Matthias Finkbeiner *Editor*

Towards Life Cycle Sustainability Management

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Preface

Towards Life Cycle Sustainability Management

The global society has undergone a paradigm shift from environmental protection towards sustainability. Sustainability does not only focus on the environmental impact, it rather consists of the three dimensions “environment”, “economy” and “social well-being”, for which society needs to find a balance or even an optimum. Sustainability has become mainstream these days. It is accepted by all stakeholders - be it multinational companies, governments or NGOs. Unfortunately, this common understanding merely relates to the general concept rather than actions. But lip service is not enough to achieve a sustainable development of our societies. If we want to make sustainability happen as concrete reality in both public policy making and corporate strategies, sustainability cannot please everybody. To make it happen, we have to be able to discern good and evil. This requires that we are able to address the question, how sustainability performance can be measured, especially for companies, products and processes. We have to be smart enough to be able to measure it or the real and substantial implementation of the sustainability concept will remain just wishful thinking.

In order to achieve reliable and robust sustainability assessment results it is inevitable that the principles of comprehensiveness and life cycle perspective are applied. The life cycle perspective considers for products all life cycle stages and for organisations the complete supply or value chains, from raw material extraction and acquisition, through energy and material production and manufacturing, to use and end of life treatment and final disposal. Through such a systematic overview and perspective, the unintentional shifting of environmental burdens, economic benefits and social well-being between life cycle stages or individual processes can be identified and possibly avoided. Another important principle is comprehensiveness, because it considers “all” attributes or aspects of environmental, economic and social performance and interventions. By considering all attributes and aspects within one assessment in a cross-media and multidimensional perspective, potential trade-offs can be identified and assessed.

This is where life cycle assessment (LCA) and life cycle management (LCM) come into play. LCA is the internationally accepted method for measuring environmental performance and LCM is in a nutshell about the application of LCA or rather life cycle thinking (LCT). It is still a relatively young concept in the environmental community with pioneering work done by a Working Group of the Society of Environmental Toxicology and Chemistry (SETAC) at the end of the last century. At that time, my definition of LCM was “a comprehensive approach towards product and organisation related environmental management tools that follow a life cycle perspective." The United Nations Environment Programme (UNEP) and SETAC later launched the Life Cycle Initiative to enable users around the world to put life cycle thinking into effective practice and introduced LCM as one of their areas of work.

While the measurement of the environmental dimension of sustainability with LCA is well established, similar approaches were developed more recently for the economic (life cycle costing – LCC) and the social (social LCA – SLCA) dimensions of sustainability. This development is crucial, because it fosters the opportunity for life cycle based sustainability assessments. Walter Klöpffer put this idea into the conceptual formula:

LCSA	= LCA + LCC + SLCA
LCSA	= Life Cycle Sustainability Assessment
LCA	= Environmental Life Cycle Assessment
LCC	= Life Cycle Costing
SLCA	= Social Life Cycle Assessment

Even though there is definitely still room to improve and expand the implementation of LCA as part of an environmental LCM approach, I believe the time has come to expand the concept to include the other pillars of sustainability in a more explicit way. This is reflected in our choice of the title of this book "Towards Life Cycle Sustainability Management". Life Cycle Sustainability Management or LCSM is the implementation of life cycle based sustainability assessment or LCSA into real world decision making processes, be it on the product, process or organisation level. In a nutshell, LCSM aims at maximising the triple bottom line (3BL) and is based on LCSA as one key element of a broader toolbox:

$$\text{LCSM} \quad = \quad f\,(\text{LCSA}) \quad = \quad \max\,(3\text{BL})$$

This book is a selection of the most relevant contributions to the LCM 2011 conference in Berlin. The Life Cycle Management conference series is established as one of the leading events worldwide in the field of environmental, economic and social sustainability. The unique feature of LCM is practical solutions for the implementation of life cycle approaches into strategic and operational decision-making. The 2011 conference motto "Towards Life Cycle Sustainability Management" was chosen to address and to focus on the implementation challenge of sustainability as outlined above. In total, 414 abstracts representing more than 1100 authors from 47 countries were submitted.

Because of the excellent overall quality of the contributions it was quite a challenge to select the 56 papers for this book. They are structured in nine Parts. The first four Parts focus on the more general, methodological topics. Part I addresses general LCSM approaches that go beyond the more traditional LCM methods and tools which are covered in Part II. Part III deals with water footprinting as specific and emerging topic. LCM applications for processes and organisations are the content of Part IV. The remaining five Parts deal with the implementation of LCM approaches in relevant industrial sectors, namely the agriculture and food sectors (Part V), the packaging sector (Part VI), the energy sector (Part VII), the electronics and ICT sectors (Part VIII) and the mobility sector (Part IX).

The authors of this volume come from 29 countries including Africa, Asia, Europe and the Americas. They represent the developed and the developing world as well as a variety of stakeholders from multinational companies, academia, NGOs to public policy. I am very grateful for their excellent and timely contributions.

In addition to the core contribution of the authors this book was only possible due to the efforts of many colleagues and friends. I am very grateful for the support of the co-chair of the LCM 2011 conference, Stephan Krinke, and all members of the scientific committee: Carina Alles, Emmanuelle Aoustin, Pankaj Bhatia, Clare Broadbent, Andrea Brown Smatlan, Maurizio Cellura, Roland Clift, Mary Ann Curran, Ichiro Daigo, James Fava, Jeppe Frydendal, Pere Fullana, Gerard Gaillard, Mark Goedkoop, Minako Hara, Michael Hauschild, Jens Hesselbach, Arpad Horvath, Atsushi Inaba, Allan Astrup Jensen, Anne Johnson, Juha Kaila, Gregory Keoleian, Henry King, Walter Klöpffer, Annette Koehler, Paolo Masoni, Yasunari Matsuno, Llorenc Mila i Canals, Nils Nissen, Philippa Notten, Erwin Ostermann, Rana Pant, Claus Stig Pedersen, Gerald Rebitzer, Helmut Rechberger, Klaus Ruhland, Günther Seliger, Guido Sonnemann, Nydia Suppen, Ladji Tikana, Sonia Valdivia, Paul Vaughan and Harro von Blottnitz. Their efforts in soliciting and selecting the right mix of contributions were extremely valuable.

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Berlin
Prof. Dr. Matthias Finkbeiner

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PART I:
LCSM Approaches

Integrating Sustainability Considerations into Product Development: A Practical Tool for Prioritising Social Sustainability Indicators and Experiences from Real Case Application

Gustav Sandin, Greg Peters, Annica Pilgård, Magdalena Svanström and Mats Westin

Abstract In this paper, a tool for prioritising social sustainability parameters in product development is described. The tool's core element is a two-step Delphi exercise carried out in the product development team. The purpose of the tool is to (i) select critical social impact indicators suitable for guiding the product development process, (ii) enhance the product development team's understanding in the field of social sustainability and (iii) engage the team in the sustainability assessment, with the further aim of ensuring the assessment's influence on the product development process. Applied in a real product development project, the tool proved successful for selecting indicators and increase understanding of social sustainability within the product development team. Selected indicators' usefulness for the product development process remains an open question to be addressed later on as the project evolves.

1 Introduction

The product development process is regarded a crucial intervention point for sustainable development in society [1,2]. In the life cycle management community, the focus on the social dimension of sustainability has increased over the past few years and is expected to further increase over the next decade [3-5]. While tools exist for considering environmental parameters in product development, there is a general lack of practical tools for considering social parameters. In 2009, UNEP and SETAC issued guidelines for social life cycle assessment (SLCA), drawing on the environmental life cycle assessment (LCA) methodology [6]. However, there is a wide range of different approaches with

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regard to the methodology of SLCA, which is a sign of the immaturity of the field [7]. For example, no standard or commonly accepted set of metrics or indicators exist for measuring the social sustainability of a product life cycle [5,7-10]. Some researchers have tried to compensate for this by stakeholder engagement in selecting and rating indicators for other aspects of sustainability, thereby claiming that social aspects have been covered by inclusion of stakeholders' opinions and values in the process [11].

According to the experience of the authors, the product development processes of today, which often involve a multitude of representatives from academia and industry in diverse fields of expertise, frequently feature a lack of understanding of social sustainability. This must be effectively managed by tools used in the sustainability assessment, as it may aggravate data collection and hinder the assessment's influence on decision-making, critical elements in order for the assessment to successfully guide the product development process.

In this paper, a tool for considering social sustainability in early product development processes is described and its usefulness in application in a real product development case is analysed. The purpose of the tool is to (i) select critical social impact indicators suitable for guiding the product development process, (ii) enhance the product development team's understanding in the field of social sustainability and (iii) engage the team in the sustainability assessment, with the further aim of ensuring the assessment's influence on the product development process. Thus, the tool aspires to contribute to resolving the above outlined issues: the non-existence of standard sets of indicators and the common lack of knowledge with regard to social sustainability within product development teams.

2 Method

The tool is based on the notion that a product's social impact is defined as its influence on people's well-being throughout its life cycle [6,12,13], and that this impact mainly depends on the conduct of companies involved in the life cycle towards their stakeholders and not on the production processes themselves [12-14]. The tool was developed with a life cycle perspective in mind, in order to facilitate integration into a more comprehensive life-cycle oriented sustainability assessment of the three pillars of sustainability (e.g. including methods such as LCA and life cycle costing). An exercise carried out in the product development team is the core element of the tool. In the exercise, the Delphi method is used to rate a set of indicators on the social impact of a certain product life cycle.

2.1 The Delphi method

The Delphi method was developed as a means of obtaining reliable consensus in a panel of experts [15]. The method was originally used as a forecasting technique but has come to be used in a broader range of applications [15]. For example, it has been used for structured group communication when dealing with complex problems including subjective judgements [16], which is the area of application in this study. In such contexts, the method's main characteristic is perhaps its ability to produce useful guidance [17].

Traditionally, the method includes the following elements: (i) individuals anonymously contribute to a group judgement (e.g. by answering a questionnaire asking for ratings of a number of options), (ii) the group judgement is assessed (e.g. as a statistical summary of the ratings) and (iii) individuals are given the opportunity to adjust their view following feedback on the group's judgement [16]. Usually, the process is repeated until the responses meet a certain level of stability (e.g. the mean of the ratings does not change significantly between rounds), which is then considered a fairly reliable measure of the group's opinion [15].

The strengths of the Delphi method primarily lie in its anonymity, which aims at avoiding social pressures within the group, and its iterative approach, which facilitates sharing and reconsideration of opinions [15]. Critiques of the Delphi method often emphasise its inability to perform better forecasts than other techniques or that it can be costly compared to other structured group interaction procedures [15].

In this study, a simplified Delphi method limited to one round of feedback is used: the two-step Delphi method. This was done due to the requirements on the tool of being practical as well as time and resource efficient. Besides, it was deemed sufficient for the purpose of the exercise.

A few cases of using the Delphi method for rating sustainability indicators can be found in literature [18-20]. However, the authors are not familiar with any previous attempts of applying the Delphi method for rating social sustainability indicators in a product development context.

2.2 Procedure of the social sustainability assessment tool

The social sustainability assessment tool consists of the following steps:

1) An oral presentation is held for the product development team, introducing the field of social sustainability and the exercise.

2) The team members anonymously rate a preselected set of indicators of how companies in the product life cycle influence stakeholders on a 1-10 scale regarding two dimensions: general importance and relevance for the specific product life cycle. Team members are also given the chance to provide written comments on their ratings.
3) The participants are given the opportunity to revise their ratings after receiving feedback on the team's mean rating on each indicator, the rating's standard deviation and the comments relating to ratings that fall outside of the standard deviation.
4) The revised rating of the general importance and the specific relevance are multiplied, generating a final score on a 1-100 scale for each indicator. This score is then used in the further sustainability assessment, e.g. for selecting indicators to be used for guiding the product development process.

3 Real case application

The tool was applied in an on-going product development project featuring applied technical research with the aim of developing a new material for consumer products. Representatives from 10 different organisations, including both academia and industry, are involved in the project. The project is organised into a number of work packages, whereof one is dedicated to the sustainability assessment. Decision-making, e.g. regarding viable routes of development, which the sustainability assessment seeks to influence, is done in everyday project work as well as in steering committee meetings. Within the project team, different members have different perspectives and various levels of previous knowledge in the field of social sustainability.

The project has a goal of contributing to a more sustainable society. The holistic perspective of the triple bottom line and the importance of a life cycle perspective are highlighted in the project's guiding documents. However, project objectives do not outline specific sustainability parameters to focus on (e.g. specific impact categories or social themes). Hence, it is part of the sustainability assessment to identify the critical sustainability parameters.

3.1 Indicator set used in the exercise

Following a literature study on social sustainability indicators proposed for sustainability assessments, 74 indicators were identified. The set was narrowed down to 36 indicators by merging, rephrasing and excluding indicators, based on four criteria acquired from Spillemaeckers et al [14]: measurability, relevance to the specific product, feasibility with regard to the resources at hand and applicability in the particular project. Selected indicators are found in Table 1. Each indicator was allocated a social theme (“social theme” is alternately referred to as “social impact category” in SLCA contexts [6]) and assigned to the most suitable of the stakeholder categories outlined in the UNEP/SETAC SLCA guidelines [6]: employees, local community, society, consumers and value-chain actors. However, a few indicators may reflect impact relevant to several stakeholders. Indicators either relate to activities of specific companies or to the general situation in the companies' sector or region of operation, which may be suitable for hotspot identification in case the firms to be involved in the product system are either unknown or difficult to assess. Some indicators are phrased precisely as in the original reference(s) listed in the table; others have been rephrased and/or merged to better fit the case application. Indicators may be yes/no questions, or require quantitative answers, qualitative descriptions, or a combination of these. Also the final score (product of the general importance and specific relevance ratings), as obtained in the case application, are displayed in Table 1.

3.2 Procedures of case application

Fourteen project partners' representatives participated in the exercise, whereof three represented industry partners and 11 represented other partners, including research institutes and academia. All participants except one were present for the introductory oral presentation, which was held in connection to a larger project meeting. The exercise was carried out by means of a Microsoft Excel questionnaire and e-mail communication. A two-page briefing on social sustainability, sustainability assessment and the exercise procedures was attached to the questionnaire.

After completion of the exercise, the participants were asked to reply to an anonymous online questionnaire, including seven yes/no/no opinion questions on how they perceived the exercise and the result's future usefulness. The respondents also had the opportunity to provide written comments to their replies.

Tab. 1: Social sustainability indicators used in the case application and final scores.

Social theme	Indicator	Refe-rence(s)	Final score
Stakeholder category: employees			
Health and safety	Presence of formal policy on health and safety	21	58.0
Health and safety	Average number of lost workdays due to injury/illness per year	22-25	58.4
Safety	Are emergency escape routes clearly marked and sprinkler systems and fire extinguishers installed in the company premises?	26	58.5
Employment security	Percentage of workforce on permanent employment contract	24	35.2
Professional development	Average number of hours employee training per year	22-25	37.6
Working hours	Clear communication of working hours and overtime arrangements and respect of contractual agreements concerning overtime	21	48.5
Child labour	Assessment of child labour reports by sector/region of operation (e.g. according to UCW country reports)	27	48.9
Child labour	Records on all workers stating names and ages or dates of birth are kept on file	21	52.5
Freedom of association and collective bargaining	Evidence of restriction to freedom of association and collective bargaining in sector/region of operation (e.g. according to International Trade Union Confederation Annual Survey of violations of trade union rights, LabourStart reports or organizations' GRI Sustainability reports)	14,21,24	42.5
Fair salary	Payment ratio (salary of upper 10% of employees/salary of lower 10% of employees)	22	35.2
Fair salary	Wages amount to at least living wage (or, if higher, minimum wage) for the concerned region at all times (e.g. according to SweetFree Communties' reports on non poverty wages)	14,21,24, 28	51.1
Equal opportunities/ discrimination	Ratio of average salary of men to women	21,24	41.4
Social benefits/social security	List of short descriptions of social benefits provided to workers (health insurance, pension fund, child care, education, accomodation, etc.)	21,29	43.2
Social benefits	Percentage of permanent workers receiving paid time-off	21	31.7
Social benefits	Average length of annually paid vacation	30	39.2
Work conditions (general)	Contracts stipulate wage, working time, vacation, terms of resignation and are kept on file	28	56.2
Work conditions (general)	All employees have the possibility to file complaints about labour practices which conflict with the principles of employment on a voluntary basis, in confidentiality and without negative consequences	28	51.9
Work conditions (general)	Does the country of operation ratify all ILO core labour standards and/or are there any known work condition issues in the region/sector of operation?	31	44.4
Respect of human rights	Existence of media reports within the last 5 years on human rights violations or discrimination	14,21,24	35.6
Stakeholder category: local community			
Community development/social justice	Community spending and charitable contributions as percent of revenues	14,22,23, 25	17.3
Community development	Number of working hours per functional unit	5,32	34.6
Secure living conditions	Strength of public security in country of operation (e.g. ranking in World Economic Forum Global Competiveness Report)	21	35.1
Respect of freedom of expression	Freedom of expression in country of operation (e.g. according to Freedom House publications or Amnesty International human rights reports)	21	33.8

Respect of cultural heritage	Is relevant organizational information available to community members in their spoken language(s)?	21	31.7
Conflicts with local community	Number of complaints from neighbours/local community during the last 5 years	14,22,25	47.1
Stakeholder category: society			
Public commitments/ transparency	Presence of publicly available code of conducts, agreements or other document on promises on sustainability issues (e.g. Global Compact, Sullivan principles, Caux Round Table, UN principles or GRI reports) and mechanism to follow-up the realisation of these promises	21	51.1
Contribution to economic/technology development	R&D costs as percentage of turnover	21,25,31	51.8
Corruption	Identification of potential corruption hotspots in region/sector of operation (e.g. by reviewing the Corruption Perception Index published annually by Transparency International)	21	39.5
Transparency	Presence of annual Corporate Social Responsibility and Environmental Sustainable Development reports	21,23	40.9
Prevention and mitigation of armed crisis	Is the organization doing business in a sector that features linkages to conflicts, e.g. where the depletion of resources allows significant profits?	21	60.1
Stakeholder category: consumers			
Consumer feedback	Presence of consumer feedback mechanism (after sale services, regular consumer satisfaction studies, etc.)	14,21	32.5
End-of-life responsibility	Do internal management systems ensure that clear information is provided to consumers about end-of-life options (if applicable)?	21	50.5
Extra customer benefits	Extra product or service benefits that enhance customer well-being (compared to competitors)	29	32.4
Stakeholder cateogry: vaule-chain actors			
Fair competition	Documented statements or procedures (policy, strategy, etc.) to prevent engaging in or being complicit in anti-competitive behaviour	21	33.1
Fair competition	Is the enterprise subject to legal actions regarding anti-competitive behavior and violations of anti-trust and monopoly legislation, or are there any known non-compliance with industry regulations?	21,24,25	36.0
Promoting social responsibility	Presence of explicit code of conduct that protects human rights and workers among suppliers and/or membership in an initiative that promotes social responsibility along the supply chain	21	56.0

4 Results and discussion

Table 1 displays the final scores of the exercise, as applied in the case study. It is imperative to keep in mind that these scores depend on the context of the specific product development project and the values, knowledge and experiences of the participants. The subjective nature renders the scores inappropriate to use for external communication in situations in which they may be interpreted as reflecting the official view of involved partners. Neither shall the scores be viewed as a final verdict on the importance of different social sustainability parameters. The result is displayed in this paper merely as an example of final scores.

The considerable differences between the final scores (spanning from 17.3 to 60.7, see Table 1) justify the exercise as a basis for defining a smaller and more manageable set of indicators. This was done by selecting the overall top 10 ranked indicators and the 10 indicators ranked highest among the industry partners, a breakdown done in order for the industry partners to have a larger influence on the selection process. This was considered reasonable as the industry partners were more experienced in the field of social sustainability than other project partners, and represent organisations that will be held responsible for the project outcome in case it is commercialised. In a few instances, indicators on the two top 10 lists were regarded too similar, e.g. representing the same social theme. In one of these instances, the indicators were rephrased and merged; in all other cases the highest rated indicator was selected. In order to cover all stakeholder categories, the top rated customer indicator was also included, although it was on none of the two top 10 lists. In this way, a set of 13 indicators was selected. This is one example of how a set of indicators could be selected based on the exercise result. However, procedures will probably have to be tailored on a case-to-case basis. This applies to the whole tool as well; it should be adapted to fit each product development project, for example depending on the resources at hand. Adaptations could include e.g. altering the indicator set used in the exercise, including more feedback rounds in the Delphi process or reformulation of the criteria used for rating. Consideration should always be given to recent developments in the field of sustainability assessment, for example to newly developed indicators.

Seven of the 14 exercise participants replied to the feedback questionnaire. Although this is a weak basis for far-reaching conclusions, a few things can be said. Six respondents stated they had, by participating in the exercise, acquired a clearer understanding of social sustainability. This indicates that the exercise did contribute to the aim of increased understanding of social sustainability within the project team. Only four respondents answered that they believe the 13 indicators selected based on the result of the exercise will be practical and meaningful (i.e. have a clear connection to the work in the project) for evaluating the project outcome's social sustainability in relation to comparable products. As considering social sustainability parameters in technology intensive product development is not yet common practice, it is not surprising that an early attempt in this area is met with some doubts. The underdeveloped methodology and lack of case application experience even in the life cycle management community, indeed justifies some degree of scepticism. Furthermore, one respondent expressed doubt regarding the practicability of using the indicators for assessing production systems which are not yet up and running. This is a persistent challenge of sustainability assessment in product development [33,34], which will have to be managed when applying the indicators for evaluating product systems.

Two respondents expressed doubt that the 13 selected indicators will adequately cover the project outcome's social impact. This emphasizes the need for keeping an open mind towards social impacts not covered by the indicators. The sustainability assessment practitioner has to be particularly cautious when it comes to coverage in case there is a stage in the product life cycle which none of the participants' is well-acquainted with.

Following the second round of rating, the average standard deviation for the team's average ratings decreased from 2.98 to 2.78 for the general importance, and from 2.04 to 1.94 for the relevance. Although a moderate change, this indicates an increased consensus following the second round of the exercise.

From follow-up talks with the participants, it turned out there were four different interpretations of the relevance rating: (i) relevance with respect to what can be influenced in the product development process, (ii) relevance for the specific processes under development, (iii) relevance for the entire product life cycle and (iv) relevance for comparisons with conventional products with a comparable functional unit. The different interpretations emphasize the importance of being clear and precise in explaining the exercise and to keep in mind that all participants may not be used to life cycle thinking.

The main criticisms of the Delphi method, i.e. its shortcomings as a forecasting technique and its resource inefficiency [15], are not considered serious drawbacks of the method as applied in this study: the tool is not used for forecasting, the exercise is limited to one round of feedback and the communication, except for the oral presentation, is managed via e-mail (which, in comparison to physical meetings, tend to be inexpensive and time efficient).

5 Conclusions

This paper reports on a tool for selecting critical social sustainability indicators early in the product development process. The following list summarizes the major outcomes and conclusions from application in an on-going product development project.

- Thirteen indicators were selected based on the result of the exercise. The suitability of these for guiding the product development process remains an open question to be addressed later on as the project evolves.
- The tool enhanced the product development team's understanding of social sustainability. Hopefully this will facilitate data collection and increase the assessment's influence on decision-making.

- The tool was practical to implement and proved time and resource efficient, valuable characteristics of sustainability assessment tools in product development processes.
- The ratings produced in the exercise are not to be seen as a final verdict on the importance of different social parameters. Also, an open mind must be kept towards social impacts not covered by the indicators.
- The tool is considered particularly useful as long as no standard sets of indicators exist for measuring the social impact of product life cycles. Even when such sets exist, the tool can be useful in selecting the most relevant indicators for the specific case.
- The tool's main drawback is the result's subjective nature. However, some degree of bias is inevitable in the art of assessing the social impact of a product, and when subjective group judgement is required the Delphi method is considered an appropriate instrument to capture the opinions.

References

[1] Ny H, Hallstedt S, Robért KH, Broman G (2008) Introducing templates for sustainable product development: a case study of televisions at the Matsushita Electric Group. J Ind Ecol 12:600-623.
[2] Maxwell D (2006) Enabling sustainable development through sustainable consumption and production. Int J Environ Sustainable Dev 5:221-239.
[3] Guinée JB, Heijungs R, Huppes G, Zamangi A, Masoni P, Buonamici R, Ekvall T, Rydberg T (2011) Life cycle assessment: past, present and future. Environ Sci Technol 45:90-96.
[4] Labuschagne C, Brent AC (2006) Social indicators for sustainable project and technology life cycle management in the process industry. Int J Life Cycle Assess 11:3-15.
[5] Kruse SA, Flysjö A, Kasperczyk N, Scholz AJ (2008) Socioeconomic indicators as a complement to life cycle assessment—an application to salmon production systems. Int J Life Cycle Assess 14:8-18.
[6] Benoit C, Mazijn B (2009) Guidelines for social life cycle assessment of products, UNEP/SETAC Life Cycle Inititative, Paris
[7] Jörgensen A, Le Bocq A, Nazarkina L, Hauschild M (2008) Methodologies for social life cycle assessment. Int J Life Cycle Assess 13:96-103.
[8] Hunkeler D (2006) Societal LCA methodology and case study. Int J Life Cycle Assess 11:371-382.
[9] Benoît C, Norris GA, Valdivia S, Ciroth A, Moberg A, Bos U, Prakash S, Ugaya C, Beck T (2010) The guidelines for social life cycle assessment of products: just in time! Int J Life Cycle Assess 15:156-163.

[10] Klöpffer W, Renner I (2009) Life-cycle based sustainability assessment of products. In: Schaltegger S, Bennett M, Burritt RL, Jasch C (eds) Environmental Management Accounting for Cleaner Production. Springer, Dordrecht
[11] Lundie S, Peters G, Ashbolt N, Lai E, Livingston D (2006) A sustainability framework for the Australian water industry. J Aust Water Assoc 33:83-88
[12] Dreyer LM, Hauschild MZ, Schierbeck J (2006) A framework for social life cycle impact assessment. Int J Life Cycle Assess 11:88-97.
[13] Dreyer LM, Hauschild MZ, Shierbeck J (2010) Characterisation of social impacts in LCA Part 1: Development of indicators for labour rights. Int J Life Cycle Assess 15:247-259.
[14] Spillemaeckers S, Vanhoutte G, Taverniers L, Lavrysen, L, Van Braeckel D, Mazijn B, Rivera JD (2004) Ecological, social and economical aspects of integrated product policy. Integrated product assessment: the development of the label 'sustainable development' for products. Belgian Science Policy, Brussels
[15] Rowe G, Wright G, Bolger F (1991) Delphi: a reevaluation of research and theory. Technol Forecast Soc Change 39:235-251.
[16] Linstone HA, Turoff M (eds.) (1975) The Delphi method: techniques and applications. Addison Wesley, Reading
[17] Linstone HA (1978) The Delphi Technique. In: Fowles J (ed): Handbook of futures research, Greenwood Press, Santa Barbara
[18] Miller G (2001) The development of indicators for sustainable tourism: results of a Delphi survey of tourism researchers. Tourism Manage 22:351-362.
[19] Hugé J, Le Trinh H, Hoang Hai P, Kuilman J, Hens L (2009) Sustainability indicators for clean development mechanism projects in Vietnam. Environ Dev Sustainability 12:561-571.
[20] Gallego Carrera D, Mack A (2010) Sustainability assessment of energy technologies via social indicators: results of a survey among European energy experts. Energy Policy 38:1030-1039.
[21] UNEP/LCA Life Cycle Initiative, Methodological sheets for 31 sub-categories of impact, <http://lcinitiative.unep.fr/default.asp?site=lcinit&page_id=A8992620-AAAD-4B81-9BAC-A72AEA281CB9> (accessed 18.08.2010)
[22] Krajnc D, Glavic P (2003) Indicators of sustainable production. Clean Technol Technol Environ Policy 5:279-288.
[23] Veleva V, Ellenbecker M (2001) Indicators of sustainable production: framework and methodology. J Cleaner Prod 9:519-549.
[24] Global Reporting Initiative, Sustainability reporting guidelines, version 3.0, 2006, <http://www.globalreporting.org/ReportingFramework/ReportingFrameworkDownloads/> (accessed 18.08.2010)
[25] Blackburn WR (2007) The sustainability handbook: the complete management guide to achieving social, economic and environmental responsibility. Earthscan, Oxford

[26] Manhart A, Grieβhammer R (2006) Social impacts of the production of notebook PCs: contribution to the development of a Product Sustainability Assessment (PROSA). Öko-Institut eV, Freiburg
[27] UCW (Understanding Children's Work), <http://www.ucw-project.org/> (accessed 20.08.2010)
[28] Dreyer LM, Hauschild MZ, Shierbeck J (2010) Characterisation of social impacts in LCA Part 2: implementation in six company case studies. Int J Life Cycle Assess 15:385-402.
[29] Schmidt I, Meurer M, Saling P, Kicherer A, Reuter W, Gensch CO (2005) SEEbalance ®: managing sustainability of products and processes with the socio-eco-efficiency analysis by BASF. Greener Manage Int 45:79–94
[30] Grieβhammer R, Benoît C, Dreyer LC, Flysjö A, Manhart A, Mazjin B, Méthot AL, Weidema B (2006) Feasibility study: integration of social aspects into LCA. Discussion paper from UNEP/SETAC Task Force Integration of Social Aspects in LCA
[31] ILO (International Labour Organisation), <http://www.ilo.org/> (accessed 20.08.2010)
[32] Azapagic A, Perdan S (2000) Indicators of sustainable development for industry: a general framework. Trans IChemE 78:243-261
[33] Baumann H, Tillman AM (2004) The hitch hiker's guide to LCA: an orientation in life cycle assessment methodology and application. Studentlitteratur, Lund
[34] Clancy G, Fröling M, Peters G, Svanström M (2010) Environmental challenges when developing renewable materials to replace non-renewable materials − receiving guidance from LCA studies. The 9th International Conference on EcoBalance 2010, Tokyo

A Life Cycle Stakeholder Management Framework for Enhanced Collaboration Between Stakeholders with Competing Interests

Christina Scandelius and Geraldine Cohen

Abstract Implementation of a life cycle sustainability management (LCSM) strategy can involve significant challenges because of competing or conflicting objectives between stakeholders. These differences may, if not identified and managed, hinder successful adoption of sustainability initiatives. This article proposes a conceptual framework for stakeholder management in a LCSM context. The framework identifies the key sustainability stakeholder groups and suggests strategic ambiguity as a management tool to harness dysfunctional conflict into constructive collaboration. The framework is of practical value as it can be used as a guideline by managers who wish to improve collaboration with stakeholders along the supply chain. The article also fills a gap in the academic literature where there is only limited research on sustainability stakeholder management through strategic ambiguity.

1 Introduction

Implementation of a life cycle sustainability management (LCSM) strategy can involve significant challenges because of competing or conflicting objectives between stakeholders. These differences may, if not identified and managed, hinder successful adoption of sustainability initiatives, which may not only harm a business's performance [1], but may also delay development to more sustainable production and consumption practices. Based on the definitions for life cycle management [2] and sustainable development [3], we suggest the following definition for life cycle sustainability management (LCSM): A strategic management system which aims at minimising an organisation's negative impact on the natural and social environment by its products or services along the entire product/service life cycle and value chain, to warrant that natural, social and economical resources are sustained for future generations.

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This definition calls for close collaboration between stakeholders along the value chain, and also interaction with stakeholders who represent the natural and social environment. Organisations will need to form relationships with a wide range of different stakeholders, and if the optimal route towards sustainable production and consumption is unclear, these relationships might be jeopardised by conflicting ideas on how to best use resources.

In the next sections we will critically review the academic literature on sustainability stakeholders, sources for tension between these stakeholders, and strategic ambiguity as a suggested tool to manage diverse stakeholders. A case study will explore how ambiguous communication could be used in a LCSM context. Finally, we will present a conceptual framework suggesting how to reduce tension between sustainability stakeholders and encourage collaboration.

2 Identification of sustainability stakeholders

The previous section introduced us to LCSM and the importance of collaboration between stakeholders to ensure sustainable development of products and services along the whole life cycle and value chain. It is therefore vital for a business to understand which stakeholder groups that have legitimacy, urgency and power [4] to collaborate in a LCSM context. A range of scholars have, based on Freeman's stakeholder theory [5], attempted to identify 'green stakeholders'. Early examples include Henriques and Sadorsky [6, 7] with their four groups of environmental or green stakeholders: regulators, organisational (including customers, suppliers, employees and shareholders), media and local communities. Fineman and Clarke [8] choose to rather name the green stakeholder groups as planet, regulatory, stakeholders with indirect interests, and internal stakeholders. In essence these groups of stakeholders are the same as the ones introduced by Henriques and Sadorsky [6,7], but they have chosen to categorise them differently and use slightly different names. For example, they [8] introduce ‘planet’ as a separate stakeholder, represented by environmental pressure groups and they also suggest the term 'stakeholders with indirect interest' to classify those stakeholders that consider environmental progress as a secondary benefit, and whose priorities are other aspects. This group includes a wide range of heterogeneous stakeholders like shareholders, customers and media. Harvey and Schaefer [9] refer to the green stakeholders as suggested by Fineman and Clarke [8] but found evidence that this classification is rather academic as many companies do not take a systematic approach to categorise and manage green stakeholders. A more recent attempt to identify green stakeholders was conducted by Rivera-Camino [10] who suggests

that there are four major green stakeholder groups: market (including customers, distributors and suppliers), social stakeholders (e.g. local communities), immediate providers (shareholders, labour unions etc) and legal stakeholders.
This section has identified the key green stakeholder groups. To better reflect the definition of LCSM, also the social aspect of sustainability should be considered and addressed more clearly, and a more appropriate term would be sustainability stakeholders. The next section will address sources for competing interests between and within these groups.

3 Sources for tension between stakeholders

The sheer number of sustainability stakeholders, as identified in the previous section, indicates that tension or even conflict is likely to arise as an effect of sustainability initiatives or lack of initiatives, or as phrased by Reynolds et al [11]: "*Whether the resources are capital, profits, effort, or time, stakeholders can and do disagree about how or where each should be utilized"*.
In addition, there are also some sources to tension specifically applicable to sustainability initiatives. Firstly, several unclear definitions of sustainability exist [12, 13] which may lead to different perspectives on the best cause of action, both internally between departments in an organisation, as well as between different organisations [14]. In addition, despite many efforts by academics and practitioners to develop processes to assess a product's impact on the social and natural environment along its life span, there is still no conclusive method available [15]. Another source to tension could be the lack of sufficient evidence that sustainability initiatives are beneficial for all stakeholders of a business [16], which may lead to resistance from certain stakeholder groups. These factors indicate that the various sustainability stakeholders are likely to have conflicting ideas on what actions need to be taken in order to achieve progress towards sustainability.

4 Strategic ambiguity as a tool to manage stakeholders

The previous section introduced us to the complex and diverse stakeholder situation that characterises LCSM. How can a business harness the possible conflicting demands and encourage constructive collaboration between the stakeholders? Davenport and Leitch [17] suggest that while clarity is important in communication when the goal is clear, they argue that "......*when the goal is less* *clear, when stakeholders are not compliant and, perhaps, have power bases from which to resist the goal, or when achievement of the goal requires a creative engagement between the organization and its stakeholders, strategic ambiguity may be more appropriate".*

One of the pioneers to introduce strategic ambiguity into the business communication literature was Eisenberg, in his seminal article of 1984 [18]. He argues that while clarity is an important aspect of communications, it might be more pragmatic to refrain from being too specific in situations where multiple contradicting goals exist. He proposes strategic ambiguity, the intentional use of ambiguous messages, as a management tool for businesses to achieve endorsement from their stakeholders. This is also supported by more recent research; for example Davenport and Leitch [16] define strategic ambiguity as:

"*...the deliberate use of ambiguity in strategic communication in order to create a 'space' in which multiple interpretations by stakeholders are enabled and to which multiple stakeholder responses are possible*".

Eisenberg's conceptual theories [18], have been empirically tested in contexts of organisational change in the public sector [17,19,20], in crisis communication in a fast food chain [1], and in the context of social marketing in a non-profit organisation [21]. Sustainability is mentioned in relation to strategic ambiguity in the context of policy documentation [12], where 'sustainability' and 'growth' were used as ambiguous keywords to facilitate collaboration between stakeholders of conflicting ideological beliefs. In addition Wexler [22] presents a conceptual model suggesting that strategic ambiguity can be beneficial to encourage collaboration between three stakeholder groups (people, profit and planet), around the ambiguous concept of the 'triple bottom line' (TBL). Wexler's work is however reducing stakeholders into these three vaguely defined groups and is limited to one reporting tool of sustainability, the TBL, and he calls for further research into the applicability of strategic ambiguity beyond an intra-organisational context.

The literature review above indicates that strategic ambiguity can be beneficial to reduce tension and encourage collaboration with stakeholders in contexts characterised by diverse stakeholders and unclear goals. These contexts share similarities to circumstances around implementation of a LCSM strategy, and lead us to believe that strategic ambiguity might be a useful tool in managing sustainability stakeholders. The next section will therefore exemplify how strategic ambiguity can be used to communicate with sustainability stakeholders through a case study.

5 Case study on strategic ambiguity to manage sustainability stakeholders around coffee brands

This section will illustrate how strategic ambiguity can be used in practice in LCSM, by studying examples of ambiguous sustainability communications to stakeholders around a coffee brand, with a specific focus on packaging. It should be noted that the research is still on-going and that the section below will present some initial observations.

