestion. Multiple data sets and modelling choices were systematically used to calculate average results and the spread in results for each disposable cup in eleven impact categories.

The LCA results of all combinations of data sets and modelling choices consistently identify three processes that dominate the environmental impact: (1) production of the cup's basic material (PS, PLA, biopaper), (2) cup manufacturing, and (3) waste processing. The large spread in results for impact categories strongly overlaps among the cups, however, and therefore does not allow a preference for one type of cup material. Comparison of the individual waste treatment options suggests some cautious preferences. The average waste treatment results indicate that recycling is the preferred option for PLA cups, followed by anaerobic digestion and incineration. Recycling is slightly preferred over incineration for the biopaper cups. There is no preferred waste treatment option for the PS cups. Taking into account the spread in waste treatment results for all cups, however, none of these preferences for waste processing options can be justified. The only exception is composting, which is least preferred for both PLA and biopaper cups. Our study illustrates that using multiple data sets and modelling choices can lead to considerable spread in LCA results. This makes comparing products more complex, but the outcomes more robust.

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4.1 Introduction

Disposable beverage cups are convenient utensils for serving hot and cold liquids. They are cheap, easy to use, and require no maintenance. Most disposable cups are made of plastic or paper. Plastic can be made from fossil fuels (i.e. oil and natural gas; petro-plastic), or from a renewable resource as corn, tapioca, sugarcane or sugar beets (biobased plastic). Petro-plastics used for disposable cups include polystyrene (PS), polypropylene (PP), polyethylene terephthalate (PET) and recycled PET (RPET). Polylactic acid (PLA) is the most used biobased plastic in disposable cups. Paper beverage cups need a lining, either from petro-plastic or biobased plastic, to prevent liquid from intruding the paper.

PLA cups and paper cups with a PLA lining, both further referred to here as biocups, are both produced from renewable resources. Renewable resources for PLA include corn, tapioca, sugarcane and sugar beets (Corbion Purac, 2013b; Galactic, 2011; NatureWorks LLC, 2011). PLA cups presently on the market only consist of PLA made from corn. Disposable beverage cups made from petro-plastic are often associated with an unnecessary use of fossil fuels and the production of waste (Butijn et al., Unpublished results). The biocups carry the compostability label (EN 13432), which indicates that the biocups disintegrate into compost within 12 weeks under industrial conditions. Petro-plastic cups made from PS, PP, or (R)PET are not compostable. The general public often considers compostable packaging material as better for the environment compared to petro-plastic materials (Jager, 2008).

Van der Harst and Potting (2013a) recently evaluated ten (comparative) life cycle assessment (LCA) studies on disposable beverage cups. Life cycle assessment (LCA) is a renowned tool for measuring the environmental performance of a product from its cradle to its grave (i.e. product system or life cycle). The selected studies in the review by van der Harst and Potting (2013) shared climate change as a common impact indicator. Van der Harst and Potting (2013a) quantitatively explored the variation in results within each cup material by calculating the ratio between the highest and lowest climate change values, and found a ratio of 1.7 for PLA cups, 3.4 for petro-plastic cups up, and 20 for paper cups. There was no consistency among the studies in their conclusion on which cup has the smallest climate change impact. Van der Harst and Potting (2013a) identified possible sources for the variation in outcomes. These were properties of the cups (e.g. material choice and weight), production processes, energy sources (e.g. fossil or renewable), and waste processing options (i.e. landfilling, incineration, recycling, composting).

Different waste treatments can lead to different LCA results of disposable cups (Häkkinen and Vares, 2010; Pladerer et al., 2008). This also holds for assumptions made about the waste process in the LCA (Franklin Associates, 2011), and the applied credits for recycled material (Franklin Associates, 2009a; Ligthart and Ansems, 2007). Other studies reported similar observations. An extensive study for Waste & Resource Action Programme (WRAP) found no agreement on the preference for plastic recycling or plastic incineration (Michaud et al., 2010). The preferred waste treatment depended on the efficiency of the incinerator, the substituted energy mix, and the credited material (Michaud et al., 2010). LCA results for paper recycling and paper incineration depended on the virgin paper production techniques, recycling technologies, incineration plant characteristics, energy use and system boundaries (Ekvall, 1999; Merrild et al., 2008; Wenzel and Villanueva, 2006). There was no unanimous preferred waste treatment for paper and cardboard (Michaud et al., 2010; Wenzel and Villanueva, 2006).

Van der Harst and Potting (2013a) additionally identified the use of different data sets as a possible source of discrepancies in LCA results. Most of the reviewed LCAs on disposable cups used only one data set per life cycle process. Data for a specific cup, made by a specific manufacturer on a specific production location can include uncertainty due to, for instance, temporal variation. For a non-specific cup, i.e. an average or typical cup, differences among manufacturers, production processes, and locations can lead to variability in inventory data. Representing an average process by only one specific data set ignores this variability among processes.

Variability among processes, due to different locations and time frames, can also influence the LCA results for PS production (van der Harst and Potting, 2014). The kind of biomass used and the applied production technologies influenced the LCA results of biobased materials (Weiss et al., 2012). Climate change results for the production of paper depended on the transport distance and means of the wood, the pulping techniques (chemically versus mechanically) and their associated type and amount on energy use (González-García et al., 2009; Manda et al., 2012; Merrild et al., 2008).

Van der Harst and Potting (2014) explored a new approach that incorporates and translates above variability in inventory data into a spread in LCA results by purposely applying multiple data sets. This approach also allows inclusion of multiple modelling choices, for instance different allocation options or waste treatment methods. The approach was previously used for a single product, i.e. a typical disposable PS cup (van der Harst and Potting, 2014). In this study we will apply the same approach in a comparative LCA.

The purpose of this study is to compare the environmental impact of disposable petro-plastic beverage cups with biocups through LCA using multiple data sets and modelling choices in line with van der Harst and Potting (2014). We included the following three cup materials: 1) PS as a representative for petro-plastic cup (Dispo International, 2012; Huhtamaki, 2012a; Krings & Schuh OHG, 2012; Papstar, 2012a), 2) PLA and 3) paper with a bioplastic liner (i.e. biopaper). The bioplastic and biopaper cups, i.e. biocups, are made from renewable sources and are compostable. Disposable beverage cups will in the rest of the document be referred to as disposable cups or cups.

4.2 Methods and means

4.2.1 Life cycle assessment (LCA)

LCA quantifies the environmental impact of a product throughout its life cycle. The life cycle runs from the extraction of resources, the production of materials and manufacturing of the product, the consumption or use of the product, up to and including the waste processing of the product. The procedure for performing an LCA consists of four main phases (ISO, 2006a). These phases are 1) goal and scope definition, 2) inventory analysis, 3) impact assessment, and 4) interpretation.

The introduction already specified the goal as comparing disposable cups from PS, PLA and biopaper by using for each of them multiple data sets and multiple modelling choices. The Netherlands is the focus area of this study. This puts limits on the possible waste treatment options of the disposed cups, as discussed in section 4.2.3. The scope of an LCA sets the methodological framework for the whole study. Several scope topics are explained in detail in the next subsections (i.e. functional unit, the product system with its boundaries, the data collection and data processing procedures for inventory analysis, and impact assessment). Inventory analysis quantifies all environmental inputs and outputs throughout a life cycle, and impact assessment translates these into their potential environmental impact. Interpretation evaluates results from inventory analysis and impact assessment in order to draw conclusions and recommendations in relation to the defined goal of an LCA.

We used a systematically approach of multiple data sets and modelling choices to evaluate the environmental impact of the cups (see van der Harst and Potting (2014) for more detail).

The approach calculates average impact results for a process, as well as the spread in these results, based on impact results from the multiple data sets. This is different compared to the mainstream approach. Traditional LCAs first determine average inventory data, distribution and confidence intervals, and next calculate the impact results and their spread based on the stochastic information. The approach here calculates spread based on the impact results, i.e. *after* impact assessment. This keeps the correlation intact between data within one inventory set. It allows for the inclusion of data sets with different formats (e.g. materials and energy use versus detailed data on resources and emissions). The approach also facilitates the inclusion of impact data if inventory data are unavailable.

The multiple data set approach comes down to:

- Making an initial LCA with one inventory data set for each process within the defined system boundaries
- Using contribution and sensitivity analysis to identify processes with major influence on the results of the initial LCA
- Collecting additional multiple inventory data sets for all processes with an influential contribution
- Applying these multiple data sets in a next LCA, together with multiple model choices, e.g. including multiple waste processing methods and multiple allocation principles for recycling
- Calculating average impact results and their spread (the highest and lowest value) for each life cycle process based on the multiple data sets and modelling choices
- Calculating and presenting the impact results and their spread (the highest and lowest value) for the total LCA.

The software program SimaPro 7.3 (PRé Consultants, 2011) was used to convert all inventory data sets and modelling choices into amounts per functional unit and next to calculate the subsequent impact results.

4.2.2 Functional unit

A comparative LCA requires a fair basis of comparison for the products involved. The functional unit provides such basis by defining their shared functionality as detailed as possible in quantitative and qualitative terms (Weidema et al., 2004). The quantitative part allows expressing environmental inputs and outputs for each product in the same functional unit. The qualitative part may constrain the product alternatives included in a comparative

LCA, because all product systems in a comparative LCA must comply with the functional unit. A functional unit including the disposability of cups, for example, excludes reusable cups from the comparison.

The functional unit in this study is the provision of a disposable beverage cup fit for serving 180 ml hot drinks by vending machines, as commonly used in organizations such as companies, offices or schools in the Netherlands. The disposable cups thus need to be suitable for vending machines with automatic cup supply. The drinks are in this type of organisations consumed at the premises and the empty cups disposed in collection bins at the same location.

The three selected cups comply with the defined functional unit. The white PS cup was chosen, as this petro-plastic cup is frequently used (Dispo International, 2012; Huhtamaki, 2012a; Krings & Schuh OHG, 2012; Papstar, 2012a). The PLA cup and paper cup with a bioplastic lining (biopaper) were selected as an alternative to the PS cup.

PLA beverage cups presently on the market are suitable for cold liquids only (PLA cold cups). These cups are typically made from PLA based on corn (NatureWorks LLC, 2011). Thermostable PLA beverage cups for hot liquids have been developed recently, but were at the time of writing this paper (end of 2013) not yet commercially available (Corbion Purac, 2013a; Purac Bioplastics, 2012; Wageningen UR, 2012). There are as yet no data available for thermostable PLA. We took PLA cold cups as the best possible approximate for the thermostable PLA cup (PLA hot cup). The PLA hot cup is expected to become available soon (Corbion Purac, 2013a; Wageningen UR, 2012).

Plastic and paper cups are claimed by cup producers and vending machine distributers to be both suitable for use in vending machines with an automatic cup dispenser (Autobar, 2012; Maas, 2012; Moonen, 2012b). The two biocups (PLA and biopaper) are made from renewable material (corn and wood respectively) and are compostable.

The weight of the cup is an important property since it influences the amount of resources, notably material and energy, used in all processes. The weight of the PS cup varies among cup manufacturers and ranges from 3.8 to 4.4 g according to cup sellers. We chose 4.2 g as representative for the weight of the average PS cup. The PLA hot cup is not commercially available yet and its weight had to be estimated. Such estimate can be based on the density of the materials, or on other material properties. PLA has a density of 1240-1270 kg/m^3 (UL IDES, 2012) versus 1050 kg/m^3 for PS (PolymerProcessing, 2012), hence PLA is 18 to 21% heavier than the same volume of PS. Companies producing cups and clamshells from both PS and cold PLA indicate a 15 to 20% heavier weight for the PLA items compared to the PS ones (Franklin Associates, 2006, 2011). The weight of PLA yoghurt cups, on the other hand, is slightly less than PS yoghurt cups due to the material properties of PLA (Kauertz et al., 2011). We assumed in this study the same weight for thermostable PLA and PS cups, i.e. 4.2 g. The weight of the (bio)paper cup varies between 4.8 and 6.2 g, depending on the cup manufacturer. We choose 5.6 g as a representative weight for the average (bio)paper cup. Sensitivity analysis on the weight of the cup is necessary to show its influence on the environmental impact.

4.2.3 Product system and boundaries

We focussed our study on the use of disposable cups in the Netherlands. The cups have thus to be available on the market in the Netherlands. The use and disposal phase of the cup both take place in the Netherlands, but cup production can take place elsewhere. Disposable cups are designed to hold liquids for human consumption and thus need to comply with the Dutch and European regulations on materials intended to come in contact with food (European Commission, 2004). One requirement states that the used material needs to be traceable. Disposable cups are therefore only produced from virgin material or traceable post-industrial material.

The life cycle of PS cups starts with the extraction of fossil fuels oil and natural gas as the basic resources for PS production. PLA can be made from renewable materials such as corn (NatureWorks LLC, 2011), sugarcane, tapioca (Corbion Purac, 2013b), or sugar beets (Galactic, 2011). The technique for thermostable PLA, as required in the PLA hot cup, has just become available and is not yet commercially applied. We assume the production of thermostable PLA to be similar as for “cold” PLA, since no information was available for the production of this thermostable PLA. PS and PLA hot cups are both thermoformed from virgin granulate.

Wood serves as resource for the paperboard used in the biopaper cups. The paperboard is produced from wood (virgin material) to ensure a clean and uncontaminated material as is required by law (European Commission, 2004). A layer of bioplastic is applied to the

paperboard as a liquid barrier, forming biopaperboard. The biopaper cup considered in our analysis is manufactured in a sequence of punching, folding, and gluing.
Disposable cups, whether they are made of PS or PLA or paper, are usually stacked in a plastic foil, and next several of these stacks are packed in a cardboard box. The cardboard boxes with cups are shipped via distributors to the customers. Among the customers are large organizations as companies, offices, and schools that often use vending machines for distribution of hot drinks. Replenishing of vending machines, e.g. with coffee and cups, can be done by the organization itself or by the distributor. The LCA includes waste processing of the packaging materials, i.e. boxes and foils containing the new cups.

The hot beverage drinker uses the cup and next disposes it in a bin, which in Dutch organisations typically is a bin for commingled waste. We assume that in large organisations almost all cups are collected in waste bins, and we therefore ignore any littering. We also ignored contamination in the cups. Commingled waste is in the Netherlands usually sent to a municipal solid waste incinerator (MSWI) (Eurostat, 2014). Most Dutch MSWI recover energy produced during the combustion of waste for electricity and heat production.

The Netherlands does no longer permit landfilling of combustible material since 1996 (De Boer, 1995). Hence this study did not consider landfilling as a waste treatment option. Other waste processing options are theoretically available, for instance recycling, or composting or anaerobic digestions in the case of biocups. We included these additional waste processing options to evaluate the potential environmental benefits of these options, although these waste treatments are in practice often not (yet) performed. We assumed for each waste treatment option a 100% inclusion of the disposed cups, as to facilitate comparison of options.

Recycling requires a homogeneous and clean waste stream. PS can be mechanically recycled via washing, shredding, drying, and regranulation (Bergsma et al., 2011; Ligthart and Ansems, 2004, 2007). Up to 5% material is lost during the recycling process (Arena et al., 2003; Shen et al., 2010). PLA can in principle be recycled in the same way as PS, but in practice still lacks the critical volume to make this feasible. Techniques for distinguishing PLA from other plastics are available (European Bioplastics, 2010c), but not often applied in practice. Chemical recycling of PLA, in which PLA is hydrolysed back to its monomer lactic acid, is only performed on an industrial pilot scale in Belgium (Galactic, 2012b; Merrild and Hedal Kløverpris, 2010). Paper cups can be recycled together with other beverage drinking cartons when the recycling system is adapted for removal of contamination (Bergsma et al., 2010). The Netherlands does not have such a nationwide system, but it is practised in the neighbouring countries of Belgium and Germany. We included mechanical recycling of PS, PLA and biopaper cups in this study.

PLA cold cups and biopaper cups with a PLA lining have the European compostability mark (EN 13432), meaning that these cups compost under industrial circumstances within 12 weeks. The cups need to be mingled with other organic material (usually vegetable, fruit and garden waste, called VFG). This means that the cups can be collected together with organic material. Composting of both biocups is included in this study as a waste processing option.

The biodegradability of “cold” PLA and biopaper makes these materials also suitable for anaerobic digestion. PLA can be co-digested with other organic waste under thermophilic conditions, i.e. temperatures between 55 and 60 degrees, where it is converted into biogas (Merrild and Hedal Kløverpris, 2010; Yagi et al., 2009). Biodegradable plastics and biobags are successfully degraded in anaerobic digesters (European Bioplastics, 2010a). Anaerobic digestion is to our knowledge not a common waste option for paper in Europe. Collected paper is mainly recycled (ERPC, 2013). In practice paper is digested as it appears as a fraction in the VFG waste. The cellulose content of paperboard digest easy, under both mesophilic and thermophilic conditions, but the lignin part is hard to digest (Bayr and Rintala, 2012; Yagi et al., 2009). The produced biogas is usually combusted to deliver electricity and heat. We included the anaerobic digestion of the PLA cold cups and the biopaper cups.

4.2.4 Inventory data collection

Initially, one readily available data set was collected for each process in the life cycles of the cups, using incineration as the waste processing option. This resulted in three initial incineration LCAs, one for each cup material. Additional initial LCAs were made for each cup material with recycling as waste processing option (initial recycling LCAs), and for the biocups with composting and anaerobic digestion of the waste (initial biotreatment LCAs). These initial LCAs facilitated to distinguish between processes with a minor and major influence on the LCA results. We considered a process influential if it contributes by at least 15% to at least five impact categories in the initial LCAs or in the sensitivity analysis. Next, multiple data sets were additionally collected for all influential processes. We also collected multiple data sets for the waste processing options, since we wanted to include their variability. The procedure is described in detail in van der Harst and Potting (2014).

The data for the cups should be representative for cups which are available on the Dutch market. We assumed PS production in the Netherlands, Belgium, or Germany, and paperboard production taking place in Scandinavian countries (Eurostat, 2012, 2013). We, therefore, excluded in this study, differently from van der Harst and Potting (2014), data for PS produced in the United States. Production of “cold” PLA takes place in Nebraska in the USA (Vink et al., 2010), Thailand (Groot and Borén, 2010), or Belgium (Galactic, 2011). We limited the cup manufacturing region to Europe. Data for the Dutch waste treatment processes were limited, so we also used European data sets which were considered representative for the Dutch processes.

Data were collected from publicly and commercially available databases, reports, articles, and companies. Company information is not recognizable published as to guard confidentiality of company specific information. Table 4.1 – 4.3 list all data sets used for the influential and waste processes in this study. Data on background processes such as resource extraction, upstream processes, electricity production, or transport mode were solely taken from the ecoinvent database (Ecoinvent Centre, 2010).

Data for the production of thermostable PLA for hot cups were not available and therefore data for “cold” PLA were used instead. PLA can be produced from corn (NatureWorks LLC, 2011), tapioca, sugarcane (Corbion Purac, 2013b) or sugar beets (Galactic, 2011). It is not clear which resource is going to be used for the PLA hot cup. Inventory information on the production of PLA from corn was obtainable, and impact data were available for the production of PLA from sugarcane (Groot and Borén, 2010; Noordegraaf et al., 2011). No information was available for PLA from tapioca or sugar beets. Information on incineration, composting, and anaerobic digestion of thermostable PLA was, similarly as for its production, put on a par with cold PLA. Recycling of PLA was modelled in accordance with recycling PS, due to the lack of available data (the process is not yet commercially taking place).

Biopaperboard for biopaper cups contains a layer bioplastic as liquid barrier. Not all collected data sets on paperboard included this coating. These sets were complemented with 10% PLA (from Vink et al. (2010)). This 10% is the average of the used amounts of bioplastic used by biopaperboard producers.

Data on packaging materials of the new cups, i.e. boxes and foils containing the new cups, were added to those cup manufacturing data sets that did not include packaging material. Inquiry among cup distributors showed that usually 100 PS cups are packed in foil, and next 30 stacks are put in a cardboard box. An average box of 3000 cups consists of 202 g PE foil and 1197 g of cardboard based on Garrido (2007), a cup seller (Papstar, 2012b), and confidential information. The packaging materials for PLA cups were assumed to be the same as for PS cups. The packaging materials for paper cups vary since the number of cups per box fluctuates from 1000 to 2600. We used averaged packaging data of 0.57 g cardboard and 0.11 g PE per paper cup, based on Papstar (2012b) and confidential information of cup manufacturers.

Table 4.1: Data sets for the influential processes in the LCAs of the PS cup. The data sets in the first row are used in the initial LCAs.

Production PS	**PS cup manufacturing**	**Incineration PS cup**	**PS recycling process**	**Credits recycled PS**
Ecoinvent database, GPPS and HIPS (Ecoinvent Centre, 2010)	Ecoinvent database, Thermoforming (Ecoinvent Centre, 2010)	Ecoinvent database, MSWI of polystyrene (Ecoinvent Centre, 2010)[b]	Bergsma et al. (2011)	60% based on economic value (PlasticNews and recyclers)
ELCD database, GPPS and HIPS (ELCD, 2008)	Ligthart and Ansems (2007)[a]	ELCD database, Waste incineration of plastics (ELCD, 2008)[b]	Ligthart and Ansems (2004) and Plastic recycler 1 (confidential)	Ligthart and Ansems (2004)
PlasticsEurope, Eco-profiles GPPS and HIPS (PlasticsEurope, 2012)	Garrido and Alvarez del Castillo (2007)	Eggels et al. (2001)	Shen et al. (2011)	Ligthart and Ansems (2007)
	Plastic cup producer 1 (confidential)	Croezen and Bergsma (2000)	Plastic granulator and Plastic recycler 1 (confidential)	Bergsma et al. (2011)
	Plastic cup producer 2[a] (confidential)		Plastic recycler 2 (confidential)	

[a] Packaging data added.

[b] Credits for energy recovery are added.

Table 4.2: Data sets for the influential and waste processes in the LCAs of the PLA cup. The data sets in the first row are used in the initial LCAs.

Production PLA	PLA cup manufacturing	Incineration PLA cup	PLA recycling process	Credits recycled PLA	Composting PLA cup	Anaerobic digestion PLA cup
Ecoinvent database, Polylactide granulate[a] (Ecoinvent Centre, 2010)	Ecoinvent database, Thermoforming (Ecoinvent Centre, 2010)	Vercalsteren et al. (2006)	Bergsma et al. (2011)	60% based on economic value (PlasticNews and recyclers)	Afval Overleg Orgaan (2002)[f] and PE Americas (2009)	Van Ewijk (2008) and Yagi et al. (2009). Degradation extent PLA 60%
Vink et al. (2010)[b]	Vercalsteren et al. (2006)	Nielsen and Weidema (2002)	Ligthart and Ansems (2004) and Plastic recycler 1 (confidential)	Ligthart and Ansems (2004)	Vercalsteren et al. (2006)	Van Ewijk (2008) and Yagi et al. (2009). Degradation extent PLA 90%
US LCI (NREL)[a]	Merrild and Hedal Kløverpris (2010)[d]	Dornburg et al. (2006)	Shen et al. (2011)	Ligthart and Ansems (2007)	Nielsen and Weidema (2002)	Merrild and Hedal Kløverpris (2010)
Groot and Borén (2010)[b]	Plastic cup producer 1 (confidential)	NatureWorks LLC (2012)[e]	Plastic granulator and Plastic recycler 1 (confidential)	Bergsma et al. (2011)	Afval Overleg Orgaan (2002)[f]	
Noordegraaf et al. (2011)[b,c]	Plastic cup producer 2[d] (confidential)		Plastic recycler 2 (confidential)			

[a] Data are based on Vink et al. (2010), but more generalized.

[b] Adjusted for biogenic CO_2 uptake in crops.

[c] Data are based on Groot and Borén (2010).

[d] Packaging data added.

[e] Credits for energy recovery are added.

[f] Data are recalculated from a dry mass percentage of 40% (for vegetable, fruit and garden waste) to a dry mass of 100%.

Table 4.3: Data sets for the influential and waste processes in the LCAs of the paper cup. The data sets in the first row are used in the initial LCAs.

Production bio-paperboard [a]	Biopaper cup manufacturing	Incineration biopaper cup	Recycling biopaper cup	Composting biopaper cup	Anaerobic digestion biopaper cup
Ecoinvent database, Production of Solid Bleached Board (Ecoinvent Centre, 2010)[b]	Vercalsteren et al. (2006)	Ecoinvent database, Disposal paper to MSWI (Ecoinvent Centre, 2010)[d]	Bergsma et al. (2010)	Nielsen and Weidema (2002)	Van Ewijk (2008) and Bayr and Rintala (2012). Thermophilic process
Paperboard producer 1[b] (confidential)	Paper cup producer 1 (confidential)	Vercalsteren et al. (2006)	Merrild et al. (2009)	Afval Overleg Orgaan (2002)[e]	Van Ewijk (2008) and Yagi et al. (2009). Thermophilic process
Paperboard producer 2[b] (confidential)	Paper cup producer 2[c] (confidential)	Sevenster et al. (2007)	Arena et al. (2004)	Boldrin et al. (2009)[e]	Van Ewijk (2008) and Bayr and Rintala (2012). Mesophilic process
Biopaperboard producer 3 (confidential)		Ecoinvent database, Disposal packaging cardboard to MSWI (Ecoinvent Centre, 2010)[d]		Brinkmann et al. (2004)[e]	Van Ewijk (2008) and Yagi et al. (2009). Mesophilic process
Biopaperboard producer 4 (confidential)		ELCD (2008)[d]			

[a] Biopaperboard consists (on average) of 90% paperboard and 10% bioplastic.

[b] 10% PLA is added as bioplastic lining.

[c] Packaging data added.

[d] Credits for energy recovery are added.

[e] Data are recalculated from a dry mass percentage of 40% (for vegetable, fruit and garden waste) to a dry mass of 100%.

Table 4.4 shows distances and transport means used in this study. The transport distances for PLA from the production site to the cup manufacturer depend on the location of the production site. Production sites can be in Nebraska in the USA, Thailand or Belgium. Transport distances for PLA from Nebraska were used in the initial LCA since NatureWorks is the largest PLA producer (Gironi and Piemonte, 2011; Vink, 2011).

Table 4.4: Transport distances and means used in the LCAs of disposable cups.

From	To	Distance (km)	Transport means
PS producers	Manufacturer PS cups	500	Lorry 32 t
PLA producer			
Nebraska USA		2000	Train
		6000	Ocean freight
		200	Truck 32 t
Thailand	Manufacturer PLA cups	300	Truck 32 t
		20,000	Ocean freight
		200	Truck 32 t
Belgium	Manufacturer PLA cups	500	Truck 32 t
Paper producer	Manufacturer paper cups	1500	Ocean freight
		200	Lorry 32 t
Packaging material	Manufacturer of cups	100	Lorry 16 t
Manufacturer cups	Distributor cups	500	Lorry 16 t
Distributor cups	Customer	100	Lorry 16 t
Customer	MSWI	150	Lorry 16 t
	PS and PLA Recycler	300	Lorry 16 t
	Paper recycler	500	Lorry 16 t
	Composter	75	Lorry 16 t
	Anaerobic digester	200	Lorry 16 t

4.2.5 Inventory data processing

All inventory data were converted as to match the functional unit, which comes down to cups of 4.2 g PS, 4.2 g PLA, and 5.6 g biopaper.

We used system expansion, i.e. substituting an output or process by another process with equal function, to credit the cups for recovered energy in their waste processing. This credit is calculated as the avoided conventional electricity and heat production equalling the recovered energy from incineration and from combustion of biogas from the anaerobic digestion of the PLA hot cup and biopaper cup. Dutch MSWIs recover energy in the form of electricity and often also heat (Otten and Bergsma, 2010). Data sets for the incineration process which did not contain recovered energy were adapted as to include this energy. Data for avoided conventional electricity production were based on the Dutch electricity production mix, and data for avoided conventional heat were based on heat from natural gas (Ecoinvent Centre, 2010).

System expansion was also used to credit the cups for the recycling into secondary material (i.e. secondary PS, PLA and pulp). PS and paper endure quantity and quality loss during the recycling process. Credits for recycled PS in the initial recycling LCA were based on avoided production of virgin PS (Ecoinvent Centre, 2010), but corrected according to economic values of virgin and recycled materials as to include the loss of quality. PLA was credited similarly as PS, since information on recycled PLA was not available. Recycled paper was credited as unbleached sulphate pulp (Ecoinvent Centre, 2010) and allocated according to its recovered mass. The actual recycling process of plastic (PS and PLA) and the credits for recycled plastic are shown as separate processes in the LCA results, since the recycling process and the use of the recycled material take place at two different locations. The recycling process of paper and the credits from the created pulp are presented as one process since the conversion of the paper into pulp and the use of the pulp take place at the same premises.

We considered the carbon dioxide uptake in crops for the production of PLA and in wood for the production of paper as short cyclic or biogenic carbon dioxide (i.e. fixation is considered equal to release). Three PLA data sets considered this uptake from carbon dioxide as non-biogenic (Groot and Borén, 2010; Noordegraaf et al., 2011; Vink et al., 2010). The carbon uptake from crops in these data sets was changed into biogenic.

Groot en Borén (2010) and Noordegraaf et al. (2011) did not provide inventory data on the production of PLA, but only provided impact results for renewable and non-renewable energy use, abiotic resource depletion (ADP), global warming potential (GWP), acidification (AP), eutrophication (EP), and photochemical oxidant formation (POCP). Groot and Borén (2010) also provided the human toxicity potential (HTP). These impact data have in adjusted form been included in this study (see next section).