5.1 Background and methodology

The chosen case study relates to Kraft Foods, and more specifically their coffee brand Kenco. Coffee brands are interesting from a sustainability perspective as the coffee value chain can have a significant impact on the social and natural environment, through its origin in regions characterised by economical poverty, and its significant water and energy consumption. Kenco coffee is of particular interest as it is positioned as a sustainability brand, and is projecting its sustainability ethos on its packaging design. Packaging can in itself become a source for stakeholder conflict through the multifaceted functions that it needs to fulfil. It needs to ensure sufficient protection and preservation of the goods the packaging is intended for, and decisions on packaging design will not only have an impact on sustainability for the packaging itself but will determine transportation efficiency and waste reduction also for the product it protects [23]. It is therefore paramount to adopt a holistic approach with close collaboration with stakeholders along the supply chain to ensure minimum adverse environmental impact along the life cycle of both product and packaging. In addition packaging serves as a promotional platform as it is an important tool to attract consumers and to support brand image [24]. Last but not least, because packaging has become a very visible symbol for waste, there is significant pressure for sustainability efforts from various external stakeholder groups. Managing stakeholders around a coffee brand is thus likely to be challenging, balancing very diverse stakeholders, requiring collaboration to achieve improved conditions. Therefore sustainability communications around a coffee brand should pose a very interesting context for research into strategic ambiguity.

The results presented below are initial observations from on-going exploratory research, applying case study design to explore the concept of strategic ambiguity in the management of stakeholders around the food and beverage industry. As there is very limited research available on strategic ambiguity in LCSM, a case study design is used combining qualitative methods such as interviews and documentary data collection to explore formal internal and external communication in depth [25]. Qualitative interviews are chosen as a suitable method in this case study design, as it provides more flexibility to follow up on new insights in areas that is not well known and it also provides more detailed answers compared to quantitative research [26]. The findings presented below are based on two in-depth semi-structured interviews, each lasting up to 2 hours, with directors of The British Packaging Federation. The interviewees were selected following their long experience (more than 30 years) in supplying packaging to the food and beverage industry, having held positions within manufacturing management and as CEOs. In addition, secondary data was collected from publicly available reports, web sites, newspapers and journal articles.

5.2 Discussion on ambiguous sustainability communications around a coffee brand

Kenco (part of Kraft Foods) is positioned as a sustainable coffee brand, announcing their social and environmental responsibility through sourcing coffee beans only from rainforest alliance certified farms, and taking initiatives to reduce packaging waste. Communications on their website and on TV ads, targeted towards consumers and natural environment stakeholders are primarily focused around their ‘eco-refill’ packaging which is claimed to reduce 97% waste compared to the traditional glass jars [27]. From a weight perspective they have reduced waste, however the eco-refill pouch is not easily recyclable and can currently not be recycled by local authorities in the UK. Instead they encourage consumers to take action to recycle by sending empty eco-refill pouches to TerraCycle, an organisation who will convert the empty bags into products like umbrellas and notebooks. The shift to the eco-refill bag is significant from a promotional aspect. In an industry where the majority of instant coffee brands are sold in glass jars, the eco-refill bag stands out on the shop shelf. Kenco has here been able to cleverly meet demands from a promotional aspect, i. e. packaging that gains attention, and also supports the sustainability image of Kenco, as the refill pouch signifies waste reduction through its size compared to the glass jars. While Kenco promote their achievements of reducing waste, other aspects of sustainability challenges, like energy and water consumption along the coffee value chain are played down or left out. This stance is interesting as one of their competitors, Nescafe (part of Nestle) take a different focus in sustainability communications. While Nescafe also offer coffee in pouches, the majority is sold in glass jars and they highlight the advantage of using glass as a natural material that is easy to recycle [28].

In communications to economic stakeholders (e.g. investors), the angle is different and sustainability is mentioned in connection with economic growth:

"*To build and sustain brands people love and trust, one must focus-not only on today but also on tomorrow. It's not easy...but balancing the short and long term is key to delivering sustainable, profitable growth-growth that is good for our shareholders but also good for our consumers, our employees, our business partners, the communities where we live and work, and the planet we inhabit"* [29].

The interviews [30,31] reveal that sustainability communications from the major consumer brands to the packaging suppliers are focusing on keywords as 'cost efficiency', using less material and/or less energy to achieve cost savings which also has a secondary benefit from an environmental perspective. Environmental initiatives, in the form of reduced packaging or more environmentally friendly processes, are considered very attractive and seen as added value and might secure a supply position, however these initiatives are expected to be sponsored by the packaging suppliers. Cost and quality dimensions will often take precedence over sustainability. It was mentioned that one underlying cause for sustainability related tension in the value chain is the often uneducated debate on waste, where the packaging industry is blamed because of the visibility of packaging [30,31]. It was highlighted that statistics suggest that as much as 30% of food is wasted in the UK, and that some of this could be reduced by more clever packaging solutions [30]. It might however pose as less attractive for some members in the value chain to address the waste problem, as less waste means less consumption and less revenue. Or, as expressed by one of the interviewees [31]:

"*The industry, at the moment, is dominated by 'greenwash', and I mean again, if you stand back, one of the biggest fundamental issues we have is that most people don't begin to understand what packaging is and what it does"*.

He further points out that:

"R*ather than educate the public, the retailers will say: We are doing the right thing - we are reducing the packaging"* [31].

Therefore, in the absence of regulations controlling food waste, the big consumer brands and retailers will continue to push their suppliers for primarily cost reducing solutions"[30, 31].

The initial observations from this case study indicate existence of competing interests in the value chain around balancing cost, quality, revenue and sustainability initiatives. The findings further suggest that strategic ambiguity is practised in sustainability communications, applying keywords of different meanings depending on the stakeholders being targeted. Further research is suggested to explore the objectives behind the choice of keywords and decisions to play down certain aspects, and also to better understand how the message is interpreted by the various stakeholder groups [18] and how it may encourage them to collaboration.

6 Presentation of a conceptual framework to manage sustainability stakeholders in a LCSM context

The previous sections have identified the key sustainability stakeholder groups, highlighting sources for conflict between these groups, and suggested strategic ambiguity as a tool to manage diverse stakeholders. Based on this and the findings from the case study we propose a conceptual framework for managing sustainability stakeholders in a LCSM context (Figure 1). This framework builds on the green stakeholder models suggested by Henriques and Sadorsky [7], Fineman and Clarke [8] and Rivera-Camino [10], however as LCSM is considering the natural and social environment as two equally important constructs, this will be reflected by classifying them as two distinct groups, and by using the term 'sustainability stakeholders'. The framework identifies five key sustainability stakeholder groups: economic stakeholders, social stakeholders, natural environment stakeholders, value chain stakeholders and internal stakeholders. The natural environment is represented by green pressure groups, regulators and legislators who pursue a green agenda, the public, media etc. The social stakeholder group is represented by social pressure groups that may or may not be the same as for natural environment, regulators and legislators from a social perspective, media and the public. In addition, as opposed to Rivera-Camino's [10] suggestion to include labour unions in 'immediate providers', the framework here suggests that labour unions belong to the social stakeholder group, as their function is to ensure its members welfare. The economic stakeholders are those who have an economic interest in the business, for example shareholders, financial institutions etc. Additionally, we add internal stakeholders to the model, as suggested by Fineman and Clarke [8]. The market stakeholders in Rivera-Camino’s model [10] are here named as ‘value chain stakeholders' to better reflect the participants in this group, which are customers, suppliers, distributors, etc.

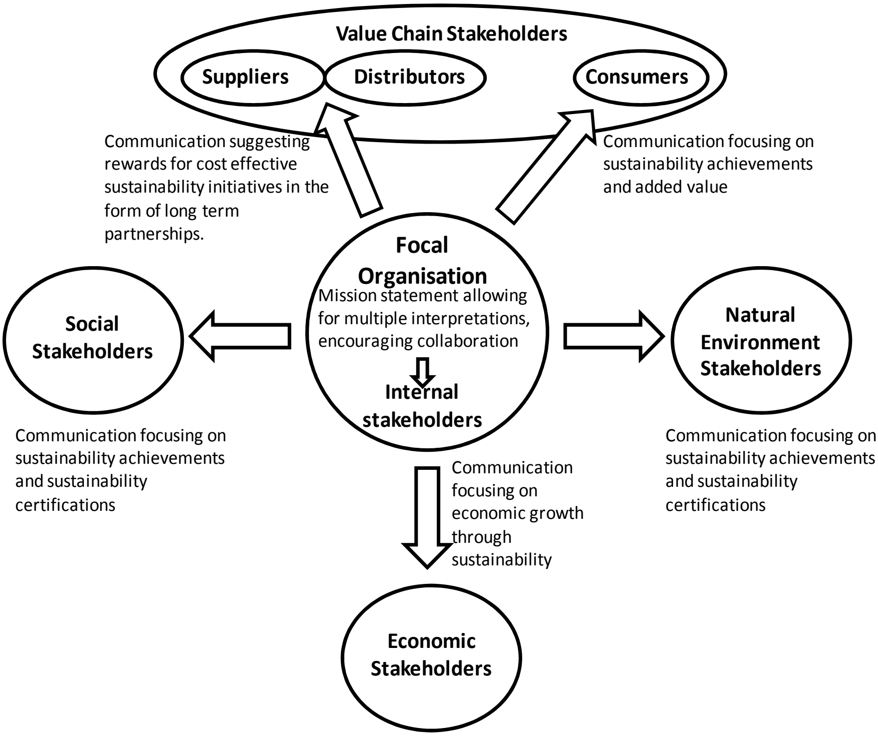

Fig. 1: Framework for sustainability stakeholder management in a LCSM context

The framework illustrates how an organisation can categorise its stakeholders according to the nature of their interests, and adapt the sustainability communications accordingly. The message may contain mission statements, keywords or metaphors that allows for multiple interpretations [12,18,19], it may leave out or play down certain aspects to create common ground [21] or to avoid confusion or indecision [17,20]. The case study also reveals that different keywords may be used to different stakeholder groups, with the purpose of stimulating a positive response to collaboration.

While many scholars suggest prioritisation of stakeholders [6,7], we suggest to rather consider them all as equally important as the power may change over time, and for a long term perspective it is therefore important to avoid giving some groups precedence over others [4,32,33].

7 Conclusions

This conceptual paper has highlighted stakeholder related challenges facing organisations implementing a LCSM strategy. The most relevant stakeholders, named sustainability stakeholders, were identified and the sources for possible tension were explained. A literature review on strategic ambiguity presented evidence on its usefulness to reduce tension and encourage collaboration in contexts characterised by diverse stakeholders and/or goals. The applicability of strategic ambiguity in a LCSM context was illustrated by a case study, giving examples of ambiguous sustainability communications around a coffee brand. Finally a conceptual framework for stakeholder management in a LCSM context was presented. The framework is of practical value as it can be used as a guideline by managers who wish to improve collaboration with stakeholders along the supply chain to optimise sustainability efforts. The article also fills a gap in the academic literature where there is only limited research on stakeholder management and strategic ambiguity in a sustainability context. Further empirical research is however suggested, to verify the validity of the proposed framework.

References

[1] Ulmer RR, Sellnow TL (2000) Consistent Question of Ambiguity in Organizational Crisis communication: Jack in the Box as a Case Study. J Bus Ethics 25:143-155.

[2] UNEP (United Nations Environmental Programme), 2007. Life Cycle Management – A Business Guide to Sustainability. <http://www.lcinitiative.unep.fr> (accessed 15.05.2009)

[3] World Commission on Environment and Development (1987) Our Common Future. Oxford University Press, Oxford

[4] Mitchell R, Agle B, Wood D (1987) Toward a theory of stakeholder identification and salience: defining the principle of who and what really counts. Acad Manage Review 22:853-86.

[5] Freeman ER (1984) Strategic Management: A Stakeholder Approach. Pitman/Ballinger, Boston

[6] Henriques I, Sadorsky P (1996) The determinants of an environmentally responsive firm: an empirical approach. J Environ Econ Manage 30:381-395.

[7] Henriques I, Sadorsky P (1999) The relationship between environmental commitment and managerial perceptions of stakeholder importance. Acad Manage J 42:87-99.

[8] Fineman S, Clarke K (1996) Green stakeholders: industry interpretations and response. J Manage Stud 33:715-730.

[9] Harvey B, Schaefer A (2001) Managing Relationships with Environmental Stakeholders: A Study of U.K. Water and Electricity Utilities. J Bus Ethics 30:243–260.
[10] Rivera-Camino J (2007) Re-evaluating green marketing strategy: a stakeholder perspective. Eur J Marketing 41:1328-1358.
[11] Reynolds SJ, Schultz FC, Hekman DR (2006) Stakeholder Theory and Managerial Decision-Making: Constraints and Implications of Balancing Stakeholder Interests. J Bus Ethics 64:285–301.
[12] Leitch S, Davenport S (2007) Strategic ambiguity as a discourse practice: the role of keywords in the discourse on sustainable biotechnology. Discourse Studies 9:43-61
[13] Smith P, Sharicz C (2011) The shift needed for Sustainability. Learn Organiz 18:73-86.
[14] Rex E, Baumann H (2007) Individual adaptation of industry LCA practice: Results from two case studies in the Swedish forest products industry. Int J Life Cycle Assess 12:266-271.
[15] Hunkeler D, Rebitzer G (2005) The Future of Life Cycle Assessment. Int J Life Cycle Assess 10:305–308.
[16] Spar DL, La Mure L (2003) The Power of Activism: Assessing the impact of NGOs on global business. California Manage Review 45:78-102.
[17] Davenport S, Leitch S (2005) Circuits of power in practice: Strategic Ambiguity as delegation of authority. Organiz Stud 26:1603-1623.
[18] Eisenberg EM (1984) Ambiguity as strategy in organizational communication. Commun Monogr 51:227-242.
[19] Leitch S, Davenport S (2002) Strategic ambiguity in communicating public sector change. J Commun Manage 7:129-139.
[20] Denis JL, Dompierre G, Langley A, Rouleau L (2011) Escalating Indecision: Between Reification and Strategic Ambiguity. Organiz Sci 22:225–244.
[21] Dickinson-Delaporte S, Beverland M, Lindgreen A (2010) Building Corporate reputation with stakeholders – Exploring the role of message ambiguity for social marketers. Eur J Marketing 44:1856-1874
[22] Wexler MN (2009) Strategic ambiguity in emergent coalitions: the triple bottom line. Corp Commun Int J 14:62 – 77
[23] Hopkins M (2010) How Sustainability Creates New Opportunity. MIT Sloan Manage Rev 51:65-69.
[24] Underwood R, Ozanne J (1998) Is your packaging an effective communicator? A normative framework for increasing the communicative competence of packaging. J Marketing Commun 4:207-220
[25] Yin RK (1994) Case Study Research, Design and Methods. Sage Publications, Newbury Park
[26] Bryman A, Bell E (2007) Business Research Methods. Oxford University Press, Oxford
[27] <http://www.kenco.co.uk> (accessed 28.01.2011)
[28] <http://www.nescafe.com> (accessed 01.04.2011)

[29] Kraft Food Responsibility Report 2010, <http://www.kraftfoodscompany.com> (accessed 01.04.2011)
[30] Interviewee A, Director of the British Packaging Federation, Interviewed in London on 15.09.2010
[31] Interviewee B, Director of the British Packaging Federation, Interviewed in London on 21.09.2010
[32] Polonsky MJ (1996) Stakeholder management and the stakeholder matrix: potential strategic marketing tools. J Market-Focused Manage 1:209-229
[33] Polonsky MJ, Scott D (2005) An empirical examination of the stakeholder strategy matrix. Eur J Marketing 39:1199-1215.

Stakeholder Consultation: What do Decision Makers in Public Policy and Industry Want to Know Regarding Abiotic Resource Use?

Marisa Vieira, Per Storm and Mark Goedkoop

Abstract There is no agreement on what the issue of concern is regarding resource use. A stakeholder consultation was carried out in order to clarify this issue. The objective was to identify decision contexts in which stakeholders would use an indicator related to resource use, and what such indicator should express. Industry representatives were interested in the short term economic consequences of depleting resources whereas policy makers were more concerned with the robustness and reliability of the indicator over a longer time horizon. Some of the aspects the indicator should cover include availability, effort increase, substitution, and societal value. The stakeholder consultation resulted in the selection of three indicators for mineral resources and two for fossil using different time horizons; the short term perspective prioritises political constraints, the midterm focuses on the increase in effort while the long term focuses on overall availability.

1 Introduction

Lack of consensus on what society understands as resource depletion is a major dilemma in life cycle assessment (LCA). There are several impact assessment methods assessing resource use [1] and yet Berger and Finkbeiner [2] demonstrated lack of correlation between them. This occurs because each method expresses a different indicator which relates to a different problem. Therefore, further clarity on the issues of concern regarding the use of abiotic resources is needed to develop a harmonised life cycle impact assessment (LCIA) method on resource depletion.

The first step towards developing a method is to understand what are the problems associated with extracting and depleting a resource. In order to gain insight into this question a stakeholder consultation was organised involving several groups of people who could be potentially affected by problems related to resource

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availability. The objective was to identify what kind of indicators would be really useful in their decision making process. This research is part of the project LC-IMPACT which is funded by the European Union FP7 programme.

2 Stakeholder consultation

A stakeholder consultation process was carried out in order to clarify the area of protection (AoP) on resource use. During this process interested parties were invited to share their views on the topic. This included a one-day workshop as well as individual interviews. The workshop took place on October 4, 2010 in Brussels. Interviews were conducted with industry and policy makers that were underrepresented in the workshop. These interviews also aimed at a bigger coverage of fossil resources expertise.

2.1 Participants

Industry representatives, policy makers, and resource experts were invited to participate in the stakeholder consultation process. Industry and policy attendees will be the users of the indicator, and provide information about the decision contexts in which they would use a resource depletion indicator; whereas experts have the knowledge to inform whether a particular indicator is feasible and if data for the different relevant aspects is available. The list of attendants of the stakeholder consultation process is presented in Table 1.

Tab. 1: Participants of stakeholder consultation process on the AoP resource.

Experts	Industry	Policy makers
ASPO International	European Aluminium Foil Association	EC DG Enterprise and Industry
Energy research Centre of the Netherlands	European Association of Mining Industries	EC Joint Research Centre
European Institute for Energy Research	Philips	French Ministry of Environment
Ghent University	Shell	
Lulea University of Technology	Umicore	
Radboud University of Nijmegen		
Raw Materials Group		
Uppsala University		

2.2 Decision contexts and type of indicator

In order to decide what type of indicator is needed for resource depletion, an inventory of situations where a stakeholder would use such an indicator was made. An exercise was performed with the stakeholders by asking them to share with us "their story". Each attendant was asked to imagine different situations in which they would use certain indicators and what they would like the indicator to express. This was achieved by answering the following questions:

- As a ____
- In order to ____
- I need an indicator that expresses ___________________________

Figure 1 shows an example of the exercise.

> *"As a corporate sustainability manager, in order to make informed decisions on current and future operations in the company, I need an indicator that expresses the costs of choosing a certain material."*

Fig. 1: Example of a post-it used for the inventory of decision contexts.

Table 2 shows the different decision contexts that were identified.

Tab. 2: Results of decision contexts.

As a:	In order to:
Consumer	- Inform on purchasing choices
Policy maker	- Foresee long-term issues regarding resource depletion - Boost innovation
Industry	- Know the impact of the corporation's operations in resource scarcity - Make informed decisions on existing or future operations - Have resources available at reasonable price

Policy makers expressed their interest to use an indicator that can support new regulations and foster innovation; whereas industry's concern is related to assuring the corporation's profit.

The main findings regarding the type of indicator were:

- **Industry** focuses on the economic consequences of extracting additional resources with a time frame of 5-10 years. This is the average time for a return on investment

- **Policy** focuses on the robustness and reliability of an indicator with a time frame of approximately 50 years. This is because the data quality and data source will be questioned.

3 Consultation on aspects to include in the indicator

In order to steer the methodology development, it is vital to identify which aspects play a key role on resource use. Consequently, in addition to the stakeholder consultation on decision contexts and type of indicator, we took the opportunity to ask experts to identify which important aspects should be taken into account when developing a method for resource use. The following aspects were identified:

- Availability
- Economic/population growth
- Increase on efforts (energy or cost)
- Historic data
- Recycling
- Substitution
- Supply risk
- Technological improvement (both supply and demand - mining technologies and emerging use of resources, respectively)
- "Value" for society

The next step involves verifying whether data is available for each of these aspects with respect to both mineral and fossil use.

3.1 Mineral resources

The following aspects were identified for mineral resources:

- **Costs** determine material choices.
- **Substitution** can be made for some metals but is not viewed as very relevant for this type of resource.
- Changes in demand due to **economic growth** and emerging technologies are relevant although quite difficult to model.
- Most metals can be recycled**.** However, because recycling is already included in the life cycle inventory stage (LCI), we should be cautious not to double count this aspect by also including it in the LCIA phase.

- **Availability** is considered an important issue, especially taking a long-term perspective. However, considering the average crust availability (as in the CML method [3]) is not eligible because not all crust is nor will be accessible for extraction.
- In order to assess future resource availability/scarcity, **historic** data should be analysed.
- **Global or European** scope is needed.

3.2 Fossil resources

The following aspects were identified for fossil resources:

- It is questionable whether an indicator that expresses fossil resource depletion is useful at all because **climate change** is driving substitution of fossil fuels by alternative sources of energy that cause lower greenhouse gas emissions. In fact stocks are not the limiting factor, climate change is. On the other hand it is seen as very useful to show policy makers what the status on depletion actually is.
- **Energy** requirements are less sensitive to political discussion than costs and besides energy is a big part of the extraction costs.
- Most fossil resources are used for energy production. As of late there are many substitution options for fossil fuels for energy production, such as biofuels or one fossil type for another, e.g. coal to liquids. For this reason, **substitution** is considered a relevant aspect for fossil resources.
- Only a small portion of the oil extracted is used as input for plastic production and the **recyclability** of plastics is also reduced, therefore this is not foreseen as an important aspect.
- **Technology development** for extraction is not perceived as a problem.
- With the **economic growth** of some emerging countries, the use of energy is increasing dramatically so this aspect should be included.
- In order to assess future resource availability/scarcity, **historic** data should be analysed.
- **Global or European** scope is needed.

4 Post consultation decisions

The stakeholder consultation resulted in the selection of three indicators for mineral resources using different time horizons; whereas for fossil resources, only two indicators were identified. The short term perspective prioritises political constraints, the midterm focuses on the increase in effort while the long term focuses on overall availability.

4.1 Mineral resource indicators

For **mineral** resources three indicators were identified for three different time horizons:

- For a short-term perspective (not greater than 5 years), an indicator that expresses availability of minerals depending on political factors.
- An indicator based on the increase in effort, expressed either in energy (as in Eco-indicator 99 [4] and IMPACT 2002+ [5]) or costs (as in the ReCiPe method [6]) as a response to a lower ore grade in a mid-term perspective (5 to 20 years). Recyclability, technology development (on both demand and supply) and economic growth should also be considered in this indicator.
- For a long-term perspective (at least 50 years), a simple resource availability indicator could be useful. It is important to include mining technological improvements and mineral reserves on this time scale.

4.2 Fossil resource indicators

Only two indicators were identified for **fossil** resources. Because mining technological developments for fossil fuels are very fast, no long-term availability problem is expected. Moreover, new regulations to reduce greenhouse gas emissions are increasingly being implemented. These will lead to increased substitution of fossil resources by alternative sources. The two indicators to be developed are:

- For short-term availability, political developments dominate so a resource availability indicator expressing political sensitivity is desired.
- A mid-term indicator that is based on the foreseen shifts in technologies for extracting fossil resources and using alternative sources. As these

alternative sources tend to consume more energy and are more expensive, the increase in effort is a good indicator.

5 Conclusions

By far most, if not all, life cycle impact assessment methods have been developed without prior consultation with stakeholders the method is developed for, the decision makers who want to integrate sustainability aspects in business decisions. This probably explains why the analysis presented in the ILCD handbook [1] found very large differences in approaches, scope, time frames, among others.
This project shows that stakeholders tend to be quite pragmatic and have a short term perspective. An important unresolved issue is whether researchers should simply accept the decision for a short term view, and develop a method according to specifications of these users. Should researchers take a more philosophic view that sustainability cannot be a short term issue?
We believe this project sets an example on how LCIA researchers can gain important insights in what decision makers would like a methodology to express, and that this should at least be considered in their work, since a method that does not address the relevant questions is likely not to be used.

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References

[1] European Commission, Joint Research Centre (2010) ILCD Handbook: Analysing of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment. European Union, Brussels
[2] Berger M, Finkbeiner M (2011) Correlation analysis of life cycle assessment indicators measuring resource use. Int J Life Cycle Assess 16:74-81
[3] Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, van Oers L, Wegener Sleeswijk A, Suh S, Udo de Haes HA, de Bruijn H, van Duin R, Huijbregts MAJ (2002) Handbook on life cycle assessment. Operational guide to the ISO standards. Kluwer Academic Publishers, Dordrecht

[4] Goedkoop M, Spriensma R (1999) The Eco-indicator 99 - A damage oriented method for Life Cycle Impact Assessment. VROM, Amsterdam

[5] Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G, Rosenbaum R (2003) IMPACT 2002+: A new Life Cycle Impact Assessment methodology. Int J Life Cycle Assess 10:324-330

[6] Goedkoop M, Heijungs R, Huijbregts M, De Schryver A, Struijs J, van Zelm R (2009) ReCiPe 2008: A Life Cycle Impact Assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. VROM, Amsterdam

Life Cycle Management Capability: An Alternative Approach to Sustainability Assessment

Thomas Swarr, James Fava, Allan Astrup Jensen, Sonia Valdivia and Bruce Vigon

Abstract There has been steady progress advancing life cycle assessment methods. However, application of LCA in business decision making has lagged. UNEP and SETAC are collaborating on development of a life cycle management capability maturity model to address this gap, particularly in small-to-medium sized enterprises (SME) with limited life cycle experience. The model provides a structured sequence of improvement actions that can speed organisational learning and deliver near- term business results. The framework also complements existing efforts to develop quantified sustainability performance measures by building the capacity of lower tier suppliers to make effective decisions based on their understanding of the local situation and according to their priorities. This should ensure the quality of the data provided as well as help further the development of sustainability indicators.

1 Introduction

There has been steady progress advancing life cycle assessment methods. UNEP published guidelines for social life cycle assessments and is currently gathering stakeholder feedback on methodological sheets for 31 subcategories of impacts [1,2]. The CALCAS project developed a blue paper on life cycle sustainability analysis [3]. There has been less progress on integrating life cycle methods into routine business decision making processes. A cursory literature search shows life cycle research has paid much less attention to implementation in business decisions compared to methodological development. Results presented in Table 1

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for a search on key terms “life cycle assessment” and either “methodology” or “business decision making” are illustrative.

Tab. 1: Results for literature search on key words (number of citations)

Database	**Life cycle assessment or LCA**	**AND methodology**	**AND business decision making**
ProQuest/ ABInform	7,824	1,184	17
EBSCO Bus. Premier	887	117	1
Science Direct	18,214	4,943	26

This paper builds on a life cycle management (LCM) capability maturity model presented at LCM 2007, which was developed to facilitate the integration of life cycle thinking into business making routines [4]. Capability models are well-established as effective tools for guiding business process improvement efforts and have been applied to numerous domains, such as software, systems engineering, product development, and personnel management [5]. The capability models can establish a common vision, help set priorities for action, guide efforts to tailor improvements to the specific needs of an organisation, and facilitate learning from the experiences of other organisations. However, the models tend to be very complex and inappropriate for lower tier SME suppliers in developing economies [6]. The LCM capability model focuses on decision making processes to develop a simpler and more practical tool for organisational assessments.

The capability approach is also a necessary complement to the multitude of sustainability performance indicators comprising various reporting, labelling and certification standards under development. There is no broad consensus on a specific set of indicators to characterise sustainability, and different regions will have different priorities that determine which indicators are relevant to that context [7,8]. Despite the noteworthy efforts of these initiatives (e.g. GRI G-3 Guidelines, ULe 880 Interim Sustainability Requirements, etc.) to engage a diverse range of stakeholders in developing the standards, it is simply not possible to reach out to all SME suppliers and their host communities. Their active participation as competent partners in evaluation and refinement of sustainability indicators is vital to ensure the continual improvement of these standards. It has been suggested that the most appropriate unit of analysis for sustainability assessments is the local region, because it is a small enough scale to be of direct interest to citizens, yet large enough to achieve a critical mass for creative solutions [8].

2 LCM capability maturity model

UNEP and SETAC launched a project to develop a capability model for use by small- to- medium sized enterprises (SME) with limited knowledge of and experience with LCM. The model is composed of 12 business processes grouped into three categories. Leadership processes set the direction for the organisation and determine if there is sufficient commitment and organisational support to achieve the desired goals. LCM processes provide the operational discipline to build, deliver, support and retire product offerings in a safe, clean, equitable and profitable manner. Enabling infrastructure processes ensure the necessary equipment, information, and people are in place over the long term. To provide a practical basis for organisational assessments, the capability framework is structured around decisions; information systems and metrics used to monitor and manage implementation; and integration of affected stakeholders into the decision process. An overview of the model is shown in Table 2.

Tab. 2: LCM capability maturity model

Maturity level	Decision process	Boundaries	Metrics
Qualified	Visible team-based trade offs	Project	Binary yes-no compliance; process outputs
Efficient	Rule-based trade-offs to achieve company goals	Enterprise	Process inputs/ outputs; eco- efficiency
Effective	Fact-based trade-offs to balance value chain goals	Value chain	Cradle to grave integrated across value chain
Adaptive	Value-based trade-offs to co-develop company goals & public expectations	Society	Sustainability, resiliency

The capability model provides a structured approach to gradually build the capability of organisations to manage more complex problems using more inclusive processes. Rather than specifying specific sustainability measures, the framework builds the capacity of actors to make effective decisions based on their understanding of the local situation and according to their values and priorities.

3 Sustainability self-assessments

A self-assessment protocol and supporting workbook were developed and tested with participating companies [9]. The protocol guides the organisation through an assessment of the 12 key business processes. Diagnostic questions and examples of indicative practices for each maturity level are provided. The supporting workbook provides additional descriptions of each process to help identify strengths of established routines that can be leveraged to speed integration of LCM considerations as well as targeting key gaps for improvement. Findings of the self- assessment can be used to define improvement projects that both meet the near term performance targets set by customers, investors, and other stakeholders while incrementally building the components of a comprehensive management system. An example of a completed assessment questionnaire for one of the 12 key processes is shown in Figure 1.

During the initial phase of the UNEP-SETAC project, some 40 individuals representing ~30 different organisations (companies, industry associations, universities, and non-governmental organisations) provided feedback on the capability assessment protocol and workbook. Language was raised as a potential barrier. Project participants strongly recommended that the assessment protocol and supporting guidance materials be translated into local languages to facilitate implementation. There was also concern about the extensive use of jargon and technical terminology in the project materials and many LCM resources referenced in the workbook. Even with translation, the same terms can mean different things in different cultures-particularly when discussing value-laden terms such as sustainability.

A more fundamental concern was that there was a tendency to treat the capability framework as a substitute for other sustainability performance measures rather than as a complementary tool to help achieve those measures. Feedback from the companies that piloted the assessment protocol reported it was difficult to avoid a compliance audit framing in discussions with management. Managers were focused on the score and specific actions needed to achieve the next higher level. The challenge of sustaining management commitment to transformational changes is similar to what was experienced by quality improvement efforts in the 1980's and 1990's [10,11]. To address these concerns, the companion workbook for the capability assessment was revised to more clearly align with the quality plan-do-check-act improvement cycle and to build on familiar supplier quality development programs and concepts.

1.3 Assess performance and communicate to interested stakeholders	Map value stream flow for key financial, environmental, and social aspects Develop company plan to achieve vision for satisfying stakeholder social and environmental concerns Define key performance indicators (KPIs) to monitor and assess progress Engage stakeholders in open dialogue to assess progress			
Diagnostic Questions: *Indicate Maturity Level Based on Observed Practices*	• Have the value streams for key stakeholders been mapped, integrated into business plans, and balanced for optimum performance? • Has the company developed a formal plan to achieve stated goals & aligned the plan with strategic business objectives? • Has a system of financial and non- financial measures been established to monitor progress toward stated goals? • Have stakeholder perspectives been adequately addressed and is the company achieving reasonable progress toward their goals? • Does the company openly communicate its progress to stakeholders in a form that allows them to make their own independent assessment of company progress?			
X	Qualified	Efficient	Effective	Adaptive
Example Practices	• Resource flows mapped to improve process efficiency • Baseline measures & improvement targets for company priorities • Performance evaluated against internal targets	• Primary flow paths simplified and aligned to value stream • Balanced set on financial & non-financial KPIs to align projects with business and goals • Performance review communicated to external stakeholders	• Future value stream defined and used to develop plans • KPIs drive optimization of value chain performance • Performance reviewed with value chain partners • Stakeholders dialogue to solicit feedback on reported performance	• Interdependencies with natural systems mapped and used to refine plans • KPIs aligned with public policy to maintain alignment with social expectations • Ongoing dialogue to assess performance and proactively guide business planning
Evidence *Observations that support rating at indicated level*	• Robust process to create KPI's and review with customer • •			
Opportunities *Key gaps to be addressed to move to next higher level*	• No consideration of user value proposition in mapping flows, strictly materials and energy • Environmental targets based on reducing wastes or incidents, no consideration of financial impacts • •			

Fig. 1: Example assessment questionnaire for LCM capability

The model is designed to provide a structured sequence of improvement projects based on the experiences of leadership companies with mature LCM programs. Building organisational maturity was represented as repetitive loops around the quality improvement cycle, and goals for each cycle are matched to the organisation's current state of understanding and practice. Hart noted that there was both a path-dependence, or sequence for corporate environmental strategies, moving from compliance to pollution prevention, and eventually to sustainable development, and an embeddedness of concepts that dictated parallel development of the necessary competencies [12]. The capability model can help resolve this apparent dilemma by applying the principles of sustainability to more narrow problems at lower maturity levels. For example, stakeholder engagement at low maturity levels is interpreted as cross-functional integration within the company. Employees learn the basic skills of defining system boundaries, identifying affected stakeholders, engaging in open and transparent dialogue, and developing collaborative solutions that achieve common goals. A commitment to quantifying performance and setting specific improvement targets holds the sustainability initiatives accountable for delivering near-term value to the business, a necessary prerequisite for sustained management support. Action-learning projects are then used to develop the procedures and systems necessary to sustain high performance. The fundamental difference (and benefit) of the capability framework is that improvement projects serve a dual purpose of gradually building components of a comprehensive management system while meeting the near- term performance targets managers and stakeholders demand.

4 Next steps

UNEP and SETAC are currently soliciting participants to pilot test the revised materials. The primary objective for phase 2 pilots is to demonstrate the usefulness to suppliers in developing effective improvement plans and to evaluate ease of integration of the framework with existing supply chain management programs and initiatives of the global customers. The pilot tests will also assess resource needs and potential partners to support the SME suppliers in implementing action plans. Additional details of the proposed pilot tests can be found on the project work site [13].

The pilot test is intended as a proof of concept and will provide only a limited validation of the framework. The project design uses an action research methodology to actively promote capability models as an effective tool for implementing LCM practices [14]. It is claimed that use of the capability framework will aid SME suppliers in selecting projects appropriate to their current state of knowledge of and practice with LCM methods, resulting in faster learning and more successful projects. A similar study of supply chains in South Australia suggests that project timeframes of six to nine months are adequate for developing action plans, but three to five years may be required to evaluate the implementation of those plans [15]. These initial pilot tests are designed to evaluate the utility of the capability framework for developing effective improvement action plans that meet both the business priorities of the SME suppliers and the sustainability requirements of their global customers.

References

[1] United Nations Environmental Programme (2009) Guidelines for social life cycle assessment of products. UNEP, Paris

[2] <http://www.estis.net/sites/lcinit/default.asp?site=lcinit&page_id=A8992620-AAAD-4B81-9BAC-A72AEA281CB9> (accessed 17.03.2011).

[3] <http://fr1.estis.net/sites/calcas/default.asp?site=calcas&page_id=E2669B0F-9DB7-4D1E-95B0-407BC7949030> (accessed 17.03.2011)

[4] Swarr TE, Fava J (2007) A capability model for life cycle management. Third International Conference on Life Cycle Management, Zurich

[5] <http://en.wikipedia.org/wiki/Capability_Maturity_Model> (accessed 17.03.2011)

[6] Sukhoo A, Barnard A, Eloff MM, van der Poll JA (2007) An evolutionary software project management maturity model for Mauritius. Interdiscip J Inf Knowl Manage 2:99-118.

[7] National Research Council (1999) Our common journey: A transition toward sustainability. National Academic Press, Washington DC

[8] van Zeijl-Rozema A, Martens P (2010) An adaptive indicator framework for monitoring regional sustainable development: A case study of the INSURE project in Limburg, The Netherlands. Sustainability Sci Pract Policy 6:6-17.

[9] Swarr TE, Fava J, Jensen AA, Valdivia S, Vigon B (2010) A capability framework to promote life cycle management practices in lower tier suppliers. SETAC North America 31st Annual Meeting, Portland

[10] Scholtes PR (1998) The Leader's handbook: A guide to inspiring your people and managing the daily workflow. McGraw–Hill, New York

[11] Sterman JD, Repenning N, Kofman F (1997) Unanticipated side effects of successful quality programs: Exploring a paradox of organizational improvement. Manage Sci 43:501-521.

[12] Hart SL (1995) A natural- resource - based view of the firm. Acad Manage Rev 20:986- 1014.

[13] <http://www.unep_cmm.airset.com/> (accessed 18.03.2011)

[14] Kaplan RS (1998) Innovation action research: Creating new management theory and practice. J Manage Account Res 10:89-118.