4.2.6 Impact assessment

The third LCA phase, impact assessment, converts the environmental input and output data from the inventory phase into their contributions to a range of environmental impact categories. We used the CML Baseline 2000 methodology (Guinée et al., 2002) and complemented its ten impact categories with the cumulated energy demand (CED) from Frischknecht et al. (2003). The ten CML Baseline 2000 categories are abiotic depletion (ADP), global warming (GWP 100) (GWP), acidification (AP), eutrophication (EP), photochemical oxidation (POCP), human toxicity (HTP), fresh water aquatic ecotoxicity (FAETP), marine aquatic ecotoxicity (MAETP), terrestrial ecotoxicity (TETP), and ozone layer depletion (ODP).

ADP data on the production of PLA from Groot and Borén (2010) and Noordegraaf et al. (2011) were provided in kg oil-equivalent and were recalculated to kg Sb-equivalent using 1 kg oil-equivalent equals 0.0201 kg Sb-equivalent (Guinée et al., 2002).

Impact results were first calculated for the separate life cycle processes and then aggregated for the whole life cycle (van der Harst and Potting, 2014). Figures 4.1 – 4.3 show the contributions of separate processes.

4.3 Results

4.3.1 PS cups

Five processes dominate the calculated environmental impacts of PS cups: PS production, cup manufacturing, PS incineration, PS recycling, and crediting of recycled PS (Figure 4.1). This is the case in both the initial LCAs as in the sensitivity analyses. These processes were, therefore, selected for collecting additional multiple data sets. No additional data sets were collected for waste processing of the packaging material and transport. Waste processing of the packaging materials, i.e. the PE foil and cardboard box, made a minor contribution to all impact categories. Sensitivity analysis on this waste process endorsed its minor influence. Transport only contributes to ODP and sensitivity analysis on transport showed no important change in its contribution to the other impact results.

The average impact results from the multiple data sets consistently pointed to the importance of the five selected processes, despite the spread in impact results from using multiple data sets (see Figure 4.1). The energy related impact categories CED and ADP have relatively consistent results across data sets, leading to a small spread. The toxicity impact categories and ODP have in general the largest spread, in particular for the incineration process.

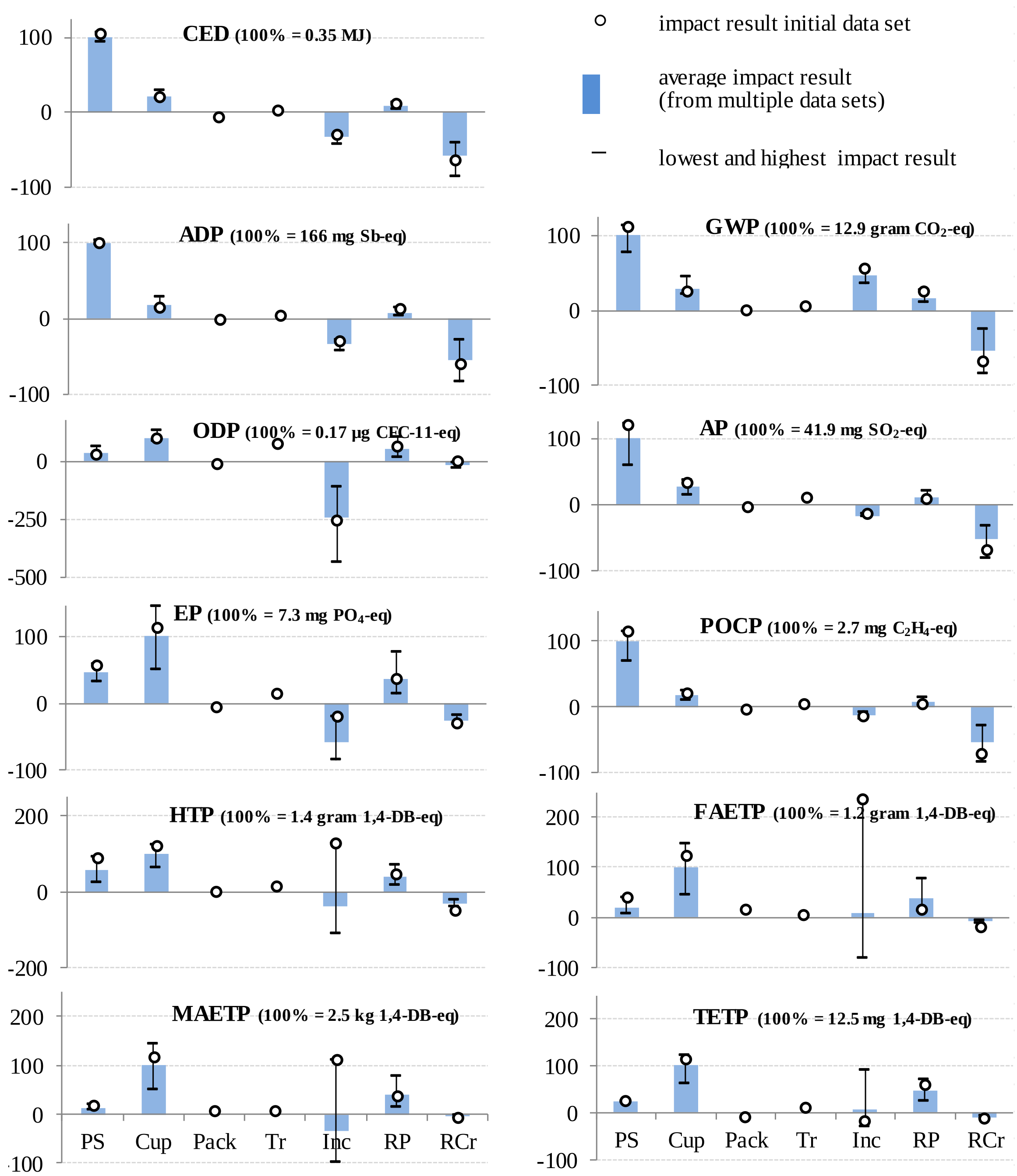

Figure 4.1: Contribution of processes in the life cycle of the PS cup to different impact categories. The graph shows the impact calculated in the initial LCA as circles, and the average impact from all data sets as bars (see Table 4.1 for data sets used). For each impact category, the process with the largest average impact is set at 100%. All other impact results are calculated relative to this 100%. The range in the impact results from the data sets is depicted as the lowest and highest result. Abbreviations: PS = production of PS, Cup = cup manufacturing, Pack = waste processing packaging material, Tr = total transportation, Inc = incineration in MSWI, RP = recycling process, RCr = credits for recycled PS, CED = cumulated energy demand, ADP = abiotic depletion, GWP = global warming potential, AP = acidification, EP = eutrophication, POCP = photochemical oxidation, HTP = human toxicity, FAETP = fresh water aquatic ecotoxicity, MAETP = marine aquatic ecotoxicity, TETP = terrestrial ecotoxicity, and ODP = ozone layer depletion.

The spread in the results of PS production is partly caused by updated data of PlasticsEurope (2012). These updated data are based on adjusted production data for benzene (precursor of PS), improved production processes, changes in energy mixes, and improved emission control processes compared to the PS data from 2002 (PlasticsEurope, 2012).

The spread in the cup manufacturing and the PS recycling process is mainly due to dissimilar amounts and types of energy use (electricity or heat). Data sets for PS incineration provide varying amounts of recovered energy, leading to subsequent varying amounts of avoided electricity or heat (or both). The incineration data sets also report different amounts of emissions (especially metals), which are the main cause for the spread in the toxicity categories.

Up to 95% of the PS cup can be recycled (weight based) (Arena et al., 2003; Shen et al., 2010). We used this percentage to credit recycled PS with avoided production of an equal mass of virgin PS, and corrected for quality loss according to economic values for secondary PS compared to virgin PS of 90% (Ligthart and Ansems, 2004), 50% (Ligthart and Ansems, 2007) and 60% (PlasticsNews, 2012), or on avoided production of other materials (Bergsma et al., 2011). These different economic values and credited materials caused the spread in the impact results.

Recycling, i.e. the recycling process plus credits for recycled PS, provides on average higher credits than incineration in five impact categories (CED, GWP, ADP, AP, and POCP). Incineration provides on average higher credits in the other six categories (EP, toxicity categories, and ODP). The incineration credits are based on avoided electricity and/or heat production in the Netherlands. Dutch electricity generation is strongly fossil-based which leads to relatively high credits for incineration. The incineration credit in ODP exceeds the impact of all the other phases together. This is mainly due to the avoided emissions of Halon during natural gas transport. The spread in all impact categories is large and overlaps for both waste processing options in most but not all categories. Recycling still has a better performance in GWP and POCP, and incineration still has a better performance in EP and ODP because the spread in results does not overlap with the other waste treatment. The overlap in most categories prevents a decisive conclusion on the preferred waste treatment.

The research of van der Harst and Potting (2014) presents the results for the PS cup in more detail. Here, we only included PS produced in Europe, while the study of van der Harst and Potting (2014) also included PS produced in the USA.

4.3.2 PLA cups

Seven processes dominate the initial LCAs of the PLA hot cup: PLA production, cup manufacturing, transport, incineration, recycling process, credited PLA, and anaerobic digestion (Figure 4.2). These processes were selected for multiple data set inquiry. Composting showed only minor contribution in GWP, but was also added to the selected processes to be consistent with the other waste processes. Sensitivity analyses for the waste processing of the packaging material confirmed its relatively small influence on the impact results.

The average impact results for the multiple data sets confirmed the importance of the selected processes. PLA production is clearly the main contributor to all impact categories. PLA production also displays the largest spread in almost all impact categories (Figure 4.2). The spread in most impact categories can be traced back to the different resources, i.e. corn or sugarcane, which are used in the production of lactic acid (the building blocks for PLA).

The spread in CED is largely caused by the energy use of PLA production, which is smaller for corn PLA compared to sugarcane PLA. Energy for corn PLA consists mainly of electricity and natural gas in the PLA production process (Ecoinvent Centre, 2010; NREL, 2011; Vink et al., 2010). Energy for sugarcane PLA is dominated by electricity, steam, and chemicals used in the production of lactic acid and its polymerisation into PLA (Groot and Borén, 2010). The total energy use (CED) is higher for sugarcane PLA, but its contribution to ADP (which includes the use of fossil fuels) is lower due to a large part of the energy being supplied via combustion of bagasse (a remnant of sugarcane plants) (Groot and Borén, 2010).

Electricity thus represents one of the main energy uses in PLA production. The used mix of fossil versus renewable energy in the production of electricity influences the impact results. Resources used for electricity production can be fairly different for dissimilar geographically location, and this difference could influence the impact categories CED, GWP and ADP. Main location for production of corn PLA is in the USA, while sugarcane PLA is mainly produced in Thailand.

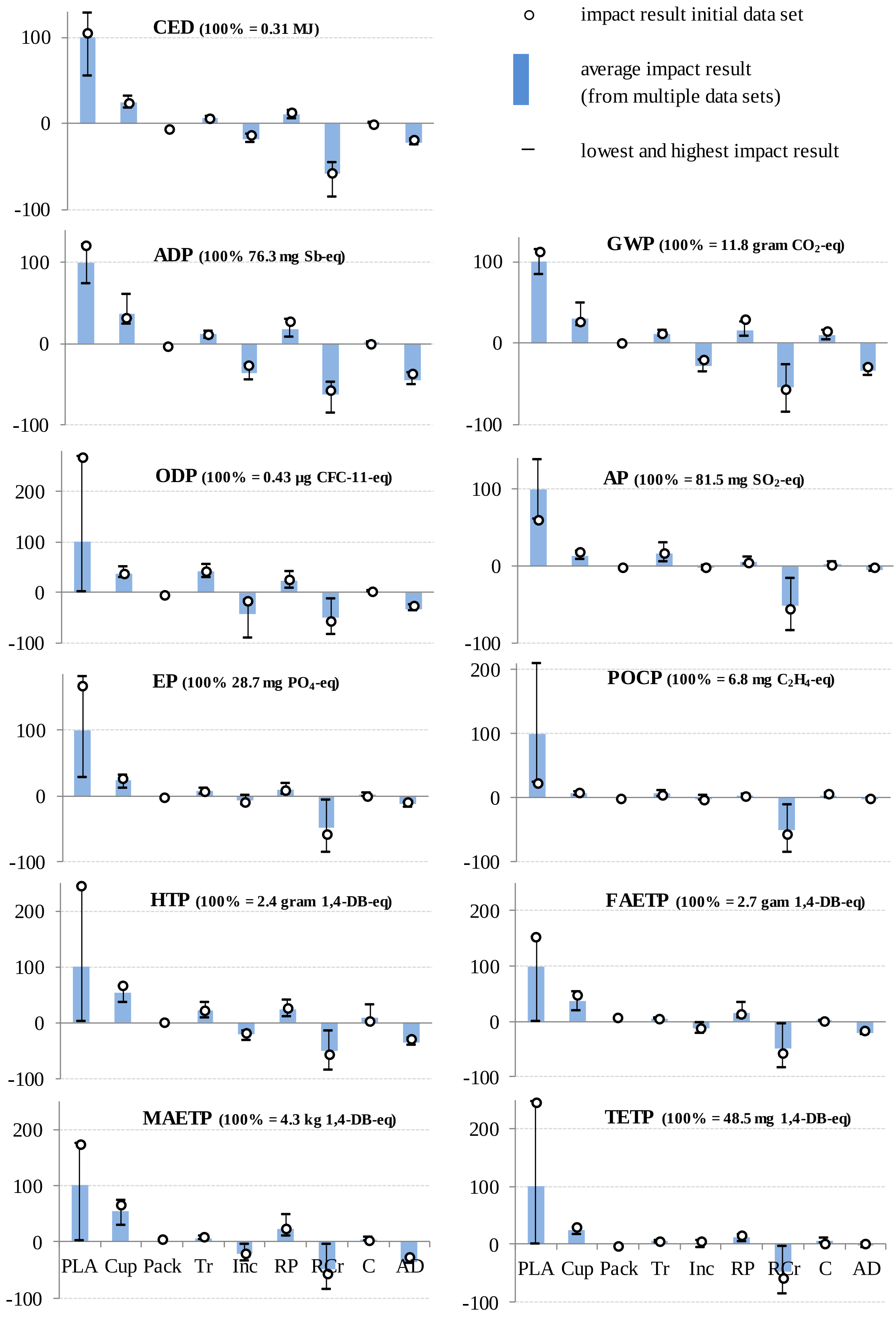

Figure 4.2: As Figure 4.1, but for the PLA cup. Abbreviations: PLA = production of PLA, C= composting, AD = anaerobic digestion, and all other abbreviations as in Figure 4.1.

AP and EP are dominated by fertilizer use during the cultivation of the crops (corn and sugarcane), the use of electricity and gas for lactic acid production and polymerisation, and from production of chemicals used in the lactic acid production. The use of electricity and natural gas also dominates the POCP impact results.

Impact results for sugarcane PLA were taken from Groot and Borén (2010) and Noordegraaf et al. (2011). They provided no impact results for FAETP, MAETP, TETP and ODP. Impact results for these categories were therefore only calculated for corn PLA. The spread in the toxicity results is big: the ratio between the highest and lowest values for HTP is 50, up to a ratio of over 700 for TETP. The large spread in TETP is caused by the insecticide cypermethrin which is included in corn production according to the ecoinvent and US LCI databases (Ecoinvent Centre, 2010; NREL, 2011), but is absent in the corn production data from Vink et al. (2010). Overall, the impact results for ODP and the toxicity categories for corn PLA from Vink et al. (2010) amount to 3% or less of the impact results of the other two corn PLA sets (Ecoinvent Centre, 2010; NREL, 2011).

As for the other processes, PLA cup manufacturing shows some spread in GWP and ADP due to differences in used energy (electricity and/or gas). These differences in energy use also led to a spread in the toxicity categories. Transport has a spread in AP due to differences in fuel use related to transport distances from PLA production locations (see Table 4.4).

PLA incineration shows a variation in ODP that is mainly based on differences in recovered energy. Nielsen and Weidema (2002) predominantly credit for recovered heat, while the other data sets mainly consider recovered electricity. The spread in the recycled PLA is, similar to PS, due to different credited materials.

Anaerobic digestion of PLA shows some spread in impact results due to the variation in degradation rates, i.e. 60% in the batch process (Yagi et al., 2009) compared to 98% in the continuous digestion process (Merrild and Hedal Kløverpris, 2010).

Comparison of averages and spread in results for the PLA waste treatment options points to composting as the least favourable. PLA does not contain nutrients, hence the compost cannot be credited for avoided fertilizer (Nielsen and Weidema, 2002; Vercalsteren et al., 2006). PLA incineration, recycling and anaerobic digestion on the other hand received credits for avoided production of material or energy (incineration directly, digestion via combustion of biogas). Recycling PLA provides on average the most credits in all but three impact categories (ODP, HTP, MAETP). The spread in recycling results, however, overlaps with the spread from incineration and anaerobic digestion in most impact categories. Recycling remains the most preferred waste treatment in CED, AP and POCP because the spread recycling results does not overlap with the other options. The average credits in our study are larger for anaerobic digestion than for incineration in almost all impact categories (not in ODP). This suggests a slight preference for anaerobic digestion above incineration, although no conclusive answer can be provided due to the overlapping impact results. Data for PLA anaerobic digestion are furthermore from lab tests and based on (theoretical) calculations. Practice has to show how much biogas is really produced.

Overall, composting is the least preferred type of waste treatment for the PLA hot cups. Average results show a preference for recycling. The most preferred option remains indecisive, however, due to large and overlapping impact results for the cups.

4.3.3 Biopaper cups

The processes with the largest impact contributions in the initial LCAs for biopaper cups include biopaperboard production, cup manufacturing, incineration, and recycling (Figure 4.3). These processes were selected for multiple data set collection. Similar to the other cups, processing of the packaging waste showed little influence. Composting and anaerobic digestion of biopaper cups had a small influence on the initial LCA results, but were added to the selected processes for multiple data set collection to be consistent with the other waste processes. Transport made a notable contribution to AP and ODP only. Sensitivity analyses indicated, in addition, that transport always made a smaller contribution than biopaperboard production or cup manufacturing.

The average results from the multiple data sets also show biopaperboard production and biopaper cup manufacturing as the main contributors to the environmental impact results, and biopaper incineration, recycling, and anaerobic digestion as the main creditors. The multiple data sets led to spread in the impact results, especially in biopaperboard production and cup manufacturing (Figure 4.3).

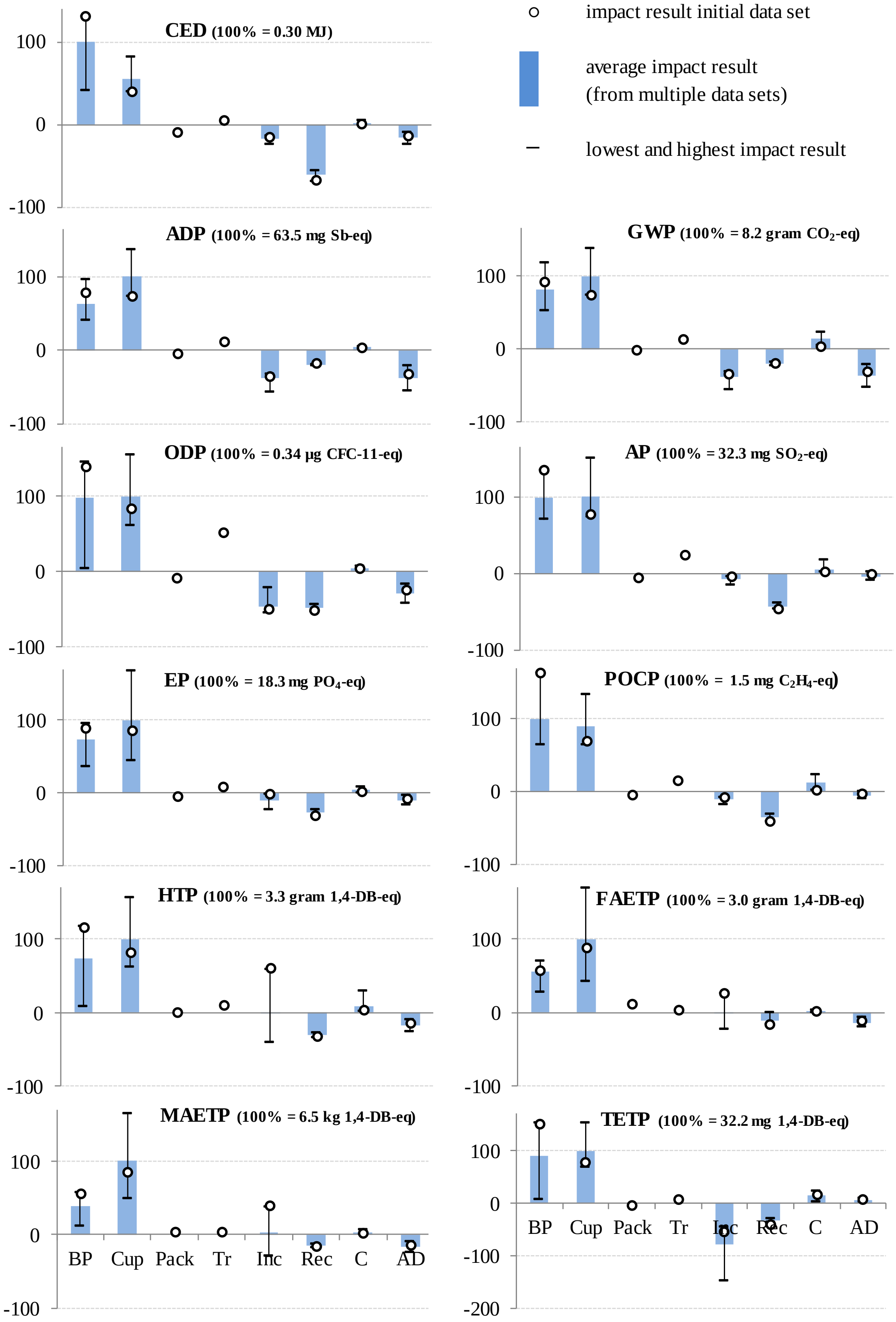

Figure 4.3: As Figure 4.1, but for the biopaper cup. Abbreviations: BP = production of biopaperboard, Rec = recycling paperboard and credits for pulp production, C = composting, AD = anaerobic digestion, and all other abbreviations as in Figure 4.1.

The spread in CED, ADP and GWP for the paperboard production correlates with the amount and type of energy used. According to Weidema et al. (2003), some reports explain the difference in energy use of paperboard production by the variation in energy efficiencies of the paper mills, while others relate it to the type of plant (i.e. pulp mills versus mills that integrate both pulp and paper production). Also the mix of fossil and wood-derived energy can substantially affect greenhouse gas (GHG) results, because the carbon dioxide emission from wood is considered carbon neutral. Different amounts and types of energy and variations in used chemicals in the pulping process furthermore caused the spread in the other impact categories. Biopaperboard consists of paperboard (90%) and a lining of bioplastic (10%). The production of bioplastic thus also contributed to the environmental impact of biopaperboard.

The sizable spread for biopaper cup manufacturing in all impact categories was traced back to the differences in the amount and type of electricity used by the three production facilities.

The waste processing options for the biopaper cup show smaller spreads in results. Differences in emissions of metals cause the spread for biopaper incineration in the toxicity categories. Ecoinvent emission data are based on the composition of paper, while others do not consider (Sevenster et al., 2007) or consider less emissions (ELCD, 2008; Vercalsteren et al., 2006) The spread in impact results for recycling biopaper is small and stems from the difference in the credited pulp amount. Differences in mesophilic and thermophilic digestion (PLA does not degrade under mesophilic conditions) and in the assumed cellulose content of the biopaperboard caused the spread in anaerobic digestion. Composting biopaper cups barely contributes to environmental impact results, similarly to composting PLA cups.

Comparison of average waste treatment results and the spread in these results shows composting as the least preferred waste treatment options compared to recycling (in all impact categories), incineration (in eight categories) and anaerobic digestion (in ten categories). Recycling biopaper is favoured in the average results in five impact categories (CED, AP, EP, POCP, and HTP), incineration in four categories (GWP, ADP, TETP, and ODP), and anaerobic digestion in three (GWP, FAETP, and TETP). Recycling receives higher credits for CED compared to incineration or anaerobic digestion, since more energy for unbleached pulp is avoided than energy recovered in biopaper incineration or anaerobic digestion. Unbleached pulp production also uses wood residues for energy, however, whereas recovered energy from biopaper incineration avoids fossil based energy. GWP and ADP therefore show a better performance for incineration and anaerobic digestion compared to recycling. The spread in the results creates an overlap between the waste treatments in several impact categories, but a few categories show no overlap. Recycling remains the best option in CED, AP, EP, and POCP, since it has no overlap with the others. Similarly, incineration remains the best option in TETP.

Overall, composting is the least preferred option for waste treatment. Average results and their spread cautiously suggest a preference for recycling. The overlapping spread in results for the waste processes, however, prevents to point to a clear preferred waste option and leads to an indecisive outcome. Note that the anaerobic digestion of biopaper cups is not a common practice, and real-life cases have to show the exact degradation possibilities and biogas formation.

4.3.4 Comparison of cups

The overall LCA results for each cup material in combination with all relevant waste processing options were expressed relative to the average PS cup results (with incineration) (Figure 4.4). This facilitated an easy comparison among the cups.

Comparison of average LCA results did not enable us to single out one cup material as the most environmentally friendly. Average LCA results show a preference for the biocups in GWP and ADP, but PS is favoured in AP, EP, ODP and the toxicity categories. The overlapping spread in LCA results between all cups further obstructs the choice for the most preferred cup material.

Comparison of different waste options across the three cups provided some common outcome. Recycling performs on average better than incineration for all three cups in CED, AP and POCP, for the two plastic cups in GWP and ADP, and for the biocups in EP, HTP, FAETP and MAETP. Incineration is on average better than recycling for all cups in ODP, for the PS cup in EP and the toxicity categories, and for the biopaper cup in GWP, ADP and TETP. Composting is the least preferred waste option for both biocups. The average results for anaerobic digestion lie between recycling and incineration for the PLA cups, but there is no trend for the biopaper cup. When taking into account the spread, however, no preferred waste management option can be identified, as indicated above.

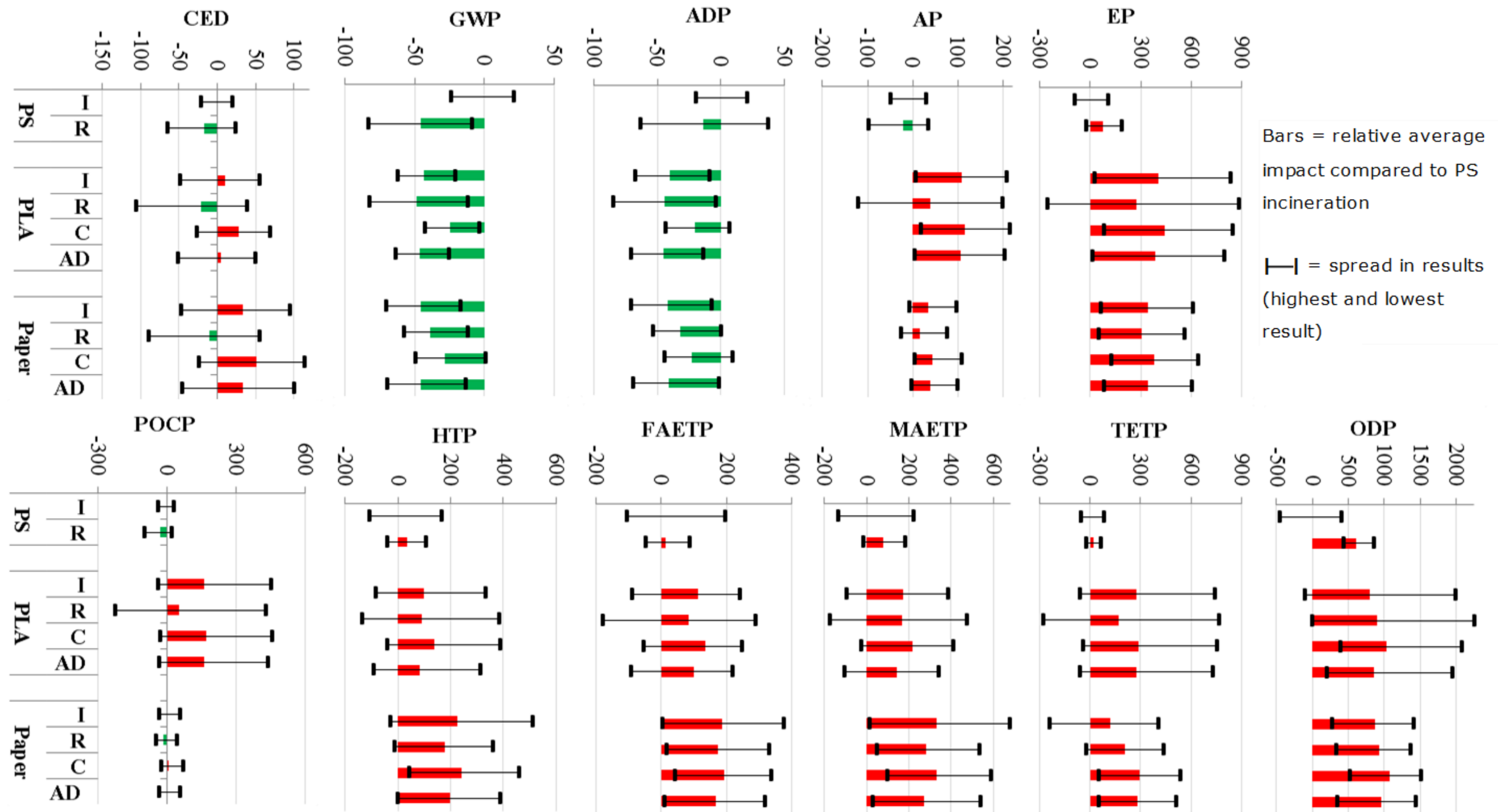

Figure 4.4: Comparison LCA results of disposable beverage cups: PS, PLA and biopaper cups. The bars show the relative impact (in %) compared to the PS cup with incineration as waste option. A negative value (green) indicates a lower environmental impact than PS-incineration, a positive value (red) indicates a higher impact. The spread in impact results reflects the highest and lowest values. PS = polystyrene, PLA = polylactic acid, I = incineration in MSWI, R = recycling, C = composting, AD = anaerobic digestion, CED = cumulated energy demand, ADP = abiotic depletion, GWP = global warming potential, AP = acidification, EP = eutrophication, POCP = photochemical oxidation, HTP = human toxicity, FAETP = fresh water aquatic ecotoxicity, MAETP = marine aquatic ecotoxicity, TETP = terrestrial ecotoxicity, and ODP = ozone layer depletion. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

4.4 Discussion

4.4.1 Limitation of the study

This study compares the environmental impact of disposable beverage cups for consumption of hot liquids from three different materials, i.e. polystyrene (PS), polylactic acid (PLA; bioplastic) and paper lined with PLA (biopaper). PS and biopaper cups for hot liquids are already on the market. The PLA cup for hot liquids is not available at the time of writing this article. Techniques for the production of thermostable PLA are available (Corbion Purac, 2013a), however, and the commercial production of PLA hot cups is expected soon (Wageningen UR, 2012). Since exact specifications of thermostable PLA were not published, we used data for “cold” PLA instead. Groot and Borén (2010) expect a higher GWP for thermostable PLA, and this would obviously influence the results for the PLA hot cup.

The ingredients and weight of the PLA hot cup were also not exactly known. Here also assumptions were made on the basis of the PLA cold cup and other PLA products. The weight of the PLA hot cup and PS cup were assumed to be the same, but the weight of a PLA hot cup may deviate depending on the properties (e.g. strength) of the cup.

This study calculates with a typical weight for each cup, even though the weight of all cups has crucial influence on the impact results. Other studies show that the results in all impact categories vary (almost) proportional to the weight of the cups (Ligthart and Ansems, 2007; Vercalsteren et al., 2006). The spread in the LCA results would increase by 15% for the PS and PLA cup and by 25% for the biopaper cup if the spread in weight was included. We did not incorporate the spread in the cup’s weight in our study since it would only increase the indecision of the preferred cup material. The cup’s weight has no influence on the comparison between the waste treatments.

The production of PS and paperboard are well-established processes. Variations in impact results in this study were largely due to dissimilar types and quantities of material and energy use in the included datasets. Different crediting approaches for recycling, i.e. based on choices made by the researchers, also created spread in impact results. Recycling issues are still debated and involve besides the scientific views also cultural customs and economic aspects (Frischknecht, 2010).

Production of PS started in the 1930s and has been improved ever since. The commercial production of PLA, on the other hand, started in the 1990s and is still a relatively new process. Potential improvements in PLA production are expected in the choice and production of the feedstock, the production of (poly-)lactic acid, and the waste treatments of PLA products. The comparison between cups made from the mature PS and the immature PLA might thus not be completely fair, but the comparison shows the performance of the materials at this moment in time.

PLA production is, contrary to PS and paperboard, a relatively new process and continues to evolve. Present resources for PLA production are corn, tapioca, sugarcane or sugar beets. It is expected, however, that these resources in time will be replaced by lignocellulosic biomass or second-generation biomass (woody or herbaceous biomass originating from wood, straw or corn stover) (Shen et al., 2009). Galactic (2012a) is even looking into a third-generation biomass based on the use of algae. The impact of the production of the second- and third-generation materials is expected to be less than for the presently used crops. The energy requirements for extracting starch or sugar from these materials, however, may be higher compared to the present crops. Whether PLA production from these new biomass materials will lead to environmental improvements is not clear at this moment and LCA studies will be needed to support any claims.

We did not include land use as environmental impact indicator. Data on land use were not always provided, but originate primarily from the production of the biomass for the biocups. The cultivation of biobased resources for PLA production competes with land use for food and feed production. The required land use per PLA cup varies between 0.005 m^2 and 0.009 m^2 (Ecoinvent Centre, 2010; Groot and Borén, 2010; Vink, 2011). This land requirement could decrease if other resources are used (second- or third generation materials) or if PLA would be recycled. Land use for the paper cup relates to the production of wood. This land is often not applied or suitable for food production, but the monoculture of a specific trees species does create a loss of biodiversity. Data on land use for the production of wood as resource for the paper cup vary between 0.001 m^2 and 0.06 m^2, based on data from paperboard producers (Table 4.3) and the ecoinvent database (Ecoinvent Centre, 2010). This land requirement decreases by 80% (from 0.0002 m^2 to 0.012 m^2) if the paper cups would be recycled.

The credits for electricity production from recovered energy in the MSWI and from the combustion of biogas (in anaerobic digestion) were based on the Dutch production mix, which consists mainly of fossil fuels. The type of energy sources (coal, gas, uranium, biomass, hydro, etc.) used in power plants has effect on all impact categories. Results can shift if electricity from another source is used instead of the Dutch mix. The WRAP study (Michaud et al., 2010) confirms the dependence of results on electricity mix from incineration credits.

We included eleven impact categories in our study. The characterisation factors for the toxicity impact categories contain large uncertainties. Toxicity impact results furthermore strongly rely on emission data, which are less consistently collected compared to energy or material data (Weidema et al., 2003). The spread in our toxicity and ODP results is higher compared to the other categories. We compared the three cups based on all eleven impact categories. The exclusion of the ODP and toxicity results could lead to more robust outcomes.

4.4.2 Waste treatment options

We considered several waste processing options, but assumed in separate LCAs that all cups (100%) enter one specific waste treatment process only. This allows a one to one comparison of waste treatment options, because it shows the environment impacts of this process.

Incineration is the only waste processing option not requiring a specific waste collection method. Impurities may strongly influence the quality of the output of recycling, composting and anaerobic digestion.

Recycling of PS cups (together with other PS objects) is performed in the Netherlands on a limited scale. Mechanical recycling of PLA is in its infancy and not yet performed on a commercial scale. Chemical recycling of PLA was not included in this study due to lack of data. This back-to-monomer procedure is a promising technique which could reduce the environmental impact of PLA cups (European Bioplastics, 2010b; Galactic, 2012b). Recycling of paper cups is not a common practice in the Netherlands but other countries do recycle paper cups together with beverage drinking cartons (Bergsma et al., 2010).

Composting biocups is complex according to Dutch composters. The turnaround time for the cups is much longer than for the regular organic waste. Furthermore, PLA cups resemble the petro plastic ones and consequently they are usually removed from incoming organic waste streams and send to the incinerator. The same applies for anaerobic digestion.

Incineration of the cups and anaerobic digestion of the biocups provide credits for recovered electricity and/or heat. Dutch electricity is mainly produced by fossil fuels, similar to many other European countries. Electricity from renewable sources will provide lower incineration credits and incline towards recycling as a preferred waste option. The energy mix in electricity production thus affects the comparison among the waste options. The government in the Netherlands and many other European member states committed to intensify the use of wind, hydro, solar and biomass energy to increase the share of renewable energy in electricity production (European Renewable Energy Council, 2011). The prospected more sustainable electricity production could alter the outcome of this study in favour of recycling.
The environmental burden caused by the production of the basic materials PS, PLA and paperboard has decreased throughout the years due to improved technologies, efficiencies and treatment of production waste. Recycling PS, PLA or paperboard includes a credit for avoided production of these materials. Improving the production of material consequently leads to decreasing credits for its recycling. This will be particularly the case for the anticipated improved production of PLA.

Other studies confirm the dependency of LCA results for incineration and recycling of PS and paper on the credited energy mix and the production of virgin material (Arena et al., 2004; Ekvall, 1999; Merrild et al., 2008; Michaud et al., 2010; Wenzel and Villanueva, 2006). There was no consensus on the preferred waste treatment for PS and paper (Michaud et al., 2010; Wenzel and Villanueva, 2006).

Few interesting messages can be taken from the discussion about recycling versus incinerating/anaerobic digestion of cups. Firstly, credits for recycling are based on the environmental impact of the production of the material. A beneficial decrease in impact of the material production lowers the benefits of recycled material. The reduced recycling credit thus decreases incentives for cleaner production. This contradicts the governmental policies where recycling of material is in general promoted above incineration. Secondly, the credits for incineration and anaerobic digestion are calculated as the avoided production of electricity and/or gas. Credits for electricity production depend on the present energy mix for electricity production and can vary depending on the country or region. These messages show the importance of not taking LCA results too easily for granted. One has to look closer *why* the results are as they are before drawing conclusions.

4.4.3 Results in relation to other studies

Figures 4.1 through 4.4 indicate the influence of each process and the spread for these processes in the impact results for disposable cups of PS, PLA and biopaper. Also the contribution of the different waste processing options is clearly visible in the figures. Average waste process results point to a slight preference for recycling over incineration of PLA and biopaper cups, for anaerobic digestion over incineration for PLA cups, and points to composting as least preferred waste processing option for PLA and biopaper cups. The spread in these waste processing option results, however, prevents a decisive outcome. Only composting remains the least preferred waste processing option for PLA cups when also the spread is included in the comparison.

The use of multiple data sets and modelling choices greatly influenced the results of this study by yielding large and overlapping spreads in all impact categories across cups. The results therefore show no clear preference for any cup material when all eleven impact categories are included. Other comparative LCA studies on disposable cups usually used only one data set (Franklin Associates, 2006, 2011; Ligthart and Ansems, 2007; Pladerer et al., 2008; Uihlein et al., 2008; Vercalsteren et al., 2006). None of these studies included all three examined cups. The studies furthermore often only incorporated GWP as environmental impact indicator. Also, landfilling or the combination of landfilling and incineration was frequently used as waste treatment, making a comparison with our results difficult.

Uihlein et al. (2008) found a lower GWP for incinerated PLA versus PS cups, consistent with our average findings for GWP. Only Franklin Associates (2011) evaluated paper cups with a PLA lining but used a landfilling/incineration waste treatment.

Pladerer et al. (2008) compared the GWP of PS, PLA and paper (PE coated) cups with incineration as waste treatment. Their outcome points to the paper cup as the best, followed by the PLA cup and PS cup as last. This order confirms our GWP outcome for the average LCAs.

Ligthart and Ansems (2007) compared recycled PS cups with incinerated PE coated paper cups. Their PS cup has, consistent with our average results, lower impact results compared to the paper cup in AP, EP, POCP, HTP, FAETP, TETP and ODP. Their paper cup has, consistent with our average LCAs, a lower impact in ADP. Our study shows an almost equal GWP for both cup systems, however, where their study shows a lower value for the paper cup. The difference can be caused by the PE coating instead of a bioplastic one. Ligthart and Ansems (2007) confirm the influence of the applied allocation method for the credits of the recycled PS on the LCA results.

In line with the results of our study, PE Americas (2009), Detzel and Krüger (2006), and Pladerer et al. (2008) also report a larger GWP for composting PLA compared to its incineration. Vercalsteren et al. (2006) claims the opposite, however, and the result of Michaud et al. (2010) is inconclusive.

Above studies seem to lead to more clear preferences for a certain cup material than our study. The large and overlapping spreads in our results from using multiple data sets and modelling choices do not support such unambiguous preferences. While this spread may provide less clear information, the results are more robust.

4.4.4 Multiple data sets

Our study adds to the current literature in that we use of multiple data sets in our LCAs. This way, we could analyse the robustness of the LCA results. We calculated considerable spread in the LCA results for most impact categories. The spread is the lowest in the energy related impact categories in CED, GWP and ADP. The variation in the other categories is partly caused by the omission or inclusion of emission data across data sets, but also by the type of energy used (fossil versus renewable). Other studies also show more consistency for energy and material inventory data than for emission data (Weidema et al., 2003).

The use of multiple data sets enabled to trace back how spread in impact results related to variations in processes, efficiencies, energy use, material use, products, etc. The spread represents the variability in the processes. These variations are unavoidable and are inherent due to companies' production methods. We need to accept that variation between processes from different facilities is a fact and part of reality. This variation is ignored if we use only one data set per process to represent an average disposable cup.

The multiple data set approach used in this study calculated the spread in LCA after impact assessment, i.e. based on the environmental impact results of processes. Other LCAs typically calculate spread in results based on inventory data. Those LCAs use average inventory data and confidence intervals to calculate the impact results and the uncertainty in these results. Averaging inventory data requires data to be in the same format (e.g. in aggregated form of resources and emissions). Our approach calculates spread based on impact results and thus eliminates this uniformity in inventory data. The new approach additionally conserves the correlation between inventory data items within a data set (e.g. between the type of energy used and its emissions). It also allows for the inclusion of impact data if inventory data are not available.

The outcome of the comparison between the cups depends on the selected data sets. The comparison can lead to biased results if single data sets are used. The multiple data sets approach tries to eliminate this bias by including the variability of the processes. Using multiple data sets requires more effort than the use of one data set, but the results include an uncertainty range which provides more robust results.

4.5 Conclusion

This paper compares the environmental impact for disposable beverage cups for hot liquids made of polystyrene (PS), polylactic acid (PLA), or paper lined with bioplastic (biopaper). We used life cycle assessment (LCA) to quantify the environmental impact on eleven categories. Those processes in the life cycle of the cups with substantial contribution to the environmental impact were quantified with the help of multiple data sets. The use of multiple data sets reflects variability which exists across similar processes on different production locations. We also considered several waste processing options, i.e. incineration and recycling for all three cups, and composting and anaerobic digestion for the two biocups (PLA and biopaper cup). Furthermore, different ways for allocation and crediting of recycling were considered. These waste options and allocation and crediting principles represent modelling choices in the life cycle of the cups. We calculated the average environmental impact for a process based on the multiple data sets for that process. Next, the spread in the process was set as the difference between the highest and lowest result for that particular process. We compared the three cups on their average results and spread in the results.

The use of multiple data sets and modelling choices consistently shows dominance in the environmental impact results for production of the basic cup material (PS, PLA, biopaperboard), the manufacturing of the cups, and the waste processes. The spread in these processes mainly relates to dissimilar amounts and types (fossil or renewable) of energy used, allocation principles, different stock material (for PLA), and reported emissions. The spreads in the toxicity impact categories are higher compared to the energy related categories.

Our study does not single out the most environmentally friendly cup material in combination with a preferred waste treatment option. The spread in the LCA results confirms this indecisive outcome. Taking all impact categories into account it was not possible to indicate PS, PLA or biopaper cups as the most preferred cup material.

Average results for the PLA cup show a preference for recycling as waste treatment, followed by anaerobic digestion and incineration. Recycling is slightly preferred over incineration for the biopaper cup. Composting biocups (PLA or biopaper cups) does not make them more environmentally friendly due to negligible credits from composting. Incineration and anaerobic digestion of the cups yield credits for recovered energy, and recycling yields credits for recovered material. The spread in the results of the waste treatment options, however, does not justify the preference for any waste treatment. Composting is the only exception: it is the least preferred option for both biocups.

The results of the LCAs with incineration and anaerobic digestion depend on the type of energy used (fossil or renewable) in the credited electricity. Recycling credits on the other hand depend on the applied allocation method and production processes of the credited material. Improvements in the production of cup materials or energy mix for electricity production may shift the preference between recycling and incineration/anaerobic digestion. One should thus be aware that the results reflect the current situation. Possible improvements can alter the comparison of waste treatment options.

Our study indicates the value of using multiple data in an LCA. It shows that a single dataset can lead to biased conclusions, when the uncertainties in input data are not considered. The additional value of using multiple datasets lies in the fact that the results include the variability of the processes and thus represent an average cup more appropriately. Additionally, the approach presents the uncertainty in the choice of the waste processing option. The average results for the cups and the spread in these results do not lead to a clear conclusion for the most environmentally preferred cup, but the results provide a more trustworthy outcome.

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5 Comparison of different methods to include recycling in LCAs of aluminium cans and disposable polystyrene cups

Abstract

Many methods have been reported and used to include recycling in life cycle assessments (LCAs). This paper evaluates six widely used methods: three substitution methods (i.e. substitution based on equal quality, a correction factor, and alternative material), allocation based on the number of recycling loops, the recycled-content method, and the equal-share method. These six methods were compared, assuming a hypothetical 100% recycling rate, for an aluminium can and a disposable polystyrene (PS) cup. The substitution and recycled-content method were next applied with actual rates for recycling, incineration and landfilling for both product systems in selected countries.

The six methods differ in their approaches to credit recycling. The three substitution methods stimulate the recyclability of the product and assign credits for the obtained recycled material. The choice to either apply a correction factor, or to account for alternative substituted material has a considerable influence on the LCA results, and is debatable. Nevertheless, we argue that incorporating quality reduction of the recycled material by either a correction factor or an alternative substituted material is preferred over simply ignoring quality loss. The allocation method focusses on the life expectancy of material itself, rather than on a specific separate product. The recycled-content method stimulates the use of recycled material, i.e. credits the use of recycled material in products and ignores the recyclability of the products. The equal-share method is a compromise between the substitution methods and the recycled-content method.

The results for the aluminium can follow the underlying philosophies of the methods. The results for the PS cup are additionally influenced by the correction factor, or credits for the alternative material, the waste treatment management (recycling rate, incineration rate, landfilling rate), and the source of avoided electricity in case of waste incineration. The results for the PS cup, which are less dominated by production of virgin material than aluminium can, furthermore depend on the environmental impact category. This stresses

the importance to consider other impact categories besides the most commonly used global warming impact.

The multitude of available methods complicates the choice of an appropriate method for the LCA practitioner. New guidelines keep appearing and industries also suggest their own preferred method. Unambiguous ISO guidelines, particularly related to sensitivity analysis, would be a great step forward in making more robust LCAs.

Based:
E. van der Harst, J. Potting, C. Kroeze. Comparison of different methods to include recycling in LCAs of aluminium cans and disposable polystyrene cups. *Waste Management* In press.

5.1 Introduction

Recycling is a well-known and widely used waste treatment to valorise the properties of wasted materials or products. The Waste Framework Directive of the European Commission prioritizes recycling in the waste management hierarchy over energy recovery and disposal options that do not include any kind of recovery (e.g. landfilling, incineration without energy recovery, emission to water bodies) (European Commission, 2008).

Recycling retains wasted materials or products by converting them into secondary materials. These secondary materials typically replace new materials and thus conserve resources. The four main recycled materials in Europe are glass, metals, paper and cardboard, and plastics (European Environment Agency, 2012). Recycling rates can within one material vary among products which are made from that material. The European recycling rate for all steel for example is 85% on average, but for steel packaging it is 70%, and for scrap cars it is 99% (TATA Steel, 2014).

The environmental and economic benefits of recycling depend the recycling process itself, the avoided production of new material, and on the market for the recycled material. Recycling is profitable from an economic perspective if the profits from the recycled material outweigh the recycling process costs. The environmental benefits of recycling can similarly be positive if the environmental credits from the recycled material outweigh the environmental burdens of the recycling process. Quantifying the benefits of recycling in the environmental assessment of products, i.e. by life cycle assessment (LCA) (ISO 14040 (ISO, 2006a)), is unfortunately not straightforward due to the ambiguous character of the recycling process.

The recycling process cannot only be considered a waste management process, but also a production process for material. Using recycled material avoids production of virgin materials. The recycling process has thus multiple functions, i.e. the recycling process is a multi-functional process. In LCA this means that the recycling process and its output, i.e. the recycled material, are at the same time part of the product system which produces the recycled material, but also of the product system which uses the recycled material.

Recycled material may be used to produce the same product as the one from which the recycled material originates. This leads to a so called closed-loop recycling system. The properties of the recycled material need to be identical to those of the original material in this case. The recycled material does not physically need to enter the same product system, but instead it is added to the stock of material with the same quality as the virgin material. Metals (e.g. steel, aluminium, copper, zinc) are typical examples of materials maintaining their quality and properties in the recycling process (Atherton, 2007). Recycled metals are obvious examples of closed-loop recycling systems.

Material can also degrade during the recycling process, leading to an open-loop recycling system in which the recycled material can only replace virgin material with a lower quality or a totally other material in the next product. The length of paper fibres, for example, is shortened during the recycling process. This gives recycled paper fibres a lower quality compared to fibres from virgin wood, although recycled paper fibres are still an excellent source for paper and board production (Merrild et al., 2008). Plastics can degrade during the recycling process, due to shortening of the polymer chains and heterogeneity of the material (Al-Salem et al., 2009), applied additives, and plain contamination during the use of plastic products. A quality drop in the recycled material reduces the application options of the recycled material, typically leading to down cycling.

Different methods are practiced in LCA to assign the environmental impacts of the recycling process and the environmental benefits of the recycled material to the product system producing the recycled material and the product system using the recycled material (e.g. Ekvall and Finnveden (2001), Ekvall and Tillman (1997), EC-JRC (2010), Guinée et al. (2002), Ligthart and Ansems (2012), Newell and Field (1998)). These methods can result in different LCA outcomes for the same product system (Azapagic and Clift, 1999; Cederstrand et al., 2014; Ekvall and Finnveden, 2001; Liu and Müller, 2012; van der Harst et al., 2014; Wardenaar et al., 2012; Weidema and Schmidt, 2010). This discrepancy in outcomes is not beneficial for the credibility and reliability of LCA studies and its use as a decision support tool.

This paper addresses three questions on the assessment of recycling in LCA. A first and main question is how and where to assign the environmental impacts of the recycling process and the environmental benefits of recycled material to the different product systems. This so-called ‘allocation problem’ is one of the most debated and controversial issues in LCA (Ekvall and Finnveden, 2001; Finnveden et al., 2009; LCA Forum, 2007; PRé Consultants, 2011, 2013; Reap et al., 2008; Weidema, 2003). Any loss in quality of the recycled material also needs to be accounted for in LCA, because the functionality of the recycled material is not the same as the original material. A second question in LCA is, therefore, how to account for loss in quality of the recycled material. Recycling methods are applied in real product systems and their actual waste treatment options. Different methods can lead to different LCA outcomes. There might be, on the other hand, additional aspects affecting the LCA results for recycling. The third question is, therefore, how sensitive LCAs results are for the choice of recycling methods compared to other factors in the recycling process.

This paper evaluates six widely used methods for modelling recycling in LCA: 1) *substitution-with-equal-quality,* 2) *substitution-with-correction-factor*, 3) *substitution-with-alternative-material*, 4) *allocation-on- number-of-recycling-loops*, 5) *recycled-content method*, and 6) *the equal-share* method. Each model is first described and then applied to two case studies: 1) an aluminium can, and 2) a disposable polystyrene (PS) cup. Next, these six methods are discussed in relation to their underlying philosophies and their influence on the case study results. Finally, one of the *substitution* methods and the *recycled-content* method are again applied to the two case studies, but now reflecting different waste management practices in several European countries.

5.2 Methods

5.2.1 Life cycle assessment (LCA)

Life Cycle Assessment (LCA) is a standardized method to assess the potential environmental performance of products or service systems (ISO, 2006a). An LCA consists of four methodological phases: 1) goal and scope definition, 2) inventory analysis, 3) impact assessment, and 4) interpretation. The goal of an LCA describes the purpose of the study and the targeted audience. The scope sets the methodological framework for the study and therewith defines how the other methodological phases are performed. The scope includes amongst others the definition of a functional unit, i.e. the function of the product under examination, and the system boundaries of the investigated product system. Also the handling of multi-functional processes in inventory analysis is laid down in the scope definition. Inventory analysis consists of the collection and processing of data about the environmental inputs (e.g. natural resources) and outputs (emissions, waste, products) for all included life cycle processes. These data are used in the impact assessment phase to calculate the contribution of the product system to a range of environmental impacts. The interpretation phase evaluates the results from the inventory analysis and impact assessment, and makes conclusions based on the goal and scope definition.

5.2.2 Research approach

This paper evaluates six methods for handling recycling in LCA and applies them on two case studies: 1) an aluminium can, and 2) a disposable PS cup. The research approach in this paper consists of the following steps:

1) Making reference LCAs for the two case study product systems (section 5.3.1). The reference LCAs are cradle-to-disposal LCAs, and hence include no waste treatment.
2) Describing the six methods to handle recycling (section 5.3.2 through 5.3.7).
3) Applying these six methods to the two case study product systems and comparing the results to the reference LCAs (section 5.3.2 through 5.3.7). The results for each method are calculated assuming a hypothetical 100% recycling rate of the product.
4) Evaluating similarities and differences in the underlying philosophies of the methods (section 5.3.8).
5) Applying a *substitution* method and the *recycled-content* method to the two case studies, but now reflecting the present recycling practices in several European countries (section 5.3.9).

Step one calculates LCA results for all ten impact categories in the CML baseline 2001 method (Guinée et al., 2002). We used SimaPro 7.3 to calculate the impacts (PRé Consultants, 2011). The elaboration of steps 2 through 5 is provided in section 5.2.5.

5.2.3 Two case studies

The two study cases are specifically chosen. Recycling of aluminium cans is a common practice in Europe and can be seen as a closed-loop product system. Recycling of disposable PS cups is not yet integrated into society, and recycling can deteriorate the quality of PS leading to an open-loop product system. The results for the reference LCAs are used a basis for evaluating the influence of different methods for quantifying the benefits of recycling.

Aluminium can

The first case focuses on the life cycle of aluminium cans. The functional unit in this study is the manufacturing of one aluminium can which can hold 330 ml liquid. A 330 ml aluminium can weights 13 gram (Amienyo et al., 2013). An aluminium can consists of two parts: a body and a lid. The life cycle of the aluminium can start with the extraction of bauxite which is used for the production of aluminium ingot (slabs, billets, T-bars, etc.) (EAA, 2013b). Aluminium ingot is next rolled into sheets, which are used in the production of the can. The can body manufacturing consists of deep-drawing and 'wall-ironing' of a piece of aluminium sheet. The can body is washed, coated and printed. The can lid is punched-out separately from a sheet. Both can components are sent to a filling station, where the can body is filled with liquid and the lid is sealed on top of the body. The production of the beverage and its ingredients is not included in the system boundary. The filled can finds its way to the final consumer via retail or other organizations. The disposed cans are collected and transported for waste management. This is where the reference LCA for the aluminium cans stops. We use all virgin aluminium as input material in the aluminium can in this reference LCA, as to enable the comparison among the effects of the six recycling methods.

Waste management options for the discarded can include recycling, incinerating, and landfilling. Recycling includes the collection of the cans and the actual recycling process. The actual recycling process consists of shredding, purifying, and remelting of the pieces into aluminium ingot.

Data sources for the life cycle processes and the assumed distances are presented in Tables 5.1 and 5.2. Data for virgin aluminium production are related to aluminium which is available on the European market. We assume that the can is manufactured and used in Europe, and the waste treatments are also performed in Europe. Waste aluminium generated during can manufacturing (post-industrial waste) is sent to the recycler, and regarded as a closed-loop recycling system. The focus in this paper is only on the recycling process of the discarded cans (post-consumer waste). The efficiency of the aluminium recycling process is 96% (EAA, 2013b). Background processes on materials, energy production, and transport modes are taken from the ecoinvent database (Ecoinvent Centre, 2010).

Table 5.1: Data sources used in the LCA of the aluminium can.

Life cycle phase	Data source	Process includes
Virgin aluminium production	Ecoinvent database (Ecoinvent Centre, 2010)	Average virgin aluminium production, based on aluminium consumed in Europe
Sheet rolling	Ecoinvent database (Ecoinvent Centre, 2010)	Hot and cold rolling of ingot into aluminium sheets, including transport to the plant and packaging material
Can manufacturing	Confidential	Manufacturing of the aluminium can body and can lid
Filling of the can and packing filled can	Amienyo (2013)	Electricity use in the filling and sealing stages, and the packaging material for the filled cans
Recycling process	EAA (2010)	Recycling of used aluminium products into ingot
Incineration	Ecoinvent database (Ecoinvent Centre, 2010)	Incineration in a municipal waste incinerator
Landfilling	Ecoinvent database (Ecoinvent Centre, 2010)	Disposal on sanitary landfill

Table 5.2: Transport distances and transport mode used in the LCA of the aluminium can.

From	To	Distance (km)	Transport mode
Aluminium sheet plant	Can manufacturer	400	Lorry 32 t
Can manufacturer	Filling station	600	Lorry 32 t
Packaging material	Filling station	100	Lorry 16 t
Filled can	Retail	200	Lorry 16 t
Used can	Recycler	200	Lorry 16 t
Used can	Incinerator	100	Lorry 16 t
Used can	Landfill	100	Lorry 16 t

Disposable PS cup

The second case focuses on PS cups. The functional unit in this case study is the provision of a disposable PS cup which is suitable for use in automatic vending machines and which can hold 180 ml of hot liquid. The weight of the cup is set at 4.2 gram (van der Harst and Potting, 2014). The life cycle of the disposable polystyrene (PS) cup starts with the extraction of crude oil and natural gas for the production of virgin PS. The manufacturing of the cup itself is a sequence of extrusion and thermoforming. The PS cup consists of a mixture of general purpose PS (GPPS), high impact PS (HIPS), and 1% titanium oxide (colouring agent) (Ligthart and Ansems, 2007; van der Harst and Potting, 2014). The readymade cups are packed and shipped to the users. After the cups are used, they are collected and transported for waste management. This is where the reference LCA for the disposable PS cup stops. Virgin PS is used as input material in the PS cup, as is required by legislation, and this enables the comparison among the effects of the six recycling methods.

Used cups can be recycled, incinerated or landfilled. The PS recycling process is a sequence of shredding, washing, drying and remelting to form recycled PS pellets (Al-Salem et al., 2009; Bergsma et al., 2011).