[15] Environmental Protection Authority (2004) Greening the supply chain: Greener business alliance project report. EPA, South Australia

The Sustainability Consortium: A Stakeholder Approach to Improve Consumer Product Sustainability

Kevin Dooley, Joby Carlson, Georg Schöner, Vairavan Subramanian and Cameron Childs

Abstract Consumers have become increasingly interested in sustainability, creating opportunities for businesses to position their products using environmental or social responsibility related information. This in turn has resulted in the widespread use of product claims, labels and other communication vehicles to deliver sustainability information to customers. To respond to this market demand, a unique industry, academic, governmental and NGO stakeholder collaborative organisation called The Sustainability Consortium (TSC) was formed in 2009. A primary objective of TSC is the development of a standardised framework for the communication of sustainability-related information throughout the product value chain. TSC's sustainability measurement and reporting system (SMRS) will enable rigorous product-level LCAs to be done at a fraction of today's time and cost, and provide a platform for sustainability-related data sharing across the supply chain. The SMRS will support ISO 14025 product claims, and is being developed with the principle of seeking to harmonise the world's leading environmental and social metrics, schemes and protocols that are currently being used by product manufacturers and suppliers.

1 Introduction

The pressure on Earth's resources, ecosystem services, and human life is increasing at a rapid pace. Much of this pressure is attributed to the growing population and its burgeoning demand for energy, and consumer goods and services [1]. In 2002, leaders at the World Summit for Sustainable Development (WSSD) in Johannesburg called for a change in the way that societies consume and produce, through a set of comprehensive programs based on life cycle analysis. In order to promote patterns of sustainable consumption & production, and increase the eco-efficiency of goods and services, WSSD's "Plan of

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Implementation" calls for the adoption of life cycle based tools, polices and assessment mechanisms [2,3].

The annual expenditure by consumers in the U.S. on goods and services has increased at an average rate of 3% per annum between 1984 and 2009 [4]. The growing consumption of consumer goods is accompanied by an increasing demand for products that have a lower environmental impact [5,6]. Globally, major institutional drivers for improving environmental performance include: (1) the U.S. federal government requiring its agencies to ensure that 95% of new contracts for products and services meet certain requirements for reduction in GHG, water and material impacts [7,10], (2) European Union's integrated product policy [8,10] and (3) France's Grenelle II law that requires carbon labelling of all products produced in or imported into France [9,10].

In "Vision 2050", the World Business Council for Sustainable Development (WBCSD) lays out a plan for the global population to live well, and for businesses to prosper within the limits of the planet. The council envisions markets that promote transparency, include externalities (such as ecosystem services and environmental impact) within the marketplace, and consumer's ability to choose products that provide overall value. The most critical element in the Turbulent Teens (2010-2020) is to have "government, academia, business and a range of stakeholders working together...on the design of systems and metrics to measure progress...shifting values and behaviours towards sustainability [11]."

As global citizens, we face extraordinary challenges: world population growth reaching 9 billion by 2050, diminished resources, degraded ecosystem services, worker safety and fair treatment, human health and safety, and the complexity of global trade and supply networks.

To address these challenges and stakeholders' interests, we need to more accurately quantify and communicate the sustainability of products. Our greatest opportunity is to work collaboratively together in developing an approach that drives better understanding, transparency, standardisation, and informed decision making.

1.1 Vision, mission and core values of TSC

In 2009, Walmart announced plans for the development of a global sustainable product index, citing consumer demands for products that provide overall value, and not just environmental value. As part of the initiative, it helped launch The Sustainability Consortium (TSC), a collaborative of universities, suppliers, retailers, NGOs and government, to develop a global database of information on the life cycle of products [12]. Since then, TSC's members have increased to include retailers, manufacturers, suppliers, governmental agencies, and NGOs from around the world. Along the way, TSC's objective has evolved into development of a standardised framework for the communication of sustainability-related information throughout the product value chain [14]. TSC sits in a unique position to fill the overlapping gaps associated with (1) the lack of institutional drivers for product sustainability, (2) the need for standardised methodologies for communicating sustainability, (3) the desire to respond to the call from WSSD, and the (4) need for such a collaborative in the turbulent teens, as identified by WBCSD.

In the long-term, TSC aims to reduce human impact on our world through more sustainable products and consumption. TSC's plans to achieve this vision by developing science and integrated tools that improve informed decision-making for product sustainability throughout the life cycle while serving the needs of members.

The core values shared amongst stakeholders are:

- Collaboration of diverse participants
- Science, applied and theoretical
- Comprehensive and holistic
- Transparency and accessibility
- Progress and solutions

2 Need for a sustainability measurement and reporting system

As mentioned before, TSC's primary purpose is to develop science and tools that improve decision making about product sustainability. Currently, decision-making is hampered by poor data quality and accessibility, inconsistent methodology, and expensive and time-consuming life cycle assessment (LCA) studies [16,17]. In order to improve the effectiveness of decision-making about product sustainability across the supply chain, TSC is developing the sustainability measurement and reporting system, or SMRS.

The SMRS is a LCA-based information system that facilitates decision-making about consumer goods and their supply chains. In large part, it draws from the world's most robust standards and protocols such as ISO 14000 series, WRI GHG Protocol, PAS 2050, and GRI. The SMRS can be used by any organisation in a supply chain to collaborate and share data with upstream suppliers and downstream customers, as well as manage internal operations and product design.

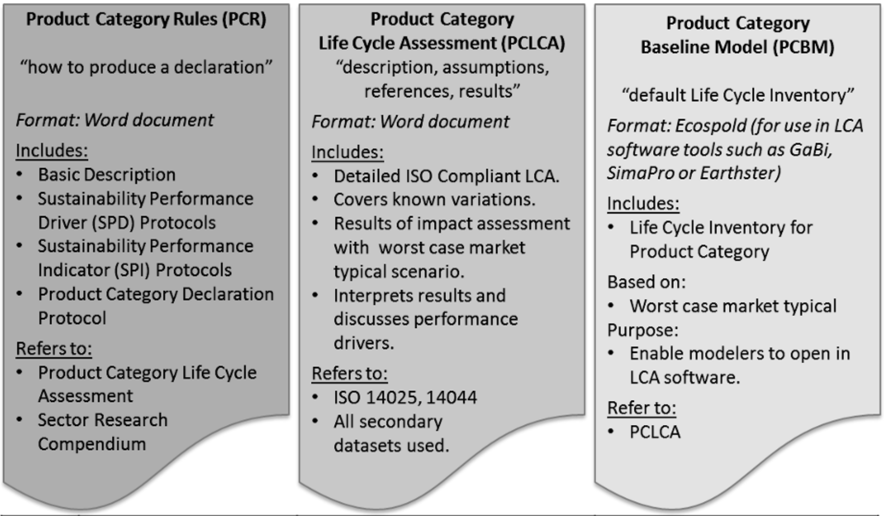
The SMRS is designed to facilitate the consistent exchange of sustainability-related information at the product level, for use by manufacturers, retailers, and eventually consumers. Other stakeholders such as NGOs, policy makers, and researchers will also use SMRS to understand the environmental and social impacts of particular products, and where innovation opportunities exist.
The SMRS will allow companies without LCA expertise to analyse and report on product information with a reasonable amount of effort, while at the same time providing an avenue for those companies with deep investments in LCA to fully utilise existing and future product level LCA work.

3 Components of the SMRS

The SMRS consists of the following components:

1) A product category rule (PCR) registry compatible with ISO 14025
2) A life cycle inventory (LCI) database linked to the PCR registry
3) A product category baseline model (PCBLM) linked to the PCR Registry
4) Computational tools and IT infrastructure to support data sharing and reporting
5) Product declarations (PD)

TSC will create and manage a PCR Registry. A PCR is a set of specific rules, requirements and guidelines for developing a Type III product declaration. PCRs will be developed in a manner compatible with ISO 14025 and ISO 14044.



Users	Brand owners, verifiers	Product and LCA Experts	LCA Modelers
Size	~20 pages	~60 Pages	1 file

Fig. 1: Component relationships and their users

A SMRS-PCR is different than a typical PCR in two ways. First, it encompasses both environmental and social impacts of the product and its life cycle. Second, it contains additional information in the form of performance indicators and drivers. These are pre-identified sustainability attributes of a product design or its supply chain that differentiate it from market-typical products.

Indicators are attributes that address known product environmental and/or social areas of significant impact (i.e. hotspots). A driver is a special type of indicator that has a pre-defined impact on the product's life cycle inventory (LCI). Thus drivers are quantitative in nature and indicators are qualitative in nature. As traceability and measurement methods in the supply chain become more robust and accurate, the performance indicators may evolve into performance drivers.

For example, "energy efficiency" is a driver for computer laptops because more energy-efficient laptops require less energy during use. This reduces their carbon footprint, leads to less greenhouse gas emissions and drops the negative impact on climate change. Conversely, a computer manufacturer's product take-back program would be considered an Indicator because while it has positive influence on environmental outcomes, the exact impact of the practice cannot be quantified.

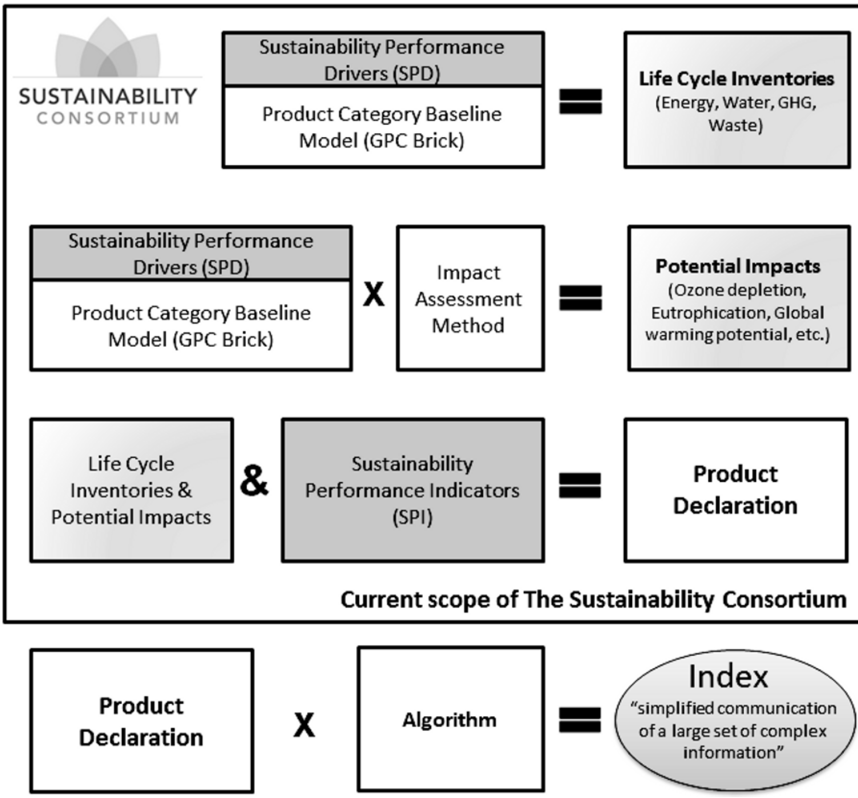


Fig. 2: How sustainability performance drivers (SPDs) and indicators create product specific declarations

TSC will act as program operator for the PCR registry, and will develop PCR creation guidelines to specify how a PCR must be built if it is to be included in the registry. The PCR creation guidelines will emphasise compatibility with existing protocols and standards, including those from ISO (14000 series), World Resources Institute/World Business Council on Sustainable Development (GHG protocol-product accounting and reporting standard), and CarbonTrust/DEFRA (PAS2050). The PCR creation guidelines will also specify reporting categories (e.g. greenhouse gas emissions, cumulative energy demand, and water consumption) that should be addressed. A full suite of impact categories will be included in the SMRS.

In order to facilitate rapid and consistent analysis of existing and new products, the SMRS will also contain product category baseline models linked to the PCR Registry. Baseline Models are constructed on best available data, or primary data from member companies. They are used to perform life cycle inventory analysis at all product life cycle stages, and to identify relevant “hotspots” of impacts. The models are built on “market typical” products, and will be subdivided (product function, geographic scope) as necessary. TSC will create a baseline model creation guideline document to specify how one must be built if it is to be included in the registry.

Finally, the SMRS will provide analytical tools to access, analyse, visualise, and share information. TSC, as well as other service providers, will develop these tools. An IT infrastructure to facilitate data sharing will be developed by external service providers, based on recommendations from the IT standards and tools working group, an internal coalition of TSC members with expertise in this area.

4 Advantages of the SMRS

Current PCRs list the requirements around functional units, system boundaries, allocation, and general assumptions. In order for those documents to enable product comparison they need to be very prescriptive. For example, a PCR needs to provide instructions on which exact datasets must be used, from which database, and where. The required rigor of the PCRs make it several hundred pages long and therefore end up being very time consuming when being applied. At the same time, most PCRs available today were developed or sponsored by one party for their own purpose and only create one product declaration.

To avoid this bias, TSC involves all interested parties in the creation process of its PCRs and baseline models. The baseline models can then be fed into a computational tool. A computational tool then restricts different unit process variables. An example of this is the product use phase and end-of-life phase of showering products that cannot be directly influenced by brand owners. Therefore the general assumptions apply for all brand owners.

SMRS creates unique value to companies of all sizes, by allowing them to:

- Use a consistent framework for benchmarking and goal tracking
- Prioritize opportunities & risks in supply chains and product life cycles
- Measure the footprint of products
- Compare products to a worst-case market typical product model and industry average
- Ask the right questions of suppliers and internal operations
- Evaluate decisions based on impact triggers
- Alter the model to suit specific supply chains and product features, or specific design alternatives
- Focus the allocation of resources where they can capture the most value

- Differentiate a product from the industry average

At this time, it is impossible to differentiate between similar products because of lack of data, uncertainty of existing data, and the variability in assumptions being made by life cycle assessment practitioners. For example TSC dealt with this issue by creating a unified substitution process for chemical ingredients in personal care products. The predefined substitution for common chemical ingredients is the first of its kind and eliminates the error resulting from an individual practitioner's subjective decision. This type of standardisation is one example of how the SMRS will reduce uncertainty in building representative models for products.

The lack of data is often caused by the very detailed information required by databases and therefore a disclosure of proprietary information. Secondary data can never be as accurate as primary data, therefore the TSC will define a system for companies to provide primary data. Within this LCI database the submitted data needs to be totally transparent in order to be vetted. But once being vetted only rolled up numbers are disclosed to the public to protect proprietary data.

5 How TSC working groups are involved

Sector working groups within TSC develop PCRs and baseline models for particular product categories, for inclusion into the SMRS. The working groups consist of TSC researchers, corporate members, NGO's, and government agencies. TSC is currently developing prototypes for seven product categories: cereal, orange juice, yogurt, surface cleaners, showering products, laundry detergent, and laptop computers. TSC's efforts will expand to other product categories in the future. Working groups around paper products, toys as well as clothing, footwear and textiles have been launched.

TSC will develop appropriate review processes for PCRs and baseline models. The sector working group votes on when they think the PCR or baseline model is complete and thus can be sent into a review process.

The measurement science working group is the body that makes recommendations concerning the methods and data used in the SMRS, while the IT standards and tools working group is the body that makes recommendations concerning the technical/IT aspects of the SMRS.

6 Conclusions

The challenges TSC set out to address are immense but a great concept has been developed to overcome the issues around product sustainability. By involving more and more stakeholders around the world in the next years TSC wants to become a major contributor to create a transparent and easy to use system. While mainly larger global corporations are currently involved in creating the SMRS, the focus is on creating this system in a way that also small and medium size enterprises can easily use it.

Creating constrained parameterised product category base line models in a format that makes them easy to use for non LCA practitioners makes the complex science based SMRS very beneficial for everybody. In addition to that the envisioned further development of live cycle inventory databases by including primary data helps to overcome significant data shortcomings that exist.

References

[1] Munasinghe M (2010) Can Sustainable Consumers and Producers Save the Planet? J Ind Ecol 14:4–6.

[2] United Nations General Assembly (2002) World Summit on Sustainable Development: Plan of Implementation. United Nations Division for Sustainable Development, New York

[3] Hertwich EG (2005) Life Cycle Approaches to Sustainable Consumption. Environ Sci Technol 39:4673-4684.

[4] <http://www.bls.gov/cex/> (accessed 14.04.2011)

[5] Grocery Manufacturers Association (2009) Finding the green in today's shoppers: Sustainability trends and new shopper insights.

[6] Furrow B, Siriani N, Childs C, Mackenzie C (2010) Consumer Science Research Compendium. The Sustainability Consortium, Tempe

[7] The President, Executive Order 13514 (2009) Federal Leadership in Environment, Energy, and Economic Performance, Federal Register, Vol. 74(194)

[8] <http://ec.europa.eu/environment/ipp/> (accessed 14.04.2011)

[9] <http://www.legrenelle-environnement.fr/-Version-anglaise-.html?rubrique33> (accessed 14.04.2011)

[10] Breisblat D, Gonnella C, He A, Nolan M, Ostman K, Reaves P, Sater K, Sifleet S, Spears E, Truckor K, Villegas J (2011) Examining Consumer and Post-Consumer Life Cycle Impacts and Policies in the United States and United Kingdom. Nicholas Institute of Environmental Policy Solutions

[11] World Business Council for Sustainable Development (2010) Vision 2050. WBCSD, Geneva

[12] <http://walmartstores.com/pressroom/news/9277.aspx> (accessed 14.04. 2011)
[13] <http://www.narcpartners.org/> (accessed 14.04.2011)
[14] <http://www.ftc.gov/opa/reporter/greengds.shtm> (accessed 14.04.2011)
[15] <http://www.sustainabilityconsortium.org/> (accessed 14.04.2011)
[16] Reap J, Roman F, Duncan S, Bras B (2008) A survey of unresolved problems in life cycle assessment. Part I: goal, scope, and inventory analysis. Int J Life Cycle Assess 13:290–300.
[17] Hur T, Lee J, Ryu J, Kwon E (2005) Simplified LCA and matrix methods in identifying the environmental aspects of a product system. J Environ Manage 75:229-337.

A Social Hotspot Database for Acquiring Greater Visibility in Product Supply Chains: Overview and Application to Orange Juice

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Abstract Social life cycle assessment (SLCA) is a technique to measure social and socio-economic impacts of product life cycles. The social hotspots database (SHDB) is an overarching, global model that eases the data collection burden in SLCA studies. It enables supply chain visibility by providing the information decision-makers need to prioritise unit processes for which site-specific data collection is desirable. Data for two criteria are provided to inform prioritisation: (1) labour intensity in worker hours per unit process and (2) risk for, or opportunity to affect, relevant social themes. This paper will present an overview of the results from a pilot study for orange juice made in the U.S. conducted with the SHDB and mandated by The Sustainability Consortium.

1 Introduction

In a world of globalised production and consumption, both positive and negative environmental and social impacts are abundant in product supply chains. With the complexity of sourcing and distributing around the world, a great deal of transparency is lost. Transparency, in an economic or business sense, is the availability of key information to help stakeholders make decisions, which in turn creates incentives for businesses to align their practices with the public's priorities [1]. Consumers are more frequently questioning where, by whom, and under what conditions their products are being sourced and produced. Most companies are not

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currently able to provide their customers with this information for the whole supply chain of their products, but many are interested in doing so.
New Earth, a non-profit organisation fostering innovative strategies and tools to help achieve sustainable development on a global level is providing a solution to acquire greater supply chain visibility. The social hotspots database (SHDB) offers an overarching, global model that eases the data collection burden in SLCA studies. The UNEP SETAC guidelines for SLCA of products recommended the development of such a resource [2]. Several other researchers also have foreseen the need for such a resource including Louise Dreyer [3] and Tomas Ekvall [4].
The Sustainability Consortium, a collaboration between retailers, manufacturers, governments, NGOs, and researchers working to develop methods and tools to accurately quantify and communicate the sustainability of products, has been one of the primary supporters of the SHDB over the last year. Consequently, New Earth was able to develop a robust database of social indicators by country and sector, as well as complete seven initial pilot studies for The Sustainability Consortium. The so-called "social scoping reports" include pilot studies of orange juice, wheat cereal, strawberry yogurt, laundry detergent, shampoo, and hard cleaner, all produced in the U.S., and laptops manufactured in China.
Results of the social scoping report for orange juice will be the focus of this paper. After a brief background on the reasoning behind and objectives of the SHDB, a literature review on the social issues existing in the orange juice supply chain is provided. The SHDB methodology and results of the social hotspot assessment on orange juice are then presented. Lastly, both the literature review and the assessment are used to offer concluding recommendations.

2 Background and social impacts of orange juice from literature

Social hotspots are production activities (i.e., unit processes) in the product life cycle that provide a higher opportunity to address issues of concern (e.g. human rights, community well-being etc.), as well as highlight potential:

- Risks of violations
- Risks affecting reputations
- Issues that need to be considered when doing business in a certain sector in a specific region/country

A tool such as the SHDB allows users to gain insights about what countries and sectors can potentially be at risk for social infringements in a supply chain and if these country-specific sectors (CSS) are likely to be labour intensive. With this type of results, site-specific assessments are the next recommended step.

Data for three criteria are used to inform prioritisation of CSS: (1) labour intensity in worker hours per country specific sector, (2) risk for, or opportunity to affect, relevant social themes related to human rights, labour rights and decent work, governance, and access to community services, and (3) gravity of the social theme. The SHDB incorporated more than 100 references to develop data tables for nearly twenty social themes and over sixty social indicators, and continues to grow. The SHDB provides results that are characterised (i.e., the level of risk or opportunity of a social indicator or issue has been assessed through a unique algorithm) in order to highlight CSS that should be investigated deeper.

Eighty percent of the oranges sourced for the production of orange juice sold in the U.S. are from Florida. U.S. orange juice production peaked in the 1997/98 season and has declined rapidly (by 47% in the 2006/07 season due to extreme weather events and crop disease, despite increased demand for orange products [5]. Most U.S. orange juice imports come from Brazil, the world's largest orange juice producer and exporter [6].

The agriculture industry, particularly fruits and vegetables, is labour intensive, even in industrialised countries like the U.S. Almost all of Florida's oranges are harvested by hand, although mechanisation equipment has started to become a potential alternative for certain parts of the harvest. Currently, labour makes up almost half of the production expenses for U.S. fruit and vegetable farms [7].

National Agricultural Worker Survey (NAWS) data for Florida suggest that approximately 75% of the workers were unauthorised for employment in the U.S. from 2002-04, significantly higher than for the U.S. overall. Authorised workers in Florida reported substantially higher wage rates than did unauthorised workers. The real hourly wage for unauthorised Florida workers in 2002-04 was $2.74 less than for authorised workers. Employment by a labour contractor reduced hourly earnings between 6-12%. Working seasonally reduces hourly earnings by 4-8% [8].

Due to lower labour costs, Brazil can produce orange juice at a lower cost than Florida [7]. During the 2007–08 season, Brazilian orange growers in Sao Paulo paid their harvest workers less than 50 cents (U.S. dollars) to harvest one box (90 pounds) of fruit. During the same season, Florida growers paid between 80 and 90 cents, at least 60% more, to harvest one box of oranges [9].

Not only are fair wages a concern, but forced labour is also an issue on both fruit and vegetable farms in the U.S. [10]. Additionally, there have been numerous reports of farm workers (often undocumented) in Florida and South Africa working excessive overtime and receiving no overtime compensation [11]. Unhealthy and unsafe working conditions are also a major concern. Illness due to

exposure to pesticides and eye injuries as well as injuries caused by accidents when using a ladder during manual picking of oranges are also important problems [12].
Child labour is still an issue in many orange-producing countries. In Brazil, Mexico and South Africa, it is estimated that thousands of minors are employed in the harvesting of oranges [12]. In 2000, it was estimated that of the 700,000 orange pickers, 15% were below 14 years old [13].
Even though Brazil has ratified 7 out of 8 core ILO Conventions and has national labour legislation favourable to employees, restrictions on trade union rights, discrimination, child labour and forced labour still exist. Brazil has not ratified the Convention of Freedom of Association and the violation of the right to organise is widespread, especially in rural areas. Collective bargaining and the right to strike are also restricted. Blacklisting and threats are common, and many workers have to join a trade union in secret [14].

3 Methodology

Social hotspots are further identified and prioritised based on an assessment using the social hotspot database (SHDB). The SHDB is a collection of tables by social issue that report quantitative indicators and/or characterised qualifiers by country and, when possible, by sector. In order to evaluate the countries and sectors that are most at risk in the orange juice supply chain, two lists of country-specific sectors (CSS's) are developed and tested in the SHDB model.
The first list is based on the share of worker hours necessary to produce US$1M of vegetables and fruits in the U.S. using a global input-output (IO) model derived from the Global Trade Analysis Project's (GTAP) general economic equilibrium model. Using data on total wages paid out per $ output by CSS from GTAP divided by data on average wages by country and sector from the International Labour Organization (ILO) and other sources, hours per dollar output are calculated. This model developed by New Earth is able to assign unskilled, skilled, and total worker hours per $ to a matrix of 57 sectors and 113 countries and regions, for a total of 6,441 CSS's.
For the orange juice model, worker hours data was determined for the production US$1M vegetables and fruits (determined to be the most relevant sector). The majority of worker hours (approximately 93%) occur in the first 200 sequential CSS, thus, only the top 200 Worker Hour (WH) CSS's were tested within the SHDB model.

The second list of CSS's is based on external research of the supply chain, defining the materials, components and resources required to produce orange juice in a Tetrapak container and assigning them to a specific sector. Next, countries that extract, refine, produce, assemble, and export these essentials are determined via the literature.

The scope of the second model based on the main production activities in the life cycle of orange juice takes into consideration the growing of oranges, the processing of orange juice, the energy used, and also the manufacture of a Tetrapak aseptic container. This model also includes business and financial services, retail and wholesale operations, or infrastructure and construction within the U.S.

The results of the social hotspot assessment are extensive for each CSS including indicators and characterisation factors for multiple social themes. Therefore, results are combined into a social hotspot index. This index is determined by dividing the sum of social issues with high and very high risk for each CSS by the total potential social hotspots a CSS could obtain (based on the number of social issues reported) and multiplying by a factor of 100. Weighting is performed in this index depending whether it is a high or very high risk and whether it is in the top 93% in the ranking of worker hours.

4 Results

4.1 Worker hours model analysis

For the worker hours assessment, a ranking of the potential 6,441 CSS was produced for the skilled, unskilled, and total workforce. The ranking is based on the greatest share of worker hours in the production of US$1M of vegetables and fruits in the U.S. Ninety-five percent of the worker hours are within the top 292 ranked CSS for total workforce.

Table 1 lists the first 10 CSS for the total workforce (skilled and unskilled rankings are similar, therefore, not shown). The top sector with the most worker hours is the vegetable and fruit sector in the U.S., which is responsible for more than one-third of the total worker hours. The sectors with the most worker hours for vegetables and fruits produced in the U.S. are all within the U.S. except for one that the literature review shows should be disregarded in the case of U.S. orange juice. The important U.S. sectors with regards to worker hours include business and retail services, financial intermediation, construction, and transport.

Tab. 1: Top ten sectors by worker hours (WH) for total, skilled, and unskilled workforce for the production of orange juice (OJ) in the United States (XAC = South Central Africa Region)

Sector	Country/ region	WH/US$1M OJ	% of total WH	Cumu-lative %	Hotspot index
Vegetables & fruits	U.S.	1.99E+004	33.70	33.70	18.46
Business services	U.S.	9.31E+003	15.79	49.49	15.38
Retail services	U.S.	3.55E+003	6.01	55.50	17.91
Financial services	U.S.	1,591.12	2.70	58.20	15.38
Paper products	U.S.	1,470.77	2.49	60.69	15.38
Construction	U.S.	1,366.7	2.32	63.01	17.91
Lumber products	U.S.	911.22	1.55	66.62	15.38
Truck transport	U.S.	751.39	1.27	67.90	18.46
Chemical products	U.S.	698.8	1.18	69.08	15.38

4.2 Supply chain model analysis

Based on a comprehensive literature review of the orange juice supply chain, the priority materials consumed in producing orange juice are oranges, fertilizers, pesticides, water, fossil fuels, and machinery. Materials for the Tetra Pak aseptic packaging include paperboard from wood, LDPE liner for liners, fossil fuels, and machinery. The primary countries that produce and export these materials (specifically to the U.S. when available) are shown in Table 2. For this assessment, it is assumed that the orange juice is grown and made for U.S. consumption; therefore, sectors such as electricity and gas, transport, water, and business services are all assessed specifically for the U.S. A similar assessment could easily be performed for any country that makes orange juice. The sectors and countries combine to offer a comprehensive list of 88 CSS that were tested within the SHDB model.

Results of the social hotspot assessment for the CSS list are shown in Table 3. CSS were eliminated if they were not in the top 95% of the worker hours in the supply chain (except for Brazil, vegetables and fruits). The worker hours ranking in the first column is out of 6,441 total CSS according to the GTAP model on worker hours per production of US$1M of vegetables and fruits. The asterisk next to Brazil's worker hours ranking indicates that this CSS is contributing only 0.01% to the worker hours. However, this assessment is performed for U.S.

vegetables and fruits in general; therefore, it was still shown in Table 2 because of its assumed importance in the supply chain of U.S. orange juice.

Tab. 2: Highest exporting countries for the materials used in orange juice production

Material	Countries (GTAP code)	Reference
Oranges	Brazil, USA, India, Mexico, China, Spain, Iran, Italy, Indonesia, Egypt	FAO Stat Online, 2008 data
Fertilisers	USA, Canada, Trinidad&Tobago, Morocco, Russia, Venezuela, Saudi Arabia, Qatar, Bahrain, Norway, China, Egypt, Kuwait	ERS/USDA Export Data, 2009
Pesticides	Germany, Switzerland, USA, Israel, Australia, Japan	ETC Group, Who Owns Nature? 2008 Report.
Oil	USA, Canada, Mexico, Saudi Arabia, Iraq, UAE, Kuwait, Venezuela, Nigeria, Russia, Algeria, Angola, Colombia, Iran, Norway	U.S. Energy Information Administration, Dept of Energy, 2009.
Coal	USA, Canada, Colombia, Venezuela, Indonesia, Australia, China, Ukraine, Russia, South Africa	U.S. Energy Information Administration, Dept of Energy, 2009.
Natural Gas	USA, Canada, Mexico, Egypt, Nigeria, Norway, Qatar, Trinidad, Yemen, Russia, Algeria, Turkmenistan, Netherlands, Indonesia, Malaysia	U.S. Energy Information Administration, Dept of Energy, 2009.
Wood	USA, Russia, Finland, Sweden	Tetra Pak Co.
LDPE Liner	Canada, China, Germany, Mexico, Japan, USA	Plastics Fact Sheet, 2009.
Machinery	USA, China	Alibaba, OJ Production Machinery

The hotspot index is based on the total potential hotspots the CSS could obtain; therefore, a higher index indicates a greater prevalence of hotspots. The Brazilian and U.S. vegetable and fruits sector fall near the bottom of this list, with hotspot indices of 24.27 and 22.68 respectively. The countries and sectors that appear to be sensitive to social hotspots are the growing of oranges in countries like India, China, Indonesia, Mexico, and Brazil; extraction of crude petroleum, coal and gas extraction in Angola, China, Nigeria, Turkmenistan, Venezuela, Indonesia, and

Algeria; and the chemical and plastic manufacturing sector in China, Venezuela, Russia, and the West Africa Region.

Tab. 3: Country-specific sectors (CSS) most at risk for social hotspots to be present based on the supply chain of orange juice

Worker Hours Rank	Country	Sector description	Hotspot index (0-100)
26	Angola	Oil extraction	78.64
30	India	Vegetables, Fruits	63
52	China	Coal extraction	54.64
12	China	Chemical, plastics	51.06
15	China	Machinery	51.06
108	China	Vegetables, Fruits	50.52
164	Turkmenistan	Natural Gas extraction	45.63
13	Venezuela	Oil extraction	42.45
51	Venezuela	Chemical, plastics	39
233	China	Petrochemicals	34.04
186	Russia	Chemical, plastics	34
62	Mexico	Vegetables, Fruits	32.04
80	United States	Electricity	23
20	United States	Water	23
1	United States	Vegetables, Fruits	22.68
365*	Brazil	Vegetables, Fruits	24.27

5 Conclusions

The social scoping assessment of the orange juice supply chain was achieved by combining different methods that offer a variety of perspectives on the potential social impacts found in the orange juice supply chain. The main source of information is the social hotspots database (SHDB) complemented by an external literature review.

The social hotspot assessment, performed for US$1M of vegetables and fruits produced in the U.S., found that the largest share of worker hours is concentrated in the production activities occurring in the U.S. In fact, the vegetables and fruits sector is accountable for nearly 40% of unskilled labour, and the business services

sector is responsible for 40% of the skilled labour. The literature review concurs with the assessment of social hotspots, which range from health and safety of workers to excessive working time during the harvest season; low wage rates, particularly for unauthorised workers; potential of forced labour in the U.S. and child labour in other producing countries; and violations of the right to organise, collective bargaining and the right to strike.
The results presented in this document indicate that while social issues do occur in the orange juice supply chain, they have the capacity to be improved. Orange juice has a limited number of ingredients and production steps are mostly occurring within the U.S., which makes improvements in the supply chain more accessible. This assessment will be followed by a stakeholder review in order to validate results and identification of sustainability performance indicators that can communicate social impacts of production of orange juice.

References

[1] Goleman D (2009) Ecological Intelligence. Doubleday Publishing Group
[2] Benoit C, Mazijn B (eds) (2009) Guidelines for Social Life Cycle Assessment of Products. UNEP/SETAC, Paris
[3] Dreyer LC, Hauschild MZ, Schierbeck J (2010) Characterisation of social impacts in LCA. Part 1: Development of indicators for labour rights. Int J Life Cycle Assess 15:247–259
[4] Ekvall T (2011) Notions in Social LCA. Int J Life Cycle Assess 16:1-2
[5] Morris RA (2010) The U.S. Orange and Grapefruit Juice Markets: History, Development, Growth, and Change. Florida Cooperative Extension Service
[6] Muraro R, Spreen T, Pozzan M (2003) Comparative Costs of Growing Citrus in Florida and Sao Paulo (Brazil) for the 2000-01 Season. Florida Cooperative Extension Service
[7] Calvin L, Martin P (2010) The U.S. Produce Industry and Labor Facing the Future in a Global Economy. USDA
[8] Emerson RD, Walters L, Iwai N, Van Sickle JJ, Roka F (2006) The Florida Farm Labor Market. Conference on Immigration and Agriculture, Washington DC
[9] Roka FM (2009) Compensating Farm Workers through Piece Rates: Implications on Harvest Costs and Worker Earnings. Florida Cooperative Extension
[10] Verite (2010) Immigrant Workers in U.S. Agriculture: The Role of Labor Brokers in Vulnerability to Forced Labor.

[11] Oxfam International (2004) Trading Away Our Rights, Women working in Global Supply Chains.

[12] www.fairfood.org, (accessed 01.2011)

[13] Amestoy M, Crosbie M (2000) Child labor in the orange juice industry in Brazil. In: Ryan L, Gasparski W (eds) Praxiology: The International Annual of Practical Philosophy and Methodology. Transaction Publishers, New Brunswick

[14] ICFTU (2004) Internationally recognized core labour standards in Brazil. Report for the WTO general council review of the trade policies of Brazil, Geneva

PART II:
LCM Methods and Tools

A Novel Weighting Method in LCIA and its Application in Chinese Policy Context

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Abstract Under given political environmental targets, if explicit and comprehensible conclusions could be reached via weighting method, LCA would play a much more crucial role for enforcement of environmental policies. A couple of distance-to-target weighting methods were proposed for this purpose, but the meaning of weighting was still questionable. Taking Chinese "Energy Conservation and Emission Reduction (ECER)" political targets as an example, a different distance-to-target weighting method was proposed, so-called ECER method. The method was tested by a LCA case study on comparison of three desulfuration technologies in float glass production, in which explicit and exclusive conclusions were delivered, i.e. dry process is better than wet process and semi-dry process. And, the differences between ECER and other "distance-to-target" methods were discussed.

1 Introduction

Many weighting methods have been proposed in order to deliver a single score and explicit conclusions in LCA studies, which is always desirable for decision making [1]. Among them, distance-to-target weighting methods, such as EDIP method [2] and ecological scarcity method [3], developed weighting factors according to international or national political environmental targets. Under given political environmental targets, if explicit and comprehensible conclusions could be reached via these weighting method, LCA would play a much more crucial role for enforcement of these environmental policies.

For example, EDIP method defines a weighting factor of a certain flow or impact as the ratio of its actual amount in reference year to its target amount in target year as stated in environmental policies. Ecological scarcity used the square of the ratio as the weighting factor. Then the weighting factors are multiplied by the life cycle results of a case study to produce a single score. The motto in such weighting

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method is "the bigger is the ratio (distance), the bigger is the weighting factor, and therefore the more important is the flow or impact category".

However, the meaning of weighting in these methods seems ambiguous. Although weighting factors were derived from policy targets, the single score or the difference of the single scores of alternative options failed to show to what extent the political targets are fulfilled. And, an opposite example can be found in China. In Chinese national "Energy Conservation and Emission Reduction" policy for 11th five-year plan (2006-2010), a much more harsh target had been assigned to reduction of SO_2 emissions than reducing energy use (see Table 1), so a bigger weighting factor would be given to SO_2 than energy use according to above weighting methods. However, the SO_2 target was easily over fulfilled in 2010, but the energy use target was almost failed. This example shows that a bigger "distance" does not always imply it's a harder target to be achieved and a bigger weighting factor should be assigned. The reason lies in the fact that policy-makers had thought through the improvement potentials and feasibility before they made the targets.

Taking Chinese "Energy Conservation and Emission Reduction (ECER)" political targets as the example, a different distance-to-target weighting method was proposed in section 2, so-called ECER method, to overcome the problems as above mentioned. The method was tested by a LCA case study on comparison of three desulfuration technologies in float glass production in section 3. Moreover, the differences between ECER and other "distance-to-target" methods were discussed in section 4.

2 ECER method

2.1 Transform original political targets into comparable targets

In the Chinese "Energy Conservation and Emission Reduction (ECER)" policy, quantitative national environmental goals are given as mandatory targets. In the eleventh national five-year plan (2006-2010), the targets are announced as in Table 1. However, those original targets are defined a little bit differently, so they need to be transformed into a comparable way first, i.e. in terms of "reduction rate per GDP in 5 years", and then be used in the ECER scoring formula.

For example, the target of GHG emission reduction per GDP (gross domestic product) by 40-45% is in the period of 2006 to 2020. Assuming the reduction rate is constant then the reduction target from 2006 to 2010 would be 17%. The reduction targets of SO_2 and COD (chemical oxygen demand) were defined as reduction of total amount instead of reduction per GDP, so they have to be adjusted according to the growth rate of GDP. From 2005 to 2010, GDP (inflation-adjusted) of China had grown 1.66 times, giving reduction rate in total of SO_2 and COD are 10%, then reduction rate per GDP can be calculated as 46% (see Table 1).

Tab. 1: Original political targets and the comparable targets after adjustment

Original political targets (2006-2010)	Comparable targets	Corresponding life cycle indicators
Reduction of GHG emission per GDP by 40-45% (in the period of 2006 to 2020)	17%	Global warming
Reduction of energy use per GDP by 20%	20%	Energy use
Reduction of water use per industrial add value by 30%	30%	Water use
Reduction of SO_2 emission by 10% in total	46%	SO_2
Reduction of COD emission by 10% in total	46%	COD

2.2 ECER scoring formula

Since the main purpose of LCA is to compare alternative options, ECER scoring formula was conceived for comparative case studies first.
When comparing one option (denoted as 1) against the baseline (denoted as 0), the difference of ECER scores is defined as

$$\Delta S = \sum_{i=1}^{5} \frac{\left(I_i^0 - I_i^1\right)/I_i^0}{T_i} \times \frac{I_i^0}{N_i} \tag{1}$$

Where I_i is the five life cycle indicators as the environmental political targets (see Table 1); T_i is the comparable political targets; N_i is the corresponding normalization references of China in 2005.
The first part in the summation is five "scores" of the option 1 according to the ratio of its actual improvement against expected improvement by political targets. Before adding up the five scores, they are weighted by the normalisation references, because the more does this indicator contributes to national total amount, the more important is this score.
According to the Equation 1, ECER single sore can be expressed as Equation 2.

$$S = \sum_{i=1}^{n} \frac{1}{T_i} \times \frac{I_i}{N_i} \quad (2)$$

3 Case study: comparison of three flue gas desulfuration technologies in float glass production

With ECER method, three flue gas desulfuration technologies are compared, namely dry process, wet process and semi-dry process. The baseline is selected as non-desulfurization. The functional unit was defined as "treatment of flue gas containing 1kg of SO_2". The product system included desulfuration process, main energy and material production upstream (cradle-to-gate type). Impact categories and inventory flows were selected according to ECER method, i.e. primary energy use, water use, global warming potential, SO_2 and COD.

The primary data of the three desulfuration processes were collected from a float glass producer in China (Table 2). The background datasets of electricity, steam, caustic soda and soda ash production were obtained from Chinese Reference Life Cycle Database (CLCD) [4]. The datasets of quicklime, slaked lime, and magnesium oxide production were from Ecoinvent database (v2.2) [5].

LCI/LCIA results and ECER score of the three processes are shown in Table 3. Benchmarked by non-desulfurization, improvement percentages of three processes were calculated.

Tab. 2: Primary dataset of three processes and data sources

Primary dataset of three processes	Non-desulfuri-zation (baseline)	Dry process	Wet process	Semi-dry process	Unit	Data source
Electricity	-	1.222	0.830	1.071	kWh	CLCD
Steam	-	1.157	-	-	kg	CLCD
Slaked lime	-	1.476	-	-	kg	Ecoinvent
magnesium oxide	-	-	1.367	-	kg	Ecoinvent
Quicklime	-	-	0.479	-	kg	Ecoinvent
Caustic soda	-	-	0.063	-	kg	CLCD
Soda ash	-	-	-	2.362	kg	CLCD
SO_2	1	0.1	0.1	0.15	kg	On site

Tab. 3: LCI/LCIA results and ECER score

Item		No desulfurisation (baseline)	Dry method	Wet method	Semi-dry method	Unit
LCI& LCIA results	Energy use	-	0.64	0.487	1.18	kgce
	Water use	-	43.8	54.2	132	kg
	GHG	-	2.82	2.84	5.11	kg CO_2e
	SO_2	1	0.109	0.107	0.165	kg
	COD	-	0.00344	0.404	0.00604	kg
ECER score	Energy use	-	6.47E-14	4.92E-14	1.19E-13	-
	Water use	-	1.39E-14	1.72E-14	4.18E-14	-
	GHG	-	1.08E-13	1.09E-13	1.96E-13	-
	SO_2	3.70E-12	4.03E-13	3.96E-13	6.09E-13	-
	COD	-	2.85E-14	3.35E-13	5.01E-14	-
	Single score	3.70E-12	6.18E-13	9.07E-13	1.02E-12	-
Improvement percent		-	83.3%	75.5%	72.4%	-

The results show that all three processes have remarkable improvements and the dry process is the best among them. Semi-dry method has relatively the lowest improvement.

Through contribution analysis, the main contributors and the reasons can be identified (Table 5). Except COD, semi-dry process has the highest scores of three methods. Energy use and GHG emission were mainly contributed by soda ash and electricity production. Water use was mainly caused by the high consumption in desulfuration process and soda ash production. High emission of SO_2 was due to the low removal rate of desulfuration. For COD, wet process has the highest score because of the high emission in magnesium oxide production.

Tab. 4: Contribution analysis of three processes in terms of ECER score

Process	Contributor	Energy use	Water use	GHG	SO_2	COD	Total
Dry process	Desulfuration	- (0.0%)	1.21E-14 (2.0%)	- 0.0%	3.7E-13 (59.9%)	- (0.0%)	3.82E-13 (61.8%)
	Electricity	4.72E-14 (7.6%)	1.28E-15 (0.2%)	4.99E-14 (8.1%)	3.04E-14 (4.9%)	2.47E-14 (4.0%)	1.53E-13 (24.8%)
	Steam	5.24E-16 (2.7%)	1.47E-17 (0.1%)	1.53E-14 (6.9%)	3.38E-16 (0.3%)	2.74E-16 (0.6%)	1.64E-14 (10.6%)
	Slaked lime	1.69E-14 (0.1%)	4.79E-16 (0.0%)	4.29E-14 (2.5%)	1.96E-15 (0.1%)	3.56E-15 (0.0%)	6.58E-14 (2.7%)
	Total	6.47E-14 (10.5%)	1.39E-14 (2.2%)	1.08E-13 (17.5%)	4.03E-13 (65.2%)	2.85E-14 (4.6%)	6.18E-13 (100.0%)

Process	Contributor	Energy use	Water use	GHG	SO_2	COD	Total
Wet process	Desulfura-tion	- (0.0%)	1.57E-14 (1.7%)	- (0.0%)	3.70E-13 (40.8%)	- (0.0%)	3.86E-13 (42.6%)
	Electricity	3.21E-14 (3.5%)	8.67E-16 (0.1%)	3.39E-14 (3.7%)	2.07E-14 (2.3%)	1.68E-14 (1.9%)	1.04E-13 (11.5%)
	Magnesium oxide	8.60E-15 (0.9%)	4.67E-16 (0.1%)	5.52E-13 (6.1%)	3.93E-15 (0.4%)	3.16E-13 (34.8%)	3.84E-13 (42.3%)
	Quicklime	7.11E-15 (0.8%)	8.49E-17 (0.0%)	1.81E-14 (2.0%)	8.19E-16 (0.1%)	1.34E-15 (0.1%)	2.74E-14 (3.0%)
	Caustic soda	1.43E-15 (0.2%)	7.69E-17 (0.0%)	1.85E-15 (0.2%)	9.17E-16 (0.1%)	7.48E-16 (0.1%)	5.02E-15 (0.6%)
	Total	4.92E-14 (5.4%)	1.72E-14 (1.9%)	1.09E-13 (12.0%)	3.96E-13 (43.7%)	3.35E-13 (6.9%)	9.07E-13 (100.0%)
Semi-dry process	Desulfura-tion	- (0.0%)	2.42E-12 (2.4%)	- (0.0%)	5.55E-13 (54.4%)	- (0.0%)	5.79E-13 (56.8%)
	Electricity	4.14E-14 (4.1%)	1.12E-15 (0.1%)	4.37E-14 (4.3%)	2.67E-14 (2.6%)	2.16E-14 (2.1%)	1.34E-13 (13.1%)
	Soda ash	7.78E-14 (7.6%)	1.64E-14 (1.6%)	1.53E-13 (15%)	2.71E-14 (2.7%)	2.85E-14 (2.8%)	3.02E-13 (29.6%)
	Total	1.19E-13 (11.7%)	4.18E-14 (4.1%)	1.96E-13 (19.2%)	6.09E-13 (59.7%)	5.01E-14 (4.9%)	1.02E-12 (100.0%)

4 Discussion

Comparing ECER method with EDIP and eco-scarcity methods, they are different in definitions. In EDIP and eco-scarcity methods, the political targets were expressed as total amount and the weighting factor was defined as the ratio ("distance") of two states (or square of the ratio in eco-scarcity), i.e. the actual amount in reference year and the target amount in target year. In ECER method, the political targets were presented as the reduction rates ("distance") and the improvement of alternative options were compared against targets.

If transforming ECER scoring formula into similar form as in EDIP method, the weighting factor in ECER is the reciprocal of the targets (Equation 4), while the weighting factor in EDIP method is totally different (Equation 3). It shows a opposite set of weighting factors for given Chinese ECER political targets, compared with EDIP and Eco-scarcity methods (see Table 3). So they wouldn't bring the same meaning to the LCA studies.

$$W_1 = \frac{F_c}{F_t} \tag{3}$$

$$W_2 = \frac{1}{T} = \frac{F_c}{(F_c - F_t)} \tag{4}$$

Tab. 5: Normalised weighting factors of the three methods for Chinese ECER targets

LCA indicator	ECER	EDIP	Eco-scarcity
GHG emission	31.88%	15.86%	12.16%
Energy use	26.84%	16.49%	13.14%
Water use	17.90%	18.85%	17.16%
SO_2 emission	11.69%	24.40%	28.77%
COD emission	11.69%	24.40%	28.77%

5 Conclusions

LCA could provide a crucial methodological basis for a wide range of political instruments, if sound and comprehensible distance-to-target weighting method based on political targets could be conceived and justified.
ECER method was proposed in this paper which was defined by a different scoring formula and its meaning could be clearly explained when alternative options were compared. Although ECER weighting factors are derived from Chinese policy, the method can be applied in other countries, provided multiple political targets are announced in a similar way.
The method was tested by a LCA case study on comparison of three desulfuration technologies in float glass production, in which explicit and exclusive conclusions were delivered, i.e. dry process was better than wet process and semi-dry process. The analysis also shows that ECER method presents an opposite set of weighting factors, compared with EDIP and Eco-scarcity method.

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References

[1] Bare J, Gloria TP (2006) Critical analysis of the mathematical relationships and comprehensiveness of life cycle impact assessment approaches. Environ Sci Technol 40(4):1104–1113.

[2] Wenzel H, Hauschild MZ, Alting L (1997) Environmental Assessment of Products. Methodology, Tools, Techniques and Case Studies, Vol. 1. Chapman & Hall, United Kingdom, Kluwer Academic Publishers, Hingham, MA, USA

[3] Frischknecht R, Steiner R, Jungbluth N (2008) Ökobilanzen: Methode der ökologischen Knappheit – Ökofaktoren 2006 (Swiss Ecological Scarcity Method). Öbu – Netzwerk für Nachhaltiges Wirtschaften, Öbu SR 28/2008, Zürich

[4] Liu XL, Wang HT, Chen J, He Q, Zhang H, Jiang R, Chen XX, Hou P (2010) Method and basic model for development of Chinese reference life cycle database of fundamental industries [J]. Acta Scientiae Circumstantiae 30(10):2136-2144.

[5] <http://www.ecoinvent.ch/>

The Usefulness of an Actor's Perspective in LCA

Henrikke Baumann, Johanna Berlin, Birgit Brunklaus, Mathias Lindkvist, Birger Löfgren and Anne-Marie Tillman

Abstract This paper is an argumentation for adding an actor's perspective to life-cycle assessment (LCA). The need for this perspective stems from a criticism about the usefulness of LCA interpretation methods comparing the relative contribution of life-cycle phases of a product. Our argumentation is based on four previously published studies providing practical examples of how value chain actors' influence may be considered in an LCA and the benefit of doing so. Manufacturing sector examples show how one company's influence can be illustrated in results and how it may relate all relevant emissions to its own processes. The food sector study shows how to assess several value chain actors' individual improvement potential. The final example, taken from building sector, explore how to consider the fact that actors in one part of the value chain can influence other actors to improve.

1 Problem and solution

Life cycle assessment (LCA) is a tool that illustrates the entire life cycle of products and services and quantifies their environmental impacts. A frequently asked question in LCAs is "which part of the life cycle contributes the most to the environmental burden of a product/service?" and the most common method used is the dominance analysis. A dominant use phase contribution to global warming is found in most products consuming energy during product application like cars, computers, and light bulbs. In the case of animal food products, such as milk and yogurt, the agricultural processes usually dominate the life-cycle environmental impact. These are typical examples of how one can learn and pinpoint so called hotspots in the product life cycle when using LCA.

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However, to what extent does such analysis underpin an improvement of the situation? Is the conclusion of the analysis relevant for the receiver of the results, or, in other words, to what extent can he or she influence? It is our experience that LCA's holistic nature often urges the analyst to define very broad goals, forgetting that no decision maker alone can influence the whole value chain of a product. Policy maker's power of influence is limited by national borders and industrial actors and consumers are limited by their location in the value chain. Results based on an analysis not taking this into account risk to mislead actors into underestimating their ability to influence and improve the product, especially if they are not acting in the dominating phase of the product life cycle. Should a worker in manufacturing stop bothering about the environmental consequences of the product he is producing if the results from an LCA contribution analysis show that the contribution of his processes is only a small fraction of the total?

The fundamental difference between an LCA with an actor's perspective and contributional analysis, as we know it, is that the sphere of influence of actors, such as organisations, companies, institutions or even specific work roles within these, are considered in the former. Contribution analysis typically compares life-cycle phases or technically defined processes.

The importance of an actor's perspective has been highlighted since the beginning of LCA and the broader field of Industrial Ecology, see e.g. [1-3]. However, developments in theory and practice including an actor's perspective are rarely found. Baumann [4] has proposed a whole new research area for understanding how organisations influence the environment. In this paper we highlight the usefulness and value of adding an actor's perspective to the LCA methodology by discussing four previously published studies where this perspective is applied. They have in common a criticism of dominance analysis in LCA but the alternative approaches vary and highlights different aspects. In the LCA textbook by Baumann & Tillman [5] an early example is given where one company's influence is illustrated in flowcharts and bar charts. Further work specifically considering the perspective of one actor in the value chain is presented in the manufacturing sector study by Löfgren [6]. He formulated a formal method to consider only the parts that are relevant for people working in manufacturing at a specific company in the value chain. The food sector study [7] introduced a way to from the perspective of several value chain actors assess their individual improvement potential. The final example is taken from building sector where Brunklaus et al. [8] explore how to consider the fact that actors in one part of the value chain can influence other actors to improve.

2 First example: Decision maker analysis

An early example where an actor perspective is applied in LCA is a "decision maker analysis" found in textbook on LCA [5]. The analysis was made for the manufacturing company SKF and was based on data from a study by Ekdahl [9]. The basis for this type of analysis is the identification of the different companies and organisations that carry out the different activities in the technical system. This can be used to identify the extent to which environmental impact is under a certain company's control. The flow chart, in Figure 1, shows the relationship between commissioner and other companies and five levels of influence where identified.

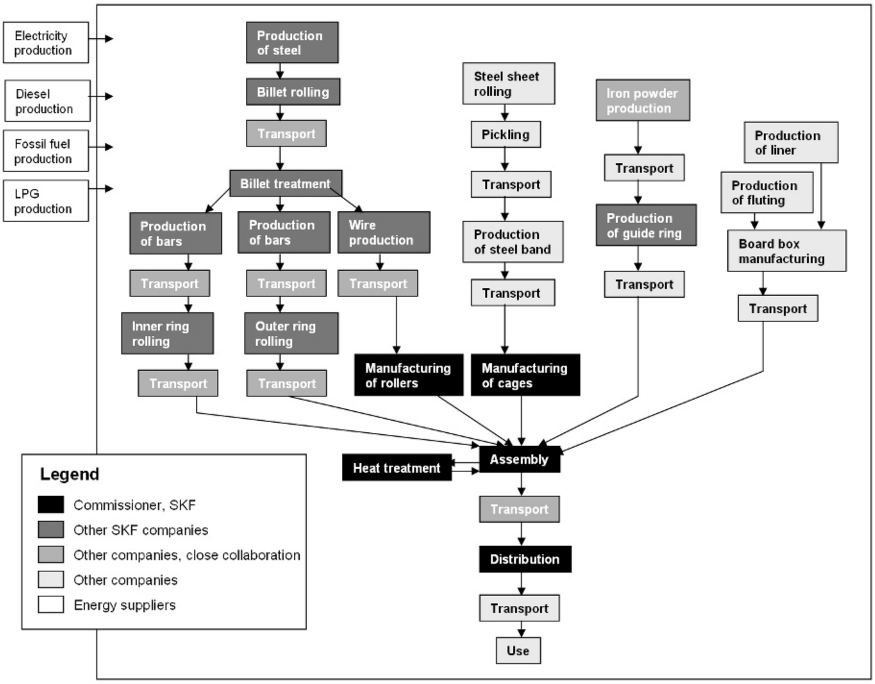

Fig. 1: Flow chart for the decision maker analysis of an SKF roller bearing [5]

Based on the information results can be presented as in Figure 2, where environmental impacts are related to the different actors. To reduce emissions to air (e.g. CO_2, SO_2, NO_x and CH_4) the decision maker needs to influence the choice of energy suppliers.

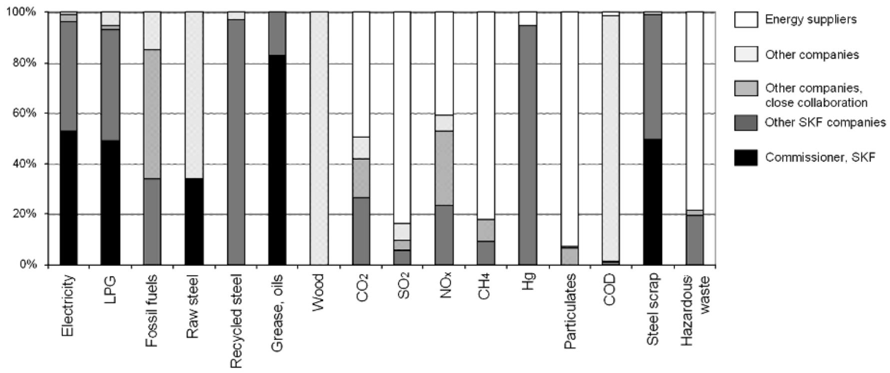

Fig. 2: Environmental impacts related to different actors [5]

3 Example 2: Relating life cycle consequences manufacturing actors

In a more recent study at SKF [6], the company wanted to better understand the environmental consequences from *manufacturing* a bearing unit. The company was specifically interested in how results could be presented to make better sense to SKF employees working in manufacturing, namely to inform them of how they, in their daily work, could increase the environmental performance of manufacturing processes.

The initial dominance analysis showing the CO_2e emissions, during the manufacturing of the bearing unit, distributed on SKF manufacturing processes, tier 1 suppliers, steel production and other suppliers, indicated that steel production was the hotspot. SKF processes' contribution was only a smaller part of the emissions.

When these results were presented to manufacturing actors in the company they were perceived as less relevant, since they did not give an immediate picture of how they could increase the environmental performance of their processes. So, to increase the relevance, the results were first reformulated by relating all upstream emissions to in-house activities, see left side of Figure 3.

The next step was to ensure that the results presented related to the work of the company's manufacturing actors; i.e. to ensure that the results only contained environmental consequences that could be improved by their actions in manufacturing. Considering that the CO_2e emissions related to the production of component material remaining in the product when it is shipped to customer, is primarily dependent on the design of the product - and *not* on how it was manufactured at SKF - this part was omitted from the results, given to the right of Figure 3.

This bar chart now illustrated a radically different picture of the environmental consequences for SKF manufacturing actors from that given by the more conventional dominance analysis. General improvement strategies for SKF manufacturing processes were clearly revealed from the bar chart as (1) designing production processes that reduces the material removed by machining, (2) increasing electricity use efficiency, and (3) decreasing scrap rate.

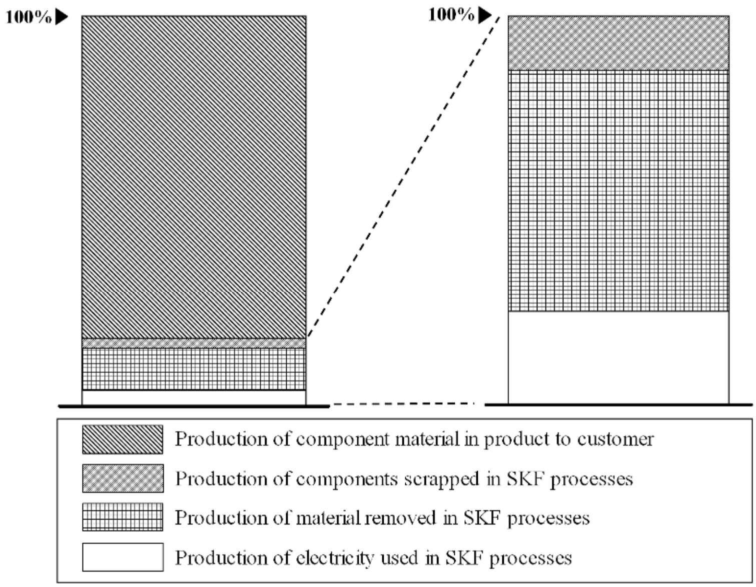

Fig. 3: To left: Distribution of CO_2e emissions for production of one bearing unit (reformulated dominance analysis) - To right: Distribution of CO_2e emissions for manufacturing of one bearing unit at SKF (dominance analysis from the perspective of manufacturing actors at SKF) [6]

Figure 4 shows how Löfgren [6] generalised the method to draw system boundaries in LCA based on a company's manufacturing actors' perspective. In this figure we see how the method disregards part of the production of direct material, and only includes the fraction equivalent to the component material losses from the actor's manufacturing system.

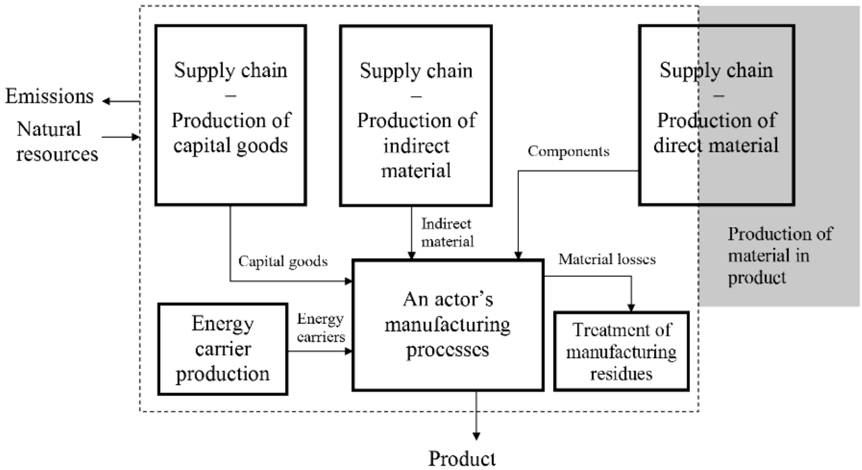

Fig. 4: System boundaries when relating environmental consequences to an actor's manufacturing processes. Supply chain production of capital goods, direct and indirect material, and energy carriers includes all upstream processes required for production. The part corresponding to material in product is partitioned away from the processes for producing direct material. [6]

4 Example 3: Assessing actions by milk chain actors

The challenge in working with environmental improvements is to select the action offering most substantial progress. As we could see in the previous examples, not all actions are open to all actors in a product chain. This study by Berlin et al. [7] demonstrates how LCA may be used to explore the environmental consequences from the actions of several actors the Swedish post-farm milk chain.

The potential improvement actions were first identified in a brainstorming session with the researchers involved in this project, which utilised their understanding of life cycle thinking and LCA methodology, combined with their experience in LCA studies of dairy products.

First, the post-farm milk chain was divided into the main actors; the dairy industry, the retailers and the consumers. Their potential actions to green the milk chain were then listed. Although the same actions were not identified for all actors, they could be sorted under the main strategies of improved energy efficiency, changed transport patterns, reduction of product losses, and use of ecological labelling. The same strategies were highlighted as areas with environmental improvement potential through an analysis of the trends in the dairy sector. It needs to be pointed out that although most of these strategies are relevant for all actors, they would not imply the same action for the different actors. The potential for the actors to improve in any of the four aspects; namely energy efficiency, transportation, product losses, and use of ecological labelling, was then quantified through literature studies and estimations.

The quantified results from the study were presented as the total life cycle environmental impact for milk, cheese and yoghurt separately in the study. The results in Figure 5 are an example of how the result looked like for the actor household and type of undertaken improvement measures for the environmental impact categories of global warming potential, energy use, eutrophication as well as POCP.

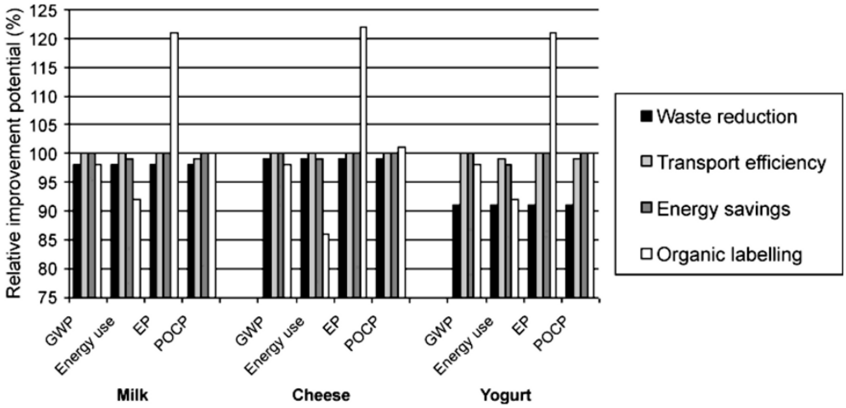

Fig. 5 The household's environmental improvement potential to reduce waste, increase transport efficiency, save energy, and buy organic products, in relation to today's environmental life cycle contributions of milk, cheese, and yogurt. 100 represents the present situation; bars lower than 100 mean improvement, and those above 100 mean impairment. GWP = global warming potential; EP = eutrophication; POCP = photochemical ozone creation potentials [7]

The most efficient improvement actions for the dairies, retailers and households are listed below:

- *Dairy:* No improvement action is clearly superior to the other, but reducing waste appears to contribute to a lower environmental impact for most impact categories for all three products.
- *Retailer:* Decreased use of energy for cold storage and display seem to be the most efficient improvement action.
- *Household (Figure 5):* Reducing waste is the improvement action that gives clearly positive results for all effect categories included. By choosing organic products, the improvements in energy use for milk and cheese appear to be even greater, but the eutrophication rises. Overall, the

household has the largest improvement potential, and yoghurt is the product that offers the greatest improvement.

Two aspects of methodology were highlighted by this study. One is the necessity of the systemic approach, the life cycle perspective, to describe the full effect of a potential improvement, in particular reducing waste. To lower waste decreases all inputs and emissions needed upstream in the system; hence waste avoided later in the chain is more important than that avoided earlier in the life cycle. The second is the usefulness and feasibility of the actor analysis.

5 Example 4: Building chain actors' relative importance

In our final example Brunklaus et al. [8] present an approach for considering the choices of value actors in LCA, similar to our previous example by Berlin et al. [7]. However, in this study LCA results are understood as the sum of choices in order to assess the total potential of greening buildings and the actors' relative importance, rather than bringing forward the most effective actions for each individual actor. This study also introduces the perspective that actors can influence each other by putting demands on each other.

The studied LCA systems (i.e. one conventional housing and two variants of passive housing) were assessed in terms of the value chain actors' green options for electricity supply, transportation and heating. The results were analysed with regard to respective actor's environmental significance for the life cycles of the studies housing cases. By comparing results for conventional and green choices in Figure 6 we see that residents appear to be the most influential environment actor – a conclusion that is different from the common view within the sector, namely that the building constructors are environmentally most important. A summary of conclusions from the 'green choices' scenarios are given below:

- *Residents* have the most environmental influence by choosing eco-labelled electricity.
- *Construction companies* have the least influence with their green transport choices. However, construction companies can recommend that residents choose eco-labelled electricity.
- *The material producer's* importance becomes even greater when residents and construction companies start making greener energy choices.

The introduction of passive house technology shifts responsibilities from district heating producers to residents, due to the technology's reduced heating requirements. This shift of environmentally most significant actor is presently not communicated, and occupants of passive houses may not be aware of the great significance of their choice of electricity supplier has for the overall environmental consequences.

The strength of the methodology applied here is that more emphasis is placed on the interpretation of results and it is therefore more usable for actors. The methodology focuses on identifying the environmentally important actors and actions instead of technology and production phases. Recommendations may also be useful to foster collaboration.

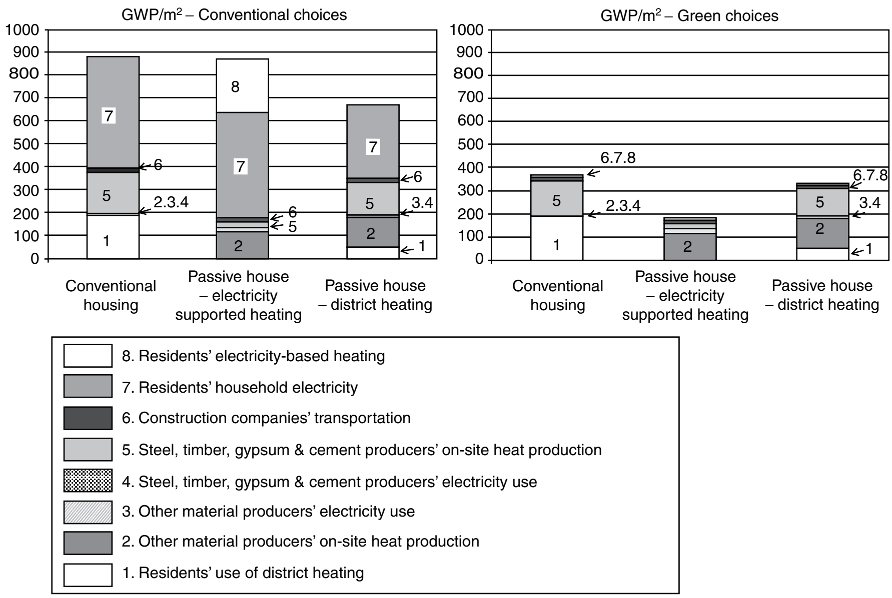

Fig. 6: Environmental impacts of actors in the buildings chain depending on their choice. All actors with either conventional choice (left) or green choice (right). GWP = kgCO_2e modified from [8]

6 Conclusions and outlook

Löfgren [6] points out that the concept life-cycle thinking includes underlying assumptions about a broadened responsibility for actors in the product chain [10]. In the LCA interpretation methods described here, this new responsibility structure becomes more apparent by assessing actors influence on environmental impact. If a company, for example, declares all direct and indirect emissions related to its processes, in accordance to an industrial standard for greenhouse gas

accounting and reporting [11], that company also acknowledges that all these emissions are consequences of its own activities. This mindset represents more than merely systems thinking, since it considers not only the cradle-to-grave implications for the environment, but also implies that the company assumes moral responsibility for them.
LCA studies often do not take this into account, when they are interpreted with dominance analyses to see what life cycle phases and particular environmental loads contribute the most to the overall results. In LCA there are seldom interpretations of the sphere of influence of the various actors along the product chain, giving a concrete example of the need for the inclusion of actors in environmental assessment, as noted by e.g. Berkel et al. [1], Andrews [2], Heiskanen [3] and Baumann [4]. The four examples in this paper has demonstrated the feasibility of such an approach, showing that the life cycle environmental implications of improvement potentials can be quantified on an actor basis and, more important, the usefulness of doing so.
The methodological contributions in the examples provide us with options for modelling LCAs to:

- Identify to which extend the environmental impact is under an actor's control [5]
- Understand how energy and material flows in a specific actor's processes relates to environmental consequences no matter where in the value chain they might occur [6]
- Focus on the environmental consequences that a manufacturing actors in a certain company is able to influence [6]
- Divide LCA results by value chain actors rather than life-cycle phases/processes, and assessing best improvement action for each actor [7]
- Evaluating most influential actor [8], and
- Evaluate the impact from actors' ability to put demand on other actors in the value chain [8].

From our experience, it is clear that these approaches to interpret and present LCA results add value to receivers of results. It is therefore important to continue learning about how this may be done and the practical implications of it. We see an increasing interest in understanding how the practice of people in organisations influences the technical systems, of which one example is the area of organising for the environment [12].

References

[1] Berkel R, Willems E, Lafleur M (1997) The Relationship between Cleaner Production and Industrial Ecology. J Ind Ecol 1(1):51-66.

[2] Andrews CJ (2000) Building a micro foundation for industrial ecology J Ind Ecol 4(3):35-51.

[3] Heiskanen E (2000) Institutionalization of Life-Cycle Thinking in the Everyday Discourse of Market Actors J Ind Ecol 4(4):31-45.

[4] Baumann H (2005) Environmental assessment of organising: towards a framework for the study of organisational influence on environmental performance. Prog Ind Ecol 1(1/2/3):292-306.

[5] Baumann H, Tillman A-M (2004) The hitchhiker's guide to LCA: an orientation in life cycle assessment methodology and application. Lund: Studentlitteratur

[6] Löfgren B (2009) Capturing the life cycle environmental performance of a company's manufacturing system. Licentiate thesis, Chalmers University of Technology

[7] Berlin J, Sonesson U, Tillman A-M (2008) Product Chain Actors' Potential for Greening the Product Life Cycle. J Ind Ecol 12(1):95-110.

[8] Brunklaus B, Thormark C, Baumann H (2010) Illustrating limitations of energy studies of buildings with LCA and actor analysis. Build Res Inf 38(3):265-279.

[9] Ekdahl Å (2001) Life Cycle Assessment on SKF's Spherical Roller Bearing. MSc Thesis, Chalmers University of Technology

[10] Heiskanen E (2002) The institutional logic of life cycle thinking. J Cleaner Prod 10(5):427-437.

[11] Ranganathan J, Corbier L, Bhatia P, Schmitz S, Gage P, Oren K (2004) The Greenhouse Gas Protocol: A Corporate Accounting and Reporting Standard - revised edition, World Resources Institute

[12] <http://www.gronorg.se/index-filer/index_en.htm> (accessed 14.04.2011)

Review on Land Use Considerations in Life Cycle Assessment: Methodological Perspectives for Marine Ecosystems

Juliette Langlois, Arnaud Hélias, Jean-Philippe Delgenès and Jean-Philippe Steyer

Abstract Land use within the life cycle assessment (LCA) methodology deals with the impacts on the environment of occupation and transformation of a piece of land for human activities. Land quality can be altered in its ability to ensure ecosystem services. The present article reviews the different methods used to assess land use impacts on ecosystem quality during life cycle impact assessment (LCIA). Details are provided on the choice of the reference state, areas of protection, indicators and methods which can be used for the assessment. Then the study focuses on the different methodological aspects which need to be investigated to take into account impacts on marine ecosystems marine use in LCA, based on the terrestrial methodological framework previously detailed.

1 Introduction

With the decreasing of fertile land availability, linked to the population pressure, use of marine ecosystems for human purposes is increasing for shipping, waste disposal, military and security uses, recreation, aquaculture and even habitation [1]. The sector of marine renewable energy is still to its infancy, but is growing fast too [2]. Coastal and marine ecosystems belong both to the most productive and the most threatened systems in the world: they are particularly sensitive to changes of use and are experiencing some of the most rapid environmental change [1]. In this context, pollution transfers from terrestrial to marine ecosystems due to habitat change and offshore occupation could increase. In the current state of life cycle assessment (LCA) no characterisation factors allow the assessment of this kind of impacts.

The goal of this study is to determine if the method developed to assess terrestrial land use impacts could be applied to marine ecosystems and to identify

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perspectives of development. Thus a review is first performed, dealing with land use impact assessment during life cycle impact assessment (LCIA). It focuses on land use, considering land as a support of ecosystem services, whose quality can be altered. It gives details on several methodological assumptions needed to: (1) define a framework and choose a reference state as a baseline, (2) define ecosystem services and corresponding areas of protection and indicators, (3) perform the impact assessment at a midpoint and/or endpoint level. Secondly perspectives for a marine application of land use are developed. Using the methodological framework defined in the first part of the study, details are provided on (1) types of human activities which could be assessed within a "sea use" impact category, (2) classification of the impacts due to marine activities identified, (3) existing methods to assess impacts of marine activities during LCIA and their possible development.