Data sources for the life cycle processes and the assumed distances are presented in Tables 5.3 and 5.4. We assume that the production of PS, the manufacturing of the cup, and all waste treatments take place in Europe. The impact of the use phase of the cup is negligible (van der Harst and Potting, 2014) and is not included in the system boundary. Background processes on materials, energy production, and transport modes are taken from the ecoinvent database (Ecoinvent Centre, 2010). The loss of the amount of material during the recycling process was set at 5% (Arena et al., 2003; Shen et al., 2010). Heat which is formed during incineration of the PS cup in a municipal waste incinerator is recovered and used for district heating and electricity production (Ecoinvent Centre, 2010). On average 4.51 MJ electricity and 9.05 MJ of heat is recovered per kg PS incinerated (Ecoinvent Centre, 2010).

Table 5.3: Data sources used in the LCA of the PS cup.

Life cycle phase	Data source	Process includes
Primary polystyrene production (GPPS and HIPS)	Plastics Europe (2012)	Average GPPS and HIPS production in Europe
Cup manufacturing	Van der Harst and Potting (2014)	Extrusion, thermoforming, packing of the cup. Includes packaging material
Recycling process	Bergsma et al. (2011)	Recycling plastic into pellets
Incineration	Ecoinvent database (Ecoinvent Centre, 2010)	Incineration of PS in a municipal waste incinerator
Landfilling	Ecoinvent database (Ecoinvent Centre, 2010)	Disposal on sanitary landfill
Energy recovery from MSWI	Ecoinvent database (Ecoinvent Centre, 2010)	Heat produced from natural gas, and average European electricity production

Table 5.4: Transport distances and transport mode used in the LCA of the PS cup.

From	To	Distance (km)	Transport mode
Producers of PS	PS cup manufacturer	500	Lorry 32 t
Packaging material	PS cup manufacturing	100	Lorry 16 t
Cup manufacturer	Retailer or customer	600	Lorry 16 t
Used cup	Recycler	300	Lorry 16 t
Used cup	Landfill	100	Lorry 16 t
Used cup	Incinerator	100	Lorry 16 t

5.2.4 Modelling recycling in LCA

The results for the reference LCAs are used a basis for evaluating the influence of different methods for quantifying the benefits of recycling. Recycling is a multi-functional process. It serves as a waste treatment of the discarded product and at the same time provides new material to be used in a next product. The question in LCA is how to divide the burdens of the recycling process and credits in the recycled material between the product system that provides the recycled material and the one using the recycled material. ISO 14044 (ISO, 2006b) defines two fundamentally distinct principles to assign environmental impacts in multi-functional processes: 1) avoidance of allocation, and 2) allocation.

ISO 14044 states the following hierarchical procedure for handling the multi-functional recycling process:

1) Allocation should be avoided by means of
    a) division of the process into sub-processes, or
    b) by means of system expansion
2) If avoidance is not possible, then allocation should be done according to
    a) the 'underlying physical relationship' (e.g. mass)
    b) based on other relationships (e.g. economic values of output products)
    c) the number of subsequent uses of the recycled material.

The above approaches each have their own limitations and weaknesses. Subdivision is not always possible, as is the case in recycling. In case of system expansion, there are not always suitable substitutes for the expanded processes or new allocation problems can occur if the expanded system also is based in another multifunctional process (Ekvall, 1999; Heijungs and Guinée, 2007), even though some claim that system expansion can always be applied (Weidema, 2003). Allocation can be based on an array of relationships, which can results in different outcomes.

The ISO guidelines do not prescribe a specific approach for handling the multi-functionality of recycling. ISO 14044 therefore suggest a sensitivity analysis if several approaches seem possible. This contradicts their preferred sequence as stated above. A sensitivity analysis would, however, indicate the dependence of the outcome on the used approach.

Special attention should be paid to the inherent properties of the recycled material. ISO 14044 distinguishes between closed-loop and open-loop recycling systems. Recycled material from a product is used in the production of the same product in a closed-loop recycling system. This means that the recycled material can fully substitute virgin material on a 1:1 ratio. ISO 14044 (2006b) suggests a closed-loop method with avoided virgin material production if the recycled material has the same inherent properties as the virgin material. The product system receives in this case credits for the avoided production of virgin material. The closed-loop method can also be applied if the recycled material is used in the production of different products, as long as the inherent properties are not changed. This is the case for metals (e.g. steel, aluminium, copper, zinc), which keep their properties during recycling and the recycled material is added to the metal-stock (Atherton, 2007).

ISO 14044 (2006b) suggests an open-loop method if the recycled material does not have the same inherent properties as virgin material. ISO does not clearly state what a change in inherent properties actually means. The question in the open-loop method is how to account for this drop in quality of the material. Recycled material can sometimes partly replace virgin material. In this case the recycled material cannot substitute virgin material on a 1:1 ratio, and a correction factor (based on the quality, mass or economical values) is used to represent the quality drop of the recycled material. Recycled material can also originate from heterogeneous materials (e.g. mixed plastics) or can plainly be contaminated and can thus no longer qualify to be used in the production of the same product or for inclusion in the stock of material with the initial quality. In this case, the recycled material ends up as resource for other products otherwise made from different material. The shortening of paper fibres and some plastics polymer chains are clear examples where recycling leads to a quality drop of the materials.

If sub-division or system expansion is not possible, then allocation is the alternative option according to ISO 14040 (2006b). Allocation is the distribution of the inputs and outputs of a process among the different multiple outputs (products) of that process. Allocation should be done first based on physical relationships, but this is hardly used in recycling because there is in fact only one output product. If physical relations do not provide a solution, then allocation should be based on other relationships, e.g. economic values (ISO, 2006b). Economic allocation is to our knowledge hardly applied in recycling. Next, allocation should be based on the number of times the material can be recycled. This is the most common manner to apply allocation in recycling, and this allocation method is included in this paper.

5.2.5 Applying methods for handing recycling to the case studies

The above mentioned ISO guidelines and additional viewpoints on the use of recycled material have led to a number of approaches and methods to solve the multi-functionality of recycling. In the second research step we describe and evaluate six of the most common methods as described in guidelines and literature, and as used in practice in LCAs. These six methods are: 1) *substitution-with-equal-quality*, 2) *substitution-with-correction-factor* for quality drop, 3) *substitution-with-alternative-material*, 4) *allocation-on-number-of-recycling-loops*, 5) *recycled-content* method, and 6) the *equal-share* method.

The six methods are next used for modelling recycling in the LCAs of the case studies. In step 3 we calculate LCA results based on a hypothetical 100% recycling rate of the product (i.e. 100% of the disposed product is sent to recycling), but taking into account a 4% loss for aluminium and 5% loss for PS due to the efficiency of the recycling process. The 100% recycling rate is used as an exercise and does not pretend to reflect a realistic situation. The 100% recycling rate is an opportunity to see the maximal effect of the different methods. The LCA results for the different methods are calculated as a percentage towards the reference LCAs to facilitate a comparison between the methods. We only use global warming potential (GWP) in the calculation of both case studies in step 3, based on the results for the reference LCAs, and as explained in the section 5.3.1.

Similarities and differences between the six methods are discussed in step 4. The main characteristics, such as the philosophy behind the methods and the strength and weaknesses of the methods, are portrayed and compared to each other.

A *substitution* method and the *recycled-content* method are in the last step (step 5) again applied to the two case studies. In this step actual average European waste treatment rates are considered, contrary to the hypothetical recycling rate of 100% in step 3. LCA results are additionally calculated for a selection of seven individual European countries. These countries are selected because they geographically spread over Europe and have different waste treatment rates for the aluminium can and the PS cup, and are used as an illustrative example. Adding more countries would not add information, but make the results more difficult to interpret.

We calculate GWP results for both case studies in this step, and additionally for abiotic depletion (ADP) in the PS cup case, based the LCA results for the European average situation, and as explained is section 5.3.9.

5.3 Results

The results section starts with the results for the reference LCAs of the two case studies (5.3.1). Next, we describe six methods and apply them to (hypothetical) scenarios in the case studies (5.3.2 thru 5.3.7). We evaluate the six methods (5.3.8), and again apply a *substitution* method and the *recycled-content* method to the two case studies, but now reflecting the present recycling practices in several European countries (5.3.9).

We use the terms ‘analysed product’ and ‘analysed product system’ if we refer to the product and product system for which we are performing an LCA.

5.3.1 Reference LCAs case studies

Reference LCA aluminium can

Figure 5.1.a. presents the results for the reference LCA for the aluminium can. The production of virgin aluminium is the main contributor (at least 65%) in all ten impact categories in the reference LCA. Recycled aluminium, as a replacement of virgin aluminium, will therefore affect all impact categories. Recycling of aluminium requires energy and these burdens can also affect the impact. The energy demand for virgin aluminium is twenty times as high compared to energy demand for producing recycled aluminium from scrap (EAA, 2013c). Only the GWP results are discussed in this paper. GWP is presently the most used impact indicator, since climate change has a prominent position on many political agendas. The influence from the used method for quantifying recycling on the results for the other impact categories follow the same pattern as GWP.

Reference LCA PS cup

The results for the reference LCA for the PS cup are presented in Figure 5.1.b. The production of the PS mix is the main contributor in the reference LCA in four impact categories (abiotic depletion, acidification, global warming potential, and photochemical oxidation) (Figure. 5.1.b). The manufacturing of the cup is the main contributor in the other categories. Credits for avoided PS production will therefore only have large effects in four impact categories. The recycling process itself requires energy and will also contribute to the impacts. The effects of the recycling modelling methods are, similar as for the aluminium cans, only discussed for GWP in the following sections as an illustrative example (5.3.2 thru 5.3.7). The effect in the other categories might deviate, but are not discussed further here. Note that for GWP the contribution of the production of PS in the reference LCA is almost equal compared to the contribution of virgin aluminium production in the aluminium can reference LCA.

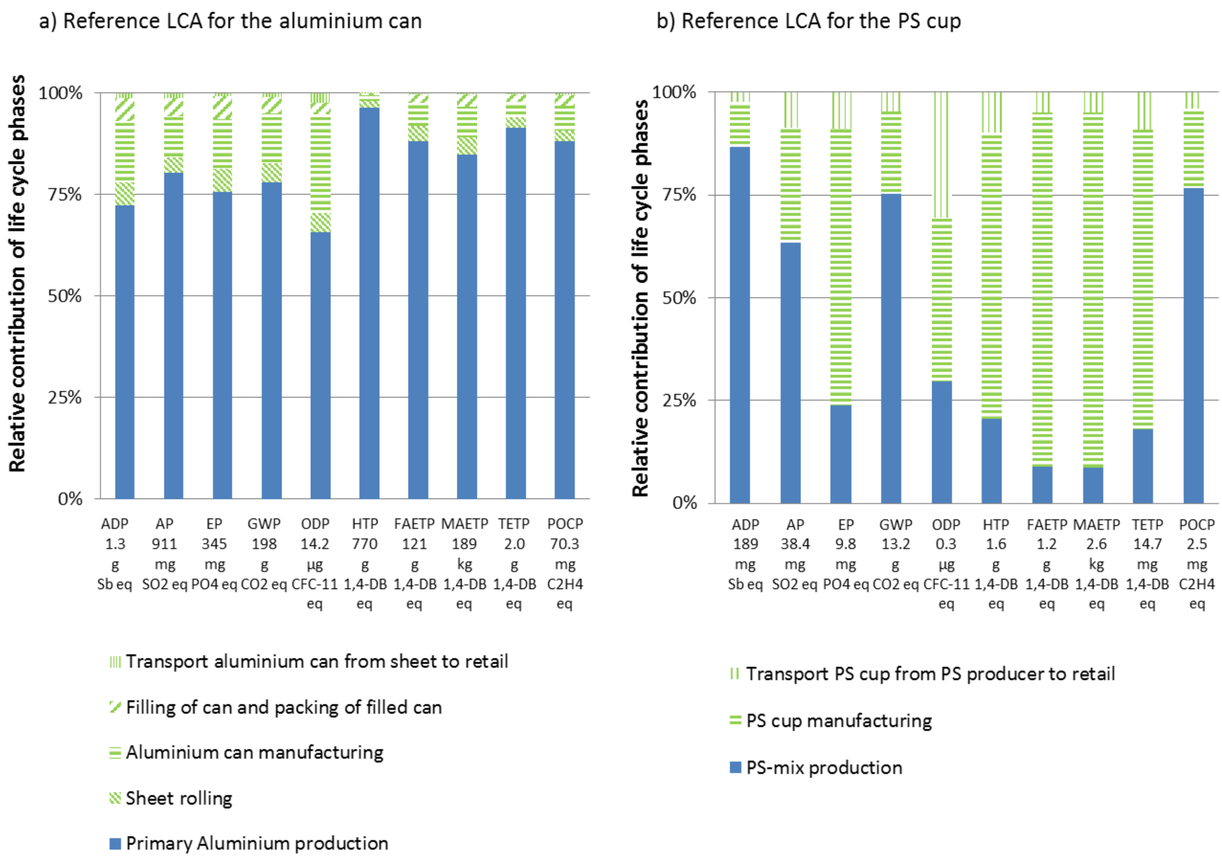

Figure 5.1: Reference LCA for a) the aluminium can, and b) the disposable PS cup. The graphs show the relative contributions of the life cycle phases up to the disposal phase to ten environmental impact categories. Both products are assumed to be completely made from virgin material. The solid blue bars refer to the production of virgin material. The green patterned bars refer to the other life cycle phases. ADP = abiotic depletion potential, AP = acidification potential, EP = eutrophication potential, GWP = global warming potential, ODP = ozone layer depletion potential, HTP = human toxicity potential, FAETP = fresh water aquatic ecotoxicity potential, MAETP = marine aquatic ecotoxicity potential, TETP = terrestrial ecotoxicity potential, POCP = photochemical oxidation potential.

5.3.2 Method 1: Substitution-with-equal-quality

System expansion for recycling involves enlarging the product system boundary of the product which is analysed by including the production of materials which are affected by recycling. In substitution methods, these additional materials are used to credit the product which produces the recycled material. Substitution methods can be applied in both closed-loop and open-loop systems. Substitution methods are often referred to as the ‘end-of-life’, ‘avoided-burden’, ‘closed-loop approximation’ methods, or the ‘recyclability substitution approach’. These methods are in fact conceptually equal to the system expansion method (Heijungs and Guinée, 2007). This paper distinguishes three different substitution methods: substitution based on 1) equal quality of the substituted material, 2) a correction factor due to quality loss, and 3) substitution by alternative material.

The *substitution-with-equal-quality* method can be applied for closed-loop product systems. A closed-loop product system requires that recycled material, which is created as the result of the waste treatment of a product, qualifies to be used as input material in the same product system (see Figure. 5.2.1). This does in practice not physically need to be the same product, but the recycled material should maintain its initial properties and quality, and is therefore added to the material pool from which the product is made. The recycled material may in principle replace the input material on a 1:1 ratio. The analysed product system gets the environmental burdens of the end-of-life recycling process of the discarded analysed product. The analysed product system also receives credits for the amount of recycled material that is obtained from the end-of-life recycling of the discarded analysed product. Only the amount of material that does not enter or gets lost in the end-of-life recycling process is accounted for as virgin material in the analysed product system. It does not matter whether input material for the product system factually consists of virgin material and/or recycled material. The method is suggested by numerous guidelines and authors (BSI, 2011; EC-JRC, 2010; Guinée et al., 2002; ISO, 2000; Pankaj Bhatia et al., 2011).

Substitution-with–equal-quality method applied to the aluminium can

The aluminium can is 100% recyclable and can be recycled infinitely without loss in aluminium quality (Ball, 2014; Crown, 2014; Rexam, 2014). Aluminium maintains its properties after recycling (EAA, 2013a; EAA and OEA, 2004) and recycled aluminium can replace virgin (primary) aluminium on a 1:1 ratio. As such, the European Aluminium Association (EAA) (2013a) prefers the term “use of aluminium” rather than its consumption, and prefers the term “cradle-to-cradle” instead of “cradle-to-grave” for aluminium products.

Recycling clearly lowers the GWP compared to the reference situation due to the incorporated credits for the avoided aluminium production (Figure 5.3). The LCA only includes a net 4% virgin aluminium production because the loss in material from the recycling process itself (recycling efficiency is 96%) needs to be compensated.

Substitution-with-equal-quality method applied to the PS cup

The method is often applied for quantifying recycling of thermoplasts (Ferreira et al., 2014; Lazarevic et al., 2010; Michaud et al., 2010; Shen et al., 2010), even though recycled PS usually has not the same properties as virgin PS due to shortening of the polymer chain length (Vilaplana et al., 2006). Recycled PS is in this case study credited as virgin GPPS. Recycling lowers the GWP due to the credits from avoided virgin PS production (Figure 5.3). Here, only a net 5% virgin PS production is included due to the loss of material in the recycling process (recycling efficiency is 95%).

5.3.3 Method 2: Substitution-with-correction-factor

The *substitution–with-equal-quality* method does not suffice if the inherent properties of the recycled material change or the quality of the recycled material is inferior to the input material. The recycled material cannot replace the input material on a 1:1 ratio in this case, which leads to an open-loop recycling systems. Two different approaches are distinguished for open-loop systems: 1) the recycled material can replace a limited fraction of input material, and 2) the recycled material is used to replace alternative materials (as explained in the next section).

If the recycled material can only substitute part of the input material, then the closed-loop approach can be used but with a correction factor expressing the quality drop (see Figure 5.2.2). The recycled material does not actually need to enter the same product system or be added to the stock of input material, as is the case in the *substitution-with-equal-quality* method above, but can be used for products made from lower quality material. The correction factor can be based on the quality, the mass, or the market value (value-corrected substitution) (EC-JRC, 2010; Guinée et al., 2002). ILCD (EC-JRC, 2010) suggest to use a value-correction factor if the new use of the recycled material is unknown. The analysed product system incurs the burden for the end-of-life recycling process of the discarded analysed product, but also receives corrected credits for the amount of recycled material that is obtained from end-of-life recycling of the discarded analysed product.

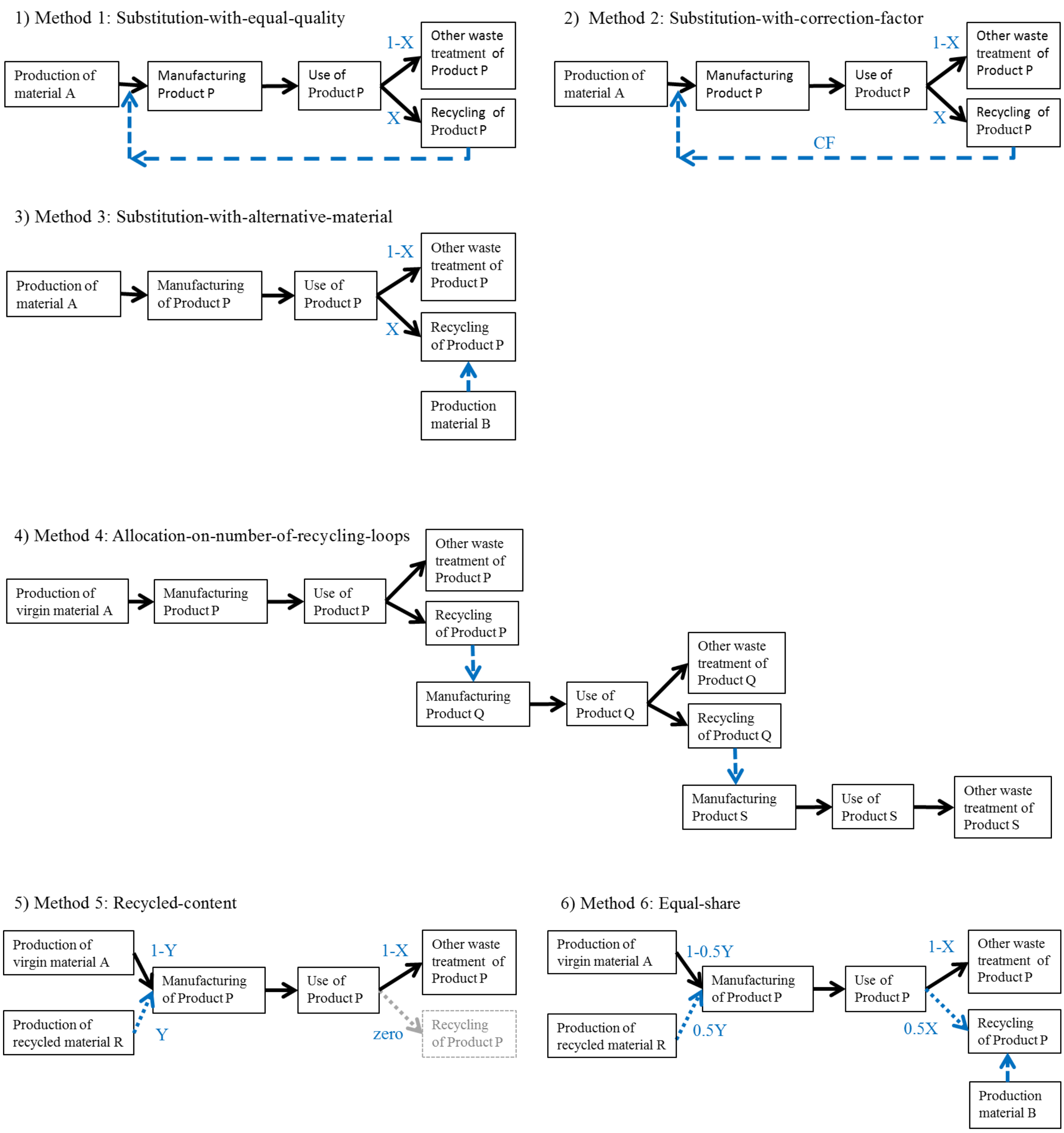

Figure 5.2: Life cycle of product P under the six recycling modelling methods. X = the fraction of discarded product P sent to the recycler. Y = the fraction of recycled material used in product P. CF = correction factor. The blue dotted lines represent the main idea on where to assign the impacts of the recycling process and the recycled material. The grey line depicts a flow which is not accounted for in the model.

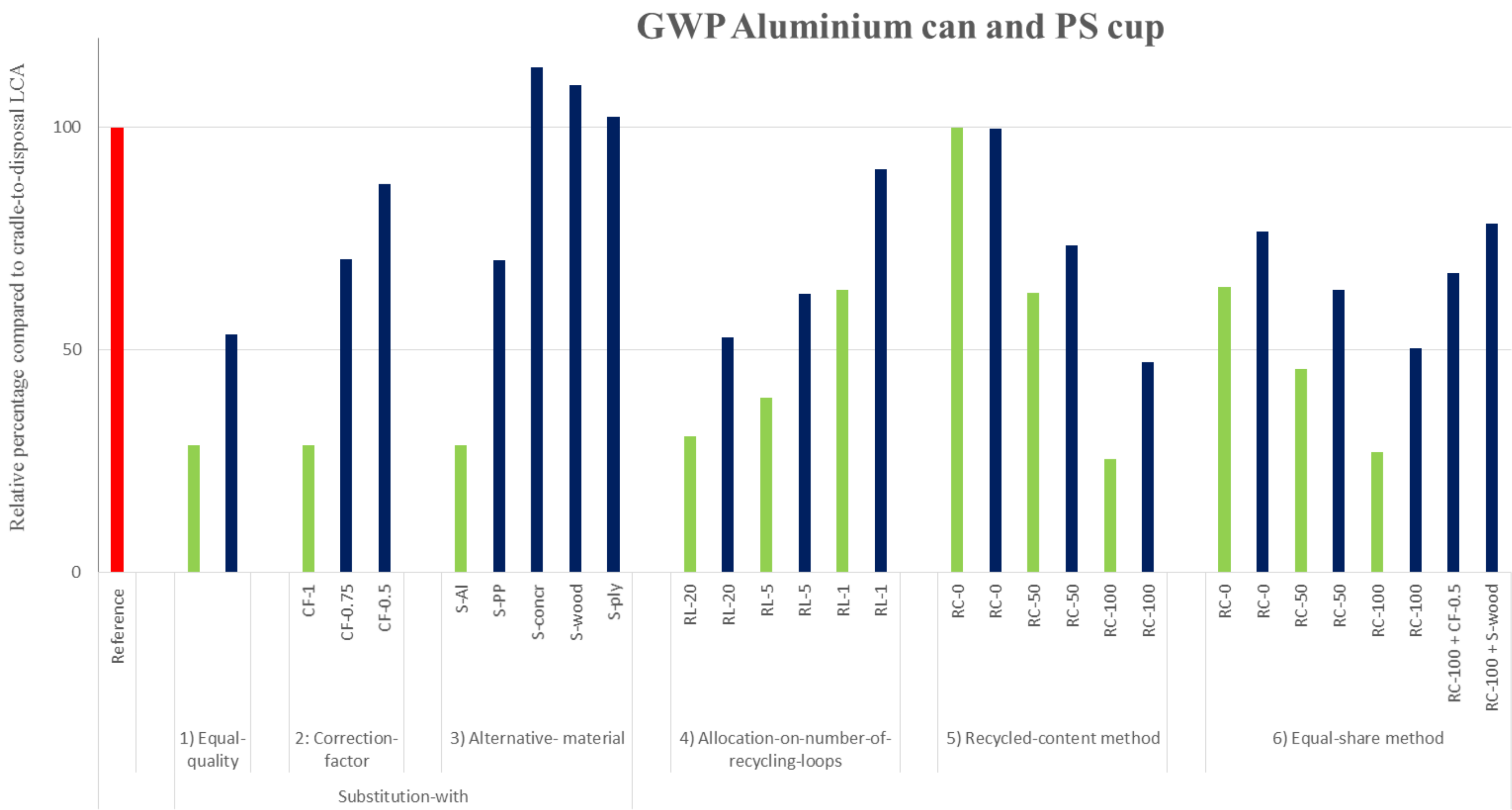

Figure 5.3: LCA results for the global warming potential (GWP) of the aluminium can (light green) and the PS cup (dark blue), using different methods to handle recycling. The reference LCAs (cradle-to-disposal LCA) for the aluminium can and PS cup are set at 100%, shown in red. The results for the aluminium can and the PS cup are presented relative to their respective reference LCAs. Products in all scenarios, except for RC>0, are made from 100% virgin material. All results (except in the reference LCA) consider a 100% recycling of the product as waste treatment. Abbreviations: Al = aluminium; PS = polystyrene; CF = correction factor; S = substituted material; RL = number of recycling loops; RC = percentage of recycled material in the product.

Substitution-with-correction-factor method applied to the aluminium can

Recycled aluminium has the same quality and properties as virgin aluminium. There is no separate market value (price) for recycled aluminium, but only for aluminium. The correction factor is in this case 1, and the GWP result in the same as in the *substitution-with-equal-quality* method (see CF-1 in Figure 5.3).

Substitution-with-correction-factor method applied to the PS cup

Recycling of PS has an effect on the properties of PS due to shortening of the polymer chain length (Vilaplana et al., 2006). If recycled PS can still replace part of virgin PS, then a closed-loop approach can be applied with a correction factor for the quality drop. Several LCAs incorporated this quality drop in plastics, which varies from 50% (Ligthart and Ansems, 2007), 60% (van der Harst and Potting, 2014; van der Harst et al., 2014), 70% (Merrild and Hedal Kløverpris, 2010), to 90% (Ligthart and Ansems, 2004). The use of a value-correction factor depends on the market prices and requires stable market prices over time. The price for virgin PS, however, relates to the price of crude oil and has fluctuated during the last decades

Results are calculated with for a correction factor of 0.75 and 0.5, to see their effect. The GWP for both correction factors (CF-0.75 and CF-0.5 in Figure 5.3) are lower compared to the reference LCA. The GWP results are, however, higher compared to the *substitution-with-equal-quality* method. The burden of the recycling process still exists, but the product system receives only a portion of the credits for the recycled PS.

5.3.4 Method 3: Substitution-with-alternative-material

If the quality and/or inherent properties are degraded in such a way that the recycled material cannot substitute the input material, then the recycled material may replace a lower grade material or alternative materials (see Figure 5.2.3). In this case we need to know or assume what material is replaced by the recycled material in the next product system. The system boundary of the analysed product system is expanded with the production of other alternative material(s), and next credited with the avoided production of the replaced alternative material (EC-JRC, 2010). System expansion increases the complexity of the studied system (Zamagni et al., 2008). Furthermore, data for the substituted material should be available, and these data should not include too large data uncertainty (Ekvall and Finnveden, 2001; Ekvall and Tillman, 1997). ILCD (EC-JRC, 2010) suggest to credit for the alternative material if the new use of the recycled material is known. The analysed product system gets the environmental burdens of the end-of-life recycling process of the discarded analysed product, similar to the other *substitution* methods. The analysed product system receives, different from the other *substitution* methods, credits for the avoided production of the replaced alternative material.

Substitution-with-alternative-material method applied to the aluminium can
The recycled aluminium of the aluminium can can replace virgin aluminium since there is no quality drop. There is no need to use alternative material to credit the recycled aluminium. The result of this method is thus the same as the previous substitution methods (see S-Al in Figure 5.3).

Substitution-with-alternative-material method applied to the PS cup
Mixing PS with other plastics, contamination with other materials such as fillers, colouring agents, dirt, or exposure to high temperatures all degrade the quality of the recycled PS (Al-Salem et al., 2009; Vilaplana et al., 2006). Recycled PS is usually not used for thin-walled products, but can be used for products with thicker walls such as flower pots or office supplies (pens, pencils and other desk items (Maharana et al., 2007)). Recycled PS and mixed plastics are also used to make sign-poles and benches (Lankhorst, 2014), plastic ‘wood’ (Kedel, 2014), or outdoor furniture (Ferreira et al., 2014). Bergsma et al. (2011) used a mix from hardwood (azobé), concrete and polypropylene (PP) as replacement for mixed plastics, and Ferreira et al. (2014) suggested plywood.

The *substitution-with-alternative-material* method is applied with credits for PP, concrete, azobé wood, and plywood as replacement for the recycled PS (see S-x scenarios in Figure 5.3). The production of plastics requires more energy compared to the production of concrete or wood products, and thus the GWP for the replacement of PP is lower compared to concrete and wood. PP production on the other hand requires less energy compared PS production, and receives less credits. Note that the GWP for a replacement by concrete or wood are close to or even higher than the reference LCA. The recycling process itself consumes energy, and this is not offset by credits for avoided production of concrete or wood.

5.3.5 Method 4: Allocation-on-number-of-recycling-loops

Allocation can be based on the number of uses or the number of times the material can be recycled, which is in this paper further referred to as the *allocation-on-number-of-recycling-loops*. This method applies allocation on the life cycle of the material itself rather than on the life cycle of the analysed product system as applied in *substitution* methods. The material is followed until it is completely disposed of as final waste (i.e. processed in another way than recycling). Material A in Figure 5.2.4 is used three times (i.e. recycled twice) before it is discarded in a waste treatment other than recycling.

One way to allocate the environmental impact of a material is to first calculate how much virgin material is replaced by the recycled material throughout its whole life span. Next, the burden of the production of virgin material, the (multiple) recycling processes, and the final disposal of the material are equally divided among the products according to the portion of material used in each product (EC-JRC, 2010; Newell and Field, 1998). The impacts from the production of virgin material A in Figure 5.2.4, the burdens of the recycling processes for the discarded products P and Q, and the disposal (other than recycling) of products P, Q and S are divided among the different products life cycles of all products, i.e. products P, Q and S. The number of times the material can be recycled is thus important. The recycling rate in this method is the average recycling rate for the material itself, and not the recycling rate of the product at stake. A correction factor (based on the market values) is needed if the quality of the material decreases during recycling (EC-JRC, 2010). It is not clear how this correction factor can be included in the method (Lindfors et al., 2012). The *allocation-on-number-of-recycling-loops* method thus allocates a proportion of the environmental burdens of virgin material production, the recycling process(es), and the final waste treatment of the material to the analysed product system. This proportion is based on the amount of material used in the analysed product, the recycling rate and the number of times the material can be recycled.

Allocation-on-number-of-recycling-loops method applied to the aluminium can

We assume a hypothetical recycling rate of 100% in the calculations. There is a 4% material loss during the recycling process itself due to its efficiency of 96%. Although aluminium theoretically can be recycled infinitely, we use three scenarios with different numbers of recycling loops to illustrate the effect of the amount of loops on the GWP results. We assume that the final waste treatment of the aluminium (after multiple recycling loops) is landfilling (50%) and incineration with energy recovery (50%). The GWP is calculated for one

(RL-1), five (RL-5), and twenty (RL-20) recycling loops. GWP systematically decreases as the number of recycling loops increases (Figure 5.3; RL-x scenarios). The biggest achievement is accounted for from no recycling to one recycling loop, when the burden of the production of virgin material is cut in half. The GWP for the *allocation-on-number-of-recycling-loops* method converges to the GWP of the *substitution-with-equal-quality* method if the number of loops increases.

Allocation-on-number-of-recycling-loops method applied to the PS cup

The recycling rate is set at 100%, but here a 5% loss occurs in each recycling process. The quality of the recycled material is, for simplification purposes, assumed to be equal to virgin PS. We assume that the final waste treatment of recycled PS (after multiple recycling loops) is landfilling (50%) and incineration with energy recovery (50%). The GWP is calculated for one (RL-1), five (RL-5) and twenty (RL-20) hypothetical recycling loops. Similar to the aluminium can case, GWP decreases as the number of recycling loops increases (Figure 5.3). The decrease in GWP (compared to the reference LCA) is small if PS is recycled only once, due to the energy consumption of the recycling process itself. The GWP for the *allocation-on-number-of-recycling-loops* method converges to the GWP of the *substitution-with-equal-quality* method if the number of loops increases.

5.3.6 Method 5: Recycled-content

The *recycled-content* method, which is also called the 'cut-off' method, considers the use of recycled material R in the analysed product system (see Figure 5.2.5). Information on the percentage of recycled material R and virgin material A entering the analysed product system is required. Virgin material A bears the complete environmental burden of its production. Recycled material R only bears the environmental burden of the recycling efforts to turn waste into recycled material, and thus this recycling process is in fact the 'production' process of material R.

Recycling as an end-of-life waste treatment of the discarded product P is outside of the system boundary in the *recycled-content* method, and hence the method does not provide credits for the recycled material which is produced from discarded product P. In fact, the end-of-life recycling process of product P and the subsequent recycled material are assigned to the next product system which uses the recycled material as input material. The next product system thus bears the burden of the recycling process and receives the credits for avoided production of new material. The analysed product thus incurs the burden of the recycling process needed to produce the recycled material used as input material, and implicitly receives ‘credits’ for avoided production of new material. The analysed product system omits any impact of the end-of-life recycling process of the discarded analysed product.

The PAS 2050 specifications (BSI, 2011) and the Greenhouse Gas Protocol (Pankaj Bhatia et al., 2011) suggest the *recycled-content* method if the recycled material does not have the same inherent properties as the virgin material. The *recycled-content* method according to PAS 2050 (BSI, 2011) is applied to both case studies.

Recycled-content method applied to the aluminium can
GWP is calculated for three (hypothetical) scenarios with recycled-content percentages of: zero percent (all virgin material; RC-0), 50% (RC-50), and 100% (all recycled material; RC-100). GWP for the can which is completely made from virgin aluminium is equal to the reference LCA, even though all aluminium cans are recycled at the end of their lives. GWP clearly depends on the percentage of recycled aluminium present in the can (Figure 5.3; RC-x scenarios). The GWP for the can which is completely made from recycled material (RC-100) is slightly lower compared to GWP in the *substitution-with-equal-quality* method. The *substitution-with-equal-quality* method accounts for the production of 4% virgin aluminium, which is lost in the recycling process (see section 5.3.2). The production of virgin aluminium requires twenty times more energy compared to the recycling of aluminium, and hence result in a higher GWP for the *substitution-with-equal-quality* method.

Recycled-content method applied to the PS cup
The PS cup contains only virgin PS, since plastic food packaging products require the use of virgin material due to concerns on food safety and hygiene standards. For the *recycled-content* method, however, we assume that the PS cup can contain recycled material for the purpose of the comparison. For the PS cup also three (hypothetical) scenarios with different recycled-content percentages are applied, i.e. RC-0, RC-50, and RC-100. Similar to the aluminium can, the GWP clearly depends on the percentage of recycled PS used in the cup (Figure 5.3; RC-x scenarios). The GWP of the cup which is completely made from recycled material (RC-100) is lower in the *recycled-content* method compared to the *substitution-with-equal-quality* method. The latter method accounts for the 5% PS lost in the recycling process. Production of PS requires more energy compared to its recycling, similar to the above case.

5.3.7 Method 6: Equal-share

The *substitution* methods promote the recyclability of the product and provide credits for the recycled material which results from the end-of-life recycling of the product. The *recycled-content* method, on the other hand, promotes the use of recycled material in the product itself, and thus provides credits for avoiding the production of new material. The methods do not promote both goals at the same time. The *equal-share* method combines both visions by equally rewarding the recyclability of the product and the amount of recycled material used in the production of the product. The results for the analysed product in the *equal-share* method include: 50% of the burden and credits from the produced recycled material from product P, and 50% of the burden and credits due to the use of recycled material R in the product P itself (Figure 5.2.6). This comes down to calculating the results for both the *substitution* method and the *recycled-content* method, and next taking the average of both results. The Product Environmental Footprint (PEF) Guide from EC-JRC (EC-JRC, 2013) prescribes this method to handle recycling in both closed-loop and open-loop recycling systems.

Equal-share method applied to the aluminium can

Recycling of the discarded aluminium cans produces aluminium with the same quality as virgin aluminium, as seen in the *substitution* methods above. The recyclability part of the *equal-share* method is therefore calculated based on the *substitution-with-equal-quality* method. We again include three scenarios with different recycled-content percentages of the aluminium can (RC-0, RC-50, RC-100). The GWP of the can which is completely made of virgin aluminium (RC-0) receives only half of the credits from the recycled aluminium which is produced from the discarded cans (Figure 5.3). The GWP decreases if the recycled-content of the aluminium can increases (RC-50 and RC-100). The lowest GWP is achieved if all discarded aluminium cans are recycled, *and* the aluminium can is completely made of recycled aluminium (RC-100).

Equal-share method applied to the PS cup

Credits for the recycled PS of the discarded PS cups can be applied in different ways, as shown in the *substitution* methods. The GWP is here first calculated with the *substitution-with-equal-quality* method representing the recyclability part of the equal-share method. We again include three (hypothetical) scenarios with different recycled-content percentages of the PS cup (RC-0, RC-50, RC-100).The GWP results follow the same pattern as in the aluminium can case (Figure 5.3). The lowest GWP is achieved for a PS cup which is recycled after it is discarded, *and* which is made completely from recycled PS. The addition of a correction factor to account for a quality drop in the recycled material, or the substitution of the recycled PS by other materials increases the GWP since less credit is received (see RC-100 + CF 0.5 and RC100 + S-wood in Figure 5.3).

5.3.8 Evaluation

Comparison of the methods

Table 5.5 summarizes the main characteristics of the six methods. All methods regard the recycling process as the 'production' process for recycled material. All methods link the burden of the recycling process itself to the credits for avoided production of new material. The main difference among the methods is *where* to assign these burdens and credits in the analysed product system: either on the input side (*recycled-content* method), or at the end-of-life side (*substitution* methods). The *recycled-content* method is therefore essentially different from the *substitution* methods. The *equal-share* method includes burdens and credits for both sides, and thus combines two contradicting viewpoints of the *substitution* and *recycled-content* methods. The *allocation-on-number-of–recycling-loops* method refrains from assigning burdens and credits on a particular side, since the calculation is essentially different from all other methods. The *allocation-on-number-of–recycling-loops* method divides the burdens of all (multiple) recycling processes of the material and the burdens of the production of the virgin material among the multiple uses of the material.

The three *substitution* methods promote the recyclability of products. *Substitution* methods include the recycled material which is made at the end-of-life of the analysed product, and provide credits for this recycled material. The *allocation-on-number-of-recycling-loops* method also rewards the recyclability, but the focus is on the life expectancy of material itself, rather than on the life cycle of the analysed product. The *recycled-content* method indirectly also encourages the recyclability of products by stimulating the use of recycled material in a product. The use of recycled material requires recycling as a waste treatment, and diverts discarded products from entering other waste treatments. The *recycled-content* method, however, provides no benefits for the recycled material which is produced at the end-of-life of the analysed product. The *equal-share* method includes only half of these credits.

Substitution methods take credits already *now* for the supposed future use of the recycled material (Frischknecht, 2010). This is debatable for products with a long lifespan in particular (e.g. aluminium used in planes, trains, industrial equipment, or buildings), because it is not sure if there will be a future market for the recycled material. The same applies for the *allocation-on-number-of-recycling-loops* method, as the quantifying of the analysed product system requires involving the expected use of the material in subsequent products. The cascade for paper and pulp is well-known, but unclear for most other materials. Limiting to the present or near future market situation of recycled material would ensure that credits can only be applied if there is a real demand for recycled material. Excluding the future use of long-lasting materials, on the other hand, undermines the durability of the material.

The *recycled-content* method rewards the *actual* use of recycled material in the product and avoids any future predictions. The *recycled-content* method furthermore does not need to consider any quality loss issues, as occur in the *substitution-with-correction-factor* method and the *substitution-with-alternative-material* method. The *allocation-on-number-of-recycling-loops* method also rewards the use of the recycled material as this method tracks the use of the material through the next subsequent product cycles. This method applies credits towards the total use of the material, and proportionally rewards them to the analysed product. The *equal-share* method includes only half of the credits for the use of recycled material in the analysed product.

Recycled material exists because virgin material was produced in the first place, and its availability depends on the recyclability and recycling rate of materials and products. These factors are not considered in the *recycled-content* method. If a specific product is (promoted to be) made from recycled material, then this means that the recycled material is not available for use in other products where its use might be more beneficial. Plastic food packaging products require the use of virgin material due concerns on food safety and hygiene standards, and are in the *recycled-content* method penalized for complying with these standards. Novel materials (e.g. bioplastics) for which a mature recycling infrastructure is not yet established also have a disadvantage in the *recycled-content* method. The *substitution* methods can include the expected future use of recycled novel material, but credits cannot be awarded if the existence of a mature market is required.

Table 5.5: Characteristics of methods used to account for recycling in LCA.

	Substitution method with			4: Allocation-on-number-of-recycling-loops method	5: Recycled-content method	6: Equal-share method
	1:Equal-quality	2:Correction-factor	3:Alternative-materials			
Philosophy	Considers the end-of-life(EOL) fate of products; Promotes the design of products for its recyclability; Is based on material stewardship			Impacts of the virgin material production is equally shared among all uses of the material	Rewards the use of recycled material in products; Stresses preservation of natural resources	Promotes the recyclability of the product and the use of recycled material in the product
Principle	Recycled material at EOL displaces production of material			Impacts from virgin material production and waste treatments are equally shared among all uses of the material	The use of recycled material does not bear any burden from virgin material production	Equally combines the credits from avoided production of material and the burden free use of recycled material
	The product system is credited with avoided production of virgin material	Same as equal-quality method, but with a correction factor	The product system is credited with avoided production of other alternative material(s)			
Waste management perspective	Recovery and recycling of material				Diversion of material from sending it to the incineration or landfill	Recovery of material and the use of recycled material
Sustainability concept	Weak; Uses grants now for uncertain future use; Shifts burden into future				Strong; No shift of burdens to unknown future; Includes burden for first use of virgin material	Mixed

	Substitution method with			4: Allocation-on-number-of-recycling-loops method	5: Recycled-content method	6: Equal-share method
	1:Equal-quality	2:Correction-factor	3:Alternative-materials			
Burden recycling process and credits for recycled material assigned to	Product which produces the recycled material as EOL waste treatment			Equal share among material consumption	Product which uses the recycled material as input material	Shared between product producing and product using recycled material
Impacts of virgin material production assigned to	Last material consumption			Equal share among material consumption	First material consumption	Shared between first and last material consumption
Guidelines using method	ISO 14040(2006a); ILCD (EC-JRC, 2010)				GHG protocol (Pankaj Bhatia et al., 2011); PAS 2050 (Pasqualino et al., 2011)	PEF (EC-JRC, 2012)
	GHG protocol (Pankaj Bhatia et al., 2011); PAS 2050 (BSI, 2011)					
Additional required data	None	Quality drop of recycled material compared to virgin material based on the quality, mass, or economic values	Production of other substituted alternative material	Average recycling rate of material; Number of times material can be recycled; Quality drop of recycled material compared to virgin material	Percentage recycled material in product	Percentage recycled material in product

	Substitution method with			4: Allocation-on-number-of-recycling-loops method	5: Recycled-content method	6: Equal-share method
	1:Equal-quality	**2:Correction-factor**	**3:Alternative-materials**			
Strength	Easy to use	Includes changes in properties (quality loss) of recycled material		Fair allocation among each use of the material	Easy to use	
Weakness	Takes credits now for unsure future use of recycled material; No credits for use of recycled material entering the product system			Difficult to use; Focus is on the material, not on the product system itself; Does not account for the recyclability of each separate product; Incorporation of quality loss in difficult	Does not reflect effect of recycling as EOL; Does not consider recyclability and recycling rate of products; Fails to stimulate use of novel materials, for which a mature recycling market is not yet established	
		Value correction needs stable market prices	Increases complexity of the product system due to inclusion of processes (system-expansion) not related to original product system; Choice of substituted product is debatable			
ISO compliant	Yes				No	

The *equal-share* method is a compromise between the contradicting viewpoints on where to assign credits for recycled material. It stimulates both the production of recycled material (recyclability of the product) and the use of recycled material in the product. The *equal-share* method shares both the benefits and draw-backs of the *substitution* methods and the *recycled-content* method.

Methods used in the aluminium can LCAs

The European Aluminium Association adopted the *substitution-with-equal-quality* method, as promoted by the metal industry, to model recycling of aluminium products if the properties of the recycled aluminium are the same as virgin aluminium (Atherton, 2007; EAA, 2013a, c). The quality of the aluminium may degrade, however, due to inclusion of alloys and/or impurities, even though the melting process of aluminium has no effect on the structure of aluminium (Dubreuil et al., 2010; Gaustad et al., 2012). The EAA (2013a), therefore, suggest a conservative approach and proposes the use of a value correction if the market price of the recycled material is lower than the price for virgin aluminium, i.e. the *substitution-with-correction-factor* method.

The use of aluminium is still increasing (EAA, 2011). The availability of recycled aluminium is, however, still limited and thus virgin aluminium has to be produced to satisfy the demand (Atherton, 2007). The recycling rate of the products should be enhanced to increase the amount of discarded aluminium available for recycling. The *recycled-content* method does not cover this potential, and the EAA (2013a) considers this method unsuitable for the use in LCA. Furthermore, the *recycled-content* method is based on the history of the material and does not stimulate a better metal management (Atherton, 2007).

LCAs on aluminium cans include the *substitution-with-equal-quality* method (Amienyo et al., 2013; Detzel and Mönckert, 2009; Gatti et al., 2008; PE Americas, 2010), the *allocation-on-number-of-recycling-loops* method (Franklin Associates, 2009b), the *recycled-content* method (PE Americas, 2010), and the *equal-share* method (Detzel and Mönckert, 2009).

Methods used in the PS cup LCAs

The plastic sector has no recycling modelling guideline similar to the metal industry. The *substitution-with-equal-quality* method with a 1:1 replacement for virgin material is often applied for thermoplasts (PS, PET, polypropylene, polyethylene) (Ferreira et al., 2014; Lazarevic et al., 2010; Michaud et al., 2010; Shen et al., 2010). LCAs for the PS cup often consider an open-loop system and apply the *substitution-with-correction-factor* method with a value-correction for the quality drop, or the *substitution-with-alternative-material* method (Bergsma et al., 2011; Ligthart and Ansems, 2004; van der Harst and Potting, 2014; van der Harst et al., 2014).

5.3.9 Country specific waste management practices

The impact results in the previous sections are calculated based on a hypothetical 100% recycling rate. In practice, however, this rate is not achieved. Figure 5.4 and 5.5 present the results calculated for the actual average European recycling fractions and the (assumed) average European incineration and landfilling fractions (Table 5.6). Figure 5.5 also presents the results for seven European countries, based on their waste management practices (Table 5.6). The results are additionally based on the actual amount of recycled material used in the products (recycled material content).

Aluminium can

The average European recycling rate of aluminium cans is 67% (EAA, 2012), indicating that 67% of the used cans are sent to the recycler. Percentages of other waste treatments for aluminium cans were not available. We assume that the amount of discarded cans not sent to the recycler is equally divided between incineration and landfilling. Average European aluminium contains 52% recycled aluminium (EAA and OEA, 2004). We therefore assume that the aluminium can contains 52% recycled aluminium. We use the *substitution-with-equal-quality* method since recycled and virgin aluminium have the same quality.

LCA results for the average European situation are presented for all ten impact categories in Figure 5.4. Note that the results are presented as relative percentages *within* each method. LCA results for the *substitution-with-equal-quality* method (Figure 5.4.a) resemble the ones for the reference LCA (Fig. 5.1.a), but with clearly visible credits for recycled aluminium which is produced from the discarded cans. LCA results for the *recycled-content* method (Figure 5.4.b) also resemble the reference LCA results, but part of the virgin material is now replaced by recycled material, i.e. resulting in lower impacts from aluminium production since less virgin aluminium is used. Landfilling and incineration inflict no impact in both methods (Figure 5.4.a and b). This means that the actual percentages of discarded cans sent to landfilling or incineration are not essential, only the recycling rate of the cans is required.

The results for the average European situation and the individual countries are further only compared for GWP, similar to the previous sections, since all impact categories follow the same pattern (Figure 5.4.a and b). The lowest GWP results are in the *substitution-with-equal-quality* method for those countries with high recycling rates (Norway, Finland, and Germany), see Figure 5.5.a. GWP results in the *recycled-content* method show the same value for all countries. Aluminium is traded throughout Europe, and thus the average fraction of recycled aluminium present in aluminium is here assumed to be equal for all countries.

GWP results show typical outcomes based on the underlying philosophies of the two methods. GWP results of specific countries are according to the *substitution-with-equal-quality* method sometimes higher and sometimes lower compared to the *recycled-content* method, depending on the recycling rate in the country. The *substitution-with-equal-quality* method clearly stimulates the recyclability and recycling of products. The *recycled-content* method does not reward the recycling waste practice, hence countries with low recycling rates perform better in the *recycled-content* method. The choice of the method is thus crucial for the absolute GWP value as well as in the comparison among the countries.

Disposable PS cup

Plastic food packaging products require the use of virgin material due concerns on food safety and hygiene standards. The PS cup consequently consists of virgin PS and thus contains no recycled PS. Specific information on the final destination of PS cups is not available, hence information on the waste treatment options for discarded post-consumer plastic waste is used instead (PlasticsEurope et al., 2012). The average European recycling rate of post-consumer plastics packaging was 33% in 2011, approximately 33% was incinerated with energy recovery, and the remaining 33% was either landfilled or incinerated without energy recovery (EuPC, 2012; PlasticsEurope et al., 2012). The fractions of discarded plastic directed to the three waste treatments vary among counties (Table 5.6). We consider the fraction which was not recycled or incinerated with energy recovery as sent to the landfill.

We assume a degradation of the quality of the recycled PS due to contamination of the discarded cups. We apply the *substitution-with-correction-factor* method due to absence of the destination of the recycled PS, and use a correction factor of 0.6 based on PS prices in January 2015 (PlasticsNews, 2015).

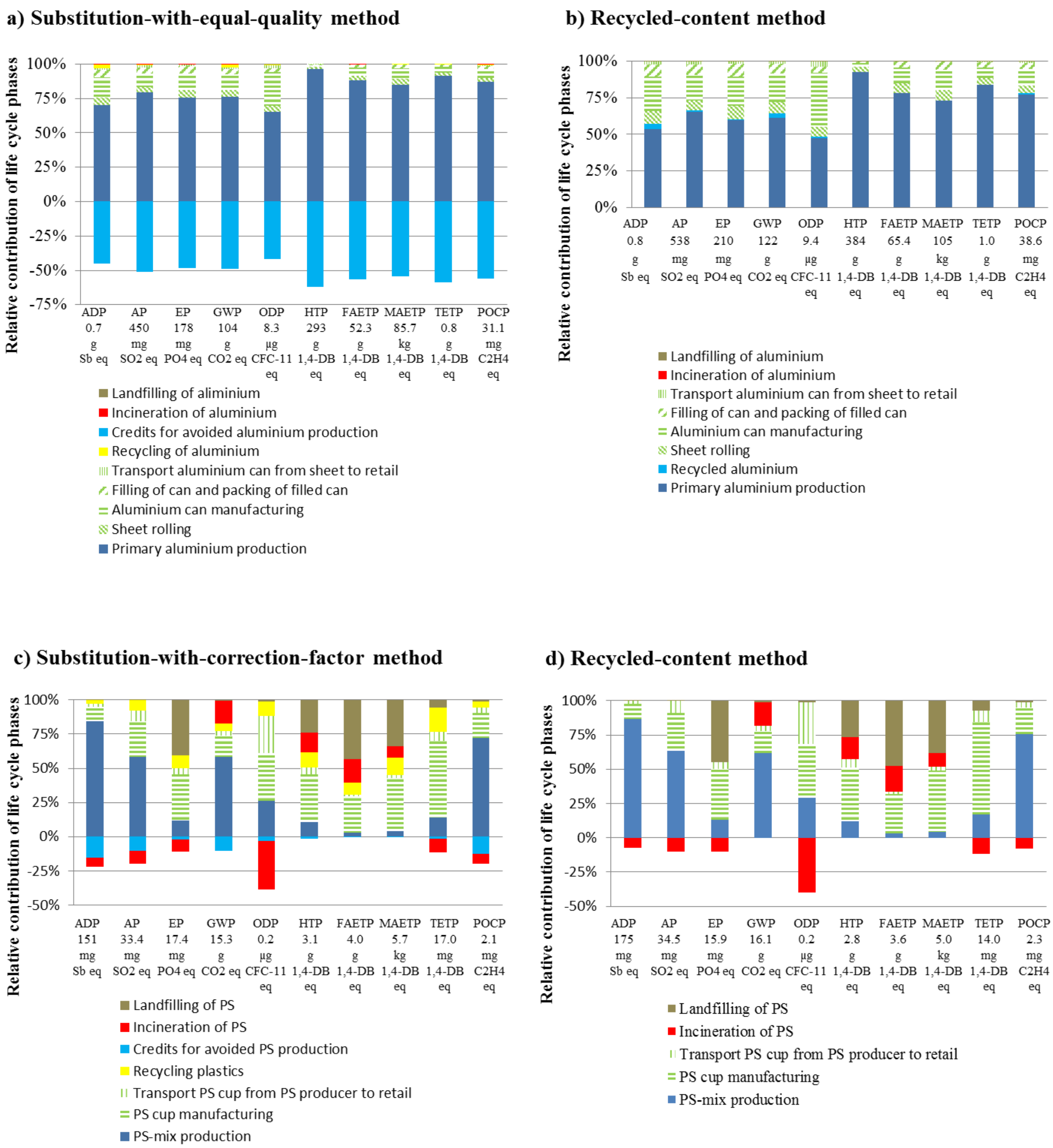

Figure 5.4: LCA results for the aluminium can (a and b) and PS cup (c and d), based on European average waste treatment rates. The graphs show the relative share of the life cycle phases in the complete life cycle of the product based on a *substitution* method (a and c) or the *recycled-content* method (b and d). The graphs show the relative contributions in ten environmental impact categories. Negative percentages indicate credits. For explanation of the abbreviation, see Figure 5.1.

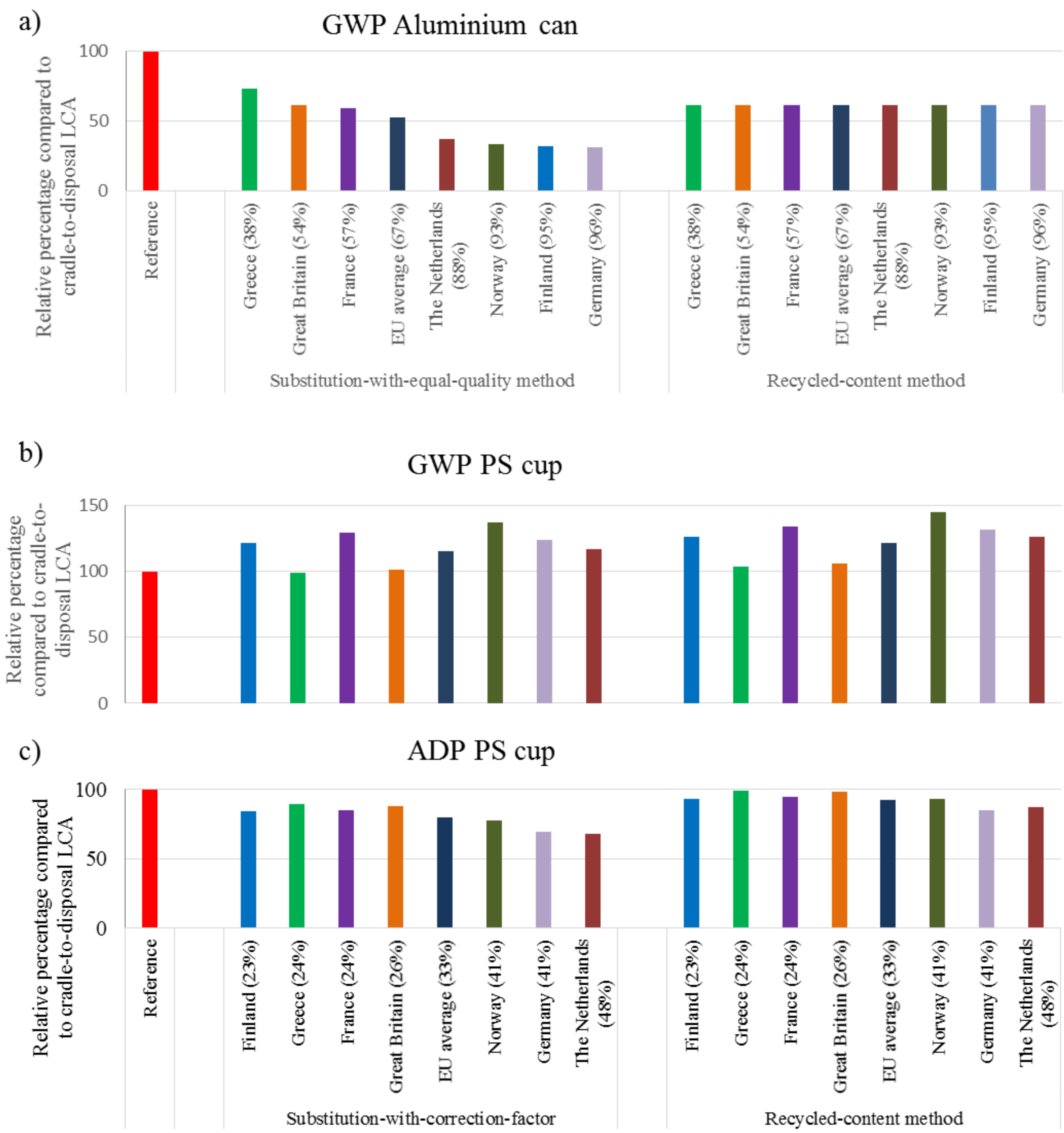

Figure 5.5: LCA results for the average European and seven country-specific waste management options for the aluminium can and the PS cup (see Table 5.6): a) GWP for the aluminium can; b) GWP for the PS cup; c) ADP for the PS cup. The countries are sorted on increasing recycling rates for the product (noted between brackets behind the countries). The results are shown using a substitution method and the *recycled-content* method to handle recycling in LCA. The *substitution-with-equal-quality* method is used for the aluminium can, and the *substitution-with-correction-factor* method, with a correction factor of 0.6, is used for the PS cup. Each country is represented by the same colours in the three graphs.