2 Overview of land use impact category in LCA

2.1 Definition of land use

Traditionally, land use consists in producing, changing or maintaining a certain land cover type for human purposes [3]. It is also a possible impact category within life cycle assessment (LCA) methodology.

On the one hand, land can be considered as a resource, with a competition for its use. Three points of view are possible [4]. (1) In the simplest form of assessment, every pieces of land are equivalent and land use impacts depending neither on the initial state nor on land characteristics. This approach, used in CML method [5] has the advantage to be applicable worldwide for any type of use. (2) Distinction of land categories can be introduced. It is performed in ReCiPe midpoint method, distinguishing agricultural, urban and natural land. (3) Another possibility is to consider land use as a resource whose stock can be reduced [4].

On the other hand land can be considered as a support of ecosystem services, whose quality can be altered. This has been the most developed approach in the LCA community. Then assessing land use impacts consists in quantifying the qualitative state of an ecosystem, i.e. its ability to perform ecosystem services [6]. In this last case, land use includes impacts of occupation and transformation [7]. It takes into account physical disturbances induced by land use, with consequences on physical and biological properties of ecosystems [8] (consequences of chemical emissions are excluded [6] as part of the inventory under other impact categories).

This last proposal is detailed in the rest of this review, to define a general framework which could be applied for marine ecosystems.

2.2 Framework: land use as a support of ecosystem services

Consensuses exist to define a framework for land use impact assessment [9-10], but are still under construction. We consider an area transformed from a land use type *(I)* to another land use type *(II)* at t_1 and is then occupied during t_{occ}. This second land use creates an amount of impacts on the environment I_{transf} due to land transformation and I_{occ} due to land occupation. Its current quality level before transformation Q_I is degraded (or enhanced) to an altered (or better) state Q_{II}. When land occupation *(II)* stops at t_2, the area recovers a quality level of reference Q_{ref} after a period of restoration t_{rest}. No consensus exists for the choice of this reference. Then differences exist within the methods. Land use impacts can be represented as shown in Fig. 1. On this graph, the *x* axis represents the time, the *y* axis is the quality index and the *z* axis is the area concerned by the use.

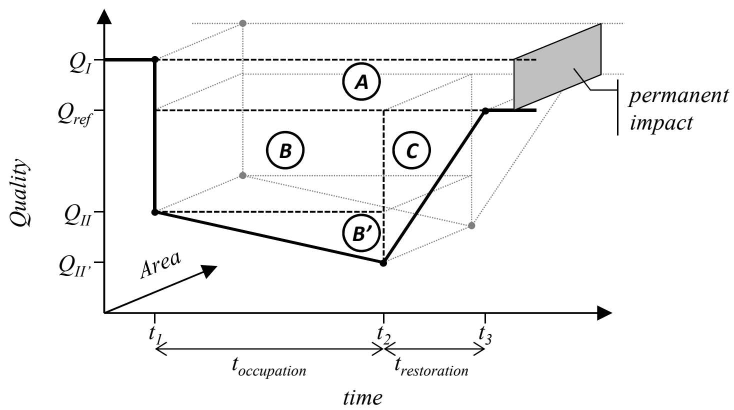

Fig. 1 : Representation of land use impacts due to transformation and occupation processes (adapted from Mila i Canals, 2007 [11])

I_{transf} can be represented by the section *a* of the volume *A* (corresponding to the permanent or irreversible impacts) and I_{occ} by the volumes *B*, *B'* and *C* [11]. Many variations exist: (1) by neglecting the initial state Q_I [8], (2) by neglecting permanent impacts and considering volumes *B* and *B'* as I_{occ} and *C* as I_{transf}, (3) by neglecting quality changes *B'* during the occupation phase (then occupation can be approximated as a simple postponement of the recovery phase).

As shown on Fig. 1, impacts strongly depend on the choice of the reference state. When considering the land use type (I) as the reference, then no permanent impacts can be defined within land use impacts. When considering the reference as a conceptual initial state without any human influence [12,13], with Q_I smaller than Q_{ref}, then permanent impacts are assessed. This conservative choice of reference causes problems of allocation within the successive previous activities. It is also possible to consider the reference as the new steady state expected after a relaxation time. It can be an actual regional [14,15] or a historical reference [6]. This choice of reference is depending on data availability but is also subjective.
Quality index calculation is depending on the functions of land use assessed in the method considered. Many different functions and corresponding indicators exist.

2.3 Choice of impact pathways and corresponding indicators

Quality index Q depends on which ecosystem functions (or ecosystem services from an anthropocentric point of view) the author focuses on [16]. Many ecosystem services exist [1] and corresponding impact pathways can be assessed during LCIA (biodiversity, life support or environmental regulations support functions). A variety of indicators can be used for the assessment. They have to be sensitive (to highlight differences between a variety of land occupations and transformations), site-specific, but applicable anywhere on the globe [9,16].
The most common is to assess the biodiversity support function within LCA, because biodiversity has positive effects on the main land ecosystem services [1]. Furthermore habitat change, loss or degradation is one of the main anthropogenic drivers of biodiversity loss [1]. Definition of biodiversity is very large [17], with more than 70 possible biodiversity indicators [18]). Species richness (SR) is the most commonly used in LCA, for vascular plants [8,10,14,19,20,21] or other taxons [22]. SR is a debatable indicator, not giving a particular value to species indigenous, endemic, invasive alien, rare, endangered or key species. The proportion of threatened species have sometimes been taken into account in LCA [6,22,23] (through the red lists), as well as the per cent coverage of invasive species in protected areas[24]. Nevertheless the lack of data availability for every land use types and for every bio-geographical region limits its use.
Scarcity and vulnerability of an ecosystem are part of its biodiversity value too [6,8,25]. Indicators can be the relative areas used within a certain ecosystem [8,19,26] or the use of a species-area relationship (SAR), expressing non-linearity of biodiversity within space [16,19,27].

It is also possible to assess the life support function of land, by measuring its biological production capacity. Free net production potential (fNPP) [13,20], or net production potential (NPP) [8] can be used. NPP is the net carbon uptake of the ecosystem (fixation through photosynthesis minus respiration). fNPP represents the NPP minus the amount of carbon sequestered for human use.
It is also possible to assess environmental regulations support functions of land. Soil quality can be assessed [6], using indicators as surplus energy needed to restore soil quality, deficit of soil organic matter [11] or cationic exchange capacity. At a more global scale, climatic regulation function can be assessed using carbon storage in soil and vegetation [28].
Land quality can also be assessed thanks to the hemeroby concept using naturalness [12], to thermodynamic concepts using thermal parameters, or to ecosystem fragmentation [20]. No operational aesthetic and cultural assessment has been built, each case being a particularity.
Depending on the choice of the functions we want to focus on and on the indicators chosen to reflect impacts of land use on this function, different types of assessments are possible. The main methods are detailed in the next section.

2.4 Main methods of impact assessment for land use

Theoretically, each type of land use could constitute a new separate impact category [6], but for more practicality they are aggregated into one or a few impact categories. As impacts depend both on the type of land use (type of coverage and intensity of use) and on bio-geographical conditions [9] (type of biome), characterisation factors should be calculated for all these cases [7,14,19]. As seen previously (sections 2.2 and 2.3), these factors depend on the type of damage assessed. Furthermore methods are developed either at midpoint or endpoint level (*e.g.* biodiversity can be either an indicator for species composition of ecosystems, mid-point for many impact chains, and as an end-point itself [8]). Thus it becomes hard to build a consensual methodology at that point.
With the recent development of endpoint damage categories in the last ten years, land use has been significantly improved. We will detail EcoIndicator99 [14] and ReCiPe-endpoint [7] methods, within the main methods of land use impact assessment used in LCA. The ecological damages of land use activities on biodiversity have been quantitatively analysed thanks to cause-effect chains modelling [19]. A meta-analysis was performed, allowing the calculation of a log-log SAR for each land use type assessed. Local damages (reflecting the impacts on species occurrence) and regional damages (expressing the vulnerability of land)

were aggregated [19]. Several operational methods are based on this method for land occupation impact assessment: Ecological scarcity 2006, impact 2002+. Some other authors [23,26], used expert knowledge to score levels of impact on biodiversity, but it is generally limited to a precise region (respectively Norwegian forests and Swiss agricultural lands). Methods considering impact pathway for biodiversity have been recently reviewed [9,18]. The global current trend is to use sets of indicator and/or to couple it with GIS [21], to ensure more exhaustive and accurate assessments.
This review shows that many questions remain to assess impacts of land use within LCA. Nevertheless it highlights which methodological framework need to be defined to apply it to other types of ecosystems: choice of a reference state, distinction of occupation and transformation impacts, availability of indicators reflecting the quality index reached by the ecosystem at performing its ecosystem services, and choice of assessment method. These methodological points are studied for marine ecosystems in the next section.

3 Is the impact category land use applicable to the sea?

Because marine activities become more and more important [1], there is a need for a tool able to assess global environmental impacts of these activities. After defining specificities of the seas in terms of types of activities, nature of their impacts, some proposals of development within the LCA framework are provided to take this concern of marine use into account.

3.1 Typology of the main marine activities

Drivers of human impacts on marine ecosystem include [30] (1) effects of land-based activities, (2) climate change, (3) ocean pollutions, (4) invasive species, (5) benthic structures (as oil rigs), (6) commercial and artisanal fishing, (7) commercial activities (shipping lanes). Within these drivers, leading to consistent impacts, only (5), (6) and (7) could be assessed under a “sea use” impact category. In land use impact assessment, effects of activities which occur outside from the area studied are not part of the assessment. Thus (1) is not relevant. Climate change as well as chemical emissions in seawater are already assed, thus (2) and (3) are not relevant neither. Increase in invasive species (4) is rather an indicator of biodiversity (see section 2.3) thus it is not included within sea use neither.

For the remaining categories, some parallels with terrestrial ecosystems can be done. The benthic structures (5) are constructions dedicated to oil extraction. In the LCA framework, this category could rather be separated in two categories. First category would be a generic construction, and then oil extraction itself would constitute another category (whose impacts sometimes belong to other impact categories). This separation could allow an extension to all kinds of constructions. Other specific effects of the structure (as petroleum emissions for oil rigs [30], impacts on birds for wind turbines [2] and so on) are not studied in this paper.
Fishing activities (6) consist in a biotic resource extraction, which is considered out from the land use framework in terrestrial assessment. Nevertheless they can be incorporated within sea use by considering some subcategories detailing the types of ecosystem (e.g. "commercial fishing in fishy area" or "commercial fishing area in non-fishy area"). It allows a distinction between the resource extraction itself and the impacts linked with cropping or picking practices (e.g. habitat degradation because of trawls).
Shipping lanes (7) can also be included within sea use classification, and extent to every type of ship (either fishing or recreational ship).
Besides these activities, aquaculture of fish, molluscs or algae can be added, considering only impacts of their practices on the environment, because substructures impacts will be taken into account in the category "construction", and resource extraction in "biotic resource extraction".
Thus in the framework of a LCIA, sea uses can be categorised by their (1) use of space for constructions or navigation (2) extraction of biotic resources through picking or cultivation. In the reality, an activity can be a mix of these categories (e.g. commercial fishing using trawls consists in a mix of "navigation" and "destructive fishing" within sea use and of "extraction of biotic resources"). Because impacts due to these activities need to be further aggregated, using common indicators, it is important to provide as precise classification of the nature of their impacts.

3.2 Sea use impacts based on water layers classes

This section is dedicated to the sea use impacts themselves. The goal is to identify the nature of these impacts. Contrary to terrestrial ecosystems, defined by the area they cover, marine ecosystems are defined both by their area and their depth, covering volumes instead of areas. Then impacts will be strongly depending on the zone where the activity occurs, because biological (biomass quantity and

quality) and physical properties (light availability) of marine ecosystems vary with their depth [30].
Marine environment can be divided into two zones: pelagic and benthic zones (respectively the water column and the seafloor). From a biotic point of view, species distribution is strongly depending on this light in marine ecosystem. Light only reaches the shallowest compartments, called euphotic or photic zones. As plant species need light to perform photosynthesis, there is not any form of vegetal life out of this photic zone. Animals are living both in the photic zone and in the depths. The rest the pelagic zone (meso-, bathy-, abysso- and remaining epipelagic zones) are not detailed and rather considered as a unique water column [30]. The same distinction between photic and euphotic benthic zone is considered.
We focus on effects on flora due to lighting modifications, and on fauna due to surface and/or volume of habitats occupation and transformation. Impacts of noise, sedimentation rates are not included. Considering human activities selected in section 3.1, impacts can be classified as described in Tab. 1.

Tab. 1: Impacts of marine activities on marine ecosystem layers

	Photic benthic zone	**Aphotic benthic zone**	**Photic pelagic zone**	**Aphotic pelagic zone**
Construction anchored on the sea floor	Seafloor_{destr} - Art. Reef - Postponement - Relaxation $\text{Habitat}_{surface}$		-	-
Construction floating in aphotic pelagic zone	-	-	-	Art. reef Habitat_{volume}
Construction floating in photic pelagic zone	Shading	-	Shading Art. Reef Habitat_{volume}	-
Navigation	Shading	-	Shading	-
Destructive fishing (trawls)	Seafloor_{destr} Relaxation	-	-	-
Non-destructive fishing	-	-	-	-
Aquaculture	Shading		Shading (Art. Reef)	

Seafloor_{destr} = transformation impact due to destruction of the seafloor
Art. reef = occupation impact due to creation of artificial reef
Postponement = occupation impact due to postponement of the relaxation phase
Relaxation = occupation impact due to the relaxation phase after destruction
Shading = occupation impact due to light depletion
$\text{Habitat}_{surface}$, Habitat_{volume} = change in the surface or volume of habitat

Table 1 highlights that construction impacts are highly variable depending on the localisation. There is no impact of transformation due to constructions in the pelagic zone because space is occupied but without any destruction (reversible change in the ecosystem quality, by displacement). When located above water layers usually highlighted they also create some shadow, as can do any floating structure. It is also important to note that marine constructions have not only

negative effects on the ecosystems: they constitute artificial reefs, and/or fish aggregation devices, increasing the fauna density around them [2].
Thus an impact category “sea use” should distinguish characterization factors for sea bottom use, euphotic water column use and non-euphotic water column use, on top of the biogeographical characteristics on the ecosystem and of the marine activities.

3.3 Review for sea use in LCA and perspectives

Within all marine activities, only fishing has been assessed through LCIA. Small size ratio, landed by-catch, discarded by-catch [31], seafloor impacts (due to the trawls) [31,32] and appropriation of NPP (aNPP, based on carbon content and trophic level of the species harvested) [33] categories have been added to classical LCA. Effects on the seafloor are assessed based on marine habitat recoverability using a fishing intensity index (FII) [32]. It has been calculated using GIS and coupled with a classification of the seafloor habitats [32], showing that fishing pressure is different between habitats (sand and gravels being potentially less affected than rocky and muddy habitats). The percentage of each habitat considered not to recover between fishing events could then be used as an indicator for the seafloor impact category within LCA. The first three indicators give information on the renewability of the biotic extracted resource. Thus it belongs rather to the “biotic resource extraction” impact category.
The indicator reflecting seafloor impacts is much more interesting to assess impacts of other marine activities. Negative impacts of constructions on the seafloor could be assessed through a high FII, and positive effects of artificial reefs could be assessed through a negative FII. Because efficiency of an artificial reef is not necessarily the same depending on its apparent area, different values of FII could be attributed in function of this parameter. aNPP has been built for fisheries activities, and is then directly applicable for aquaculture. As any horizontal surface stops penetration of light within euphotic zones or floating constructions, aNPP could be approximated a certain amount of the maximal NPP in the area for floating constructions, depending on their depth.
In 2000, SR-based LCA methods could not be applied to the sea by lack of data [20]. In marine and coastal ecosystems, biodiversity is a lot less known than it is in terrestrial ecosystems [1]. Knowledge on threatened species exists only for chondrichtyan species [1]. Fragmentation would not be relevant either. At the current state of knowledge marine use impacts on biodiversity could only be assessed at the ecosystem scale, with scarcity and vulnerability.

There is no consensus on how to classify marine ecosystems [30]. It could be limiting to build typology of uses and characterisation factors, as impacts are highly depending on bio-geographical conditions. For the reference state, regional average data would not be relevant, as marine ecosystems are already strongly altered by human activities [30]. Thanks to the development of protected areas data could be available through this way, using protected areas as reference.
To conclude, it could be possible to assess marine use impacts of constructions, shipping, fisheries and aquaculture at a midpoint level using the framework developed for terrestrial ecosystems, using FII and/or aNPP. Inventory data would be more important, reflecting the complexity of an environment in three dimensions: horizontal area occupied in euphotic pelagic zone and depth of occupation, apparent area occupied in euphotic and aphotic zones, volume occupied in pelagic zone, area of occupied and transformed benthic zones, time of occupation and time of relaxation.

4 Conclusion

The review on land use impacts highlights that there is no consensual framework within the LCA community. As it exists at its current stage, land use impact category could theoretically be applied to marine ecosystems. After defining the types of marine activities and of impact which could be assessed within a sea use impact category, it appears that some indicators (fishing intensity index or appropriation NPP) could be used as generic indicators. Only a lack of data to build the characterisation factors could limit perspectives of development.

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References

[1] Millenium Ecosystem Assessment (2005) Ecosystems and Human Well-being: Current State and Trends, Island press
[2] Inger R, Attrill MJ, Bearhop S, Broderick AC, Grecian WJ, Hodgson DJ et al (2009) Marine renewable energy: potential benefits to biodiversity? An urgent call for research, J Applied Ecol 46(6):1145–1153.

[3] FAO/UNEP (1999) The future of our land - Facing the challenge, Rome
[4] Heijungs R, Guinée J, Huppes G (1997) Impact categories for natural resources and land use. CML Report (138)
[5] Guinée J, Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A. et al. (2001) Life Cycle Assessment - An operational guide to the ISO standards, CML. Bureau B&G, Delft University of Technology, Netherlands
[6] Müller-Wenk R (1998) Land-use-The Main Threat to Species, How to Include Land Use in LCA. IWÖ–Diskussionsbeitrag
[7] Goedkoop M, Heijungs R, Huijbregts M, De Schryver A, Struijs J, van Zelm R (2009) ReCiPe 2008, A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Ministerie VROM, Den Haag
[8] Weidema BP, Lindeijer E (2001) Physical impacts of land use in product life cycle assessment. Technical University of Denmark
[9] Mila i Canals L, Bauer C, Depestele J, Dubreuil A, Knuchel R, Gaillard G et al. (2007) Key Elements in a Framework for Land Use Impact Assessment Within LCA. Int J Life Cycle Assess 12(1):5-15.
[10] Bare J (2010) Recommendation for land use impact assessment: first steps into framework, theory, and implementation. Clean Technol Environ Policy 13(1):7-18.
[11] Schmidt JH (12008) Development of LCIA characterisation factors for land use impacts on biodiversity. J Cleaner Prod 16(18):1929–1942.
[12] Mila i Canals L, Romanya J, Cowell S (2007) Method for assessing impacts on life support functions (LSF) related to the use of « fertile land » in Life Cycle Assessment (LCA). J Cleaner Prod 15(15):1426–1440.
[13] Brentrup F, Küsters J, Lammel J, Kuhlmann H (2002) Life cycle impact assessment of land use based on the hemeroby concept. Int J Life Cycle Assess 7(6):339–348.
[14] Blonk H, Lindeijer E, Broers J (1997) Towards a methodology for taking physical degradation of ecosystems into account in LCA. Int J Life Cycle Assess 2(2):91-98.
[15] Goedkoop M, Spriensma R (2001) The Eco-Indicator 99, A damage oriented method for Life Cycle Impact Assessment. Ministerie VROM, Netherlands
[16] Koellner T (2000) Species-pool effect potentials (SPEP) as a yardstick to evaluate land-use impacts on biodiversity. J Cleaner Prod 8(4):293–311
[17] Müller-Wenk R, Brandao M (2010) Climatic impact of land use in LCA-carbon transfers between vegetation/soil and air. Int J Life Cycle Assess 15(2):172-182.
[18] United Nations (1993) Text of the Convention on Biological Diversity. Rio de Janeiro
[19] Curran M, de Baan L, De Schryver A, van Zelm R, Hellweg S, Koellner T et al. (2011) Toward Meaningful End Points of Biodiversity in Life Cycle Assessment. Environ Sci Technol 45(1):70-79.
[20] Lindeijer E (2000) Biodiversity and life support impacts of land use in LCA. J Cleaner Prod 8(4):313–319.

[21] Geyer R, Stoms DM, Lindner JP, Davis FW, Wittstock B (2010) Coupling GIS and LCA for biodiversity assessments of land use. Int J Life Cycle Assess 15(5):454-467.

[22] Koellner T, Scholz R (2008) Assessment of land use impacts on the natural environment. Part 2: Generic Characterization Factors for Local Species Diversity in Central Europe. Int J Life Cycle Assess13(1):32-48.

[23] Jeanneret P, Baumgartner DU, Knuchel RF, Gaillard G (2008) A new LCIA method for assessing impacts of agricultural activities on biodiversity (SALCA-Biodiversity). Life Cycle Assessment in the Agri-Food Sector Conference, Zurich

[24] Schenck RC,(2001) Land use and biodiversity indicators for life cycle impact assessment. Int J Life Cycle Assess 6(2):114–117.

[25] Cowell S (1998) Environmental life cycle assessment of agricultural systems: Integration into decision-making, Surrey

[26] Michelsen O (2007) Assessment of land use impact on biodiversity. Int J Life Cycle Assess 13(1):22-31.

[27] Nelson E, Mendoza GJ, Polasky RS, Tallis H, Cameron Dr et al. (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ 7(1):4-11.

[28] European Commission, Joint Research Centre - Institute for Environment and Sustainability (2010) International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. Luxembourg

[29] Halpern BS, Walbridge S, Selkoe KA, Kappel CV, Micheli F, D'Agrosa C et al. (2008) A Global Map of Human Impact on Marine Ecosystems. Sci 319(5865):948-952.

[30] Ziegler F, Emanuelsson A et al. Extended LCA of southern pink shrimp (Penaeus notialis) products exported to Europe from Senegalese artisanal and industrial fisheries. Submitted.

[31] Nilsson P, Ziegler F (2007) Spatial distribution of fishing effort in relation to seafloor habitats in the Kattegat, a GIS analysis. Aquatic Conservation: Mar Freshwater Ecosyst 17(4):421-440.

[32] Pelletier NL, Ayer NW, Tyedmers PH, Kruse SA, Flysjo A, Robillard G et al. (206) Impact categories for life cycle assessment research of seafood production systems: Review and prospectus. Int J Life Cycle Assess 12(6):414-421.

Visual Accounting

Andreas Moeller and Martina Prox

Abstract Computers can support life cycle management in different ways. Normally, personal computer software in this field like Gabi, Simapro or Umberto are characterised as stand-alone solutions. They are normally not integrated into the computer-based information systems infrastructure of corporations. It would be better to have a fully integrated module within standard enterprise resource planning systems to provide decision support in the field of life cycle management. The question is why the stand-alone software tools are successful while still no software modules for enterprise resource systems are available. This paper examines the reasons for the success of computer tools today. Therefore, the first two chapters take a critical look at the theoretical background of instruments in the field of life cycle management, mainly life cycle assessment (LCA): the assumption that the instruments are decision support instruments.

1 Introduction

The application domain of tools and instruments in the field of life cycle management seems to be established: "LCA is a decision support tool" [1]. The perception behind is that decision makers use tools and instruments to take rational decisions. They should select the alternative (e.g. product variant, production technology, supply chain) with the lowest impact on the environment. If all decision makers incorporate aspects like climate change or resource scarcity, the overall eco efficiency of societies can be improved substantially. Consequently, it is of high importance to integrate the instruments into the IT infrastructure of companies so that all everyday decisions and actions are supported by these instruments. The most important type of software in this domain is enterprise resource planning systems. However, after more than ten or fifteen years of research in this field, the application of these approaches is marginal.

On the other hand, software tools for life cycle assessment are quite successful. The metaphor "tool" is used to characterise this kind of computer utilisation:

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computers extend the capabilities of the users; they help to design, support experiments with the models (e.g. sensitivity analyses) and visualise the results in an appropriate way. The focus is on better understanding complex situations, identifying current or future problems and on finding innovative solutions.

2 Related works

As mentioned in the introduction, the application domain of instruments like life cycle assessment, material flow cost accounting and others are rational decision making. This embedding is based on specific assumptions about human problem solving. Winograd and Flores call this the rationalistic tradition [2]: Complex problems of societies, natural environments and non-sustainable developments in this relationship are transformed in to management problems.

In fact, the methodological structure of life cycle assessment with steps like goal definition, inventory analysis, impact assessment, valuation and interpretations can be interpreted as a formalised and standardised information processing systems (IPS), presented by Newell and Simon in the early 1970ties: "Our theory of human thinking and problem solving postulates that the human operates as an information processing system" [3].

However, Winograd and Flores point out that "nearly every writer on management and decision making draws a dichotomy between two kinds of managerial situations: 'programmed vs. non-programmed decisions' [4], "structured vs. unstructured problems' [5], 'established' vs. emergent situations' [6]" [7]. The rationalistic tradition emphasises mainly established situations, structured problems and programmed decisions. In such a case, it is self-evident to implement decision support instruments as modules of enterprise resource planning systems. And the methodological structure of life cycle assessment helps to design the required software components (e.g. calculation engines) and to specify the data demand of the instruments.

In emergent situations with unstructured problems the situation is quite different. Winograd and Flores introduce a new language to deal with these constellations. "Instead of talking about 'decisions' or ' problems' we can talk of 'situations or irresolution', in which we sense conflict about an answer to the question 'What needs to be done?'…The process of reaching resolution is characteristically initiated by some claim that generates a mood of irresolution" [8]. Claims in the field of sustainability are for example the Stern report, eco taxes, national carbon dioxide emission reduction targets etc. The life cycle assessment community

provides a set of instruments and procedures to incorporate these new claims into decision making.
However, such an enhancement of systemic mechanisms cannot be conducted directly. The question is how societies and organisations deal with the emergent situation of sustainable development and resulting challenges and problems. In this regard, Habermas' theory of Communicative Action [9, 10] could provide new insights, in particular the application of the theory on the rationalisation of society ("occidental rationalism" [11]). Habermas' distinguishes lifeworld and subsystems like the society's economy. In his concept, economic efficiency, cost cutting and profit maximisation are generalised action orientations in the subsystem economy. Societal subsystems the result of a "social evolution as a second-order process of differentiation: system and lifeworld are differentiated in the sense that the complexity of the one and the rationality of the other grow" [12]. The process of occidental rationalism is triggered by communicative action, as an attempt of the society or organisation to deal with the new claims and the resulting "breakdown" of conventional patterns of action. In such a situation, societies and organisations exploit the "potential for rationality in communicative action. Action oriented to mutual understanding gains more and more independent from normative contexts. At the same time, even greater demands are made upon this basic medium of everyday language; it gets overloaded in the end and replaced by delinguistified media" [13]. That's the paradox of communication: It is a co-evolutionary process and "modern societies attain a level of system differentiation at which increasingly autonomous organisations are connected with one another via delinguistified media of communication: these systemic mechanisms – for example, money – steer a social intercourse that has been largely disconnected from norms and values, above all in those subsystems of purposive rational economic and administrative action that, on Weber's diagnosis, have become independent of their moral-political foundations" [14]. We can call such a phase, in which communication plays an important role to deal with breakdowns and situations of irresolution as a transition phase. Societies and organisation establish new generalised action orientations and systemic mechanisms. With regard to sustainable development, generalised action orientations may include that all economic processes have to take place within a carbon-neutral society.
As a result, life cycle management can be interpreted as a new management approach that focuses the transition phase of companies to more corporate sustainability. As long as sustainability is not part of daily routine operations, this new kind of management requires a special kind of software support (and enterprise resource planning systems cannot provide this new kind of support): Not only information support enhancing rational decision making but also support for effective communication [15]. First life cycle thinking (and other images of

life cycle assessment and life cycle management) must become part of the social reality before they are self-evident background of routine operations.
Software that supports the transition phase has to visualise the novel problems and challenges of sustainability (scarcity of resources, climate change etc.). And they should provide good arguments in communication processes. Therefore, the tools implement the new “language” of life cycle management, including e.g. life cycle, process, inventory, impact category etc. But the language includes as well images: flow charts, Sankey diagrams, typical bar charts for impact assessment, the cost matrix of material flow cost accounting etc. So, an important feature of software tools should be to make it possible to speak about the challenges of sustainability.

3 Languages and visualisations

The most important conclusion from Habermas' theory is that the emerging situations of irresolution result from communications processes and that effective communication processes help to transform emerging situations into established. In this regard computer instruments play a completely different role. They are involved into the communication processes within the organisations.
In such a constellation we can characterise the application domain of software support as follows:

- Software systems serve as tools to deal with situations of irresolution. The question is: What needs to be done?" [8]. It is necessary to develop a (linguistic) meta model. The meta model describes how users apply the software systems in the field of life cycle management: What are typical semi-structured situations, problems? How can we make the situations visible? How can we transform them step by step into programs and maybe later programmed problems?
- The software tools have to support effective communication. As a consequence, the software system has to speak the same language as the application domain and they have to carefully introduce the new languages of the instruments, e.g. with aid of descriptions and examples. So, the user interface of the tools specifies the language of new instruments and methodological frameworks - like for example life cycle assessment.
- Therefore they introduce words or provide new interpretations: 'process', 'flows', 'material', 'life cycle' or 'database'. Often, we have to distinguish different languages and dialects. For example, life cycle assessment tools can be applied to calculate product carbon footprints. However, it would

help the "carbon footprint dialect" to support the users better. For example the users expect the results in a specific form (tables and charts). Material flow cost accounting - even if the same core algorithms can be applied - speaks a different language: Quantity centers instead of processes, cost classifications, cost matrix, negative products etc.

- They recommend typical visualisations: stacked bar charts for life cycle impact assessments, typical flow charts, e.g. in form of Sankey diagrams, and typical table like the cost matrix in case of material flow cost accounting (MFCA). Visualisations are important because the situations of irresolution are as well a situation of missing images. We do not have a clear picture of the situation. Flow charts, Petri nets, Sankey diagrams, Pie chart etc. should fill the gap.

LCA tools should speak the language of life cycle assessment, MFCA tools the language of material flow cost accounting, the language of carbon footprinting should be a dialect of LCA. That does not mean that it is necessary to develop different software tools. All these tools use the same core algorithms, data handling and databases. But the user interface is different. To integrate all the instruments into one software framework, a meta data repository is required. Such a repository describes (1) the application domains by meta models and a dictionary (table 1), (2) the work flows (typical modelling steps as specified for example in the methodological framework of life cycle assessment) and (3) the required tool chains and algorithms.

Tab. 1: Dictionary within the meta data repository

Life cyle assessment	Cost accounting	Material flow cost accounting
Process	Cost center	Quantity center
Material	Cost type	Cost type
…	…	…

We use material flow cost accounting (MFCA) as an example to illustrate how the meta models have consequences on the design of software instruments. Figure 1 shows the modelling user interface of an MFCA software prototype. Form a life cycle assessment this figure shows nothing special. But the application domain is different. We are in the world of managerial accounting and in this world the lists and tables dominate the mental models. So, for cost accounting experts such a flow chart is a big step forward. To be compatible with the "cost accounting" language it is necessary to speak this language (see Table 1): The flows are called costs, the names of the objects are cost types or cost items. A next learning step is that the processes are not called cost centres. The material cost accounting experts

want to emphasise the strong relationship between the cost centres and the physical processes. So they decide to call the processes "quantity centres".
Dictionaries as presented in Table 1 should not contain only direct translations but as well descriptions and tooltips. These additional fields of such a table help to integrate a domain-specific help system. That requires that the meta data repository is not paper-based or a word documents. It needs to become a component of the software framework.
Figure 2 shows another consequence of filling the meta model repository and in particular the repository with information: the gaps. For example cost accounting provides modelling objects that can be called "cost centre groups". Cost centre groups are collections of cost centre. Often these modelling elements are used to represent infrastructures like buildings or transport vehicles. So, the cost type group "bakery" is not only a free rectangle on the canvas. There is a logical net element behind: It is possible to assign costs, input and output flows to these elements. Special allocations algorithms, derived from cost accounting approaches, are applied to assign the resulting flows and impacts to the embedded processes.

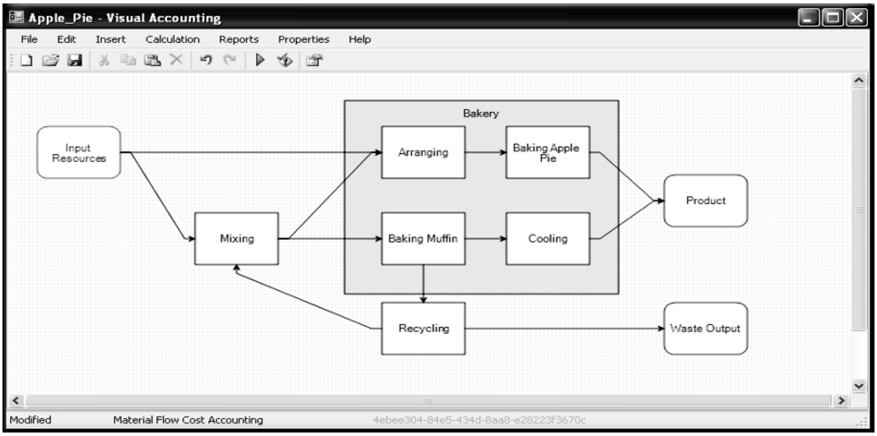

Fig. 1: Material flow cost accounting software prototype

Sometimes it is required to introduce additional concepts, which are required to present the results in an appropriate way. In case of life cycle assessment, life cycle costing and carbon footprinting the life cycle phases play an important role. For instance stacked bar charts are used to present the contributions of different life cycle phases to environmental impacts. So the life cycle phase becomes a new attribute of flow chart elements like processes, links and flows. Figure 2 shows the introduction of life cycle phases as a new element of the network editor.

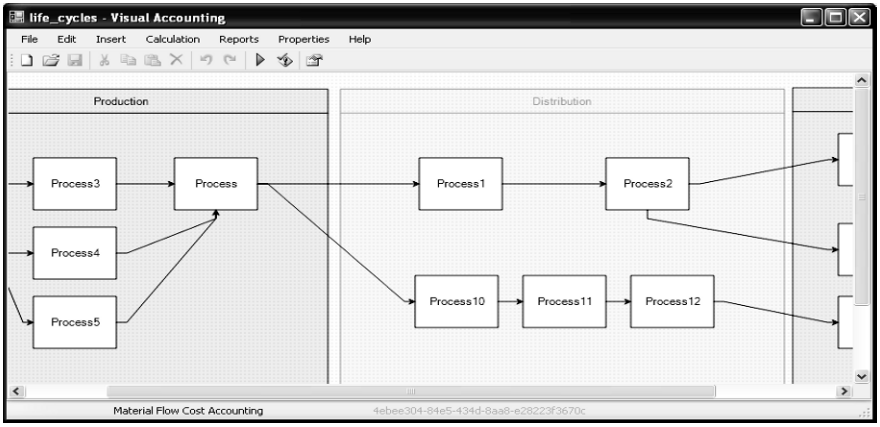

Fig. 2: Visualisation of life cycle phases within the flow chart editor

The next figure shows typical tables and charts. They are embedded into the flow chart editor. The model consists not only of the required model elements like processes and links. The editor can handle with objects, which are not part of the theoretical model: frames, lines etc. A special kind of these free elements are tables and charts. The software tool updates the content of these editor elements automatically after finishing the calculations.

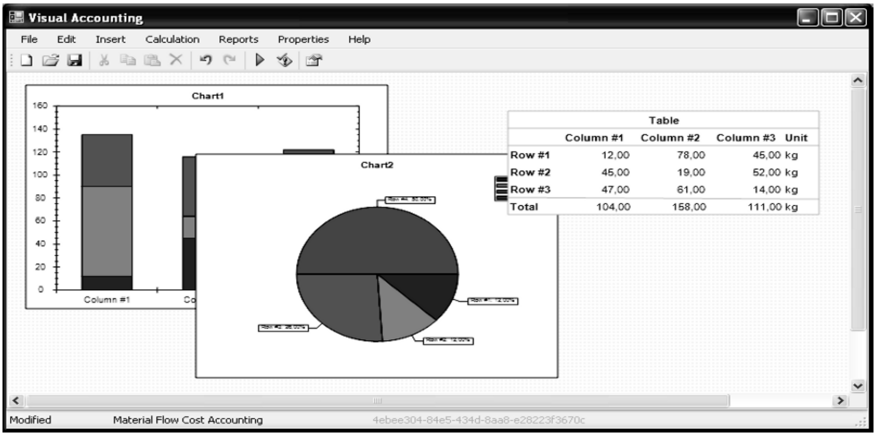

Fig. 3: Charts and tables as flow chart editor elements

The main purpose of integrating tables and charts into the network editor is that this solution better support interactive modelling: (1) a better link between the steps specification, calculation and evaluation and as a consequence (2) an enhanced support for experiments with the model. This is an important aspect

concerning the semi-structured and not well-understood situation. The experiments give hints to the question "what needs to be done". Such a design of human-computer interfaces this is called direct manipulation [16]. Users think they manipulate directly the processes and flows - and they see the effects and impacts.
Another step to support interactive modelling is to support better the input side. Normally, the data input seems to be fixed. We discuss the data quality of real existing processes, flows and stocks. However, the question "what needs to be done" results in future-oriented models: beside status-quo analyses mainly different scenarios. System dynamics software tools normally provide the required editor elements: switches, sliders for input parameters, buttons to start the calculation etc. Life cycle assessment tools do not have these elements

4 Conclusions

Integration of instruments of life cycle management like life cycle assessment is a persistently hot issue for many researchers in the field of business informatics. Often we read about time and effort to conduct the assessments and the poor connection to available data. The consequences of poor integration into the IT infrastructure can be significant: rejection of the whole framework, frustration after first tests and experiments, and lacking images how to integrate the aspects into daily routine. In our own research on this subject we have turned up new insights that are very helpful especially if we interpret the introduction of life cycle management as a transition phase, in which effective conversation plays an important role. The problem in such a transition phase is not efficient decision support; the problem is to bring forward effective conversations in the field of life cycle management.
Software support in this filed has to address the question, how it affects communication processes and how such kind of IT support should be designed. Much of the literature on communication support focuses on Web 2.0 and social networks. However, these software platforms are not designed to provide trustable arguments about the impacts of human action on the environment. Here instruments like life cycle assessment come into play. In other words, we interpret the results of these instruments as "good arguments" in conversation processes. And these processes are reactions on the question: What needs to be done [8] ?
So, the software design in this field begins with the realisation that the instruments are used in conversations. And the purpose of the software tools is to become a facilitator of those conversations. As a consequence, we have to embed them by

speaking the same language and by presenting the results as arguments which can be appreciated by the participants.