Table 5.6: Country-specific recycling rates for the aluminium can (EAA, 2012), waste treatment rates for post-consumer plastic waste (PlasticsEurope et al., 2012), and the composition of electricity production (Ecoinvent Centre, 2010) for the average European situation and seven countries. The waste treatment rates for the plastic waste are estimated from bar charts. We consider the plastic fraction which is not recycled or incinerated with energy recovery (MSWI) to be landfilled.

Country	**Aluminium can**	**Plastic waste**	**Plastic waste**	**Plastic waste**	**Main energy sources for electricity production**
	Recycling rate (%)	Landfilling rate (%)	MSWI rate (%)	Recycling rate (%)	
Europe (EU-27)	67	33	33	33	
Finland	95	40	37	23	27% nuclear; 19% coal; 18% hydro; 15% natural gas; 12 co-generation
France	57	34	42	24	78% nuclear; 11% hydro; 4% coal; 3% natural gas
Germany	96	1	58	41	27% nuclear; 25% lignite; 23% coal; 10% natural gas; 4% hydro; 4% wind
Great Britain	54	67	7	26	40% natural gas; 33% coal; 20% nuclear
Greece	38	73	3	24	59% lignite; 15% natural gas; 14% oil; 8% hydro
Great Britain	54	67	7	26	40% natural gas; 33% coal; 20% nuclear
Norway	93	4	55	41	98% hydro
The Netherlands	88	4	48	48	61% natural gas; 23% coal; 4% nuclear

LCA results for the average European waste management are presented for all ten impact categories in Figure 5.4.c and d. The results for the *substitution-with-correction-factor* method (Figure 5.4.c) resemble the results for the *recycled-content* method (Figure 5.4.d) since the PS cup is made from virgin material, but the *substitution-with-correction-factor* method includes impacts from the recycling process and credits for the avoided PS production. Incineration and landfilling inflict, contrary to the aluminium can, impacts in both methods. Incineration of PS releases emissions to the environment (e.g. carbon dioxide), but also provides credits for recovered electricity and heat. Incineration of PS ends up as a burden in several impact categories (including GWP), but as a credit in others (e.g.

abiotic depletion (ADP)). Landfilling also contributes to the environmental impact in several impact categories.

The results for the average European situation and the individual countries are further only compared for GWP and abiotic depletion (ADP), since these two impact categories react different to incineration of PS. For each country, credits for recovered electricity during incineration are based on the production of electricity in the corresponding country (Table 5.6).

GWP and ADP results provide conflicting outcomes. Greece and Great Britain have the lowest GWP of the included countries in *both* methods (Figure 5.5.b), even though the recycling rate of the Netherlands is almost twice as high. The Netherlands has a high incineration rate, which causes a burden in GWP. This implies that a higher recycling rate does not automatically lead to lower impact results. Diverting PS waste from the incinerator to the landfill and/or recycler does lower the GWP. For ADP, on the other hand, the countries with the highest recycling rates (Norway, Germany, the Netherlands) display the lowest results in both methods (Figure 5.5.c). Here, countries with high landfilling rates (Greece and Great Britain) have the highest ADP, because they do not receive credits from recycled PS and/or recovered energy.

Other interesting results stem from the credits for avoided electricity production, which are accounted for in both methods. Electricity production based on fossil fuels generates higher GWP and ADP compared to its production based on nuclear or renewable sources. Cleaner electricity production provides therefore fewer credits for recovered electricity compared to polluting fossil based power production. Norway and Germany have comparable waste rates for plastics, but the Norwegian GWP and ADP results are higher, because the credits for incineration are lower. That is due to Norwegian electricity mainly being produced from hydro power and German electricity being fossil oriented. GWP and ADP results thus also depend on the composition of the power production. Note that the country-specific electricity production is here only included for incineration, and average European electricity production is considered in all other processes. The material production and product manufacturing require considerable larger amounts of electricity compared to the received credits for incineration. The use of cleaner electricity in these processes will reduce the absolute GWP and ADP results even more.

The GWP results for each individual country are somewhat lower in the *substitution-with-correction-factor* method compared to the *recycled-content* method. The difference is more prominent for ADP results, because ADP has no emissions from incineration of PS. The use of a higher correction factor will lower the results in the *substitution-with-correction-factor* method, while a possible inclusion of recycled PS in the cup will lower the *recycled-content* method results. An increase of the recycling rate does not necessary lower the results in the *substitution-with-correction-factor* method, as is clearly presented for GWP results. This seems to contradict the philosophy of the *substitution* methods as discussed in the previous sections, and as presented in the aluminium can case. GWP results actually decrease with increasing recycling rates, but the influence of incineration surpasses this effect. The results for the *recycled-content* method are different for all countries, even though the amount of recycled material is the same for all countries. This also deviates from the aluminium can study. The GWP and ADP results for the PS cup highly depend on the fractions entering incineration and landfilling. The impact results for the PS cup thus depend on the applied method, the waste treatment management (recycling rate, incineration rate, landfilling rate), and the power production composition. Furthermore, the contributions of the different life cycle stages and waste treatments options of the PS cup differ in each of the ten impact categories. This stresses the importance to consider other impact categories besides GWP in the environmental evaluation of products.

5.4 General discussion

This paper evaluates six methods to handle the multi-functionality of recycling in LCA. In practise there are more methods proposed and applied, (e.g. Ekvall and Tillman (1997), ISO 14049 (ISO, 2000), the French AFNOR BP X30-323 (AFNOR, 2011), or the EPD guidelines (International EPD system, 2013)). We limit ourselves to a selection of frequently used and debated methods.

Our results for the aluminium can case study show a dependence on the applied method, the recycling rate of the discarded can, and the amount of recycled material used in the can. Aluminium can studies of PE Americas (2010) and Gatti et al. (2008) confirm the influence of the recycling rate on the results. PE Americas (2010) found a lower result for the *recycled-content* method (with 67% recycled material in the can) compared to the *substitution* method (with a 52% recycling rate), which is in line with our findings. Interesting is also that differences in absolute GWP results among aluminium can studies cannot be attributed to the used method or recycling rate alone (Amienyo et al., 2013; Detzel and Mönckert, 2009;

Franklin Associates, 2009b; Gatti et al., 2008; PE Americas, 2010). Other factors, for instance differences in data for aluminium production or can manufacturing, must also contribute to these differences. The electricity mix e.g. is an important issue in the production of aluminium (Liu and Müller, 2012).

Our results for the PS cup case study also show a dependency on the applied method, the recycling rate of the discarded cups, and the amount of recycled material used in the cup. In contrast to the aluminium can, however, also the other waste treatments (incineration, landfilling) and the composition of the credited energy in incineration are relevant in the country exercise. Other PS cup studies only used *substitution* methods, and we found no other PS cups studies which applied the *allocation-on-number-of-recycling-loops, recycled-content,* or *equal-share* method. Ligthart and Ansems (2004) reported a reduction of nearly 50% in GWP credits when the value-correction factor was lowered from 90% to 50%. Bergsma et al. (2011) found similar to our study variation in results when using different alternative materials for the recycled PS and suggested to credit with the material which provides the most environmental benefits. Van der Harst and Potting (2013a, 2014) and van der Harst et al. (2014) quantified the influence of the different ways to credit the recycled material in the *substitution* methods, and additionally point to the properties of the cups, the used data and waste treatments as important players in the LCA of PS cups. A *substitution* method which incorporates the actual quality reduction of the recycled material and the further possible use of the recycled material seems, despite its problems, more appropriate than ignoring a quality drop and bluntly adapting a closed-loop system (i.e. *substitution-with-equal-quality*).

The two case study are intentionally chosen because they differ in the input material (mix of recycled and virgin aluminium versus all virgin PS), the quality of the recycled material at the end-of-life (equal versus reduced quality), and the impact of other waste treatments (landfilling and incineration) on the discarded products. This study shows, other than single product cases, the influence of the methods on diverse materials and makes a comparison of the effect on the materials possible. Our results clearly show for the PS cup case that additional factors can play a (more) important role than the applied recycling method. This study also looks beyond GWP as the only environmental indicator. The PS cup case clearly shows that recycling affects other impact categories in different ways.

ISO 14044 procedures to handle recycling are ambiguous and have led to numerous guidelines. EC-JRC prepared several guidelines how to model recycling in LCA. The ILCD, published in 2010, promotes the use of the *substitution* methods (EC-JRC, 2010), but the PEF, published in 2013, promotes the use of the *equal-share* method (EC-JRC, 2013). PAS 2050 (BSI, 2011) and the Greenhouse Gas Protocol (Pankaj Bhatia et al., 2011) use both the *substitution* and *recycled-content* method, depending on the properties of the recycled material. The guidelines keep changing and evolving due to new insights into the recycling methods and recycling perspectives. Industrial sectors (metal industry, aluminium association, paper industry) made recommendations on the preferred method for their specific material (Atherton, 2007; Cederstrand et al., 2014; EAA, 2013c). This abundance of (conflicting) guidelines is not helpful for the LCA practitioner. Is it possible to solve the multi-functionality problem by providing only one method? As long as there are multiple methods available and used, we should at least be transparent in the method we apply. Nevertheless, there is a clear need for unambiguous guidelines. Guidelines need to include sensitivity analysis of the methods. Guidelines also need to incorporate the reaction of the recycling market on the supply and demand for recycled material. This means applying credits only if there is a market for the additional recycled material, and also applying credits based on the actual quality of the recycled material. New and unambiguous ISO guidelines would be a great step forwards in making more robust LCAs.

5.5 Conclusion

Modelling the impacts of recycling and the use of recycled material in products has led to recurring debates in the LCA community. This paper evaluates six widely used methods to handle the multi-functionality of recycling in LCA: 1) *substitution-with-equal-quality*, 2) *substitution-with-correction-factor*, 3) *substitution-with-alternative-material*, 4) *allocation-on-number-of-recycling-loops*, 5) *recycled-content* method, and 6) *equal-share* method. These methods are applied to two case studies: an aluminium can and a disposable polystyrene (PS) cup.

The main difference among the methods stems from the underlying assumption on where to apply credits for recycled material. The three *substitution* methods stimulate the recyclability of the product, and provide credits for recovered material to the product system producing recycled material. The *allocation-on-number-of-recycling-loops* method tracks the use and reuse of the material throughout the life cycle of the material, rather than the life cycle of a specific product. The benefits of the reuse of the material are proportionally rewarded for each product using the material. The *recycled-content* method stimulates the use of recycled material in products, and hence credits material entering the product using recycled material. The *equal-share* method makes a compromise, and equally credits the use of recycled material in the product and the production of recycled material as end-of-life waste treatment of the product.

The quality of the recovered recycled material at the end-of-life of a product might be reduced compared to the quality of the material entering the product. A correction factor or credits for alternative material can be applied to account for such quality loss, but the choice of an appropriate correction factor and alternative material is debatable, especially when correcting with (volatile) economic values of virgin and recycled material. Accounting for quality reduction is incorporated in all methods except the *substitution-with-equal-quality* and *recycled-content* methods which do not require any accounting for quality loss.

The six methods are applied to the two case studies, and involve hypothetical scenarios on the recycling rate and recycled content of the products. The results for the aluminium can case study are straightforward and clearly follow the underlying principles of the methods. The results depend on the applied method, the recycling rate of the product, and the amount of recycled material used in the product. The results for the PS cup case study are more complex. The correction factor used for the quality drop of the recycled PS, as well as the choice of alternative material that can be replaced by the recycled PS considerably influence the results. Incorporating quality loss is, however, preferable over bluntly ignoring any possible degrading of PS during recycling.

We again applied the *substitution* method and the *recycled-content* method to the two case studies, but now for seven European countries and their respective waste management practices. The results for the aluminium can again clearly depend on the applied method, the recycling rate of the product, and the amount of recycled material used in the product. The results for PS cup additionally depend in both methods on the fractions of the discarded cups sent to the landfill and incinerator. The assessment of PS incineration is important, because it provides credits for energy recovery which depends on the type of fuel used to credit the avoided production of electricity. Recycling PS, furthermore, affects several environmental impact categories in different ways. This stresses the importance to consider more impact categories than global warming alone. The PS cup case study shows that no general conclusions can be made on the effects of the recycling modelling methods on the

LCA results. Influences from other waste treatments and credits for alternative substituted materials can surpass the influence of the method selection.

This study illustrates that the multitude of methods, changes in guidelines, and separate guidelines for specific materials do not make it easier for the LCA practitioner to choose an appropriate recycling modelling method. We need guidelines that include sensitivity analysis of the methods. Incorporation of the reaction of the recycling market on an increased demand for supply of recycled material would also be a great improvement. New and unambiguous ISO guidelines would be a great step forwards in making more robust LCAs.

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6 General discussion

6.1 Introduction

LCA results require among others to be robust and trustworthy if they are used in decision making in industry and policy making (Finnveden and Ekvall, 1998; Geisler et al., 2005; Guinée et al., 2002; Ingwersen and Stevenson, 2012). Robust results are expected to be insensitive to most known uncertainties (IPCC, 2001). In practice, LCA results for the same product can considerably deviate (see e.g. Chapter 2, Finnveden and Ekvall (1998), Lazarevic et al. (2010), Michaud et al. (2010), Padey et al. (2012), Price and Kendall (2012), von Falkenstein et al. (2010), Weiss et al. (2012), and Wenzel and Villanueva (2006)).

Uncertainties in LCA can occur in a number of places (locations), for instance in the model system definition representing the life cycle of the product, in the philosophy behind the LCA method, in the model translating relationships between substances and their effect on the environment, or in the used data (Table 6.1). Types of uncertainty include variability, choices, and unreliability (Table 6.1). Uncertainty management in LCA typically focusses on data uncertainty (Finnveden et al., 2009; Heijungs and Huijbregts, 2004; Lloyd and Ries, 2007; Ross et al., 2002). Variability among data sets from different databases is sporadically addressed. Choices are usually addressed separately from data uncertainty, namely by means of sensitivity analysis or scenarios (Heijungs and Huijbregts, 2004; Huijbregts, 1998a). The simultaneous handling of different types of uncertainty is sporadically performed in LCA.

The aim of this thesis is ***to evaluate whether the use of multiple data sets and multiple modelling options can increase the robustness of LCA results.***

The aim of the thesis is addressed through three research questions:

1) What are reasons for differences in LCA results for the same product?
2) Can the use of multiple data sets for a process increase the robustness of LCA results?
3) Can the inclusion of multiple modelling options increase the robustness of LCA results?

Question three is further divided into modelling options for waste treatments (a representation of the life cycle of a product), and options in the modelling of recycling in LCA (underlying modelling philosophies). The research questions are addressed by means of case studies for disposable beverage cups and aluminium beverage cans.

Table 6.1: Locations and types of uncertainty. ● denotes a relationship. The grey areas represent the focus areas of this thesis.

Location of uncertainty	Type of uncertainty		
	Variability	***Choices***	***Unreliability***
Model			
Modelling the product life cycle	●	●	●
Underlying modelling philosophy	●	●	
Modelling environmental impacts	●	●	●
Data			
Inventory data	●	●	●
Available databases	●	●	

This final chapter reflects on all of the previous chapters and provides answers to the research questions. I discuss the results of the previous chapters in a broader perspective, and address the feasibility of the use of multiple data sets and multiple modelling options in uncertainty management. This final chapter ends with a conclusion of this thesis.

6.2 Reasons for differences in LCAs for disposable beverage cups

There is a raising interest in meta-studies reviewing several comparative LCAs for the same product to clarify and understand the effect of different technologies, products and materials on LCA results (Brandão et al., 2012). Frequently observed reasons for discrepancies among LCA results for the same product include different technologies, products and materials, but also efficiencies, amount and type of energy use, and geographical coverage (Michaud et al., 2010; von Falkenstein et al., 2010; Weiss et al., 2012; Wenzel and Villanueva, 2006). Chapter 2 comes to similar insights.

Chapter 2 presents a review of ten existing (comparative) LCA studies for disposable beverage cups. The review quantitatively and qualitatively compared the ten LCAs to each other to identify reasons for differences in LCA results, and to assess the robustness of these LCA results. The quantitative review reveals considerable variation in the LCA results for cups made from the same material, and especially the paper cups show a great variation in results. No cup material can be consistently identified as the most environmentally friendly one. This inconsistency is also observed in (comparative) studies of other packaging products, e.g. carrier bags (Khoo et al., 2010), or beverage packaging (von Falkenstein et al., 2010). Although this was not considered in Chapter 2, even the choice of the selected LCA software package can influence the results and ranking of beverage packaging materials (Speck et al., 2015).

The transparency in the ten reviewed LCAs facilitated the analysis of the underlying characteristics of these studies. This transparency is one of the major achievements of the harmonisation and standardisation of LCA. The LCA harmonisation and standardisation process was triggered in the mid-nineties by discrepancies in conclusions across studies on the same product. These discrepancies were difficult to trace back to methodology and data due to little transparency in the reporting. ISO 14040 and 14044 (ISO, 2006a, b) provide standardisation and harmonisation in the LCA procedure, and to some level as well for selected methodological parts and reporting of outcomes. The goal and scope definition, one of the four well-defined phases in LCA, requires clear specification of methodological choices and data requirement for all other phases, and includes among others a clear description of the studied product system, the functional unit, system boundaries, allocation procedures, impact categories, data requirements, assumptions, and limitations (ISO, 2006a). The description of these aspects makes it possible to compare the underlying methods, choices, assumptions, and results of different studies.

Several of the disposable cup LCAs contain scenarios for among others the weight and material of the cups, waste treatment options and conditions, production processes, and transport distances. The availability of multiple scenarios for a specific cup *within* individual studies to some extent facilitated the examination of the influence of a particular parameter, assumption, or modelling choice on the LCA results. This allowed identification of possible reasons for differences in the LCA results for disposable cups. The identified reasons include the properties of the cups, production processes, waste treatment options, and allocation options. The comparison of LCA characteristics and methodological aspects *among* the ten studies provided additional reasons for differences in LCA results, such as choices in system boundaries, additional allocation options, impact indicators, and potentially also data sets. These reasons were only observable by comparing studies, i.e. not from scenarios *within* individual studies. Other reviews of LCAs for paper, biomaterials, and beverage cups also identified reasons which were not visible from or addressed in single LCA studies (Michaud et al., 2010; von Falkenstein et al., 2010; Weiss et al., 2012; Wenzel and Villanueva, 2006).

Modelling results are considered robust if they are insensitive to most known uncertainties (IPCC, 2001). The identification of uncertainty locations and types of uncertainty is thus a crucial part of the robustness process. The challenge is to recognise these locations and types, but the difficulty is how to recognise them. Tools exist to identify the uncertainty locations and types, for example by means of an uncertainty matrix (van der Sluijs et al., 2003; Walker et al., 2003). Personal and normative believes influence the identification of uncertainties (van Asselt and Petersen, 2003). The ten reviewed studies in Chapter 2 identified and performed uncertainty analysis on obvious or easy recognisable locations of uncertainty in the modelling process. These locations related mostly to the properties of the included cups or the covered waste treatment options of the disposed cups. The weight of the cup, for instance, is an obvious uncertainty and has a proportional influence on the LCA results. The ten reviewed studies also identified the production of the cup material and the manufacturing of the cup as important contributors to the results. Almost none, however, performed uncertainty analysis on these influential processes. Restricting uncertainty analysis only to obvious locations diminishes the opportunity to include additional uncertainties and thus limits the possibility to increase the robustness of the results.

Data uncertainty is the most widely addressed uncertainty in LCA, although it is not exactly clear why. Possible reasons are a general consideration that it is the most important type, or plainly because data and methods are available to perform easily the uncertainty assessment (Lloyd and Ries, 2007). The main examined uncertainty locations in the ten reviewed LCAs include the weight of the cups (inventory data), assumptions in waste treatments (inventory data), and covered waste treatment options (model of the life cycle of the cups). The simultaneous handling of several uncertainty locations and types is not addressed in these studies, and is only sporadically performed in LCAs. The review of the ten LCAs identified the use of different data sets as a possible uncertainty location. The variability among data sets is also only sporadically performed in LCA. Choices are usually addressed by means of sensitivity analysis or scenarios, and independently from data uncertainty. The next sections discuss a method to include these locations and types of uncertainty in LCAs.

6.3 Influence of multiple data sets on the robustness of LCA results

Chapter 3 presents a new developed method to incorporate amongst others multiple data sets from different data sources representing the same or a similar process. Applying the same model with multiple data sets has lately been used in integrated assessment (Uusitalo et al., 2015) and in uncertainty management in integrated assessment, but is new in LCA.

An important step in the developed method in Chapter 3 is to first localise the main uncertainty locations before collecting multiple data sets. The identification of important uncertainty locations is in Chapters 3 and 4 done by means of a contribution analysis of an LCA with default data sets, and subsequent sensitivity analyses of processes and parameters. Although not used in Chapters 3 and 4, uncertainty locations and types can additionally be identified by stakeholders and experts by means of an uncertainty matrix (van der Sluijs et al., 2003; Walker et al., 2003). In this thesis, only those processes with influential contributions to the LCA results are selected for multiple data sets search. These data sets are collected from various data sources, e.g. from organisations, institutions, companies, databases, articles, and reports. Next, the collected data sets are used to calculate the average impact results and the spread in these results for each of the selected processes. Hence, the variability in data retrieved from different sources is projected. The results of the separate processes are next combined into average results for the total LCA, and the spread in the results for the total LCA. The data variability of each separate uncertainty location is thus combined into a variability for the total LCA results. The inclusion of the variability in the results increases the robustness of the results.

The use of the multiple data sets method presents the LCA result for each impact category as an average value (single absolute value) and as a spread in the result (value range). The comparison of single value LCA results are easier to interpret, but the comparison of these single values can lead to conflicting outcomes across studies, as shown in Chapter 2. Although the use of multiple data sets increases the complexity of interpreting results, the results provide insight on *where* uncertainties in results stem from, and the *relative importance* of the uncertainty in the overall uncertainty and overall outcome. The results of the disposable cups furthermore consistently identify the same processes as the influential contributors to the LCA results (Chapters 3 and 4).

The use of multiple data sets is suitable when the use of generic data is more appropriate compared to the use of specific data, e.g. for an LCA about producing a product in general. Representing a general process by only one specific data set ignores any variability which exists among manufacturers, production sites, production processes, or geographical and temporal differences. Variability among data from different data sets is not limited to technical aspects only, but may also relate to the included number of inventory data entries, the allocation of the inventory data to a process, and the translation of measured data into the database (Jiménez-González and Overcash, 2000; Peereboom et al., 1998). Variability is a fact of life (van Asselt and Rotmans, 2002; van der Sluijs et al., 2003; Walker et al., 2003). The use of multiple data sets, however, tries to eliminate any possible bias for the use of a specific data set when generic data is more appropriate. The use of multiple data sets for general processes includes uncertainty caused by variability among data sets, and therefore increases the robustness of LCA results.

The use of specific data or a specific data set for a process is more suitable if the exact process specifications are known. Cup producers, for instance, can use data from their own cup manufacturing site(s) if they want to perform an LCA for one of their own products. Uncertainty within these specific inventory data can be addressed by means of quantitative methods such as scenarios, stochastic modelling, Monte Carlo analysis, Latin Hypercube sampling, uncertainty propagation, Taylor series, Bayesian analysis, or Fuzzy set theory (Coulon et al., 1997; Finnveden et al., 2009; Heijungs and Huijbregts, 2004; Huijbregts, 1998b; Lloyd and Ries, 2007; Zamagni et al., 2008).

Available data for a process can be based on a single observation, an average of multiple observations, or a mixture of several ones. The ecoinvent database, for example, offers average PS data based on data from several production facilities, PLA data from one producer, and solid bleach board data as a mixture of data from one other database and data from one producer (Ecoinvent Centre, 2010). Uncertainties in data entries in the ecoinvent database are often unknown, since there is regularly only one data source with only mean values (Frischknecht et al., 2005). The uncertainties for the data entries are then quantified using a pedigree matrix and expert based uncertainty factors (Frischknecht et al., 2003; Weidema et al., 2013). The use of the multiple data sets method, on the other hand, reflects the *actual* variability among data sets. The variability among data sets is usually not taken into account in LCAs, although the influence of this variability can be considerable (this thesis), and even higher compared to the effect of data variability within data sets (Steinmann et al., 2014).

Individual data entries in data sets can be related to each other. There exists, for instance, a relationship between the used amount and type of energy, and the emissions such as carbon dioxide. When multiple data sets for the same process are available, then the data in these sets are often combined to form a data set with average values. The data set with average values is next used to calculate the environmental impact of the process. Existing correlations among data entries in the individual data sets can get lost when data from several data sets are averaged. The multiple data sets method first calculates the environmental impacts for each individual data set, and then calculates the average of these impact results, and thereby conserves the correlations which exist within the data sets.

The increase in the robustness of the results is at the expense of an increased demand for available resources (time, money, people, expertise), and an increased complexity of the results. A discussion on the trade-offs among the robustness, the required resources, and the complexity is included in the uncertainty management section (section 6.7).

6.4 Influence of waste treatment options on the robustness of LCA results

The complete cradle-to-grave life cycle of a product includes the waste treatment of the disposed product. LCA practitioners and stakeholder make choices in the representation of the life cycle by including selected waste treatment options. This choice depends on the intended goal of the LCA and the interest of the stakeholders (e.g. actual waste management practice or alternative waste treatment options). The choice for the inclusion of a given waste treatment can be influenced by a certain type of interest, customs, regulations, preferences, assumptions, but also on ethics, biases, or ignorance.

The goal of the study thus determines the considered waste treatment options. If the goal intentionally specifies or limits the included waste treatment options, then the addition of extra options has no added value and can even cause confusion, if not error. This is different if the waste management practice is unknown, i.e. if there is limited or no available knowledge on waste fractions entering divers waste treatment options. In this case, presenting a number of waste scenarios with only one waste treatment per scenario, as used in Chapters 3 and 4, provides an opportunity to explore the uncertainty in waste management practice from different mixtures of waste treatment. Inclusion of several waste treatment options can also be used for explorative purposes, e.g. reflecting different possible waste management practices or upcoming waste treatments, and thereby provide valuable information for decision makers. Including multiple waste treatments options can, therefore, *in certain cases* increase the robustness of LCA results. Once again, the goal or intention of the LCA determines the usefulness of different scenarios.

The actual waste management practice of disposed products involves one or more waste treatments. The contribution of the waste phase to the LCA results is often presented as one comprehensive impact. This one value disguises the individual contribution of the different options if the waste management practice consists of several waste treatments. Concealing the results of the individual options lowers the transparency of the study, and hinders the comparison of LCAs for the same product. This thesis evaluates purposely separate waste treatment options as to examine the influence of the individual options on the LCA results, and to facilitate the comparison of these options (Chapters 3, 4, and 5).

The environmental impacts of various waste treatments for disposable packaging materials are divers (Arena et al., 2004; Boldrin et al., 2009; Merrild et al., 2008, 2009; Michaud et al., 2010; Wenzel and Villanueva, 2006). The waste treatment is an influential part of the LCA results for disposable beverage cups and aluminium cans, and consequently these LCA results depend on the included waste treatment options (Häkkinen and Vares, 2010; Pasqualino et al., 2011; Pladerer et al., 2008; Vercalsteren et al., 2006), and as confirmed by Chapters 2, 3, 4, and 5. The choice of the included waste treatment options thus plays an important role in the LCA results for disposable cups and aluminium cans.

The identification of the preferred waste treatment for paper and plastic products is not unanimous across studies (Laurent et al., 2014; Michaud et al., 2010; Villanueva et al., 2004; Wenzel and Villanueva, 2006). The environmental impacts of waste treatments for packaging materials such as paper, plastics and metals depend on key assumptions in those treatments (Arena et al., 2004; Astrup et al., 2009; Damgaard et al., 2009; Ekvall, 1999; Fruergaard et al., 2009; Merrild et al., 2008; Villanueva et al., 2004). The variability in each waste treatment process can be addressed by means of the multiple data sets approach, as described in Chapter 3. The use of multiple data sets only makes sense if the waste treatment displays an influential contribution in the total LCA results, as is the case for disposable cups and aluminium cans (Chapters 2, 3, 4, and 5).

Overlap in waste treatment results for the disposable cups prevents identification of a preferred option for each individual material. The results, however, give a clear message that composting is the least desirable option for the PLA and biopaper cups (Chapter 4). The sellers of PLA and biopaper cups praise the compostability of the cups, and assume that this property makes the cups more sustainable. This suggestion is, however, not supported by the results in Chapter 4.

The multiple data sets for recycling in Chapters 3 and 4 are based on the variation in quantifying credits for recycled material, i.e. based on economic values or replacement of other materials. There is still a scientific debate on the modelling of recycling in LCA, hence a number of recycling modelling methods exist, as discussed in the next section.