The design metaphor we present in this contribution is "Visual Accounting". Based on many years of experiences in this field and the surprising of software tools for designing Sankey diagrams, we found out that languages and visualisations play an important role in designing software tools in this field. This is in line with. So the challenge is how to express a semi-structured situation and problem with aid of words, numbers and especially with aid of visualisations. This practical experiences and insights are in line with recent research in the field of software engineering [17].

As mentioned above, the software tools play an important role in the transition phase. It's the same paradox as the paradox of communication: The purpose of the software tools is to replace them as soon as possible by higher efficient modules and components of enterprise resource planning systems.

References

[1] Werner F (2005) Ambiguities in Decision-oriented Life Cycle Inventories. Springer, p. 41

[2] Winograd T, Flores F (1988) Understanding Computers and Cognition. 3rd edn. Addison-Wesley, p. 14

[3] Newell A, Simon HA (1965) Human Problem Solving, Prentice-Hall, 2, p. 19

[4] Simon HA (1965) The Shape of Automation for Men and Management. Harper and Row

[5] Keen PGW, Scott-Morton MS (1978) Decision Support Systems: An Organizational Perspective. Addison-Wesley

[6] Boguslaw R (1965) The New Utopians: A Study of System Design and Social Change. Prentise Hall

[7] Winograd T, Flores F (1988) Understanding Computers and Cognition. 3rd Ed. Addison-Wesley, p. 153

[8] Winograd T, Flores F (1988) Understanding Computers and Cognition. 3rd Ed. Addison-Wesley, p. 147

[9] Habermas J. (1984) The Theory of Communicative Action. Volume 1, Reason and the Rationalization of Society. Beacon Press

[10] Habermas J. (1984) The Theory of Communicative Action. Volume 2, Lifeworld and System: A Critique of Functionalist Reason. Beacon Press

[11] Habermas J. (1984) The Theory of Communicative Action. Volume 1, Reason and the Rationalization of Society. Beacon Press, p. 143

[12] Habermas J. (1984) The Theory of Communicative Action. Volume 2, Lifeworld and System: A Critique of Functionalist Reason. Beacon Press, p. 153

[13] Habermas J. (1984) The Theory of Communicative Action. Lifeworld and System: A Critique of Functionalist Reason. Beacon Press, p. 155
[14] Habermas J. (1984) The Theory of Communicative Action. Lifeworld and System: A Critique of Functionalist Reason. Beacon Press, p. 154
[15] Winograd T, Flores F (1988) Understanding Computers and Cognition. 3rd edn. Addison-Wesley, p. 157
[16] Sheiderman B (1987) Designing the User Interface: Strategies for Effective Human-Computer Interaction. 1st edition. Addison-Wesley
[17] Zuellighoven H et al (2094) Object-Oriented Construction Handbook. dpunkt.verlag

International Reference Life Cycle Data System (ILCD) Handbook: Review Schemes for Life Cycle Assessment

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Abstract Quality and consistency of life cycle based decision support are essential in public policy and business context. Those can and should be supported by 'critical review' of underlying data and of life cycle assessment (LCA) studies themselves. A review can help avoiding errors, assuring that all options or method requirements have been appropriately taken into account, and increasing acceptance by stakeholders. The principle requirements for reviews are briefly addressed in the ISO 14040 series. While other LCA-based standards define some review requirements in more detail, information on how to conduct the review or what qualifications are required from reviewers is scarce. Building on these standards, more specific requirements and guidance on review are provided in the International Reference Life Cycle Data System (ILCD) handbook. Distinctions are made depending on the application context. Differentiations are made with respect to the intended audience, the complexity and broadness of the assessment, and the need of stakeholder involvement. In result, two different review types were identified (with/without stakeholder involvement). The handbook also provides minimum requirements for the qualification of reviewers. An upcoming document will provide more detail on scope ("what") and the method ("how") of review.

1 Introduction

Life cycle thinking (LCT) and life cycle assessment (LCA) are the scientific approaches behind modern environmental policies and business decision support related to sustainable consumption and production (SCP), Figure 1 presents the relationship between the objectives, LCA and related methods, and policies and business instruments. The concept of LCT helps to avoid the possibility of resolving one environmental problem while creating another, avoiding the so-called "shifting of burdens", e.g. from one part of the life cycle to another or amongst different types of impacts on the natural environment and on human

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health. This concept is supported by the quantitative tool life cycle assessment (LCA), standardised in ISO 14040 and 14044 [1,2].
Building on and further specifying these broad ISO standards, the international reference life cycle data system (ILCD) provides a common basis for consistent, robust and quality-assured life cycle data and studies. Such data and studies support coherent SCP instruments, such as among others ecolabelling, ecodesign, carbon footprinting, and green public and private procurement [3].

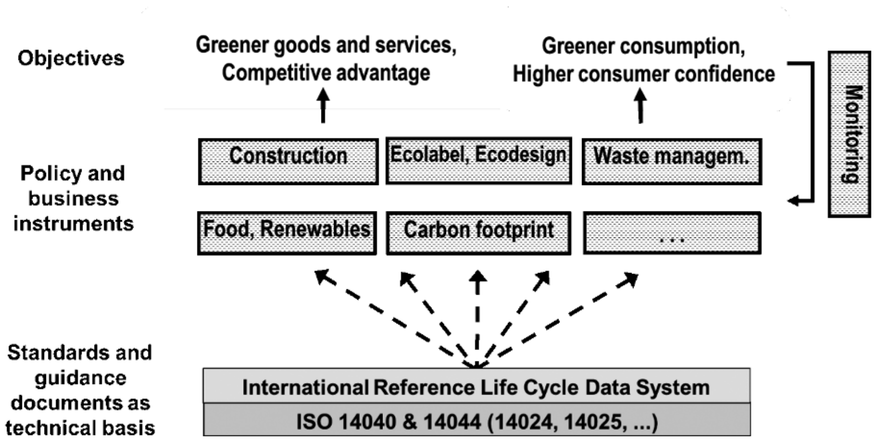

Fig. 1: Life cycle thinking and assessment - coherence and quality-assurance support for EU SCP/SIP policies

The principle requirements for reviews are briefly addressed in the ISO 14040 series and ISO 14025 including the definition of some review requirements. However, none of them provides information on how to conduct the reviews, or the required qualifications of reviewers. Therefore, more specific requirements and guidance on reviewing life cycle inventory and life cycle assessment studies are given in the ILCD handbook. The ILCD review requirements conform to the LCA-based ISO standards [4].

2 The international reference life cycle data system (ILCD)

The international reference life cycle data system (ILCD) has been established for guiding the development of consistent and reproducible, quality-assured life cycle data and robust assessments for use in the public and private sectors. This system consists primarily of the ILCD handbook and the ILCD data network: The handbook is a series of technical guidance documents (see Figure 2). It has been developed through peer review and consultation and is in line with the ISO 14040 and 14044, while it provides further specified guidance for better reproducibility and quality-assurance than the broader ISO framework can offer. To facilitate this development, links have been established with national LCA database projects in all parts of the world, and with the World Business Council for Sustainable Development (WBCSD) and the United Nations Environment Programme (UNEP).

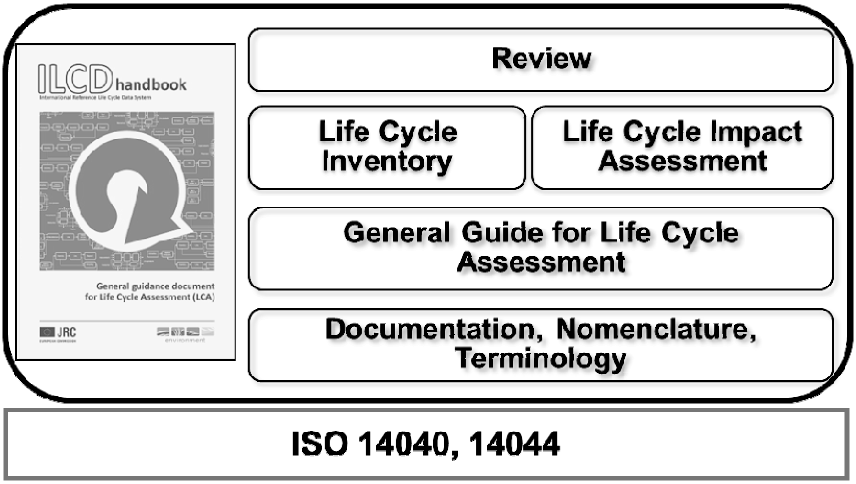

Fig. 2: ILCD handbook guidance documents

3 ILCD- review scheme for LCA

A critical review is one of several important elements to ensure the quality of LCA studies and its applications. Therefore it shall be performed by qualified experts that have not been involved in the performance of the LCI/LCA study. This is generally necessary to assure the quality and credibility and hence value of the study [3].

According to ISO 14044, the critical review shall assure that the methods used to carry out the LCA are consistent with the ISO standard, the methods used to carry out the LCA are scientifically and technically valid, the data used are appropriate and reasonable in relation to the goal of the study, the interpretations reflect the limitations identified and the goal of the study, and the study report is transparent and consistent [5,6]. However, very limited additional information is provided.

Building on the ISO standard framework, the ILCD handbook provides three separate documents i.e. "review schemes for life cycle assessment (LCA)", "reviewer qualification for life cycle inventory (LCI) data sets", and "review scope, methods and documentation" (in development), to address critical review in LCA study and its applications.

3.1 Review type

There are three review types defined in the ISO 14040, ISO 14044 i.e. critical review by "internal expert", by "external expert" and by "panel of interested parties". The ISO 14025 has one additional review type which is "third party panel"; consisting of at least of a chair and two further members [7]. The named "third party" is a person or body that is recognised as being independent of the parties involved, as concerns the issues in question.
The ILCD handbook further specifies this based on the ISO 14044 and ISO 14025. It provides guidance on minimum review type, reviewer qualification, and how to review. The first step is to identify the suitable review type as a minimum requirement for twelve applications ("cases"). For this the ILCD handbook considers five criteria/factors: extent of stakeholder involvement, complexity and broadness of the case, ISO standards requirement, knowledge/experience of audience, and cost. The result of analysis shows that only two review types “independent external review" and "independent panel review" are required and are hence recommended as minimum requirement, depending on the case [4]. In addition to the reviewer, for some cases, interested parties (stakeholders) shall be openly invited to participate in the process. The accredited reviewer from a fourth party is not foreseen for the ILCD system. Table 1 provides an overview of the minimum requirements related to the review type and the need of stakeholder involvement.

Tab. 1: Example for minimum review requirements of each LCA work for ILCD system based on stakeholder involvement, and technical knowledge of the audience

Application	Review type		Required involvement of interested parties
	Independent external review	Independent external panel review	
Micro level LCI data sets and LCA study	x		No
Comparative assertions on micro-level (e.g. products) disclosed to the public		x	Yes
Meso/macro level decision support LCA studies and meso/macro life cycle based accounting indicators		x	Yes
Meso/macro level LCA studies	x		Yes
LCA studies for identifying Type I ecolabel criteria and eco-design key environmental performance indicators (KEPIs)		x	Yes
Indirect aspects in environmental management schemes (EMS)	x		No
Micro level LCA studies/ Micro level monitoring indicator	x		No
Environmental product declarations (EPD)	x		Yes
Environmental product declarations (EPD) (B2B)	x		No
Product category rules (PCR) for type III, product-group and sector-specific guides		x	Yes

Note: The organisation-internal application is excluded from the scope of ILCD review scheme.

3.2 Required qualification of reviewers

For all review types independency, expertise and experience of the reviewer(s) are required. To ensure these aspects are treated appropriately and in a comparative manner, the ILCD handbook provides related requirements. The four main qualification aspects are: LCA methodology expertise, knowledge of applicable review rules, review or verification experience, and technical know-how. If one reviewer alone does not fulfil all the above requirements, the review framework allows for having more than one reviewer to jointly fulfil the requirements, forming a "review team" [8].

To help commissioner(s) of reviews to identify appropriate reviewer(s), a scoring system for eligible reviewers is introduced in the ILCD handbook. The approach taken allows different mechanisms and means of qualification (e.g. work experience, formal qualifications, conducted review or verifications). Differentiation of the level of qualification is achieved using a scoring system with minimum ‘hurdles’ in the four above-mentioned qualification aspects. Figure 3 illustrates the ILCD score. The detail information is provided in the "ILCD handbook: reviewer qualification for life cycle inventory data sets".

The online ILCD reviewer self-registry is under implementation to facilitate the use of LCA in policy and business. The reviewer self-registry is a web application that allows competent LCA-reviewers to register in the registry database and to create their own profile as reviewer.

They will be able to declare themselves using the ILCD handbook reviewer qualification scoring system. The application is a multiple domain system, similar as the upcoming ILCD data network system. The Joint Research Centre provides the IT application without cost and use restrictions so that any organisation can use it if they would like to adapt this system for their own needs.

<table>
<tr><td colspan="3"></td><td rowspan="2">Barrier level (minimum required)</td><td colspan="4">Score (points)</td></tr>
<tr><td colspan="3"></td><td>1</td><td>2</td><td>3</td><td>4</td></tr>
<tr><td rowspan="6">Mandatory</td><td>Topic</td><td>Criteria</td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td rowspan="2">Verification and audit practice</td><td>Years of experience</td><td>3</td><td>3 – 4</td><td>5 – 8</td><td>9 – 14</td><td>> 14</td></tr>
<tr><td>Number of reviews</td><td>3</td><td>3 – 5</td><td>6 – 15</td><td>16 – 30</td><td>> 30</td></tr>
<tr><td rowspan="2">LCA methodology and practice</td><td>Years of experience</td><td>3</td><td>3 – 4</td><td>5 – 8</td><td>9 – 14</td><td>> 14</td></tr>
<tr><td>"Experiences" of participation in LCI work</td><td>5</td><td>5 – 8</td><td>9 – 15</td><td>16 – 30</td><td>> 30</td></tr>
<tr><td>Technologies or other activities represented by the LCI data set</td><td>Years of experience</td><td>3 (within the last 10 years)</td><td>3 – 5 (within the last 10 years)</td><td>6 – 10 (within the last 20 years)</td><td>11 – 20</td><td>> 20</td></tr>
<tr><td rowspan="5">Optional</td><td>Verification and audit practice</td><td>Optional scores relating to audit</td><td colspan="5">▪ 2 points: Accreditation as third party reviewer for at least one EPD Scheme, ISO 14001, or other EMS.
▪ 1 point: Attended courses on environmental audits (at least 40 hours).
▪ 1 point: Chair of at least one review panel (for LCA studies or other environmental applications).
▪ 1 point: Qualified trainer in environmental audit course.</td></tr>
<tr><td>LCA methodology and practice</td><td>Relating to LCA methodology and practice</td><td colspan="5">▪ 1 point: At least 5 methodological articles on LCA in peer-review journals or books.
▪ 1 point: Active participation in at least 3 research projects on LCA-related methodological issues or case studies.</td></tr>
<tr><td rowspan="3">Technologies or other activities represented by the LCI data set</td><td>Formal scientific qualification:</td><td colspan="5">▪ 1 point: At least one PhD obtained.
▪ 0.5 point: At least one Masters thesis or equivalent completed.</td></tr>
<tr><td>Other work experience outside the private sector:</td><td colspan="5">▪ 1 point: At least 3 years work experience</td></tr>
<tr><td>Other work experience within the private sector:</td><td colspan="5">▪ 0.5 point per one additional industry sector (up to 2.5 points in total.</td></tr>
</table>

Fig. 3: Scoring system for eligible reviewers/review teams and for qualification as a potential member of a review team (adapted from [8])

The Figure 4 presents the overall self-registry application. This application also supports building up the quality data network of the ILCD [9].

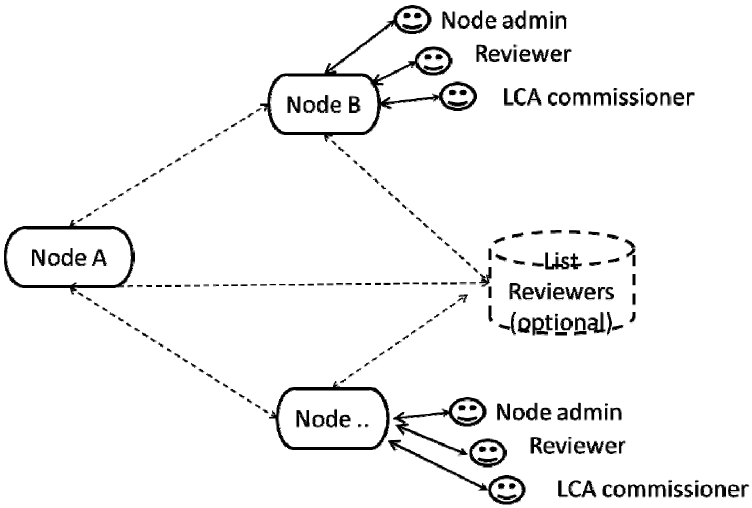

Fig. 4: LCA reviewer self-registry application.

3.3 Review scope and method (upcoming)

Typically the selected reviewer(s) need to review the LCA study or LCI data set along goal definition, scope definition, life cycle inventory, life cycle impact assessment, interpretation and report against the ISO 14044 and additional requirements, if any. Therefore it is useful at least during the scope definition, as 2nd step of LCA, to decide whether a critical review will be done, and if so which form of review and performed by whom.

This early decision will allow the data collection, documentation and reporting of the LCI/LCA study to be tailored to meet also the requirements of the review, typically shortening and lowering the overall effort. An early decision also allows running an inter-active concurrent review process: In a concurrent review the reviewers are given the opportunity to comment already e.g. on the goal and scope definition prior to the onset of the inventory analysis, and possibly on interim results of the impact assessment and interpretation before the reporting. This way their comments can guide the process of the LCA and can often prevent unpleasant surprises at the end of the project, e.g. additional data needs or even unsuitable comparisons that can set back a comparative assertion study by many months. A concurrent review is understood to generally also further improve the credibility of the study [3].There is no fixed requirement given in ISO 14044 and 14025 on how to review in each step of an LCA. Therefore a separate document,

foreseen to be part of ILCD handbook, provides a suitable set of guidelines for carrying out the actual life cycle assessment (LCA) review, according to the different kinds of LCA applications.
Examples of methods to apply in order to achieve a good review for each scope item are 1) evidence collection by means of available documentation, 2) cross-checks with other sources, other similar processes or product system and legal limits, 3) calculation of energy balance, mass balance, and chemical element balances, 4) sampling review (the number of random data selected should be representative) 5) plant visit, interview, 6) expert judgment. Table 2 provides the draft overview of the methods for review and how they should be implemented.

Tab. 2: Draft overview of methods used for review

Method		Implementation
Evidence collection by means of available documentation		Analysis of the documentation produced during the LCA work
Cross-checks	Cross-check with one or more other sources	Comparison with data and/or information on the same issue, from another, independent source (can be both database and literature)
	Cross-check with one or more other processes or product systems	Comparison with data and/or information on similar processes or product systems, from the same or from other sources (can be both database and literature)
	Cross-check with legal limits	Comparison with applicable legal limits
Verification and review of data source		Analysis of data source declared, checking its context-specific correct use as well as its relevance and quality
Calculations	Energy balance	Calculation and analysis of energy balance
	Mass balance	Calculation and analysis of mass balance
	Element balances	Calculation and analysis of the context-specific relevant chemical element balances
	Other calculations	Verification and analysis of other calculations
Evidence collection by means of plant visits and/or interviews		Interviews and/or plant visits should be performed in case of relevant inconsistencies, uncertainties, or doubts. Persons to be interviewed need to have detailed technical expertise on the analysed process and issue

Method	Implementation
Expert judgement	Analysis by means of expert opinions. The expert needs to have methodological and/or detailed technical expertise on the item to be verified and the process or product system in question, as required to obtain a qualified expert judgment
Conformity with ISO 14040 and 14044	Review against the requirements set forth in ISO 14040 and 14044. Note: Conformity with the ILCD Handbook implies conformity with ISO 14040 and 14044, but should be explicitly verified and reported.

4 Conclusion and outlook

LCA is moving more and more into the core of modern environmental policies and business decisions. Therefore, reliable means are needed for the users of LCA data and studies to assure the quality of the results they are using to make better informed decisions. The critical review does not only increase the credibility of a study but also helps improving its quality. The close interaction between the LCA practitioner and the reviewer is vital for an efficient and effective review process.
Recognising the relevance of critical review, in line with the ISO 14040 series frame, the ILCD Handbook provides minimum review type requirements for different cases of LCA applications, the detail for selecting qualified reviewer and will address how to review properly in a forthcoming document.
The upcoming reviewer registry will allow LCA experts to demonstrate their qualification and to register independently in the different participating countries and regions. The self-reviewer registry is under development and the beta test of this system is foreseen to start in 2011.

References

[1] Pant R et al (2008) International Reference Life Cycle Data System (ILCD). 10th International Ecobalance Conference, 9-11 December, Tokyo
[2] Wolf M et al (2009) Review of LCA work – the accredited certification scheme of the International Reference Life Cycle Data System (ILCD). SETAC Europe 19th Annual Meeting 31st May - 4th June, Goteborg
[3] European Commission - Joint Research Centre - Institute for Environment and Sustainability (2010) International Reference Life Cycle Data System

(ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. First edition March 2010. EUR 24708 EN. Publications Office of the European Union, Luxembourg

[4] European Commission - Joint Research Centre - Institute for Environment and Sustainability (2010) International Reference Life Cycle Data System (ILCD) Handbook - Review schemes for Life Cycle Assessment. First edition March 2010. EUR 24710 EN. Publications Office of the European Union, Luxembourg

[5] ISO 14040:2006. Environmental management - Life cycle assessment-Principles and framework

[6] ISO 14044:2006. Environmental management - Life cycle assessment-requirements and guidelines

[7] ISO 14025:2006. Environmental labels and declarations - Type III environmental declarations - Principles and procedures

[8] European Commission - Joint Research Centre - Institute for Environment and Sustainability (2010) International Reference Life Cycle Data System (ILCD) Handbook - Reviewer qualification for Life Cycle Inventory data sets. First edition March 2010. EUR 24379 EN. Publications Office of the European Union, Luxembourg

[9] <www.lct.jrc.ec.europa.eu>

Time and Life Cycle Assessment: How to Take Time into Account in the Inventory Step?

Pierre Collet, Arnaud Hélias, Laurent Lardon and Jean-Philippe Steyer

Abstract Life cycle assessment is usually an assessment tool which only considers steady state processes: the temporal and spatial properties of extractions, usage and emissions are lost during the life cycle inventory step. This approach significantly reduces the environmental relevance of some results. As the development of dynamic impact methods is based on dynamic inventory data, it seems essential to develop a general methodology to achieve a temporal life cycle inventory. This study presents a method to select steps, in the whole network tree, for which dynamics have to be considered while the others are approximated by steady state representation. The selection procedure is based on the main contributors in term of impact. The approach is illustrated by the life cycle assessment of simplified rapeseed oil production as biofuel system.

1 Introduction

Life cycle assessment (LCA) is usually a static assessment tool which only considers steady state [1]. Temporal and spatial aspects are usually ignored in a classical LCA. Although the integration of a spatial dimension is more and more frequently pointed out as a hotspot of LCA and has been subjected to methodological developments, little attention has been given to the temporal aspect of LCA, either in the life cycle inventory (LCI) or in the life cycle impact assessment (LCIA). Since a few years, dynamic has been identified as one of the main unresolved problem in LCA [2].

During the LCI step, the dynamic of the emissions is lost by aggregation. The temporal course of the emissions and the ensuing concentrations in the environment cannot be known. At the LCIA step, the time is only taken into account by considering timescales over which the effect of the emissions has to be integrated for some impacts (e.g. climate change). Despite it is generally assumed

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that the biological processes respond linearly to environmental disturbances and that threshold effects can be neglected [3], temporal factors such as the timing of emissions and the rate of release can potentially modify the impact of pollution [3, 4]. For instance, some impacts are subject to seasonal variations, like aquatic eutrophication which is higher in the summer than in winter [5].

The introduction of time in LCA can pursue different objectives: the assessment of future technologies, the development of dynamic inventories, and the development of dynamic impact methods, including the special case of long term impacts. To perform dynamic LCA, a dynamic LCI is required. However the systematic introduction of the dynamic in the inventory is a very tedious work. It means the collection and the tuning of at least one model for each process whereas a usual LCA can involve hundreds of processes.

All this work is time-consuming and can be at the root of many approximations. The resulting values could then be less accurate than the classical steady-state ones. The objective of this work is to set up a methodology to determine which steps - in a whole process network - have to be considered as dynamic ones and which could be approximated by a steady state representation. This selection is done by limiting the introduction of dynamic models to only the main contributors in term of impact.

First the general methodology is presented. Then a practical case study based on production and self-consuming of biodiesel from rapeseed is used to explain the approach. Finally, the conclusions highlight the main contributions of this work and identifies the research developments to pursue.

2 Methodological framework

To take into account spatial differentiation, authors in [6] propose to firstly start with a generic approach, and then to develop spatially differentiated factors, where “large variations of fate and exposure or of effect variables are observed”. The same methodology is applied here to temporal differentiation by getting a global overview of the system and then introducing dynamic where it is relevant. A contribution analysis is performed to assess which steps have to be considered as dynamic and which ones could be approximated by a steady state representation.

2.1 Granularity of the contribution analysis

In the LCA approach, the system is described as a process network composed by unit processes. A substance can be emitted to the environment by one or by several processes. In the same way, a unit process can generate one or several emissions. So the identification of the main contributors can not be based on a contribution analysis of exclusively the emissions and/or exclusively the processes. The developed approach here proposes to consider the most impacting couples {process, emission}.

The classical model for the inventory analysis of a LCA with p processes, u economic flows and e emissions is the following:

- A the technology matrix is known. It is a square matrix of order p.
- B the intervention matrix is known. It is a e by p matrix.
- f the final demand vector is known.

The scaling vector s is calculated with the Equation 1.

$$s = A^{-1} \times f \tag{1}$$

The inventory vector g is obtained with the Equation 2.

$$g = B \times s \tag{2}$$

In a classical LCA, the inventory vector g represents the set of all environmental flows corresponding to the considered reference flow. In our approach we need to define a new e by p matrix C describing the detail of emissions j for each process i according to the scaling vector.

$$C = \begin{pmatrix} c_{1,1} & \cdots & c_{1,i} & \cdots & c_{1,p} \\ \vdots & \ddots & & & \vdots \\ c_{j,1} & \cdots & c_{j,i} & \cdots & c_{j,p} \\ \vdots & & & \ddots & \vdots \\ c_{e,1} & \cdots & c_{e,i} & \cdots & c_{e,p} \end{pmatrix} \tag{3}$$

$$\text{with } \forall (i,j) \in [1;p] \times [1;e] \quad c_{j,i} = b_{j,i} \times s_i \quad \text{and} \quad \sum_{i=1}^{p} c_{j,i} = g_j$$

Each $c_{j,i}$ corresponds to the emission j in the process i. The environmental flows are then converted into impacts with the endpoint characterisation factor CF^k_j relative to the impact k for the emission j.

Comparing the relative contribution of the different impacts requires to express them in the same unit. We choose to express all the impacts with the ReCiPe method [7] in Point (i.e. the impacts are normalised and weighted). For the l impacts considered, the impact score $isc^k_{j,i}$ of each element is equal to:

$$\forall (i,j,k) \in [1;p] \times [1;e] \times [1;l] \;\; isc_{j,i}^{k} = c_{j,i} \times CF_{j}^{k} \tag{4}$$

The single score elements $ssc_{j,i}$ corresponding to the emission j of the process i are computed as described by the Equation 5:

$$\forall (i,j) \in [1;p] \times [1;e] \quad ssc_{j,i} = \sum_{k=1}^{l} isc_{j,i}^{k} \tag{5}$$

The set of the single score values of all the couples {process, emission} is used to select the most contributing processes. The $isc^{k}_{j,i}$ elements are sorted by descending order. The first element selected is the one with the highest $isc^{k}_{j,i}$ value.

We assume that if the addition of an extra couple induces a variation of more than 1% of the cumulative single score (i.e. the sum of all the previous single scores of the couples selected plus the single score of the couple being tested), then the couple is selected. If not, the couple is not selected and the iterative selection process is stopped.

2.2 Relevance of introducing model(s) in the selected couples

Once the couples are selected, the relevance of introducing a dynamic has to be estimated. Emissions from a process are characterised by their own timescale θ_e. These emissions contribute to specific impact previously identified. As underlined by [8] each impact is associated with a natural timescale θ_i. This timescale θ_i defines the time step on which the emissions and removals are aggregated. To be consistent, the introduction of time in the LCI should consider both the timescales θ_e and θ_I.

- If the emission's timescale θ_e is very short compared to the impact's timescale θ_i, the dynamic of the emissions does not affect significantly its impact.
- If θ_e is very long compared to θ_i, the introduction of dynamic should be considered. Indeed in this case the dynamic of the impacts is driven by the slow but existing dynamic of the emissions. The dynamic of the impacts is then the same than the dynamic of the emissions shifted by the timescale θ_i.
- Finally if the timescales θ_e and θ_i of the selected couple have the same order of magnitude, the introduction of models is also relevant.

Table 1 presents the inherent timescales θ_i of the impacts which are considered in our study. The timescales of the impacts can globally be divided in three parts, which correspond to the three areas of protection describes in ReCiPe (2008) [7] - human health, ecosystems and resources. The timescale of the impacts related to human health (and more generally to the toxicity and the ecotoxicity) is the day, the one of the impacts related to ecosystems is the month and the one of the impacts related to resources is the year.

Tab. 1: Time scales associated with the impacts

Impact categories	Time scale: θ_i
Photochemical oxidant formation (POF)	Hour
Human toxicity (HT)	Day
Particulate matter formation (PMF)	Day
Terrestrial ecotoxicity (TET)	Day
Freshwater ecotoxicity (FET)	Day
Marine ecotoxicity (MET)	Day
Ionising radiation (IR)	Day
Terrestrial acidification (TA)	Month
Freshwater eutrophication (FE)	Month
Marine eutrophication (ME)	Month
Water depletion (WD)	Month
Climate change (CC)	Year
Ozone depletion (OD)	Year
Agricultural land occupation (ALO)	Year
Urban land occupation (ULO)	Year
Natural land transformation (NLT)	Year
Mineral resource depletion (MRD)	Year
Fossil resource depletion (FD)	Year

Figure 1 sums up the main steps of the selection of processes. This methodology leads to the selection of a given number of couples, based both on their major impact and their single score. Once these couples are selected, the relevance of introducing time is assessed by comparing the timescales θ_e and θ_i of each couple. If the timescale θ_e is very fast compared to the timescale θ_i, taking into account the dynamic is not relevant. In the other cases, (θ_e very slow compared to θ_i or timescales are similar), a model could be developed to generate distributed emissions over the period considered.

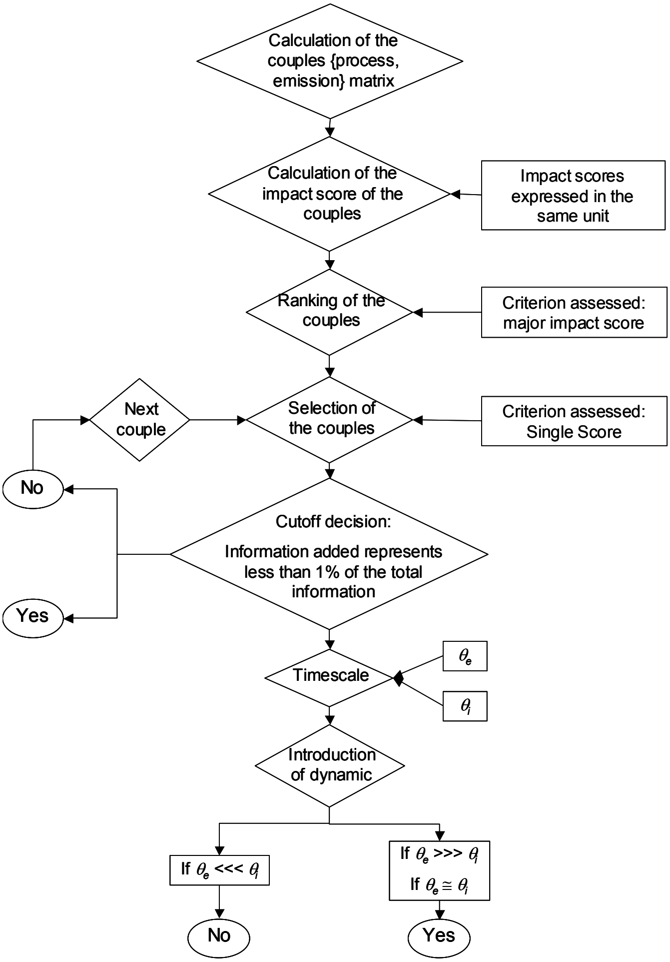

Fig. 1: Main steps of the introduction of time in the LCI

3 Case study: biodiesel production from rapeseed

3.1 Overview of the system

To illustrate this approach, we consider a practical case study based on production and self-consuming of biodiesel from rapeseed. The functional unit is one MJ burned in an internal combustion engine. According to the principles of LCA, the inventory includes production, harvesting, storage, oil extraction, and combustion of the oil. An energetic allocation method has been used to handle co-products at the oil-press stage. The inventory is based on figures derived from academic resources and processes described in the Ecoinvent database [9]. The location of the system is in the southern Europe; the energy mix for electricity is based on the European average. Figure 2 shows an overview of the system, which is not described more precisely for sake of concision. The system considered is composed of 37 unit processes and 660 emissions or removals are listed. The number of potential couples is 24 420, but the real total number of couples of the system is 5249. Table 2 presents the materials flows required to construct the agricultural machines needed to realise the functional unit.

Tab. 2: Materials fluxes required to construct the agricultural machines

	Tractor	**Plow**	**Cultivator**	**Speader**	**Seed drill**	**Sprayer**	**Combine harvester**
Steel (kg)	1.7685x10^{-4}	1.0808x10^{-4}	1.31x10^{-5}	1.1808x10^{-2}	3.4799x10^{-6}	3.4089x10^{-2}	1.4036x10^{-4}
Polypropylene (kg)	9.825x10^{-6}	6.0042x10^{-6}	7.2778x10^{-7}	6.56x10^{-4}	1.9333x10^{-7}	1.8938x10^{-3}	7.7976x10^{-6}
Synthetic rubber (kg)	9.825x10^{-6}	6.0042x10^{-6}	7.2778x10^{-7}	6.56x10^{-4}	1.9333x10^{-7}	1.8938x10^{-3}	7.7976x10^{-6}
Steel manufacturing (kg)	1.7685x10^{-4}	1.0808x10^{-4}	1.31x10^{-5}	1.1808x10^{-2}	3.4799x10^{-6}	3.4089x10^{-2}	1.4036x10^{-4}
Injection moulding (kg)	1.9769x10^{-5}	1.208x10^{-5}	1.4643x10^{-6}	1.3199x10^{-3}	3.8899x10^{-7}	3.8105x10^{-3}	1.5689x10^{-5}
Disposal steel (kg)	1.7685x10^{-5}	1.0808x10^{-5}	1.31x10^{-6}	1.1808x10^{-3}	3.4799x10^{-7}	3.4089x10^{-3}	1.4036x10^{-5}
Steel recycling (kg)	1.5917x10^{-4}	9.7268x10^{-5}	1.179x10^{-5}	1.0627x10^{-2}	3.131x10^{-6}	3.0680x10^{-2}	1.2632x10^{-4}
Rubber incineration (kg)	9.825x10^{-6}	6.0042x10^{-6}	7.2778x10^{-7}	6.56x10^{-4}	1.9333x10^{-7}	1.8938x10^{-3}	7.7976x10^{-6}
Polypropylene incineration (kg)	9.825x10^{-6}	6.0042x10^{-6}	7.2778x10^{-7}	6.56x10^{-4}	1.9333x10^{-7}	1.8938x10^{-3}	7.7976x10^{-6}

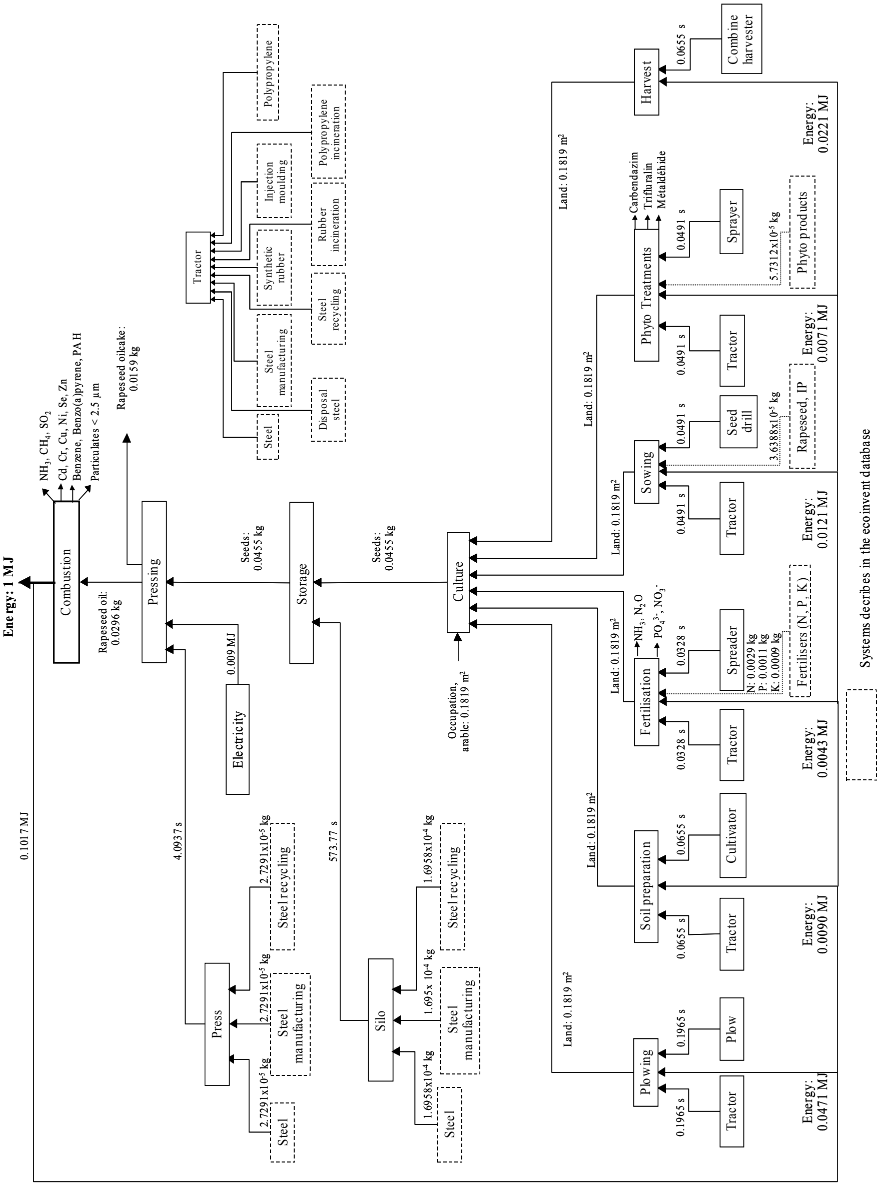

Fig. 2: Overview of the system of biodiesel production from rapeseed. For the sake of clarity agricultural machines are not described. An example is given for the tractor (see the insert).

3.2 Results

Figure 4 represents the normalised cumulative sum of the single scores of the ordered couples {process, emission}, and the normalised single score results for each couple. The procedure described above allows to obtain 82.5% of the total normalised cumulative sum. The number needed to satisfy the criterion selection previously defined is 14 couples {process, emission}, which represent 0.27% of the total number of couples. This analyse highlights the fact that only a limited number of couples is enough to describe in a proper way the needed information to introduce time in the LCI. The histograms of Figure 3 correspond to the normalised single score values of each couple. It is interesting to notice that these values are not necessarily ranked in the same order that the major impact values. Indeed the relative contribution of the major impact to the single score could strongly vary from a couple to another, because for some couples the emission can contribute to more than one impact.

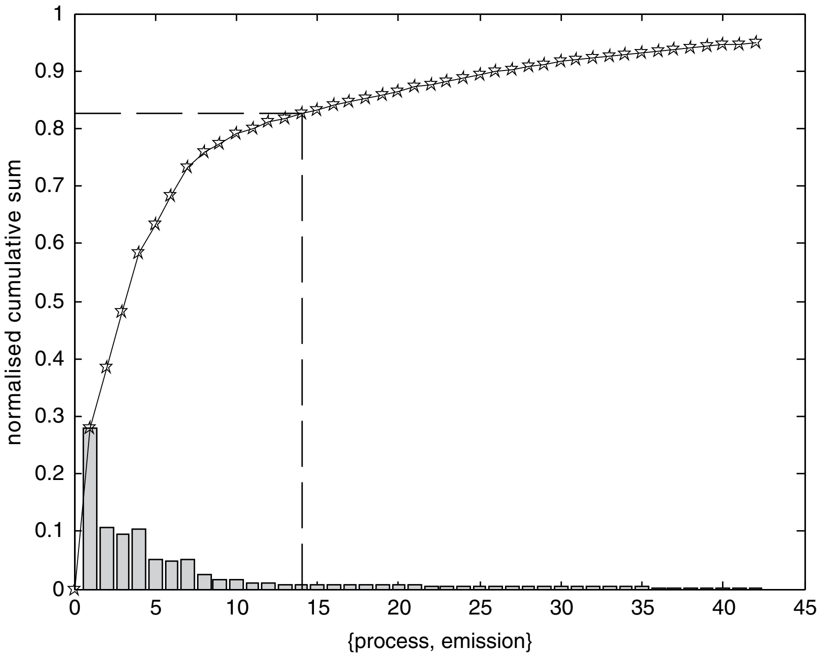

Fig. 3: Selection of the couples {process, emission}

Table 3 presents the main information for each selected couple {process, emission}. According to the process where it occurs, an emission can have different θ_e. For instance θ_e of "dinitrogen monoxide" flows of the process “fertilisation French agricultural practices” is linked to the crop growth, so θ_e is the month. But the timescale θ_e of “dinitrogen monoxide” flows of the process “Ammonium nitrate as N at regional storehouse” is determined by the evolution of technologies, so θ_e is the year.

Tab. 3: Results

% of the impact score	Process	Emission	Impact	θ_i	θ_e	Selection
27.85	Rapeseed production French agricultural practices	Occupation, arable, non-irrigated	ALO	Year	Year	Yes
9.60	Fertilisation French agricultural practices	Dinitrogen monoxide	CCH	Year	Month	-
9.49	Ammonium nitrate as N at regional storehouse	Gas, natural, in ground	FD	Year	Year	Yes
9.25	Ammonium nitrate as N at regional storehouse	Dinitrogen monoxide	CCH	Year	Year	Yes
4.99	Combustion internal combustion engine	Particulates, < 2.5um	PMF	Day	Day	Yes
4.95	Ammonium nitrate as N at regional storehouse	Oil, crude, in ground	FD	Year	Year	Yes
4.46	Ammonium nitrate as N at regional storehouse	Carbon dioxide, fossil	CCH	Year	Year	Yes
2.37	Fertilisation French agricultural practices	Ammonia	PMF	Day	Month	Yes
1.67	Single superphosphate as P_2O_5 at regional storehouse	Oil, crude, in ground	FD	Year	Year	Yes
1.47	Single superphosphate as P_2O_5 at regional storehouse	Carbon dioxide, fossil	CCH	Year	Year	Yes
1.14	Fertilisation French agricultural practices	Dinitrogen monoxide	CCE	Year	Month	-
1.09	Ammonium nitrate as N at regional storehouse	Dinitrogen monoxide	CCE	Year	Year	Yes
1.02	Potassium sulphate as K_2O at regional storehouse/RER S	Oil, crude, in ground	FD	Year	Year	Yes
0.98	Ammonium nitrate as N at regional storehouse	Nitrogen oxides	PMF	Day	Year	Yes

The comparison of θ_e and θ_i allows us to select 12 couples on 14 to introduce a dynamic. The couple {occupation arable non-irrigated, Rapeseed production French agricultural practices} is the one with the highest impact score compared to the single score of the whole system. The impact score of this couple is influenced by the yield parameter. Consequently we need to introduce a model determining the yield.

Another case is the couple {gas natural in ground, ammonium nitrate as N at regional storehouse}. The impact score is mainly due to the way of producing ammonium nitrate. For that reason the model to introduce at this level should be one modelling the evolution of future technologies. The same reasoning should be done in all the other couples selected where the process involved corresponds to the production of fertilisers.

Then the impact score of the couple {particulates <2.5um, combustion internal combustion engine} is driven by the farmer’s choices about the organisation of his

day's work. This is more difficult to model because it is strongly correlated with organisational schemes and agronomic decisions.
Finally the impact score of the couple {ammonia, fertilisation French agricultural practices} is influenced by the use of fertilisers in the field. Consequently we need to introduce a model determining the need in fertilisers and their nutrient fates.

4 Conclusions

This work develops a methodology which allows us to specify where the introduction of dynamic is the most relevant in a whole process tree. It is based on a level of granularity, the couple {process, emission}, more accurate than in a classical contribution analysis. The introduction of dynamic depends on the relation between two different timescales θ_e and θ_i. θ_e is directly linked to the process where the emissions occur, whereas θ_i can be defined more generally. The definition of θ_i could change according to the impact assessment selected or to the cultural perspective chosen. For instance, in a individualist perspective [10], based on short-term interest, the timescale of the ionising radiation impact would be the day, whereas in an egalitarian one, which takes into account the longest timeframe, it would rather be the year or the century. The description of proper timescale for each impact could be a subject of future developments in LCIA methods.
The time distributed LCI which is generated by this approach can be coupled with dynamic LCIA methods like the one developed by [11] in order to have a full dynamic LCA. It could be noticed that the time introduction in the LCI can also be done in the technologic flows between unit processes. Finally selection criterion leads to the exclusion of some couples with an important impact score. It could be also interesting to develop models for these couples because the generation of a dynamic LCI will lead to more precise evaluation of emissions that could be integrated over time and hence be used in a classical LCA approach.

References

[1] de Haes HU (2006) How to approach land use in LCIA or, how to avoid the Cinderella effect? Int J Life Cycle Assess 11:219-221.

[2] Reap J, Roman F, Duncan S, Bras B (2008) A survey of unresolved problems in life cycle assessment. Int J Life Cycle Assess 13:374-388

[3] Owens JW (1997) Life-Cycle Assessment in Relation to Risk Assessment: An Evolving Perspective. Risk Anal 17:359-365.

[4] Field F, Kirchain R, Clark J (2000) Life-Cycle Assessment and Temporal Distributions of Emissions: Developing a Fleet-Based Analysis. J Ind Ecol 4:71-91.

[5] De Haes HU, Finnveden G, Goedkoop M, Hauschild M, Hertwich EG, Hofstetter P, Jolliet O, Klopffer W, Krewitt W, Lindeijer E, Mueller-Wenk R, Olsen SI, Pennington DW, Potting J, Steen B (2002) Life-cycle impact assessment: striving towards best practice, Society of Environmental Toxicology and Chemistry (SETAC), Brussels

[6] De Haes HU, Jolliet O, Finnveden G, Hauschild M, Krewitt W, Mueller-Wenk R (1999) Best available practices regarding impact categories and category indicators in life-cycle impact assessment. Int J Life Cycle Assess (4):66-74.

[7] Goedkoop M, Heijungs R Huijbregts M, De Schryver A, Struijs J, van Zelm R (2008) ReCiPe 2008 - A life cycle impact assessment method which comprises harmonised category indicators at themidpoint and the endpoint level.

[8] McKone TE, Nazaroff WW, Berck P, Auffhammer M, Lipman T, Torn MS, Masanet E, Lobscheid A, Santero N, Mishra U, Barrett A, Bomberg M, Fingerman K, Scown C, Strogen B, Horvath A (2011) Grand Challenges for Life-Cycle Assessment of Biofuels. Environ Sc. Technol 45:1751-1756.

[9] Frischknecht R, Althaus HJ, Doka G, Dones R, Heck T, Hellweg S, Hischier R, Jungbluth N, Nemecek T, Rebitzer G, Spielmann M (2007) Overview and methodology, final report ecoinvent v2.0 No. 1. Swiss Centre for Life Cycle Inventories

[10] Thompson M, Ellis R, Wildavsky A (1990) Cultural Theory, Westview Print Boulder

[11] Levasseur A, Lesage P, Margni M, Deschène L, Samson R (2009) Considering Time in LCA: Dynamic LCA and Its Application to Global Warming Impact Assessments. Int J Life Cycle Assess 10:3169-3174.

A Method of Prospective Technological Assessment of Nanotechnological Techniques

Michael Steinfeldt

Abstract Nanotechnology is frequently described as an enabling technology and a fundamental innovation, i.e. it is expected to lead to numerous innovative developments in the most diverse fields of technology and areas of application in society and the marketplace. Nanotechnologies are regarded as a substantial element for environmental reliefs. As a result the following questions arise: How large are the possible relief effects on the environment by nanotechnological techniques? This contribution describe a new method of prospective technological assessment of nanotechnological processes and gives a current overview of existing studies of published LCAs of the manufacture of nanoparticles and nanocomponents.

1 Introduction

It is not – or only to a very limited extent – possible to steer technological development by means of political intervention; it is rather the interaction of the various players, which usually leads to a developmental path that can be concurrently shaped "en route." Inasmuch as nanotechnology is for the most part still in an early phase of development, there still exists, at least in principle, a large degree of freedom, thus allowing research efforts to be steered towards sustainable development. In order to use these purposefully methods of the prospective technological assessment in particular for the early development phases of nanotechnologies are necessary.

2 Prospective eco-profiles as a new approach for prospective technological assessment of nanotechnological techniques

As an assessment approach, the prospective eco-profiles are based on the life cycle assessment methodology. The life cycle assessment (LCA) is the most

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extensively developed and standardised methodology for assessing the environmental aspects and product-specific potential environmental impacts associated with the complete life cycle of a product. It has the advantage that by means of comparative assessments, an analysis of eco-efficiency potential in comparison to existing applications is possible.

In the early stage of the development of new nanotechnological techniques great gaps of quantified material and energy flow data are present. Likewise, when comparing established or mature technologies with those still in development, one must recognise that a new technology is at the beginning of its “learning curve” and holds the potential for significant increase in efficiency.

Due to the interdisciplinary nature of nanotechnology, an enormous wealth of methods for the production of nano-scale products can be found in the literature. Products can for example be differentiated according to their nano-scale basic structure: particle-like structures (e.g. nanocrystals, nanoparticles, and molecules), linear structures (e.g. nanotubes, nanowires, and nanotrenches), layer structures (nanolayers), and other structures such as nanopores, etc. Materials can also be produced from the gas phase, the liquid phase, or from solids in such a way that they are nano-scalar in at least one dimension.

In the new approach manufacturing processes are modelled on the laboratory experiments and mini plants level. On this basis extrapolating estimations in different scenarios for the scale-up and optimisation of the manufacturing process are investigated. The different influences of the inputs and outputs of the process are considered by appropriate conversion factors (e.g. change of yield, change of energy efficiency, change of efficiency of operating supplies). So it is possible to performed cradle-to-gate assessment of nanotechnological techniques for the production of nanoparticles and nanocomponents (see also Table 1).

3 Evaluation of specific manufactured nanoparticles

The largest groups of manufactured nanoparticles for industrial applications are inorganic nanoparticles (e.g. TiO_2, ZnO, SiO_2, Ag), carbon-based nanomaterials (carbon nanofiber, multi wall carbon nanotubes (MWCNT), single wall carbon nanotubes (SWCNT)), and quantum dots (semiconductor nanoparticles with a specific size (e.g. CdSe, CdS, GaN)). Beside qualitative environmental assessments of the different manufacturing methods [1,2] unfortunately quantified material and energy flows data exist also within this range only for a very small number of manufacturing processes and/or for individual nanomaterials. A summary of published studies is shown in Table 1. Particularly remarkable is that the majority of the studies investigate the production of carbon-based nanomaterials.

Tab. 1: Overview of studies of published LCAs of the manufacture of nanoparticles and nanocomponents (Source: based on [3] and own data)

Nanoparticle and/or nanocomponent	Assessed impact(s)	Ref.
Metal nanoparticle production (TiO_2, ZrO_2)	Cradle to gate energy assessment, global warming potential	[4]
Nanoclay production	Cradle to gate assessment, energy use, global warming potential, ozone layer depletion, abiotic depletion, photo-chemical oxidant formation, acidification, eutrophication, cost	[5]
Several nanomaterial syntheses	E-factor analysis	[6]
Carbon nanoparticle production	Cradle to gate energy assessment	[7]
Carbon nanotube production	Cradle to gate assessment with SimaPro software, energy use, global warming potential, …	[8]
Single-walled carbon nanotube (SWCNT) production	Cradle to gate assessment with SimaPro software, energy use, global warming potential, …	[9]
Carbon nanofiber production	Energy use, global warming potential, ozone layer depletion, radiation, ecotoxicity, acidification, eutrophication, land use	[10]
Nanoscale semiconductor	Cradle to gate assessment, energy use, global warming potential	[11]
Nanoscaled polyaniline production	Cradle to gate assessment with Umberto software, energy use, global warming potential, …	[12]
Multi-walled carbon nanotube (MWCNT) production	Cradle to gate assessment with Umberto software, energy use, global warming potential, …	[12]
Carbon nanofiber production	Cradle to gate assessment with Umberto software, energy use, global warming potential, …	[13]
Silver nanoparticle production	Cradle to gate assessment with Umberto software and economic analysis, cost, energy use, global warming potential, …	[14]

Osterwalder and co-workers performed cradle-to-gate assessments of titanium dioxide (TiO_2) and zirconia dioxide (ZrO_2) nanoparticle production [4]. The goal of the study was to compare energy requirements and greenhouse gas emissions for the classical milling process with that for a novel flame synthesis technique.

Flame based nanoparticle production using organic precursors. The functional unit of the study was 1 kg of manufacturing materials.
Roes and co-workers evaluated the use of nanocomponents in packaging film, agricultural film, and automotive panels [5]. The goal of the prospective assessment was to determine if the use of nanoclay additives in polymers (polypropylene, polyethylene, glass fibre-reinforced polypropylene) is more environmentally more advantageous than conventional materials. Specific material and energy flows of the nanoclay production were collected. The manufacture of nanoclay includes several processes, e.g. raw clay (Ca-bentonite) extraction, separation, spray drying, organic modification, filtering, and heating.
Eckelman and co-workers have performed an E-factor analysis of several nanomaterial syntheses [6], as the E-factor is a measure of environmental impact and sustainability that has been commonly employed by chemists. The E-factor (or waste-to-product ratio) includes all chemicals involved in production. Energy and water inputs are generally not included in E-factor calculations, nor are products of combustion, such as water vapour or carbon dioxide. The results are not comparable unfortunately with the other studies.
Kushnir and Sanden modelled the requirements of future production systems of carbon nanoparticle and used also a cradle-to-gate perspective, including all energy flows up to the production and purification of carbon nanoparticles [7]. All calculations are made for a functional unit of 1kg of nanoparticle. Several production systems (fluidised bed chemical vapour deposition (CVD), floating catalyst CVD, HiPco, pyrolysis, electric arc, laser ablation, and solar furnace) are investigated and possible efficiency improvements are discussed. Carbon nanoparticles are found to be highly energy-intensive materials, on the order of 2 to 100 times more energy intensive than aluminium, given a thermal to electric conversion efficiency of 0.35.
Singh and co-workers performed environmental impact assessments for two potential continuous processes for the production of carbon nanotubes (CNT) [8]. The high–pressure carbon monoxide disproportionation in a plug-flow reactor (CNT-PFR) and the cobalt-molybdenum fluidised bed catalytic reactor (CNT-FBR) were selected for the conceptual design. The CNT-PFR reactor has catalytic particles formed in situ by thermal decomposition of iron carbonyl. The CNT-FBR process employs the synergistic effect between the cobalt and molybdenum to give high electivity to carbon nanotubes from CO disproportionation.
Healy and co-workers have investigated in the life cycle assessment the three more established SWNT manufacturing processes: arc ablation (arc), chemical vapour deposition (CVD), and high pressure carbon monoxide (HiPco) [9]. Each method consists of process steps that include catalyst preparation, synthesis, purification, inspection, and packaging. In any case, the inspection and packaging

steps contribute minimally to the overall environmental loads of the processes. Although the technical attributes of the SWNT products generated via each process may not always be fully comparable, the study provides a baseline for the environmental footprint of each process. All calculations are made with a functional unit of 1 g of SWNT.

Khanna and co-workers [10] have performed a cradle-to-gate assessment of carbon nanofiber (CNF) production. The goal of the assessment was to determine the non-renewable energy requirements and environmental impacts associated with the production of 1 kg of CNFs. Life cycle energy requirements for CNFs from a range of feedstock materials are found to be 13 to 50 times higher than primary aluminium on an equal mass basis.

Krishnan and co-workers have presented a cradle-to-gate assessment and a developed library of materials and energy requirements and global warming potential of nanoscale semiconductor manufacturing [11]. The goal of the study was to identify potential process improvements. The functional unit selected was 1 silicon wafer with a 300mm diameter that can be used to produce 442 processor chips. The total energy required for the process is 14,100 MJ/wafer including 2,500 MJ/ wafer accounts for the manufacture of fabrication equipment. The greenhouse potential is 13kg CO_2e /wafer.

Steinfeldt and co-workers have performed in several in-depth life cycle assessments of processes and products also cradle-to-gate assessments of the production of nanoscaled polyaniline and of the production of multi-walled carbon nanotube (MWCNT) and carbon nanofiber [12,13]. With the cooperation of firms, it was possible to produce detailed models of the manufacturing processes for nanoscaled polyaniline, for MWCNT and for carbon nanofiber to generate specific life cycle assessment data.

Kück and co-workers have investigated green silver nanoparticle production using micro reactor technology method in comparison with water-based batch synthesis [14]. The aim of the study was to reveal the potential and constraints of this approach and to show, how economic and environmental costs vary depending on process conditions. Because of the lower energy consumption and lower demand of cleaning agents, micro reactor is the best ecological choice.

The data above can provide some insight regarding the potential burdens that must be addressed if the large-scale use of these types of nanoparticles and nanocomponents is to continue. For this purpose the data from the studies is expressed in a common mass-based unit. Accordingly, energy demand is presented in MJ-equivalents/kg material and global warming potential is expressed as kg CO_2e/kg product. Energy consumption during the product life cycle is very important because it relates to the consumption of fossil fuels and the generation of greenhouse gases. Therefore, it is desirable to design manufacturing

processes that minimise the use of energy. The data for energy consumption of the materials discussed above is shown in Figure 1 and 2. Additionally into the comparison data of conventional materials is included.
The represented cumulative energy requirements for various carbon nanoparticle manufacturing processes differ very strongly from each other. The various processes for the production of SWCNT (excluding equipment fabrication) are by far the most energy intensive processes as compared with the production of other carbon nanoparticles. A cause for the very large differences between the examined studies lies in the different process conditions (temperature, pressure) of the manufacturing processes. Furthermore large differences are found in the assumptions of reactions and purification yields. The relative small reference value appears remarkable for the mass production of carbon black by means of flame synthesis. The production of MWCNT based on catalyst CVD surprises also here with a relative small cumulative energy requirement.

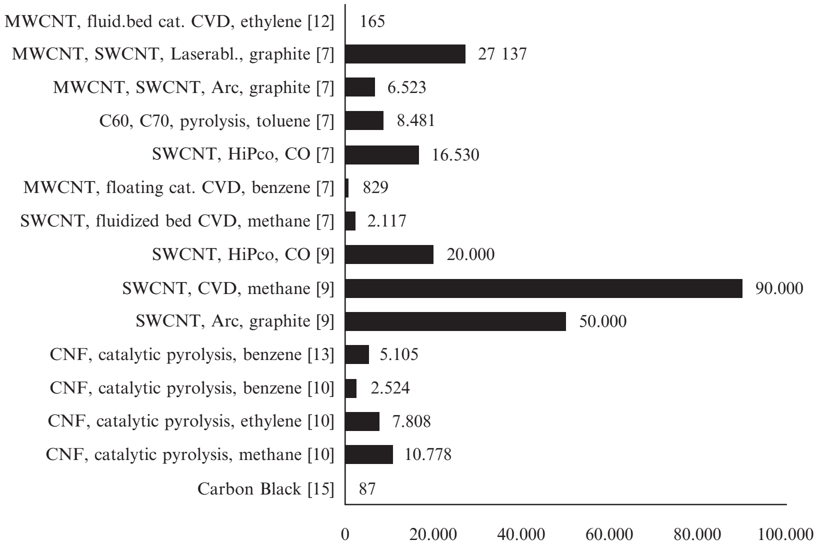

Fig. 1: Comparison of the cumulative energy requirements for various carbon nanoparticle manufacturing processes [MJ-equivalents/kg material]

The comparison of the cumulative energy requirements for the production of other conventional and nanoscaled materials and components make clear that the production of nanosemiconductors is also a very energy intensive process. Only the extraction of the precious metal platinum as example is still more complex. The production of nanoscaled polyaniline is likewise very energy-intensive.

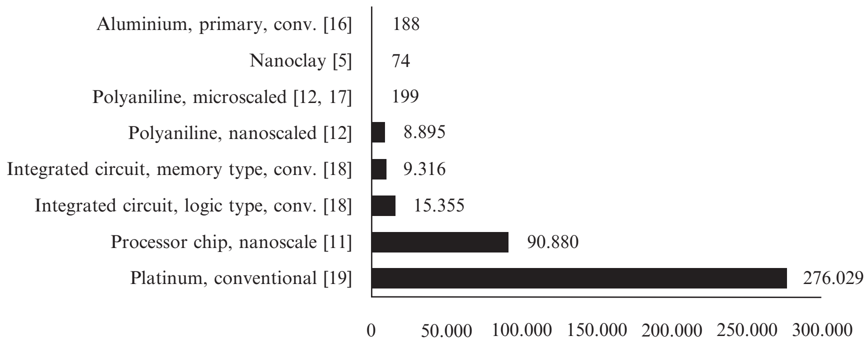

Fig. 2: Comparison of the cumulative energy requirements for the production of various conventional and nanoscaled materials and components [MJ-equivalents/kg product] (in parts own calculation)

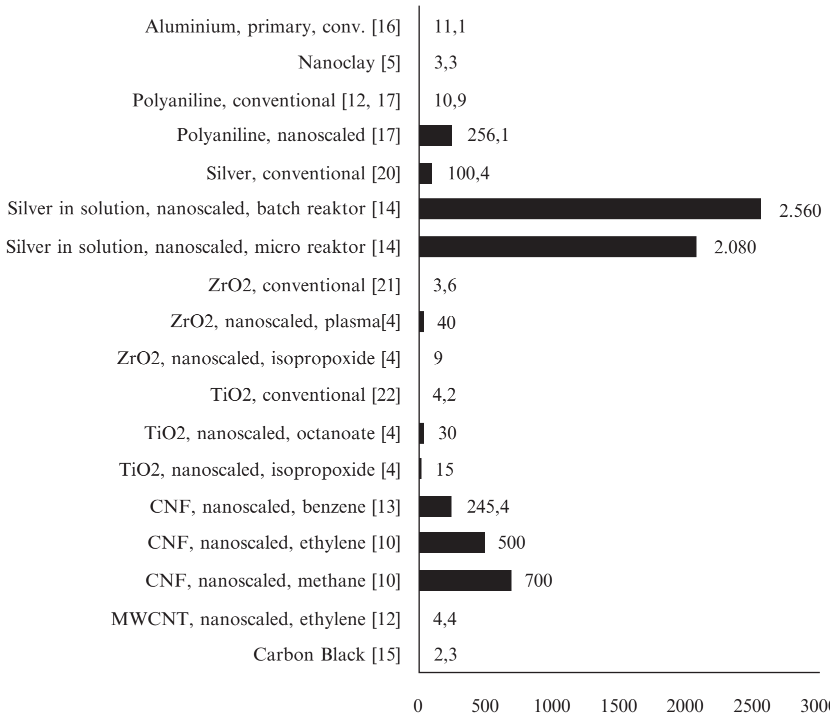

Fig. 3: Comparison of the global warming potential for the production of various conventional and nanoscaled materials [CO_2e/kg product] (in parts own calculation)

A comparison of the global warming potential for the production of various conventional and nanoscaled materials is shown in Figure 3. The production of silver nanoparticles has the largest impact when compared to the other materials. However, the production of carbon nanofiber and nanoscaled polyaniline demonstrates also a high global warming potential. The reason for the larger global warming potentials for silver nanoparticle, CNF and polyaniline manufacturing is the much larger energy requirements when compared to other nanoparticle production. Also the cleaning agents in the manufacturing processes have a large influence of the global warming potentials. The production of MWCNT based on fluidised bed catalyst CVD surprises also here with a small global warming potential.
This represented results place no comprehensive life cycle assessments, the results do offer useful insight when considering the environmental impact of various nanomaterials and nanotechnology-based applications. It is commonly pointed out that the nanocomponent is only a fraction of the total product (often only 2, 3 or 4 per cent) implying that only a small fraction of the environmental impact of a nanoproduct can be attributed to the nanocomponent and its manufacture. The high specific energy demand for the production of nanoparticle relates itself then in nanoproduct.

4 Summary

The prevalence of diverse manufacturing routes for nanoproducts is a significant driver for nanotechnological innovations. All nanoproducts must proceed through various manufacturing stages to produce a material or device with nanoscale dimensions. These techniques can be classified based on the type of approach in top-down or bottom-up. Top-down processes achieve nanoscale dimensions through carving or grinding methods (e.g., lithography, etching, and milling). Bottom-up methods assemble matter at the atomic scale through nucleation and/or growth from liquid, solid, or gas precursors by chemical reactions or physical processes (e.g. gas-phase deposition, flame-assisted deposition, sol-gel process, precipitation, and self-organisation techniques).
Quantitative investigations of anticipated or still to be realised environmental benefits arising from specific nanotechnological products and processes, as well as further-ranging environmental innovations such as product and production-integrated environmental protection or energy related solutions have so far been the exception. Currently, a large number of data gaps exist when considering the application of LCA to nanoproducts. Specifically, only minimal data detailing the

material and energy inputs and environmental releases related to the manufacture, release, transport, and ultimate fate of nanocomponents and nanoproducts exists. The presented new approach of prospective technological assessment of nanotechnological processes throughout their life cycle based on prospective scenarios and scaling-up models can help to close the existing data gaps.
Additional thought must also be given to address risk assessment and socio-economic impacts/benefits which should be integrated with the LCA framework to provide a more comprehensive assessment tool for decision making when considering the use of nanoparticles for the manufacturing of nanoproducts.

References

[1] Steinfeldt M, von Gleich A, Petschow U, Haum R (2007) Nanotechnologies, Hazards and Resource Efficiency. Springer, Heidelberg
[2] Sengül H, Theis TL, Ghosh S (2008) Toward Sustainable Nanoproducts: An Overview of Nanomanufacturing Methods. J Ind Ecol 12(3):329-359.
[3] Meyer DE, Curran MA, Gonzalez M (2009) An examination of existing data for the industrial manufacture and use of nanocomponents and their role in the life cycle impact of nanoproducts. Environ Sci Technol 43(5):1256-1263.
[4] Osterwalder N, Capello C, Hungerbühler K, Stark WJ (2006) Energy consumption during nanoparticle production: How economic is dry synthesis? J. Nanopart Res 8(1):1–9.
[5] Roes A, Marsili E, Nieuwlaar E, Patel MK (2007) Environmental and Cost Assessment of a Polypropylene Nanocomposite. J Polym Environ 15(3):212–226.
[6] Eckelman MJ, Zimmerman JB, Paul T, Anastas PT (2008) Toward Green Nano: E-factor Analysis of Several Nanomaterial Syntheses. J Ind Ecol 12(3):316-328.
[7] Kushnir D, Sandén BA (2008) Energy Requirements of Carbon Nanoparticle Production. J Ind Ecol 12(3):360-375.
[8] Singh A, Lou HH, Pike RW, Agboola A, Li X, Hopper JR, Yaws CL (2008) Environmental Impact Assessment for Potential Continuous Processes for the Production of Carbon Nanotubes. Am J Environ Sci 4(5):522-534.
[9] Healy. ML, Dahlben LJ, Isaacs JA (2008) Environmental Assessment of Single-Walled Carbon Nanotube Processes. J Ind Ecol 12(3):376-393.
[10] Khanna V, Bakshi BR, Lee J, Carbon Nanofiber Production: Life Cycle Energy Consumption and Environmental Impact. J Ind Ecol 12(3):394-410
[11] Krishnan. N, Boyd S, Somani A, Raoux S, Clark D, Dornfeld D (2008) A. Hybrid Life Cycle Inventory of Nano-Scale Semiconductor Manufacturing. Environ Sci Technol 42(8):3069–3075.

[12] Steinfeldt M,, von Gleich A, Petschow U, Pade C, Sprenger R-U (2010) Entlastungseffekte für die Umwelt durch nanotechnische Verfahren und Produkte (Environmental Relief Effects through Nanotechnological Processes and Products). UBA-Texte 33/2010, Dessau <http://www.umweltdaten.de/publikationen/fpdf-l/3777.pdf> (accessed 05.04.2011)

[13] Steinfeldt M, von Gleich A, Henkle J, Endo M, Morimoto S, Momosaki E (2010) Environmental relief effects of nanotechnologies – factor 10 or only incremental increase of efficiency. Proceeding of the 9th International Conference on EcoBalance Towards & Beyond 2020, Tokio, Japan

[14] Kück A, Steinfeldt M, Prenzel K, Swiderek P, von Gleich A, Thöming J (2010) Green nanoparticle production using micro reactor. J Phys, Conferences Series NANOSAFE 2010, Grenoble, (forthcoming)

[15] Hischier R (2007) Carbon Black. In: Althaus HJ, Chudacoff M, Hellweg S, Hischier R, Jungbluth N, Osses M, Primas A Life cycle inventories of chemicals. Ecoinvent report No 8. Swiss Centre for Life Cycle Inventories, Dübendorf

[16] Althaus HJ (2009) Aluminium. In: Althaus H, Blaser S, Classen M, Emmenegger MF, JungbluthN, Scharnhorst W, Tuchschmid M Life cycle inventories of metals. Ecoinvent report No 10, part I. Swiss Centre for Life Cycle Inventories, Dübendorf

[17] Hischier R (2007) Life cycle inventories of packaging and graphical papers. Ecoinvent report No 11, part II. Swiss Centre for Life Cycle Inventories, Dübendorf

[18] Hischier R, Lehmann M (2007) Electronic components. In: Hischier R, Classen M, Lehmann M, Scharnhorst W Life Cycle Inventories of Electric and Electronic Equipment: Production, Use and Disposal. Ecoinvent report No 18, part I, Swiss Centre for Life Cycle Inventories, Dübendorf

[19] Classen M, Althaus HJ (2009) Platinum group metals (PGM). In: Althaus H, Blaser S, Classen M, Emmenegger MF, JungbluthN, Scharnhorst W, Tuchschmid M (2009) Life cycle inventories of metals. Ecoinvent report No 10, part V, Dübendorf

[20] Classen M, Tuchschmid M, Emmenegger MF, Scharnhorst W (2009) Gold and Silver. In: Althaus H, Blaser S, Classen M, Emmenegger MF, Jungbluth N, Scharnhorst W, Tuchschmid M Life cycle inventories of metals, Ecoinvent report No 10, part IX. Swiss Centre for Life Cycle Inventories, Dübendorf

[21] Primas A (2007) Zirconium oxide production from mineral sands. In: Althaus HJ, Chudacoff M, Hellweg S, Hischier R, Jungbluth N, Osses M, Primas A Life cycle inventories of chemicals. Ecoinvent report No 8. Swiss Centre for Life Cycle Inventories, Dübendorf

[22] Chudacoff M (2007) Titanium dioxide. In: Althaus HJ, Chudacoff M, Hellweg S, Hischier R, Jungbluth N, Osses M, Primas A Life cycle inventories of chemicals. Ecoinvent report No 8. Swiss Centre for Life Cycle Inventories, Dübendorf

State of the Art Study - How is Environmental Performance Measured for Buildings/ Constructions?

Anne Rønning and Kari-Anne Lyng

Abstract Several studies have used life cycle assessments to measure the impacts of energy consumption in different building stocks in a quantitative way. The use of LCA as the assessing tool has become commonly used in this respect. Today, greenhouse gas emissions from buildings are mostly linked to energy consumption during its operation period. Through increasingly stringent energy requirements and other changes, energy use for the operation is likely to decrease over time. On the basis of a literature reviews, an assessment is carried out with the focus on explaining the methodological platforms the different studies are based on, and thereby explaining why the results vary and / or may not be comparable.

1 Introduction

For more than a decade life cycle assessment (LCA) has been developed as a tool for assessing environmental aspects of different building products and constructions during its lifetime. However, we see a lot of LCA and environmental product declarations (EPDs) of building materials which to a great extent have been limited to the environmental impacts associated with the building materials or products – cradle to gate. On the other hand as the energy for operation decreases as passive or low energy houses are built, the relative contribution to the total emissions in an LCA-perspective from building materials will increase.

Today, greenhouse gas emissions from buildings are mostly linked to energy consumption during its operation period. Through increasingly stringent energy requirements and other changes, energy use for the operation is likely to decrease over time. If so, this means that the energy consumed during production, transportation and construction of the building to a larger extent can be relatively more important in a life cycle assessment.