The simultaneous use of multiple models to include choices in the waste treatment options, and the use of multiple data sets to include the variability in these processes, clearly provides more robust results compared to the results of the separate waste scenarios for the disposable cups (Chapters 3 and 4). The simultaneous handling of several uncertainties is already frequently applied in integrated assessment studies, and shows there to provide more robust results compared to the results of separate models (Delle Monache and Stull, 2003; van Loon et al., 2007). The simultaneous handling of multiple uncertainties is only sporadically performed in LCA (Huijbregts et al., 2003; van Zelm and Huijbregts, 2013). The use of multiple data sets and modelling choices is a transparent method to address simultaneously the variability among data sets and the choices in modelling options in LCA.

6.5 Influence of recycling modelling options on the robustness of LCA results

Allocation in recycling is a recurrent and highly debated topic in the LCA community (Reap et al., 2008; Russell et al., 2005). Different methods exist to include recycling in LCA. Chapter 5 evaluates six recycling modelling methods: three substitution methods (based on substitution with equal quality material, a correction factor, and alternative material), an allocation method based on the number of recycling loops, the recycled-content method, and the equal-share method. These methods are based on different underlying philosophies on sustainability concepts and waste management perspectives. The substitution methods promote the recyclability of products, while the recycled-content method stimulates the use of recycled material in products. The equal-share method is a combination of both viewpoints. The main differences in the methods is the focus on *where* and *how* to assign burdens and credits for recycled material in the life cycle of a product. All six methods are applied in LCAs, and all six methods show validity in their approach. The choice of the method has, different from the waste treatment choice, consequences for methodological issues such as system boundary, included processes, and allocation principles.

Recycling of metals is a common practise, and the quality of the recycled metal often equals the quality for virgin metal (Atherton, 2007). Paper recycling is also a common practise, but the recycling process reduces the quality of the paper fibres (Merrild et al., 2008). Recycling of plastics is increasing, but the quality of recycled material compared to virgin plastic is debatable (Al-Salem et al., 2009; Lazarevic et al., 2010; Vilaplana et al., 2006). Differences in the provided credits for recycled plastics clearly affect the LCA results (Chapters 3, 4, and 5). The inclusion on multiple ways of crediting for recycled plastics creates a spread in the recycling results, but this inclusion increases the robustness of these results.

The choice of the modelling method clearly influences the LCA results of the aluminium can and the PS cup (Chapter 5). These choices give some room to LCA practitioners or stakeholders to steer results into a favourable direction. Conflicting viewpoints on sustainability cannot be eliminated, however, and divers recycling modelling methods will continue to exist. The choice of the applied method is decided by LCA practitioners, preferable with consent of the stakeholder. Including several recycling modelling methods in the LCA incorporates these various standpoints, makes them transparent, and thus makes the results more robust. ISO 14044 (ISO, 2006b) proposes a sensitivity analysis if several recycling modelling methods seem applicable. This thesis goes one step beyond the ISO suggestion, and purposely includes a multiplicity of methods for modelling recycling in LCA.

This thesis covers six methods for the modelling of recycling in LCA, but even more methods exist. Several industrial branches declared their preferences on a specific modelling method or suggested useful methods (e.g. the metal industry (Atherton, 2007; EAA, 2013c), and paper branch (Cederstrand et al., 2014)). Modelling recycling of a product which contains materials from different branches, each with their own preferred modelling method, would consequently face the problem of incorporating different and possibly opposing methods. The applied allocation principles should be defined in the scope definition of the LCA study. Incorporating multiple recycling modelling methods for all the different materials of a product requires additional time and effort. Here arises, again, a trade-off between the added value and the effort it takes to perform the study/calculations. The contribution analysis of the LCA results can indicate whether the inclusion of additional recycling methods may make sense in such a situation.

Different recycling modelling methods can provide conflicting LCA results (Chapter 5). One way to eliminate the uncertainty from these methods is to establish a standardised method, i.e. a single method to account for recycling. This would, however, restrain scientific freedom in the use of existing or introduction of new sustainability viewpoints and related recycling principles. Standardisation can moreover only be achieved if there is a consensus by many parties. Issues with new or conflicting viewpoints are, therefore, hard to standardise. This is visible in e.g. the introduction of new impact categories. New methodological insights, the call for additional impact categories, and continuous new developments in impact assessment keep emerging and should not be restricted by standardised procedures. As for the recycling modelling methods, the recycled-content method, which provides credits for the use of recycled material in products, does in theory not comply with ISO 14044. The method is, nevertheless, increasingly used in LCAs, especially for products made from recycled material.

6.6 Added value of the overall approach

The multiple data sets method presented in Chapter 3 uses contribution and sensitivity analysis to identify potential uncertainty locations. This is a fairly objective procedure based on environmental impact results. Uncertainty locations can also be identified by the LCA practitioner and stakeholders. Stakeholders usually only work in a specific part of the life cycle of a product, and often possess less knowledge on uncertainties in the other life cycle stages, or do not recognise these uncertainties. The personal and normative believes of stakeholders furthermore influence identification of uncertainties, hence these uncertainties have a subjective element (van Asselt and Petersen, 2003). Uncertainty analysis is then often limited to prejudices or the obvious and easily recognisable locations. Including only trivial uncertainties while ignoring others may lead to misleading results for the decision makers, and can lead to unintended and undesirable management decisions. Robust results should hold under a variety of approaches, methods, and assumptions (IPCC, 2001). The identification of the most influential uncertainty locations is thus a desirable step, and is included in the multiple data sets method as described in Chapter 3.

Tools to handle uncertainties are abundant, and include statistical modelling, sensitivity analysis and scenario analysis (Heijungs and Huijbregts, 2004; Huijbregts, 1998a; van der Sluijs et al., 2004). Data uncertainty is the most addressed uncertainty in LCA (Finnveden et al., 2009; Heijungs and Huijbregts, 2004; Lloyd and Ries, 2007; Ross et al., 2002). Statistical methods often focus on variability *within* data sets, e.g. Monte Carlo analysis. The multiple data sets method (Chapter 3) tackles the variability *among* data sets from different databases. This is basically different from variability *within* data sets itself. The variability among data sets leads to a greater accuracy of the results (i.e. how close the results are to the actual values or targets), whereas the variability within data sets leads to a higher precision of the results (i.e. how close the results are to each other). Precise results do not imply to be accurate, and accurate results do not indicate to be precise. The use of multiple data sets increases the accuracy of the results. The statistical methods and the multiple data sets method serve different purposes, and can supplement each other.

Calculation of the uncertainty associated with variability within a data set requires stochastic information of inventory data entries, e.g. distributions and confidence intervals, to calculate the impact results and the uncertainty in these results (Heijungs and Huijbregts, 2004). Correlations among input variables can be incorporated, but these correlations are usually poorly known, and in practice hardly included (Lloyd and Ries, 2007). Ignoring correlations may lead to over- or underestimation of uncertainty in the outcome (Refsgaard et al., 2007). The multiple data sets approach first calculates the environmental impact for each individual data set, and next calculates the average and spread of the process based on the impact results of the individual data sets. Correlations in the data sets are preserved by calculating the spread *after* impact assessment has been carried out, as is done in the multiple data sets method. Calculating the variability on inventory level and after impact assessment both aim to display the variability in the inventory data. The two different approaches serve, again, different purposes.

The calculation of the spread in results on impact assessment level, as is done in the multiple data sets method, allows for different formats for inventory data sets. Inventory data can be presented as aggregated inventory data of resources and emissions, bill of materials (a list of materials and the quantities needed to manufacture a product), or even impact assessment data. This lowers the requirement on the inventory data format. Companies often have their own specific accounting system, or are reluctant to release their confidential information if the exact company data be can traced back from this information. The free choice in inventory data format enables the inclusion of these data.

The spread in results for a particular process, due to the use of multiple data sets, reflects the variability in that process. The spread in the results are ideally for benchmarking purposes in the industry, and can identify manufacturers with the best performance (i.e. the lowest environmental impact) in their field. The results can encourage competition among manufacturers and stimulate improvements in the production processes. The mature PS production process, for instance, shows a much lower spread compared to the fairly new PLA production process (Chapter 4). The production processes of the best performers can be designated as “best practises”. Benchmarking can thus be the basis to set up standards for products and processes, and governmental regulations where branches should live up to. The use of multiple waste treatments can point to undesirable waste practices. Specific waste treatments might be forbidden, for example landfilling of combustible material is banned in the Netherlands (Jorritsma-Lebbink, 1997).

The use of multiple data sets is actually the running of one model with multiple variations of data values (i.e. single-model ensemble) (Parker, 2013; Uusitalo et al., 2015). The inclusion of choices in the model is performed by running multiple models with the same data (i.e. multi-model ensemble) (Parker, 2013; Uusitalo et al., 2015). The combination of both types of ensemble modelling is in integrated assessment used as a form of uncertainty assessment and has already proven its usefulness in weather forecasting, climate change predictions, and air quality forecasts (Delle Monache and Stull, 2003; IPCC, 2000; Parker, 2013; van Loon et al., 2007). Results from ensemble modelling are more robust compared to the results of single models (Delle Monache and Stull, 2003; van Loon et al., 2007). The simultaneous handling of variability and modelling choices, which is currently hardly performed in LCA, also seems to lead to more robust results in LCA, as illustrated in Chapters 3 and 4. The method provided in Chapter 3 fills this gap and provides a transparent tool to capture these uncertainties.

The multiple data sets method is in this thesis used for inventory data sets. Finding available data is challenging due to confidentiality of company data. For the case studies in Chapters 3 and 4, every data set was given the same weight in the calculation of averages and spreads. Data obtained from small companies represent only a limited fraction of the complete market. Data retrieved from (commercial) databases can represent a specific company, but also a major part of the companies in that sector. The importance of the market share of the data sets used in this thesis was unknown. The addition of a weight factor showing the market share of the included (company) data could balance the representation of the data. This thesis presents the spread in results as the lowest and highest values (Chapters 3 and 4). Additional statistical procedure can furthermore be used to calculate standard deviations, confidence intervals, or distributions of the results. The approach of Chapter 3 is now used for inventory data, but is also applicable in other LCA phases, such as impact assessment (characterisation factors).

The multiple data sets and multiple modelling options method was in this thesis only applied for disposable beverage cups and aluminium cans, and these case studies are not necessarily representative for other products and materials. Packaging products normally have a short life span, and endure limited impact from the use phase. The impact results for packaging products are basically from the production of the packaging material, the manufacturing of the product, and the disposal of the product. Transport vehicles and electronic equipment, for instance, have a longer lifespan and the use and maintenance phases can be considerable contributors to the environmental performance (Del Pero et al., 2015; Hischier and Baudin, 2010; Spielmann and Althaus, 2007). Nevertheless, the method can be applied to any product.

Summarizing, the added value of the approach taken in this thesis is the identification of possible uncertainty locations, the increase of the accuracy and robustness of results, thereby preserving the correlations within data sets and lowering the requirement on the inventory data format, and providing a tool to handle multiple uncertainties in LCA.

6.7 Uncertainty management

LCA is widely used to support decision making in business and policy making (Lazarevic, 2015; Lloyd and Ries, 2007; Tillman, 2000; UNEP, 2003). The desired level of robustness of the LCA results is defined by the stakeholders and the goal of the study. Simplified LCAs might be well-suited for e.g. internal use in companies. Full-blown LCAs are often demanded for comparative studies which are disclosed to the public. In both cases, inclusion of uncertainty in LCA results can provide essential information for decision makers, hence uncertainty should be incorporated in these LCA results (Ciroth et al., 2004; Finnveden et al., 2009; Geisler et al., 2005; Heijungs and Huijbregts, 2004; Huijbregts, 1998a; Notten and

Petrie, 2003; Ross et al., 2013). The inclusion of variability among data sets and modelling choices can thus provide vital information, and is an asset in the decision-making process. The use of multiple data sets and multiple modelling options increases the robustness of the results, but at the expense of an increased demand for available resources (time, money, people, expertise), and an increased complexity of the results (Chapters 3, 4 and 5). Complex results may need to be translated into more manageable information before delivered to decision makers. Simplifying LCAs could decrease the complexity, for instance through focussing on only one impact category. Huijbregts et al. (2006) examined the use of the cumulated energy demand as indicator for the environmental performance of products, and concluded that it can be used as a screening indicator. Uncertainties, however, are high for waste treatments and non-fossil energy use. Results of this thesis show that one should be careful with focusing on a single impact indicator only, because this may not reveal the complete environmental picture of the products, as shown in the comparison among PS and biobased materials (Chapter 4). ISO 14044 (ISO, 2006b) furthermore requires the assessment for a set of impact indicators when comparative LCA results are disclosed to the public.

There is a call from LCA practitioners and companies for simplified assessment methods. The Product Carbon Footprint (PCF), for instance, focusses on climate change results only, while full LCAs strive for a broad environmental assessment of different impact categories (Feifel et al., 2010). Streamlined LCAs limit or omit upstream and/or downstream stages, impact categories, and/or inventory data (Todd and Curran, 1999). Streamlined LCAs require less resources, but are usually more geared towards internal use in companies, and less for external use (Todd and Curran, 1999). The trade-off between simplified and detailed LCAs depends on the goal of the study, and moreover on the intended use of the results.

Several types of uncertainty can be recognised, and these can occur in numerous locations in the LCA (see Table 6.1). This thesis specifically addresses choices made in the representation of the waste treatments in life cycle of the product, choices in modelling methods for recycling, and variability in available databases. This thesis does not address the uncertainty in the modelling of the effect of substances on the environment, although the uncertainty in characterisation factors can be very high (Huijbregts, 1998a). This thesis furthermore does not investigate data uncertainty *within* the data sets. Tools to handle uncertainties within data sets comprise a number of statistical modelling techniques, sensitivity analysis and scenario analysis (Coulon et al., 1997; Finnveden et al., 2009; Heijungs and Huijbregts, 2004; Huijbregts, 1998b; Lloyd and Ries, 2007; Zamagni et al., 2008). The method described in

Chapter 3 is an additional tool to handle uncertainty, specifically for the variability among data sets, and the simultaneous handling of this variability and modelling choices.

Uncertainty assessment should aim to bring the LCA results closer to reality, reduce uncertainty as much as possible, and minimize the extent to which uncertainties affect conclusions (Maxim and van der Sluijs, 2011). Uncertainty in data and parameters may be reduced by means of additional measurements, more precise measurements, or collection of better or missing data (Björklund, 2002; Lo et al., 2005; Refsgaard et al., 2007). Standardisation and harmonisation on measurement procedures of emissions and inventory data, data quality requirements, or life cycle impact assessment models can also reduce uncertainties (e.g. Hauschild et al. (2008)).

Acquiring additional knowledge on the product or product system can decrease uncertainty, as well as discussing potential uncertainties with stakeholders (Finnveden et al., 2009). Choices are, however, unavoidable in LCA. Choices are subjective decisions based on the interest of LCA practitioners or decision makers, the available knowledge of the subject, etc., and are normative decisions. Standardisation may reduce the uncertainty caused by choices (Björklund, 2002; Huijbregts, 1998a). The standardisation and harmonisation of LCA in ISO standards led to more uniform LCA procedures, framework and terminology, and indicates where methodological choices need to be made. Some of the applied methodological choices depend on the intended goal of the LCA, for instance an attributional or consequential LCA, a screening or detailed LCA, partial of full LCA, or from a product or actors standpoint (Baumann and Tillman, 2004; Brandão et al., 2014; Löfgren et al., 2011; Tillman, 2000). The selection of included waste treatments for disposed products (Chapters 2, 3, 4 and 5) also cannot be standardised, as the selection is based on the goal of the LCA. Specific waste treatments, on the other hand, can be mandatory or forbidden in particular regions. Restricting the handling of recycling in LCA through standardised methods disregards the different perceptions of recycled material, and restrains new methodical insights and sustainability perspectives. Hence, standardisation can be, but is often not, a desirable solution for uncertainty caused by choices.

Variability cannot be reduced by further or additional measurements (Björklund, 2002). Variability is a reflection of reality (van Asselt and Rotmans, 2002; van der Sluijs et al., 2003; Walker et al., 2003). Differences in among others equipment, applied technologies, efficiencies, materials, and geographical coverage all contribute to variability in similar processes (Chapters 3 and 4). Variability among different data sets for similar processes can also be due to the included number of inventory data entries and the translation of measured data into the database (Jiménez-González and Overcash, 2000; Peereboom et al., 1998). This latter variability can be reduced by means of standardisation or harmonisation of data requirements. The multiple data sets method, on the other hand, requires no standardised format for inventory data, and simplifies the inventory collection. This does not mean that these inventory data have a lower data quality requirement. The non-standard format lowers the effort for inventory acquirement and aims to increase the willingness of companies to provide data.

Uncertainty is a fact of life and is present in all scientific fields. A better understanding of uncertainties and their incorporation in LCA outcomes can create more trustworthy information. The robustness of LCAs cannot be assessed without an uncertainty analysis. The use and willingness for inclusion of uncertainties in research depends more on the nature and personal preferences of the scientists, than on the scientific discipline of the work field (van Asselt and Petersen, 2003).

6.8 Conclusion

LCA results need to be robust if they are used in decision making. In practise, however, LCAs for the same product can provide different and even conflicting outcomes. This thesis evaluates such inconsistent LCA results for disposable beverage cups. Reasons for these differences include variation in the properties of the cups, production processes, waste treatment options, allocation options, choices in system boundaries, impact indicators, and potentially also the used data sets. These reasons could be identified relatively easily as a result of the transparency of the LCAs, a beneficial effect of the standardisation of LCA procedures.

This thesis focuses on two potential sources of uncertainties in LCA results: variability among data sets, and choices in modelling options. In existing LCA studies, different types of uncertainties are typically addressed one by one. No integrated approach exists in LCA to incorporate multiple uncertainties simultaneously. LCA results can be considered robust if they are insensitive to such combinations of uncertainties. This thesis evaluated whether the combined use of multiple data sets and multiple modelling options can increase the robustness of LCA results.

The results of this thesis indicate that including multiple datasets can make the results of an LCA more robust. This thesis presents a method to perform an LCA using multiple data sets for processes in the life cycle of a product, and simultaneously include multiple modelling options. The variability among data sets is only sporadically addressed in the existing literature. Conventional data uncertainty is geared towards variability or uncertainty within a data set. The use of multiple data sets, as in this thesis, reduces possible bias from the use of a specific single data set. The method furthermore preserves the correlations among individual data entries within a data set. The results of the disposable cups consistently point to the same processes as the influential contributors to the LCA results. Using multiple data sets creates a spread in the results. The LCA results might be harder to interpret, but the results can also be considered more robust compared to the use of single data sets.

The results of this thesis show that including multiple waste treatments options can in certain cases increase the robustness of LCA results. Various waste treatment options exist for disposed products. The inclusion of multiple waste treatment options has no added value and can even cause error if the goal of the LCA is intentionally limited to only specific waste management options. Including multiple waste treatment options in an LCA makes sense if the waste management practice is unknown, or the goal of the LCA is to explore possible options. The choice for one of these waste treatment options can affect the LCA results, as this thesis shows for disposable cups and aluminium cans. The goal or intention of the LCA thus determines the usefulness of including multiple waste treatment options.

The results of this thesis indicate that the combined inclusion of the choices in waste treatment options and the variability among data sets for the waste treatments increases the robustness of the LCA results for the disposable cups. The additional use of multiple data sets and allocation principles for the waste treatment options reduces potential bias in the use of a specific data set or a specific assumption. The spread in the results for different waste treatment options for the disposable cups overlaps, and provides no clear preference for one of the waste treatments. Composting, however, is clearly identified as the least preferred option for compostable cups.

Integration of various recycling modelling methods into the LCA results captures the different viewpoints on recycling, and thus increases the robustness of the LCA results. Recycling is a multifunctional process; it is a waste treatment for disposed products and a production process for material which can be used in a next product. The distribution of environmental impacts among the two functions (allocation) is a recurring and unsolved topic in LCA. Various methods exist to model recycling in LCA. Again, different methods can lead to different LCA results, as this thesis shows for a PS cup and an aluminium can. The modelling methods that were analysed here represent various viewpoints on sustainability and waste management, and differ in the approach on where and how to assign credits for recycled material. The choice of using a specific method is decided by the LCA practitioner and stakeholders and depends on the goal of the study, and the uncertainty associated with this choice cannot simply be reduced. Incorporation of various recycling modelling methods into the LCA results makes the related uncertainty transparent by the spread in outcomes.

This thesis demonstrates the added value of the use of multiple data sets and modelling options in LCA. The use of multiple data sets is especially useful when general data for processes are used. The combined approach of the use of multiple data sets and modelling options tackles both the variability among data sets and uncertainty due to the choice. The simultaneous handling of several uncertainties is fairly new in LCA. The use of multiple data sets method and the combined approach are, therefore, additional tools for uncertainty management. The trade-off between an increase in the robustness of the results and the additional required resources (time, money, effort) should be assessed. The increased complexity and spread in the results can reduce clear-cut conclusions for decision support. However, my research shows that inclusion of the uncertainty in the results provides the decision maker with valuable information. This thesis thus provides a useful method to increase the robustness of LCA results.

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Summary

Life cycle assessment (LCA) is a well-established method to evaluate the potential environmental impacts of product and service systems throughout their life cycles. An LCA consists of four well-defined phases: 1) Goal and scope definition, 2) Inventory analysis, 3) Impact assessment, and 4) Interpretation. LCA standards exist that provide guidelines on the LCA procedures, but there is still room for methodological choices. In practice, it can happen that LCAs for the same product have different and even conflicting outcomes. This is not beneficial if LCA results are used in the decisions-making process. LCA results need to be robust and trustworthy if they are used in decision making.

Results are considered robust if they are insensitive to most known uncertainties. Uncertainties can occur at different locations in all four LCA phases. Locations may refer to the definition of the life cycle of the product, the followed philosophy in the underlying methods, the selection and modelling of environmental impacts, or in the data used. In this thesis, I make a distinction in three types of uncertainties that may be relevant for each location: variability, choices, and unreliability. Variability refers to observable variation as a result of natural randomness or heterogeneity, e.g. data from several production facilities producing the same or similar materials or products. Choices refer to the normative choices which are taken by the stakeholders and/or LCA practitioners. I bundled all other types of uncertainties under the type unreliability, e.g. inaccurate, inexact, and unrepresentative depiction of data or a model. The variability *within* data sets is often addressed in LCA studies. The variability *among* different data sets representing the same product or process is relatively new in LCA. The simultaneous inclusion of this variability among data sets and the inclusion of various modelling options is sporadically performed in LCA.

The aim of my thesis is: *to evaluate whether the use of multiple data sets and multiple modelling options can increase the robustness of LCA results.* To achieve this aim I formulated three research questions: 1) what are reasons for differences in LCA results for the same product, 2) can the use of multiple data sets for a process increase the robustness of LCA results, and 3) can the inclusion of multiple modelling options increase the robustness of LCA results. For the third question I evaluated modelling options for waste treatment, and modelling options for recycling. I used case studies for disposable beverage cups and aluminium cans to support my research.

In Chapter 2, I reviewed ten existing LCA studies for disposable beverage cups to find reasons for possible differences in LCA results. The ten (comparative) LCA studies include beverage cups made from petro-plastics (i.e. form oil and natural gas), biobased-plastics (i.e. from renewable resources), and paperboard. The results from these studies were compared qualitatively and quantitatively. Climate change is the only common impact category across these studies, and hence the comparison was restricted to the global warming potential (GWP) impact indicator. I compared the absolute GWPs for the cups after a correction for the cup volume was done. The quantitative comparison shows no consistent best or worst cup material. The GWPs within each cup material group varied. The ratio between the highest and lowest GWP value is 3.4 for the petro-plastic cups, 1.7 for the biobased-plastic cups, and nearly 20 for the paperboard cups. I next evaluated the underlying methodological choices of each individual study to find reasons for these differences in GWPs.

Several of the studies reviewed in Chapter 2 present multiple cup scenarios, where each scenario contains different data, choices, assumptions, or methodological methods. These scenarios made it possible to identify the influence of these choices on the GWP results. I also compared the methodological choices among the ten studies, which revealed reasons not identifiable within the separate studies. Reasons for differences in GWPs include the variation in the properties of the cups, production processes, waste treatment options, allocation options, choices in system boundaries, impact indicators, and potentially also the used data sets. These reasons represent different types of uncertainties. The review of the studies identified the used data as a possible source for differences in the outcomes. The variability among data sets was not addressed in these studies and in LCA studies in general. The ten studies usually only treat one uncertainty at a time. The simultaneous inclusion of multiple uncertainties has proven to provide more robust results in integrated assessment modelling of climate change. This simultaneous inclusion of uncertainties might also provide more robust LCA results, and is addressed in the next chapter.

In Chapter 3, I purposely used different data sets and different modelling choices to evaluate their influence on the LCA results. I tested a method to include these multiple data sets and multiple modelling options in the LCA. The method consists of a number of steps, as will be described next, with an LCA of a disposable polystyrene (PS) beverage cup as a vehicle. I first made an initial LCA of the PS cup with one (default) data set per process and incineration as waste treatment. I made an additional LCA with the same data sets but now with recycling as waste treatment, i.e. recycling is an additional waste treatment modelling option.

Contribution and sensitivity analyses identified the production of PS, the manufacturing of the cup, and the incineration and recycling of the cup as the influential contributors to the LCA results. Next, additional data sets were only collected for these selected influential processes. In the next step, the environmental impact results for each separate data set were calculated. This was followed by the calculation of the average result and the spread in this result for each process, based on the multiple data sets and modelling choices for that process. Calculating the spread after impact assessment preserves any correlation which exists within the individual data set. Finally, the results of the individual processes were combined into average results and their spread for the total LCA results.

The results from the different data sets and modelling options confirm that the production of PS, the manufacturing of the cup, and the waste treatments (incineration and recycling) are the main contributors to the LCA results. The variability in these processes all lead to considerably spread in the results. This spread is caused by differences in the amount and type of the used resources and energy, and in reported emissions. The spread in the PS production furthermore relates to the origin of the PS production location and the time period of data collection. Different choices in the crediting of recycled PS created the spread in the recycling process. The spread in the impact categories related to energy use was lower compared to the spread in the toxicity categories. The overlapping spread in incineration and recycling results prevented a decisive conclusion on the preferred waste treatment for the PS cups. The use of multiple data sets and modelling options increases the insight in the relative contribution of processes to the overall LCA results and the uncertainty in these results. The spread in the results increases the complexity of the results. Although the results might be less easy to perceive, the outcome is more certain and provides decision makers with valuable information. The multiple data sets represent the variability which exists among the same or similar processes, and the waste treatments represent choices in the modelling of the life cycle of the PS cup. The simultaneous inclusion of these uncertainties provides more robust results compared to the individual handling of uncertainties.

The multiple data sets and modelling choices method was in Chapter 3 applied on a disposable PS beverage cup. Chapter 4 tested the usefulness of this method in a comparative LCA of disposable beverage cups. The comparison includes three cups: a PS cup, a polylactic acid cup (PLA, a biobased plastic), and a cup made from biopaper (paper with a lining of biobased-plastic). I included incineration and recycling as waste treatment options for the three cups, and additionally composting and anaerobic digestion for the PLA and biopaper cup. I also included multiple crediting options for the recycled plastics. The influential contributors to the LCA results are the production of the cup materials (PS, PLA, biopaper), the manufacturing of the cups, and the waste treatments. The spread in the results of these processes are, again, related to the amount and type of energy used, allocation principles, and reported emissions. The spread is lower for the mature PS production compared to the still fairly new production of PLA. Variation in PLA production results is mainly caused by differences in the stock material. Toxicity impact categories, again, display a higher spread compared to the energy related impact categories.

Based on the average results, I was not able to point to the most environmentally friendly cup material when I considered eleven impact categories. The spread in the results for the cups endorses this decision. Average results suggest for the PLA cup a preference for recycling over the other waste treatments. The spread in the waste treatment results, again, overlaps for each of the cups, and prevents a clear preference for one of these options. I clearly identified composting, however, as the least preferred option for the PLA and biopaper cups.

The use of multiple data sets represents variability in processes, and in this study represents more appropriately a general cup. The use of a single dataset or a specific modelling option can steer the results of the comparison of the three cups into a favourable direction. The use of multiple data sets and modelling choices has led to a considerable spread in the results of the cups. This made the comparison of the cups more complex, but the overlap in results provides more robust information to decision makers. The combined inclusion of the variability among data sets and the waste treatment options makes the results more trustworthy.

The LCA results for the plastic cups in Chapters 3 and 4 show a spread in the recycling results, caused by various ways of crediting recycled plastics. Modelling of recycling is an unsolved issue in LCA, and leads to recurring debates in the LCA community. Recycling is a multifunctional process. It is a waste treatment for disposed products and a production process for material which can be used in a next product. In Chapter 5, I described and evaluated six widely used recycling modelling methods: three substitution methods (i.e. substitution based on equal quality, a correction factor, and alternative material), allocation-on-number-of -recycling-loops, the recycled-content method, and the equal-share method. The main difference among the methods lies in the assumption on where and how to apply credits for recycled material. The three substitution methods stimulate the recyclability of the product, and give credits to the product system producing recycled material. The allocation-on-number-of-recycling-loops method divides the impact of the production of virgin material throughout the life cycle of the material, rather than to the life cycle of a specific product. The recycled-content method, on the other hand, promotes the use of recycled material in products, and gives credits to the product system using recycled material. The equal-share method is a compromise between the use of recycled material in the product and the production of recycled material due to the end-of-life waste treatment of the product.

I applied these six recycling modelling methods in two case studies (a disposable PS beverage cup and an aluminium beverage can) to evaluate the effect of the methods on LCA results. I assumed a hypothetical 100% recycling rate for the PS cup and the aluminium can, as to see the maximal effect of the different methods. Aluminium retains its quality after it is recycled, and can be seen as a closed-loop product system. Recycling of PS, on the other hand, can deteriorate the quality of PS, and is seen as an open-loop system. The results of both case studies clearly depend on the applied recycling modelling method, the recycling rate of the product, and the amount of recycled material used in the products. The results for the PS cup furthermore depend on the consideration of a quality drop of the recycled PS. This quality drop can be expressed by a correction factor, or by means of an alternative material that can be replaced by the recycled PS. The representation of the quality drop is debatable.

I again applied a substitution method and the recycled-content method in the PS cup and aluminium can studies, but now used the actual waste management practices for the average European situation and for selected European countries. The results for the aluminium can depend in the substitution method on the recycling rate of the can, and the results differ among the countries. The results are equal for all counties, on the other hand, when the recycled-content method is used. The choice of the method thus plays an important role for the aluminium can results. The results for the PS cup are for both methods influenced by the other waste treatments (landfilling and incineration) for the cups. Incineration of PS allows for the recovery of energy, and this recovered energy is credited to the PS cup. Variation among the waste management practices of the countries and the variation in the type of energy used in these countries influence the results for the PS cup.

The results furthermore differ across impact categories. This stresses the importance to consider other impact indicators besides the most commonly used global warming potential.

The different modelling methods for recycling represent various underlying modelling philosophies. The choice of the method clearly influences the LCA results of the aluminium can and the PS cup. The choice of the applied method gives some room to steer the results into a favourable direction. Conflicting viewpoints on sustainability cannot be eliminated, however, and divers recycling modelling methods will continue to exist. Including several recycling modelling methods in the LCA incorporates these various standpoints, and thus makes the results more robust.

This thesis demonstrates the added value of the method to include multiple data sets and multiple modelling choices in LCA. The identification of the most influential uncertainty locations is a crucial step in the procedure. The spread in results is calculated after impact assessment has been carried out, and preserves any correlations in the data sets. Data collection allows for different formats for inventory data sets. The use of multiple data sets is especially useful if general processes instead of specific processes are used in the representation of the product system. The use of multiple data sets increases the accuracy of the results, and is a supplemental tool next to statistical methods which increase the precision of the results. The simultaneous handling of variability among data sets and modelling choices is hardly performed in LCA. The method in Chapter 3 fills this gap and provides a transparent tool to capture these uncertainties. The trade-off between an increase in the robustness of the results and the additional demand for resources (time, money, effort) should be assessed, and depends on the goal of the study and on the intended use of the results. My research shows that inclusion of the uncertainty in the results provides the decision maker with valuable information. This thesis thus provides a useful method to increase the robustness of LCA results.

Samenvatting

De levenscyclusanalyse (LCA) is een veel gebruikte methode om de mogelijke milieubelasting van producten te bepalen gedurende de gehele levenscyclus van de producten. De levenscyclus van een product bestaat uit alle processen vanaf de winning van grondstofwinning, via de productie van materialen, de fabricage van het product, het gebruik van het product tot en met de afvalverwerking.

Een LCA bestaat uit vier goed afgebakende fasen: 1) Vaststellen van doel en reikwijdte, 2) Inventarisatie, 3) Effectbeoordeling, en 4) Interpretatie. In de eerste fase wordt de doelstelling van het onderzoek beschreven en de beoogde toepassing, dus waarom het onderzoek wordt uitgevoerd. Daarnaast wordt ook vastgelegd hoe het onderzoek wordt uitgevoerd. Dit omvat onder andere een overzicht van de verschillende fasen in de levenscyclus van het product (het product systeem) en wat er wel of niet wordt meegenomen (de systeemgrenzen). De tweede fase (inventarisatie) bestaat uit het verzamelen en verwerken van inventarisatie gegevens voor elk proces in de levenscyclus van het product. Deze inventarisatie gegevens (verder aangeduid als data) bevatten enerzijds de benodigde hoeveelheid grondstoffen, materialen en energie om een product te maken en daarnaast milieudata over gewonnen natuurlijke grondstoffen en de hoeveelheid aan emissies en afval die worden geproduceerd. De geïnventariseerde data worden in de derde fase (effectbeoordeling) omgezet in het effect op het milieu in één of meerdere milieueffectcategorieën. Effectcategorieën zijn bijvoorbeeld klimaatverandering, uitputting van grondstoffen, verzuring, vermesting, of toxiciteit. In de laatste fase (interpretatie) worden de resultaten van de inventarisatie en de effectbeoordeling geëvalueerd en conclusies en aanbevelingen geformuleerd. In deze laatste fase wordt ook de robuustheid van de resultaten onderzocht door middel van gevoeligheids- en onzekerheidsanalyses.

De LCA procedure, dus welke stappen moeten worden uitgevoerd, is beschreven in standaards. Handleidingen beschrijven daarnaast de LCA methodologie, dus hoe die verschillende stappen moeten worden uitgevoerd. Ondanks deze standaards en handleidingen is er nog altijd ruimte voor het maken van keuzen hoe deze stappen exact uit te voeren (de methodologische keuzen). In de praktijk komt het voor dat LCAs van hetzelfde product tot verschillende en zelfs tegenstrijdige resultaten leiden. Dit is niet bevorderlijk indien de LCA resultaten onderdeel uitmaken in de besluitvorming over producten. Besluitvorming is bij voorkeur gebaseerd op robuuste en geloofwaardige LCA resultaten.

Resultaten worden als robuust beschouwd als ze ongevoelig zijn voor de meeste bekende onzekerheden. Onzekerheden kunnen op diverse locaties in alle vier LCA fasen voorkomen. De locaties kunnen betrekking hebben op de definitie van de levenscyclus van een product, de achterliggende filosofie van de gevolgde methode, de gebruikte data, de selectie van milieueffectcategorieën, of het modelleren van de milieueffectcategorieën.

In dit proefschrift maak ik onderscheid in drie typen van onzekerheden die relevant kunnen zijn voor alle onzekerheidslocaties, te weten: variabiliteit, keuzen en onbetrouwbaarheid. Variabiliteit heeft betrekking op de observeerbare variatie in waarden als gevolg van natuurlijke willekeurigheid of verscheidenheid. Een voorbeeld hiervan is het verschil in data voor eenzelfde proces dat op diverse verschillende productielocaties plaatsvindt. Keuzen hebben betrekking op de keuzemogelijkheden die de stakeholders en LCA uitvoerder maken. Alle andere vormen van onzekerheden heb ik gebundeld onder het type onbetrouwbaarheid. Hiertoe behoren onnauwkeurige, incorrecte, of niet-representatieve voorstellingen van data of modellen. De variabiliteit *binnen* een data set wordt nu al vaak meegenomen in LCA studies. Het meenemen van variabiliteit *tussen* verschillende data sets welke hetzelfde product of proces vertegenwoordigen is redelijk nieuw in LCA. Het gelijktijdig behandelen van zowel deze variabiliteit tussen data sets en modelleringsopties is sporadisch toegepast in LCA.

Het doel van mijn proefschrift is: *evalueren of het gebruik van meerdere data sets en meerdere modelleringsopties leidt tot robuustere LCA resultaten*. Om dit doel te bereiken heb ik drie onderzoeksvragen beantwoord: 1) wat zijn redenen voor verschillen in LCA resultaten voor hetzelfde product? 2) kan het gebruik van meerdere data sets voor een proces de robuustheid van LCA resultaten verhogen? En 3) kan het gebruik van meerdere modelleringsopties de robuustheid van LCA resultaten verhogen? Voor het beantwoorden van de laatste vraag heb ik naar verschillende opties in de afvalverwerking van producten gekeken en naar verschillende manieren voor het modelleren van recycling in LCA. Ik heb bij het beantwoorden van de vragen gebruik gemaakt van case studies voor wegwerpbekers en aluminium blikjes.

Mijn onderzoek begint met een review van tien bestaande LCA studies van wegwerpbekers (Hoofdstuk 2) om de mogelijke verschillen in LCA resultaten op te sporen. De tien (vergelijkende) studies bevatten wegwerpbekers gemaakt van fossiel-plastics (plastic afkomstig van olie en aardgas), van biobased-plastic (plastic gemaakt van hernieuwbare grondstoffen) en van karton. De resultaten van deze studies zijn zowel kwalitatief alsook kwantitatief met elkaar vergeleken. Klimaatverandering is de enige gemeenschappelijke effectcategorie in deze studies en de vergelijking was daarom gelimiteerd tot de resultaten voor klimaatverandering (afgekort tot GWP, Global Warming Potential). De absolute GWP waarden zijn eerst gecorrigeerd voor het volume van de wegwerpbekers. De resultaten van de studies wijzen niet consistent naar eenzelfde materiaal als het beste of slechtste beker materiaal. De GWP waarden variëren daarnaast ook binnen elke beker materiaal groep. De verhouding tussen de hoogste en laagste GWP waarden is 3,4 binnen de fossiel-plastic bekers, 1,7 binnen de biobased-plastic bekers en bijna 20 binnen de kartonnen bekers. Daarna heb ik de onderliggende methodologische keuzen die in elke studie zijn gemaakt met elkaar vergeleken om mogelijke redenen voor deze verschillen te achterhalen.

Een aantal van de onderzochte studies maakt gebruik van meerdere beker scenario's, waarbij elk scenario andere data, keuzen, aannames, of methodologische opties bevat. Deze scenario's maakten het mogelijk om de invloed van deze keuzen op de GWP resultaten te achterhalen. Ik heb de methodologische keuzen tussen deze studies vergeleken, waaruit redenen naar voren kwamen die niet identificeerbaar waren aan de hand van de scenario's binnen de individuele studies alleen. Redenen voor verschillen in GWP resultaten omvatten de variatie in de eigenschappen van de bekers (bijvoorbeeld het gewicht), de gebruikte productie processen, de gekozen afvalverwerkingsopties en verder keuzen in allocaties, systeemgrenzen, de geselecteerde effectcategorieën en mogelijk ook in de gebruikte data. Allocatie verwijst naar het verdelen van de milieubelasting van een proces over meerdere producten. Deze redenen representeren verschillende typen van onzekerheden.

De review van de tien studies identificeert dus ook de gebruikte data (sets) als een mogelijke bron voor onzekerheden in de uitkomst. De variabiliteit tussen data sets is in de tien studies niet onderzocht en komt ook in het algemeen niet in LCA studies aan bod. Daarnaast keken de tien studies doorgaans maar naar één onzekerheid tegelijkertijd. Het vakgebied van de geïntegreerde modellen voor grensoverschrijdende milieuproblemen, zoals verzuring of klimaatverandering, gebruikt vaak meerdere data sets en modellen voor het berekenen van resultaten. De ervaring daar leert dat het gelijktijdig gebruiken van verschillende data sets en modelkeuzen, teneinde meerdere onzekerheden tegelijkertijd mee te nemen, leidt tot meer robuustere resultaten. Het simultaan behandelen van meerdere onzekerheden zou mogelijk ook tot meer robuustere LCA resultaten kunnen leiden. Dit simultaan gebruikt komt in het volgende hoofdstukken aan de orde.

Doorgaans wordt in de inventarisatie fase van een LCA voor elk proces maar één data set gebruikt. In Hoofdstuk 3 maak ik opzettelijk gebruik van meerdere data sets voor een proces en evalueer ik de invloed hiervan op de LCA resultaten. Tegelijkertijd maak ik opzettelijk gebruik van meerdere modelleringsopties en evalueer ook de invloed hiervan op de LCA resultaten. Ik heb een methode getest om het gebruik van deze meerdere data sets en modelleringsopties in LCA te integreren. De methode bestaat uit een aantal stappen, die worden uitgelegd aan de hand van een LCA van een polystyreen (PS) wegwerpbeker. Eerst heb ik een initiële LCA van de PS beker gemaakt. Hierbij is voor elk proces één (gangbaar) data set gebruikt. Verder is verbranden als inititiële afvalverwerkingsoptie gemodelleerd. Daarna heb ik een additionele ´initiële´ LCA gemaakt met dezelfde data sets, maar nu met recyclen als afvalverwerkingsoptie. Dit betekent dat recyclen een additionele optie is voor het modelleren van de afvalverwerkingsfase in de initiële LCA. Op basis van zwaartepunts- en gevoeligheidsanalyses identificeerde ik de volgende processen als invloedrijke factoren voor de LCA resultaten: de productie van PS, het maken van de PS beker, het verbranden van de beker en het recyclen van de beker. Alleen voor deze invloedrijke processen zijn additionele data sets verzameld. Van elk verzameld data set is vervolgens het milieueffect berekend. Voor elk proces is daarna het gemiddelde milieueffect en de spreiding in het milieueffect berekend aan de hand van de data sets en modelleringsopties die bij het betreffende proces horen. De spreiding in de resultaten is dus berekend nadat de effectbeoordeling is uitgevoerd en zodoende behouden de resultaten de eventuele correlaties die binnen individuele data sets aanwezig kunnen zijn. Als laatste werden de effectbeoordelingsresultaten van de afzonderlijke processen met elkaar gecombineerd om de gemiddelde milieueffecten en de spreiding hierin voor de gehele LCA uit te rekenen.

De LCA resultaten op basis van meerdere data sets en modelleringsopties bevestigen dat de productie van PS, het maken van de PS beker en de afvalverwerkingsopties (verbranden en recyclen) de belangrijkste bijdragen leveren aan de milieueffecten over de levenscyclus. De variabiliteit in deze processen resulteerde in een aanzienlijke spreiding in de LCA resultaten. Verschillen in hoeveelheden en typen van de gebruikte materialen en energiebronnen, alsook verschillen in gerapporteerde emissies veroorzaken deze spreiding. De spreiding in de productie van PS is verder toe te schrijven aan verschillen in productie locaties en de perioden waarin de data zijn verzameld. De resultaten voor verbranden versus recyclen van de PS bekers behoeven nadere toelichting.

Het verbranden van PS levert energie op in de vorm van elektriciteit en warmte. Deze teruggewonnen energie vervangt het produceren van energie elders en hiervoor ontvangt het verbranden van de PS beker krediet. Recyclen van de PS beker levert gerecycled PS op. Dit gerecycled materiaal kan op zijn beurt weer als grondstof dienen voor de fabricage van een volgend product. De PS beker wordt in dit geval als het ware beloond voor het maken van gerecycled materiaal en ontvangt hier krediet voor in de LCA. De spreiding in de resultaten van de beloning van het gerecycled PS is afkomstig van de verschillende manieren van het toewijzen van dit krediet. De spreidingen in de resultaten voor het verbranden en het recyclen van de PS beker overlappen elkaar, waardoor er geen overtuigende voorkeur kon worden gemaakt voor één van beide afvalverwerkingsopties.

De spreiding in de resultaten van de totale LCA was relatief hoog voor de toxiciteitscategorieën. De spreiding was relatief laag in milieueffectcategorieën waaraan energiegebruik een dominante bijdragen levert. Het gebruik van de meerdere data sets en modelleringsopties verhoogt het inzicht in de relatieve bijdrage van de afzonderlijke processen in de totale LCA resultaten en in de onzekerheid in deze resultaten. De ontstane spreiding in de resultaten verhoogt daarmee wel de complexiteit van de resultaten. Ondanks dat de resultaten hierdoor mogelijk minder eenvoudig te leggen zijn, zijn de resultaten meer betrouwbaar en verschaffen de beslissers daarmee waardevolle informatie. Het gebruik van de meerdere data sets vertegenwoordigt immers de variabiliteit die in de praktijk bestaat tussen dezelfde of soortgelijke processen. Het gebruik van meerdere afvalverwerkingsopties vertegenwoordigt de keuzen in het modelleren van de levenscyclus van de PS beker. Het simultaan behandelen van deze onzekerheden leidt tot robuustere resultaten vergeleken met het individuele adresseren van onzekerheden.

De methode voor het gebruik van meerdere data sets en modelleringsopties is in Hoofdstuk 3 toegepast op een PS wegwerpbeker. Hoofdstuk 4 gaat over het testen van de bruikbaarheid van deze methode in een vergelijkende LCA studie van drie wegwerpbekers. De drie onderzochte bekers zijn gemaakt van: 1) polystyreen (PS), 2) poly-melkzuur (PLA) en 3) biokarton. Poly-melkzuur (polylactic acid, PLA) is een biobased plastic, gemaakt van hernieuwbare grondstoffen. Biokarton is karton met een laagje biobased plastic. Voor alle drie bekers zijn verbranden en recyclen als afvalverwerkingsopties opgenomen. Tevens zijn composteren en anaeroob vergisten meegenomen voor de PLA en biokartonnen beker. Daarnaast zijn er meerdere manieren gebruikt om het gerecycled plastic (PS en PLA) te belonen. De LCA resultaten van de drie bekers wijzen wederom naar dezelfde invloedrijke processen op de LCA resultaten, namelijk de productie van het beker materiaal (PS, PLA en biokarton), het maken van de bekers en de afvalverwerking. Ook nu weer wordt de spreiding in de resultaten van deze processen beïnvloed door de hoeveelheid en type van energie gebruik, de beloning voor het gerecycled materiaal, de beloning voor teruggewonnen energie en de gerapporteerde emissies. De productie van PS is een volgroeid proces en de spreiding is in dit proces lager vergeleken met de productie van PLA dat nog in de ontwikkelingsfase verkeert. De variatie in PLA productie komt voort uit de verschillende basismaterialen die als grondstof dienen voor de productie van melkzuur (mais en suikerbieten). De spreiding is in de toxiciteit effectcategorieën, net als in Hoofdstuk 2, wederom hoog in vergelijking met de energie gerelateerde effect-categorieën.

De gemiddelde resultaten voor de bekermaterialen geven geen duidelijke indicatie over het meest milieuvriendelijke bekermateriaal indien elf verschillende effectcategorieën worden meegenomen. De gemiddelde resultaten voor de afvalverwekingsopties van de PLA beker suggereren een voorkeur voor recyclen ten opzichte van de andere opties. De spreiding in de resultaten voor de afvalverwerkingsopties overlappen elkaar binnen elk van de drie bekers. Deze overlap belet een duidelijke identificatie van de meest geprefereerde afvalverwerkingsoptie voor elk van de drie bekers. Alleen composteren van de PLA en de biokartonnen bekers komt duidelijk als minst gewenste optie naar voren.

Het gebruik van meerdere data sets voor een proces representeert de variabiliteit in deze processen. In dit onderzoek representeren deze meerdere data sets een wegwerpbeker zoals gemiddeld in Nederland gebruikt, en dus niet een specifieke beker gemaakt door een specifiek bedrijf. Het gebruik van een specifieke data set of een specifieke modelleringoptie zou de uitkomst van de vergelijkende studie voor de drie bekers in een bepaalde richting kunnen sturen. Het gebruik van meerdere data sets en modelleringsopties leidde tot een aanzienlijke spreiding in de resultaten van de bekers. Alhoewel deze spreiding de complexiteit van de vergelijking van de bekers verhoogt, geven de resultaten meer robuustere informatie voor de beslissingsnemers. Het simultaan gebruik van de variabiliteit in de data sets en de meerdere afvalverwerkingsopties verhoogt de betrouwbaarheid van de resultaten.

De LCA resultaten van de plastic bekers (PS en PLA) vertonen een spreiding in de resultaten voor het recyclen van de bekers (Hoofdstuk 3 en 4). Deze spreiding wordt veroorzaakt door verschillende manieren waarop het gerecycled plastic wordt beloond. Het modelleren van recyclen in LCA studies is een onopgelost probleem en leidt tot herhaalde discussies binnen de LCA gemeenschap. Recyclen is een zogenoemd multifunctioneel proces. Recyclen is enerzijds een afvalverwerkingsoptie voor weggegooide producten, maar anderzijds ook een productie proces voor gerecycled materiaal dat in een volgend product gebruikt kan worden. In Hoofdstuk 5 beschrijf en evalueer ik zes veelgebruikte methoden om de multifunctionaliteit van recyclen te modelleren: drie substitutie methoden, een allocatie methode, de recycled-content methode en de gelijke-aandelen methode. De drie substitutie methoden gaan ervan uit dat het gerecyclede materiaal het originele materiaal kan vervangen op basis van een gelijke kwaliteit (methode 1), met in acht neming van een correctie factor voor een verlies in kwaliteit (methode 2), of een substitutie met een alternatief materiaal (methode 3). De allocatie methode is in deze studie gebaseerd op het aantal recycling loops dat het materiaal kan ondergaan. De recycled-content methode is gebaseerd op het aandeel gerecycled materiaal in het product. De gelijke-aandelen methode is een mix van de substitutie methode en de recycled-content methode, waarbij beide methoden een gelijk aandeel hebben. Het belangrijkste verschil tussen deze zes methoden is de opvatting over waar en hoe de beloning voor het gerecycled materiaal toe te passen. De drie substitutie methoden stimuleren het recyclen van het product als afvalverwerkingsoptie en geven de beloning aan het product dat het gerecycled materiaal produceert. De allocatie methode (gebaseerd op het aantal recycling loops) verdeelt de milieubelasting van de productie van het primaire materiaal over de gehele levenscyclus van het materiaal zelf, in tegenstelling tot de levenscyclus van een product. De recycled-content methode stimuleert het gebruik van gerecycled materiaal in producten en beloont het product dat gerecycled materiaal als grondstof gebruikt. De gelijke-aandelen methode is een compromis tussen het gebruik van gerecycled materiaal in het product (recycled-content methode) en de productie van gerecycled materiaal als afvalverwerkingsoptie van het product (substitutie methode).

Ik heb de bovenstaande zes methoden voor het modelleren van recyclen toegepast in twee case studies met als doel het effect van de verschillende methoden op de LCA resultaten te onderzoeken. De twee case studies zijn een PS wegwerpbeker en een aluminium blikje. In dit onderzoek ben ik uitgegaan van een hypothetisch 100% recycling percentage om het maximale effect van de methoden te zien. Aluminium behoudt zijn eigenschappen nadat het gerecycled is en wordt daarom vaak als een gesloten systeem beschouwd. De kwaliteit van

PS kan tijdens het recyclen achteruit gaan en daardoor kan gerecycled PS mogelijk niet het originele materiaal vervangen, maar alleen PS van een andere kwaliteit of een heel ander materiaal. PS recycling wordt daarom als een zogenaamd open-loop systeem beschouwd. De LCA resultaten van beide case studies hangen duidelijk af van de toegepaste recycling modelleringsmethode, het recyclingpercentage en de hoeveel gerecycled materiaal dat in het product is gebruikt. Daarnaast hangen de resultaten van de PS beker ook af van de veronderstelde kwaliteitsverlaging van het gerecycled PS. Deze kwaliteitsverlaging kan worden uitgedrukt in een correctie factor of door te belonen met alternatief materiaal dat door het gerecycled PS kan worden vervangen. Het vaststellen van de hoogte van de correctiefactor en het te vervangen materiaal zijn discutabel.

De substitutie methode en de recycled-content methode zijn nogmaals toegepast op de PS beker en het aluminium blikje, maar nu gebaseerd op de actuele combinatie van afvalverwerkingspraktijken in Europa en in enkele Europese landen. De LCA resultaten van het aluminium blikje hangen in de substitutie methode af van het recyclingpercentage van het afgedankte blikje en deze resultaten verschillen daardoor tussen de diverse landen. De resultaten voor het blikje zijn gelijk voor alle landen indien de recycled-content methode wordt toegepast. De keuze van de toegepaste methode speelt daarom een grote rol in de resultaten van de aluminium blikjes. De resultaten voor de PS beker worden in beide methoden beïnvloed door de overige afvalverwerkingsopties, namelijk verbranden en storten. Tijdens het verbranden van PS kan energie terug gewonnen worden en de PS beker wordt daarom beloond met deze teruggewonnen energie. Variatie in de afvalverwerkingspraktijken tussen de landen en variatie in het type energie dat in de landen wordt gebruikt (i.v.m. de beloning van de teruggewonnen energie) beïnvloeden de resultaten van de PS beker. Verder is de uitwerking van deze twee variaties en van de toegepaste methode verschillend voor de diverse effectcategorieën. Het is daarom belangrijk om niet alleen naar het effect op de veelgebruikte klimaatverandering te kijken, maar ook naar de uitwerking in andere effectcategorieën.

De verschillende methoden om de multifunctionaliteit van recyclen in LCA te modelleren representeren onderliggende modelleringsfilosofieën. De keuze van de toegepaste methode heeft een duidelijk effect op de resultaten van het aluminium blikje en de PS beker. Door het kiezen van een bepaalde methode kunnen de resultaten in een specifieke richting worden geleid. Conflicterende meningen over duurzaamheid kunnen niet geëlimineerd worden waardoor er altijd verschillende visies zullen blijven bestaan op de toe te passen modelleringsmethode voor recyclen. Door gebruik te maken van diverse recycling modelleringsmethoden in een LCA worden deze verschillende standpunten meegenomen in de resultaten, waardoor de resultaten robuuster worden.

Mijn proefschrift toont de toegevoegde waarde van de methode om meerdere data sets en meerdere modelleringsopties mee te nemen in LCA studies. De identificatie van de meest invloedrijke onzekerheidslocaties is een cruciale stap in de gehele procedure. De spreiding in de resultaten is in mijn aanpak berekend nadat de effectbeoordeling heeft plaatsgevonden en daardoor worden correlaties binnen de individuele data sets behouden. Daarnaast is er geen specifiek formaat nodig voor de presentatie van inventarisatie data in de data sets. Het gebruik van meerdere data sets is vooral bruikbaar indien data voor een gemiddeld product nodig zijn en niet voor specifiek product van een specifieke producent. Het gebruik van meerder data sets verhoogt de juistheid van de resultaten en is een aanvullende tool naast de statistische methoden die de precisie van de resultaten beogen te verhogen. Het simultaan behandelen van meerdere data sets en modelleringsopties is nog nauwelijks toegepast in LCA studies. De methode in Hoofdstuk 3 vult deze leegte en beschrijft een transparante aanpak om deze onzekerheden mee te nemen. De afweging tussen de verhoging van de robuustheid van de resultaten en de additionele benodigde middelen (tijd, geld, inzet) moet afgewogen worden en hangt af van het doel van de studie en het beoogd gebruik van de resultaten. Mijn onderzoek toont aan dat het zichtbaar maken van onzekerheden in de resultaten de besluitnemers van waardevolle informatie kan voorzien. Dit proefschrift biedt een bruikbare methode om de robuustheid van LCA resultaten te verhogen.

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About the author

Eugenie van der Harst-Wintraecken was born on 22 March 1961 in Bunde, the Netherlands. After high school she studied chemistry at the Zuid-Limburgse Laboratoriumschool and received a Bachelor in Analytical Chemistry. Her first job was as a chemical researcher and development analyst at a chemical factory. During that period her interest in computers grew. She switched jobs and worked as a programmer and systems analyst for the municipality of Heerlen, and later as an information analyst at Rijkswaterstaat. In 1993 she and her husband moved to the Atlanta area in Georgia (USA), where her husband was offered a job. She went back to study and received a BSc in Mathematics at Kennesaw State University. After spending five years in the Atlanta area, she moved with her husband to the Gothenburg area in Sweden. Her interest for nature and the environment rekindled during her stay in Sweden and she started taking courses in natural sciences at the Open University. After spending more than five years in Sweden, Eugenie and her husband continued their foreign journey and stayed for nearly a year in Mexico City (Mexico), and two years in Cary, North Carolina (USA). In the end of 2006 they returned to the Netherlands. Eugenie picked up her study at the Open University and received a MSc in Environmental Science (cum laude) in 2010. She made her first Life Cycle Assessment (LCA) during her MSc internship. After her graduation she revised exams for one of the courses from the Open University. In 2011 she started working as a researcher in the Biocup project at Wageningen UR. After a year she was offered the opportunity to works towards a PhD. The topic of her PhD thesis is Life Cycle Assessment. She developed a method to increase the robustness of the LCA results. During her stay at Wageningen UR she supervised MSc students during their thesis work and assisted in several BSc and MSc courses. Presently she is looking forward to a new challenge in her professional life.

List of publications

Helder M., Chen W-S., van der Harst E.J.M., Strik D.P.B.T.B., Hamelers H.V.M., Buisman C.J.N., Potting, J. 2013. Electricity production with living plants on a green roof: environmental performance of the plant-microbial fuel cell. *Biofuels, Bioproducts and Biorefining* 7: 52-64.

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Van der Harst E., Potting J., Kroeze C. 2015. Comparison of different methods to include recycling in LCAs of aluminium cans and disposable polystyrene cups. *Waste Management* In press.

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- Environmental Research in Context (2013)
- Complex dynamics in Human-Environment systems (2013)

Other PhD and Advanced MSc Courses

- Supervising MSc thesis, Wageningen University (2013)
- Practical uncertainty analysis in LCA, International Life Cycle Academy (2013)
- Reviewing a Scientific Paper, Wageningen University (2014)
- Uncertainty and Quality in Science for Policy, University of Bergen (2014)
- Techniques for writing and presenting a scientific paper, Wageningen University (2014)
- Environmental Impact Assessment Livestock Systems, Wageningen University (2015)

Management and Didactic Skills Training

- Supervising seven MSc thesis students (2011-2015)
- Assisting practicals of the MSc course 'Analysis and Management of Sustainable Organic Production Chains' (2011-2012)
- Teaching and co-organising the BSc course 'Introduction to Environmental Systems Analysis' (2012-2014)
- Co-organising the 7th International Symposium on Non-CO_2 Greenhouse Gases (NCGG7) Symposium, Amsterdam (2014)
- Supervising projects for the BSc course 'Environmental Project Studies' (2015)

Oral Presentations

- *Spread in LCA results from using multiple data sets and modelling choices*. Life Cycle Management Conference (LCM 2013), 25-28 August 2013, Gothenburg, Sweden
- *Wie van de drie: Wil de meest milieuvriendelijke beker opstaan?* Masterdag Open Universiteit, 1 June 2013, Utrecht, The Netherlands
- *Comparison of disposable cups*. Sense symposium: Carbon climate interactions, 12 January 2015, Wageningen, The Netherlands

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