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The Norwegian Ministry of Local Government and Regional Development has in this connection commissioned Ostfold Research to conduct a literature study, which will provide an overview and assessment of existing literature / research reports that describe various building materials' global warming potential and how this translates into a life cycle perspective (LCA – life cycle assessment), and thereby describe the knowledge platform these assessments are based on. Moreover, it entails a description of the factors which affect the climate and environmental impacts, including the parts of the life cycle that are important.

2 The review study

The literature study was carried out through searches in scientific databases (Springer Link, Science Direct, Google Scholar, EPD Norway's database of environmental product declarations). The literature search is limited to studies that are based on LCA as a methodology for calculating the climate impacts associated with buildings and building materials.
Further, on the basis of the literature reviews, it is carried out an assessment with the focus on explaining the methodological platforms the different studies are based on, and thereby explaining why the results vary and / or may not be comparable. To illustrate this, this assessment is based on two statements that are strong in the public debate about the environmental impacts of building materials and buildings throughout their lifetimes; 1) Climate impacts of today's buildings are linked to the operational phase and 2) In low energy buildings the production phase becomes as important as the operational phase.
The methodological foundations that are used in the different studies are further applied in relation to the two statements.

3 Results

There are several LCA-studies that compare building materials, such as wood against concrete or steel. Many of the studies exclude phases or activities throughout the life cycle of the building [1]. The argumentation of the exclusion is not always clear, however some argue that "energy for heating is equal for both building systems" [2, 3]. This is clearly a weakness when comparing building systems based on different building materials.

Although numerous studies conclude that "the LCA shows that energy use during use phase is most important", it is simultaneously concluded that a building based on one particular building material is better than the other, without discussing if there are significant differences between the systems.
The different approaches of performing LCAs and of excluding certain life cycle phases or activities, affect the results in two main directions; a) what life cycle phases that has the largest impact and b) the influence of material choice on the results.
Haapio and Viitaniemi [4] have performed a literature review on different calculation tools for environmental evaluations based on LCA of entire buildings. The study shows that LCA results are dependent on the tool used, and that a comparison between results from different tools is impossible.

3.1 Are climate impacts of today's buildings linked to the operational phase?

One important result from the LCA analyses is that energy use for operation contributes to 70-90% of the total during the lifespan. This is a general conclusion and assertion that is verified whether analysing heavy or light constructions [1, 5, 11].
Several factors are influencing the relative importance between production of building materials and operation phase. They can be divided in three categories - LCA methodology, localisation and building technical aspects.
Geographical and climate conditions with respect to the localisation of the buildings, how they are designed, fitting in the terrain and how they are used will influence the total energy use and embodied energy. On the other hand these factors are not highlighted in the literature; it is mainly LCA methodological aspects that are focused.

3.1.1 Data

The selection of data is of vital importance for the results. First of all, an LCA can be approached methodologically from two different perspectives: bottom-up, based on process life cycle assessment (PLCA) or top-down, based on input-output life cycle assessment (IO-LCA) analysis. A combination is hybrid approaches, which link process information collected in physical life-cycle inventories with monetary flows in economic models.

In the building sector PLCAs have been the most usual approach. This approach is calculating emissions from the inputs by its masses, which represents challenges for several reasons. Firstly, the construction sector does not have a tradition to evaluate their projects on mass basis, only in economic terms. Thus, one does not have key figures or experienced based calculations to lean on. Secondly, in a feasibility phase one doesn't know which materials will be chosen. And last but not least, there are not environmental data available for all building materials and components.

The combination of LCA and input-output models has shown value as a complementary tool to traditional inventory methods in LCA. The reason is bipartial. Firstly, the total embodied energy is not included when using PLCA. Input-Output life cycle assessments for typical US industries indicate that on average up to 75% of total emissions were overlooked when only looking upon the industries direct emissions and not include services etc. [12].

Secondly, in a feasibility phase one does not know which materials will be chosen and the construction sector does not have a tradition to evaluate their projects on mass basis, only in economic terms. Especially in US one see the approach of IOLCA and Hybrid LCA utilised when analysing a construction project to overcome the lack of data and to include embodied emissions [3, 13-15].

3.1.2 Electricity model

When considering greenhouse gas emissions, the choice of energy model is essential, including the assumption of what energy carriers are generating the electricity consumed in the building. The assumption considerably affects the relative difference in impact from each life cycle phase. Reviews of variation in greenhouse gas emissions for different electricity models used in Norwegian calculations differ from 0 - 1.400kg CO_2e/kWh [16].

This implies that the choice of electricity model may overturn the conclusion that the use phase is the most important phase when it comes to greenhouse gas emissions, as for cumulative energy demand. This proves the importance of considering more environmental indicators than greenhouse gas emissions to obtain a holistic information basis.

3.1.3 Definition of user phase or operation

The perception and understanding of the term operation is not unambiguous. Figure 1 shows a typical environmental profile as a result of an LCA [9]. The phases and terms are in accordance with the MOMD-term – management, operation, maintenance and development and in addition end of life is included.

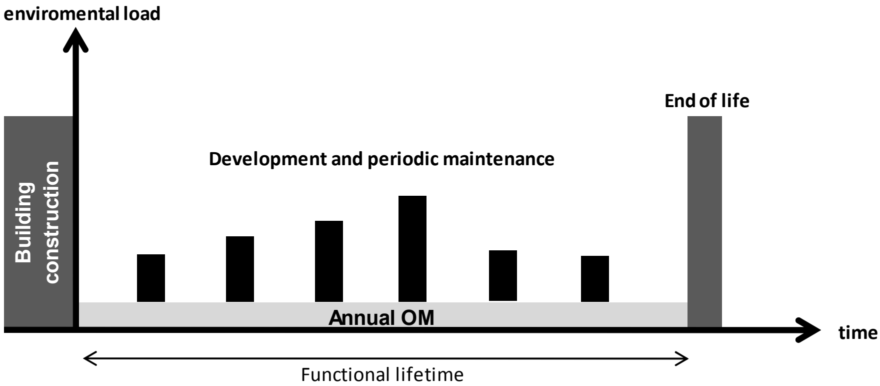

Fig. 1: Life cycle phases of a building [9]

Some studies are defining operation as use of energy for heating and cooling [17]. Other includes lighting, use of technical equipment and appliances in addition [18]. In the first case one can question whether lighting and use of technical equipment are explicitly included since it is hard to address the total energy used to different purposes. Other operation related activities like e.g. cleaning and inspection technical equipment are a part of daily operation of the building, but seldom included in LCAs. At the same time will design and choice of materials and equipment affect the need for and type of e.g. cleaning [15].

It is a challenge for the credibility of LCA results that the phases maintenance, replacement and development are excluded from the analysis. With that the total environmental performance through the life span of the building is under estimated. And the results from those studies are not presenting in real terms what life cycle phases are important and for what conditions they are important [7, 17]. Thus, design focused upon adaptability and use pattern is of more important than the building materials and products itself [15, 19-22].

The average rental period for office buildings in Norway is 7 years. Every seventh year the building is undertaken an extensive rebuilding due to new renters needs and requirements [15]. How extensive the rebuilding processes will be, is depending on the building's degree of adaptability.

This review study found no LCAs simulating consequences different use patterns have on the total energy use or other environmental aspects. This is interesting due to the fact that use pattern affect the energy use vitally. Comparison of five two-person households living in exactly equal houses in the same area shows that the variation between lowest and highest energy use for heating was 4.000-14.600 kWh/yr [23].

3.1.4 Factor of time

Estimated time, for which the LCA is undertaken, is of vital importance for the results due to a relatively long service life [19, 24, 25]. The choice of service life period (SLP) influence the climate impact vitally and the range in SLP is found to be 20-100 years in the literature. SLP is defines as the period where no changes of the building occur.

With the exception of service life predictions the factor of time has not been dealt with in LCA until now [25]. Changes over time, being changes in the building and changes in technology, are not mentioned. If SLP will last during the whole life of the building, then there is no need for high adaptability, e.g. an opera building. But, in those cases where SLP is short e.g. hospitals or private houses were changes occur due to new technology or changes in household and use pattern the different those aspects should be included in an LCA. So far mainly static factors such as energy for operation and maintenance are taken into account in LCAs [25]. Aspects related to adaptability; re-design, changes in use pattern and functionality are often neglected in LCAs. Adaptable buildings give lower refurbishments cost and then total low life cycle costs [25].

On the other hand LCAs are often based on technical service life for products given by producers. Then this will again be basis for defining scenarios for maintenance, repair, replacement etc. to be included in the LCA. This often leads to prediction of use phase activities which not reflect real life. Studies of SLP in practice, shows that the real SLP differ considerably from the SLP given by producers [27, 28]. The building product itself can satisfy the given requirements, but it may be other factors that define how and when changes occur, e.g. design and colour trends influence the need for paining and not necessarily the need for maintenance the set the frequency for panting.

3.1.5 Transport

Several studies conclude when doing LCA of buildings for planning purposes, especially regional planning, one should include transport activities related to use of the building. Transport may contribute to as high as 50 per cent of the total energy use [29, 33]. Transport of construction workers are not insignificant and are excluded in most studies [34]. On the other hand, LCA standards do not require transport of users (or workers) as a consequence of localisation to be included in LCAs.

3.2 Will the production phase become as important as the operational phase for low energy building?

A building material's impact on an entire buildings total energy use throughout the use phase has large influence on the results, as shown in earlier chapters. As the buildings become more and more energy efficient, the importance of the emissions from the production of the building materials increases [35-41]. Hubermann and Pearlmutter [41] give an example where upstream energy use is responsible for 60% of the total energy use. Dependent on type of building, location and type of model applied for the calculations of CO_2 factor for electricity, the balance between the impacts from each life cycle phase may shift even more. Simultaneously, awareness should be raised with regard to interaction between the life cycle phases. A narrow focus on the importance of each individual life cycle stage must not lead to ignorance of the significance of choices done when designing the building, such as choice of building materials, as these choices has large impact life cycle of a building [17].
Sartori and Hestnes have performed a literature review where the objective was to clarify the relative importance of energy use during operation opposed to energy use in upstream processes, including energy use to extraction of raw materials, production of building materials and on site construction and transport to site - especially related to low energy buildings [39]. Most of the 60 cases studied concluded that energy use during operation represent the largest contribution, and that low energy buildings are more energy efficient than conventional buildings although energy use for production of (upstream) materials increases.

4 Discussion

To ensure greater use of life cycle considerations, focus should be on the challenges along two axes: on the one hand, to strengthen the credibility of the underlying data and calculation methods of LCAs and on the other hand facilitate the use of results in actual construction processes, companies' product development and overall priorities at the state and municipal levels. There are a number of measures that could increase the use of lifetime considerations along the two axes. Examples of such measures could be:

Methodology – strengthen the credibility of calculations:

- Ensure equal calculation methodologies for LCAs of building materials, though the development of product category rules (PCR) for building materials and composite building elements such as external wall solutions, roof structures and floors
- Develop and make data available; establish key values or databases with realistic lifetimes for maintenance and development phases and investigate the relationship between user patterns and energy consumption
- Clarify the relationship between the building's adaptability and consequences on maintenance and replacements.
- Establish consensus on how the environmental data for different building materials can be calculated on the entire building's lifecycle; i.e. how to connect the material properties and technical properties different materials have, singularly or in combination with other materials

Encourage increased use of LCA in decision making processes and policy formulation:

- Clarify what environmental information decision-makers need in the various phases of the construction process
- Increase knowledge in the industry about the relationship between choices in the construction process and environmental performance through, for example, training courses, education and other outreach
- Integrate LCA results in existing tools that are traditionally used in the construction process (e.g. BIM)
- Encourage the private sector to increase their focus and knowledge of their own products by requiring the use of LCAs in the tender processes, in relation to new construction processes, rehabilitation and maintenance
- Encourage increased use of interaction processes in public development projects, where LCA can be used as a communication forum though simulation of the environmental consequences of choice
- Require LCA documentation with future scenarios for buildings of a given size in building permit procedures

5 Conclusion

Based on the reviewed LCA literature, we highlight the following main findings;

- The environmental impacts and energy consumption associated with the operation, maintenance and development phases (OMD) are of great and greater importance than the production of various materials.
- For low energy buildings, the relative importance of the production of building materials will increase.
- There is no basis to claim that one kind of building material should be prioritised over another with regard to environmental impacts.
- Through the inclusion of the overarching choices of solutions, which means that more phases and activities will be incorporated in the LCA, the total environmental loads through the building's life span will increase. The importance and scope of the various phases will depend on the purpose of the analysis, the type of construction, user patterns and more.
- LCA as a method makes it possible to assess the environmental consequences of different choices during the early planning stages, the design phase and the MOMD stage.
- Because it within the LCA modelling is given opportunities to make large variations in terms of calculation methods, it will be possible to get different results with regard to environmental impacts. The variations are explained in relation to the purpose of the study, the available data used and the quality of the data used as well as how the system boundaries are determined (which phases to include / exclude).
- Excising models and methods for calculating the LCA presents results in a form that is not necessarily adapted to the specific actors in the construction industry's need for environmental information. Neither are they adapted to existing tools which traditionally are used in the building process.
- LCAs are mostly used for documenting the consequences of already established choices and decisions or completed construction projects, and are to a lesser extent used as a planning tool for simulation of consequences of different choices in various phases of the construction process or though the lifetime of a building.

References

[1] Ortiz O, Castells F, Sonnemann G (2009) Sustainability in the construction industry: A review of recent developments based on LCA. Constr Build Mater 23:28-39.

[2] Upton B, Miner R, Spinney M, Heath LS (2008) The greenhouse gas and energy impacts of using wood instead of alternatives in residential construction in the United States. Biomass Bioenergy 32:1-10.

[3] Guggemos AA, Horvath A (2005) Comparison of Environmental Effects of Steel- and Concrete-Framed Buildings. J Infrastruct Syst 11(2):93-101

[4] Haapio A, Viitaniemi P (2008) A critical review of building environmental assessment tools. Environ Impact Assess Rev 28:469-482.

[5] Rønning A, Vold M, Nyland CA (2001) As a producer in an early stage in the value chain – how to effect decisions in the user phase? The 9th SETAC Europe Conference, Leiden

[6] Vold M, Hanssen OJ, Rønning A, Bjørberg S (2006) Environmental documentation for the life cycle of entire building constructions – Lessons learned from the building sector in Norway and from international methodology review. OR 17.06, STØ, Fredrikstad

[7] Fernandez NP (2008) The influence of construction materials on life-cycle energy use and carbon dioxide emissions of medium size commercial buildings, Master Thesis (Building Science). School of Architecture, Victoria University of Wellington

[8] Barrett J, Wiedmann T (2007) A comparative Carbon Footprint Analysis of On-Site Construction and an Off-Site Manufactured House. Research report 07-04, Stockholm Environment Institute

[9] Rønning A, Nereng G, Vold M, Bjørberg S, Lassen N (2007) JOMAR - A Model for Accounting the Environmental Loads from Building Constructions. OR.07.07, Ostfold Research, Fredrikstad

[10] Dimoudi A, Tompa C (2008) Energy and environmental indicators related to construction of office buildings. Resour Conserv Recycl 53:86-95.

[11] Bribián IZ, Usón AA Scarpellini S (2009) Life cycle assessment in buildings: State-of-the-art and simplified LCA methodology as a complement for building certification. Build Environ 44:2510-2520.

[12] Matthew HS, Hendrickson CT, Weber CL (2008) The importance of carbon footprint estimation boundaries. Environ Sci Technol 42(16): 5839-5842.

[13] Sharrard AL, Scott Matthews H, Ries RJ (2008) Estimating Construction Project Environmental Effects Using an Input-Output-Based Hybrid Life-Cycle Assessment Model. J Infrastruct Syst 14:323-336.

[14] Treloar GJ, Love PED*,* Faniran OO, Iyer-Raninga U (2000) A hybrid life cycle assessment method for construction. Constr Manage Econ 18:5-9

[15] Rønning A, Vold M, (2008) Miljøvurdering av nytt hovedkontor for Sparebank 1 SMN. Sammenligning av to alternative løsninger"*, Østfoldforskning*, OR.10.08, Fredrikstad

[16] Nereng G (2010) Presentasjon Bellonas Energiforum. 26. November, Oslo
[17] Blengini G, Di Carlo T (2010) The changing role of life cycle phases, subsystems and materials in the LCA of low energy buildings. Energy Build 42(6):869-880.
[18] Scheuer C, Keoleian G, Reppe P (2003) Life cycle energy and environmental performance of a new university building: modeling challenges and design implications. Energy Build 35(10):1049-1064
[19] Erlandsson M, Borg M (2003) Generic LCA-methodology applicable for buildings, constructions and operation services-today practice and development needs. Build Environ 38:919-938.
[20] Davison N, Gibb AG, Austin SA, Goodier CI (2006) Adaptables, international conference on Adaptable Building Structures. Juli 2006, Nederland
[21] Gibb A, Austin S, Dainty A, Davison N, Pasquire C (2007) Towards adaptable buildings: pre-configuration and re-configuration – two case studies. ManuBuild 1st International Conference, The transformation of the Industry, Open Building Manufacturing, April 2007, Rotterdam
[22] Manewa A, Pasquire C, Gibb A, Schmidt R (2009) Towards economic sustainability through adaptable buildings. Adaptable Futures, Loughborough University, UK
[23] Gram-Hanssen K (2010) User practices and variations in energy consumption of households, PHD –Course, Sustainable buildings - technical innovations and user practices. Universitetet i Aalborg
[24] Haapio A, Viitaniemi P (2008) Environmental effect of structural solutions and building materials to a building. Environ Impact Assess Rev 28:587-600
[25] Klunder G (2002) The factor of time in life cycle assessment of housing. Delft University of Technology, the Netherlands
[26] Valen MS, Bjørberg S (2006) The dynamic and adaptable building in a life cycle perspective, CIB W70 - Changing User Demands on Buildings - Needs for Lifecycle Planning and Management. Trondheim
[27] Bjørberg S (2009) Levetidstabeller kan misbrukes – og det gjøres, Juristkontakt nr. 9
[28] Haagenrud SE (2006) Praktisk bruk av standarder for livsløpsplanlegging og levetidsprosjektering. presentasjon på seminar om Bærekraftig byggevirksomhet Standard Norge, 24. April 2006
[29] Norman J, MacLean HL, Kennedy CA (2006) Comparing High and Low Residential Density: Life-Cycle Analysis of Energy Use and Greenhouse Gas Emissions. J. Urban Plann Dev 132(1):10-21.
[30] Rønning, A., Modahl IS, Nordahl SM (2006) Miljømessige effekter av lokalisering av IR-avdeling v/HIØ'. Ostfold Research OR 07.06, Fredrikstad
[31] Selvig E (2007) Klimagassregnskap for utbyggingsprosjekter. Utvikling av er beregningsverktøy for eksempel på et klimagassregnskap for en del av Fornebu". Civitas, Oslo

[32] Selvig E, Cervenka Z (2008) Holistic life cycle GHG emissions associated with buildings. presentasjon på World Sustainable Building Conference, Melbourne
[33] Statsbygg: klimagassregnskap.no - beregningsverktøy for klimagassutslipp fra byggeprosjekter, <www.klimagassregnskap.no>, 2011
[34] Blom I, Itard L, Meijer A (2010) Environmental impact of dwellings in use: Maintenance of façade components. Build Environ 45:2526-2538.
[35] Treloar GJ, Fay R, Ilozor B, Love PED (2001) An analysis of the embodied energy of office buildings by height. Facil 19(5-6):204-214.
[36] Thormark C (2002) A low energy building in a life cycle—its embodied energy, energy need for operation and recycling potential. Build Environ 37:429 – 435.
[37] Yohanis YG, Norton B (2002) Life-cycle operational and embodied energy for a generic single-storey office building in the UK. Energy 27:77-92.
[38] Thormark, C., The effect of material choice on the total energy need and recycling potential of a building. Build Environ 41:1019–1026.
[39] Sartori I, Hestnes AG (2007) Energy use in the life cycle of conventional and low-energy buildings: A review article. Energy Build 39:249-257.
[40] Brunklaus B, Thormar C, Baumann H (2008) Passivhus och konventionella hus – en miljöjämförels. Bygg & Teknik 5
[41] Hubermann N, Pearlmutter D (2008) A life-cycle energy analysis of building materials in the Negev desert. Energy Build.40:837-848.

PART III:
Water Footprint

Comparison of Water Footprint for Industrial Products in Japan, China and USA

Sadataka Horie, Ichiro Daigo, Yasunari Matsuno and Yoshihiro Adachi

Abstract Recently, water scarcity has received attention. With the development of industries and the growth of population, the amount of water use has increased. In order to evaluate the water use of industrial products, the method of estimating water footprint (WF) has been developed. WF is defined as the amount of water use during the lifecycle of products or services. In this study, we estimated WF of industrial products in Japan, China, and the U.S. using input-output analysis. It was found that WF for BOF crude steel in Japan was estimated as 0.62 m^3/t, whereas WF for EAF crude steel in Japan was estimated as 0.85 m^3/t. WF of crude steel in China was estimated as 0.99 m^3/t. In the U.S. the pig iron, crude steel and ferroalloy cannot be divided into each sector, so we cannot compare the results of the U.S. to those of Japan and China. In WF for a passenger car, the indirect water use dominated their WF in all countries. To compare the results in each sector between countries appropriately, consistency of industrial sector in the data for water use is required.

1 Introduction

Water is indispensable for life. According to Japan's statistics for 2006, approximately 15.7 billion m^3 of water was used in private households, while approximately 12.6 billion m^3 was used in industries, and 54.7 billion m^3 was expended in the agricultural sector. Looking at worldwide figures, as populations grow and industries continue to develop, it is clear that we will require larger quantities of water in the future. Thus, potential shortages are cause for concern. This situation highlights the increased importance of accurately evaluating water withdrawals during product manufacturing. To facilitate such evaluations, the water footprint (WF) concept is being put forward as a means for quantifying the water amounts required to produce a certain product, as well as the quantity that a product will require over its lifecycle. When determining the WF, we designate the quantity of water required in the various manufacturing processes as the

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quantity directly used (direct withdrawal), and the quantity indirectly used (indirect withdrawal). Kondo et al. calculated the water quantities used directly in various industries, and then, using input-output analysis, calculated the WF of various Japanese industrial products [1]. Blackhurst et al. calculated the WF for the primary, secondary and tertiary industry segments of the U.S., using input-output analysis [2]. Zhao et al. used a comparable method to calculate the WF for 23 segments in China [3].

In this work, we focus on water consumption for industrial products, especially, in the iron and steel industry. In order to investigate limitations to the water necessary for future steel production demands, it was first necessary to calculate the WF for various regions where steel is produced and compare them. However, no detailed analysis has been conducted to determine China's steel WF. Furthermore, no studies have yet been performed that compare the WF of the same industrial products in different regions and countries. Thus, the objective of our research was to calculate the WF of industrial products (primarily steel) in Japan, China and the U.S., and then to conduct a comparative evaluation.

2 Method

In our research, we used input-output (I-O) analysis to calculate the WFs of various industrial products. Using I-O analysis we can calculate the WF with the Equation (1):

$$W = d(I-(I-M)A)^{-1} \tag{1}$$

Here, I is the identity matrix, M is the import coefficient matrix, A is the input coefficient matrix, and d is the vector for direct water withdrawals.

The data availability of water withdrawal varied from one country to another. Therefore, when calculating the vector d for direct water withdrawals, we used different methods for each country.

For Japan, according to the industrial statistics report by Industrial Site and Water [4], a publication of the Ministry of Economy, Trade and Industry, direct withdrawals in various industries have been determined for the water quality such as industrial water, public water consumption, groundwater, other freshwater, and seawater. Also the withdrawals have been determined for the water usage such as cooling usage, boiler usage, water for material use, product treatment, rinsing, and other applications. In our research, in order to calculate freshwater withdrawals, we considered both direct and indirect withdrawals for industrial and private water consumption, in addition to groundwater use. In contrast, seawater withdrawals were not considered. Because the number of industrial sectors in the statistics [4] is larger than that in the I-O table in 2005 [5], we aggregated the industrial sectors (560 sectors) into 246 in accordance with the literature [6].

For China, data for industrial waste water were obtained [7]. Then, we assumed that the water withdrawals could be equivalent to the waste water quantities. The industrial waste water data for China were obtained for 38 industrial sectors. Therefore, we allocated them into 89 corresponding industry sectors in the Chinese input-output table in 2007 [8], based on the transaction value for the "Production and Distribution of Water" item in Chinese input-output table in 2007.

For the U.S., obtained were water withdrawal data [9] for eight different items (public supply, domestic, industrial, irrigation, livestock, aquaculture, mining and thermoelectric). So, these were allocated to 279 industry types in the U.S. input-output table in 2002 [10] according to the previous study [3]. WF per economic value of each industry was calculated by Equation (1). Then, we multiplied this by the producer price of each product to find the per-ton or per-unit WF.

Tab. 1: Comparison of data sources and estimation methods in the three countries

Country	Data on direct water withdrawal	The number of sectors for industry in I-O tables	Method of allocation or aggregation
Japan	Industrial water withdrawals for 560 sectors	246 sectors	Using the code correspondence table
China	Industrial waste water for 38 sectors	89 sectors	Using the transaction value for the "Production and Distribution of Water"
The U.S.	Water withdrawals for 8 sectors	279 sectors	Referring to the former study [3]

3 Results

Figure 1 and 2 show the results of the WF calculated for Japan, China and the U.S. using the technique described in section 2.

Figure 1 shows the results of the WF for the iron and steel industry in each country. As shown in Figure 1, the industrial classifications are different in each country. For example, classification of the industry in the U.S. I-O tables includes pig iron, crude steel and ferroalloy, so we can't compare the result to other countries. Figure 1 shows that the WF for the pig iron and crude steel in Japan were smaller than those in China. Especially for the pig iron, the WF in China is about twice as large as that in Japan.

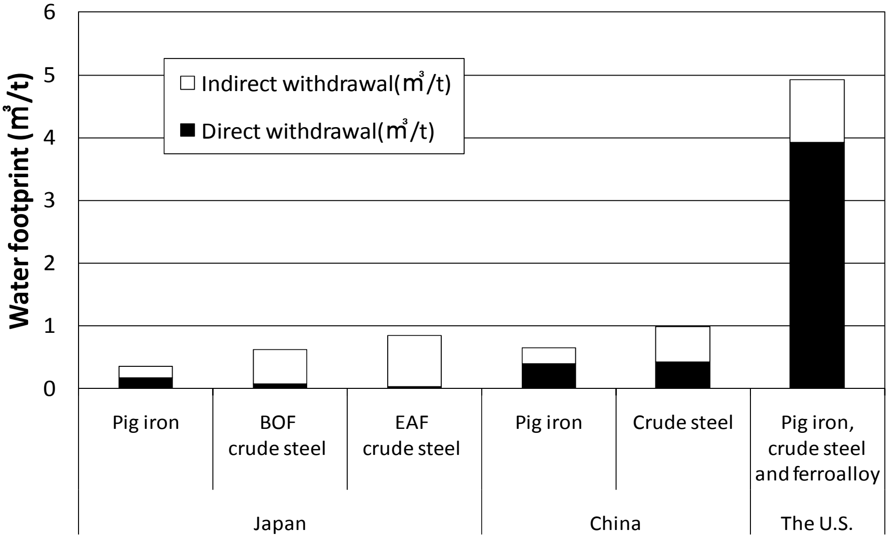

Fig. 1: The WF for iron and steel industry in Japan, China and the U.S.

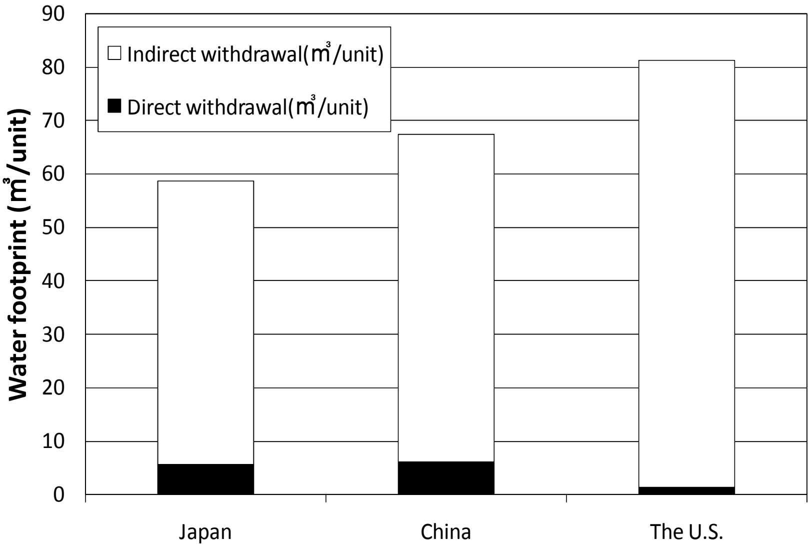

Fig. 2: The WF for a passenger car in Japan, China and the U.S.

Figure 2 shows the WF for a passenger car in each country. Every country has a sector related to passenger car in I-O tables. The Chinese I-O table has a sector named "Automotive", which includes trucks, buses, passenger cars, motor cycles, and bicycles. Therefore, we converted all of them into passenger car equivalent on the basis of the price ratio in Japan. The WF in the U.S. is about 1.4 times as large as that in Japan. For all three countries, we found that indirect withdrawals accounted for almost all WF in the automotive sector. However, there are uncertainties due to the data availability, so further study is required with more accurate data in detail.

4 Discussion

Using the calculated WF, we estimated the total amount of water withdrawal for upstream life cycle until producing crude steel in Japan and China. Table 2 shows the WF for crude steel and the amount of crude steel production [11]. The results are shown in Table 3. As Table 3 shows, 73 million m^3 of water was used for crude steel in Japan, and 620 million m^3 in China. If we can produce the amount of Chinese annual crude steel production with the water efficiency in Japan, we can save one-third of water use for crude steel in China.

Tab. 2: The WF for crude steel and the amount of crude steel production in Japan and China.

		Annual crude steel production in 2007 (1,000t)	**WF for crude steel (m^3/t)**
Japan	BOF crude steel	85,756	0.62
	EAF crude steel	23,845	0.85
China		626,654	0.99

BOF: basic oxygen furnace
EAF: electric arc furnace

Tab. 3: Total amount of water withdrawal for upstream life cycle until producing crude steel in Japan and China

		Total amount of water withdrawal for upstream life cycle until producing crude steel (million m^3)
Japan	BOF crude steel	53
	EAF crude steel	20
China		620

5 Conclusions

In our study, we calculated the WFs of industrial products (such as iron, steel and passenger cars) for Japan, China and the U.S. and compared them. It is expected that the needs for steel products and cars will be larger. We quantified the water needs for producing process of industrial products. However the water data from different sources was used when calculating the WFs of the three countries, as were differences in the industrial classifications in the input-output tables of the countries. Therefore the WF in three countries were compared under some assumptions. In the future, we believe it will be necessary to determine the WFs more precisely, by use of various other methods, such as compiling more detailed data concerning each classification.

References

[1] <http://hydro.iis.u-tokyo.ac.jp/Info/Press200905/> (accessed11.06.2010)

[2] Zhao X, Chen B, Yang ZF (2002) National water footprint in an input-output framework – A case study of China 2002. Ecol Modell 220:245-253.

[3] Blackhurst M, Hendrickson C, Sels i Vidal J (2010) Direct and Indirect Water Withdrawals for US Industrial Sectors. Environ Sci Technol 44(6):2126-2130.

[4] <http://www.meti.go.jp/statistics/tyo/kougyo/result-2.html> (accessed 02.07. 2010)

[5] <http://www.e-tat.go.jp/SG1/estat/List.do?bid=000001019588&cycode=0> (accessed 18.06. 2010)

[6] Ministry of Internal Affairs and Communications (2009) the Input-Output Tables in 2005 Data Report (2). Research institute of economy, trade and industry

[7] Zhenhua X (2008) China Circular Economic Yearbook 2008. China Financial & Economic Publishing House

[8] National Bureau of Statistics National Development and Reform Committee (2009) Input- Output tables of China, China Statistic Press

[9] <http://water.usgs.gov/watuse/data/2000/index.html> (accessed 02.12.2010)

[10] <http://www.bea.gov/industry/io_benchmark.htm> (accessed 02.12.2010)

[11] <http://www.jisf.or.jp/data/iisi/index.html> (accessed 05.02.2010)

Assessment of the Water Footprint of Wheat in Mexico

Carole Farell, Sylvie Turpin and Nydia Suppen

Abstract Water footprinting is becoming a popular way of understanding the total water input to consumer products along its life cycle. Nevertheless, when the water footprint is defined only in terms of volume it can conceal important impacts on the environment caused by the use of water. When the water footprint is assessed with a life cycle approach and normalised with the water stress index, it is possible to integrate to a water footprint accounting, both the environmental impacts caused by the use of water and the water scarcity caused by the process that is being evaluated. The proposed methodology was applied to the growth of irrigated wheat in Mexico. Results show not only the volume of water used, but also the impact on ecotoxicity and depletion of the resource that agricultural activity causes in México. Results are also compared with the water footprint of wheat in México assessed with different methodologies.

1 Introduction

In order to address the unsustainable use of environmental resources several indicators have been developed. The term “footprint” originates from the ecological footprint discussion [1], and it implies the measure of the total amount of environmental impacts caused directly or indirectly by persons, organisations or products in all stages of their life cycle. Just as the ecological footprint is a measure of human demand on the Earth's ecosystems, the concept of "water footprint" should be a measure of human demand on the Earth's water resources. In both concepts the environmental impacts are implicit. Such environmental impacts are measured by first conducting an emissions inventory. Once you know the size of the footprint, it is possible to implement a strategy to reduce it.

On the other hand, we can find some strengths and weaknesses in the Water Footprint Network (WFN) methodology that defines the water footprint as “the total volume of freshwater that is used to produce the goods and services

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consumed by the individual or community or produced by the industry" [2]. The advantages of this framework are that its meaning is intuitive and the required data is available in most cases; it can be an effective public-awareness building tool; nevertheless, with this methodology it is not possible to evaluate the depletion neither the resource nor the environmental impacts caused by the use of water. Reference to the quality of the water is not made either, which can damage human health. This framework details the origin of the resource and reports three types of water individually [2]: blue water (surface and groundwater), green water (soil water originating from rainfall), and grey water (volume of water that is required to dilute pollutants to such an extent that the quality of the ambient water remains above agreed water quality standards).

The concept of neutral water [3,4] implies to reduce the water footprint of an activity as much as reasonably possible and offsets the negative externalities of the remaining water footprint. This concept has not been fully developed yet, since in order to compare the water footprint and, if it is the case, reduce its impacts, a standardised methodology that measures the footprint in these terms must be established first. The lack of a standardised methodology that helps report the water footprint, has made the International Organization for Standardization (ISO), start the process to develop a new standard that will establish the requirements and guidelines to report the water footprint within the principles of life cycle assessment [5].

The life cycle assessment (LCA) methodology assesses the potential environmental impacts throughout a product's life cycle: from raw material extraction through production, use and disposal. Nevertheless, the LCA methodology assesses water only in the life-cycle inventory phase, where resource use, energy and material consumption are recorded, but little differentiation is made between the types of water use. Recent research focuses on the design of an LCA method which, from the inventory, identifies the necessary data to characterise and assess the environmental impacts caused by the use of water [6-9]. The UNEP –SETAC Life Cycle Initiative is currently performing a review of existing methods linked to water assessment within a LCA framework. The aim is to identify the key elements to be considered when modelling the cause effect chains in water assessment and provide guidelines to derive operational characterisation methods and factors to assess water use in LCA. [10].

This article suggests to consider the proposal of the WFN methodology [2], and complete it with the LCA technique [11], with which it is possible to assess, within the grey water component (concept within the WFN), the environmental impacts generated by the use of water. The grey water is normalised with the water stress index of the region where the resource is being extracted from. Results are compared with the water footprint of wheat assessed with other methodologies. A preliminary discussion of the proposed methodology and conclusions are provided.

2 Methodology

The methodology proposed consists on the individual evaluation and analysis of the blue, green and grey components as described below. The grey water is normalised with the water stress index in order to reflect the impact in terms of water depletion. It is not recommended to add the three components as a total to provide a metric for water footprint. The aggregated blue-green-grey metric can be misleading and does not clearly communicate the problems of water use.

2.1 Calculation of blue water

The blue component of the production of irrigated wheat in Mexico is obtained from the sum of the volume of water used in irrigation of the field, divided by the wheat yield. The data is obtained from statistics published by the National Water Commission of Mexico [12]. These statistics quantify individually the volume of freshwater withdrawals at each irrigation district from different sources: surface water (dams, rivers, lakes) and groundwater which is distributed by gravity or pumping.

2.2 Calculation of green water

A large number of empirical or semi-empirical equations have been developed for assessing reference evapotranspiration from meteorological data. Numerous researchers have analysed the performance of the various calculation methods for different locations. As a result of an Expert Consultation held in May 1990, the FAO Penman-Monteith method is now recommended as the standard method for the definition and computation of the reference evapotranspiration ETo [13]. In this research the green component is quantified with the FAO 56 Penman-Monteith method, using the definition of the reference crop as a hypothetical crop with an assumed height of 0.12 m having a surface resistance of 70 s/m and an albedo of 0.23, closely resembling the evaporation of an extension surface of green grass of uniform height, actively growing and adequately watered. It is then

multiplied by the specific crop coefficient of wheat (Kc) that corresponds to the climate of the irrigation district that is being assessed. The weather parameters required are: a) localisation (altitude, latitude altitude), b) wind speed, c) maximum and minimum air temperature, d) maximum and minimum relative humidity, and e) sunshine hours. The afore variables are obtained with a time step of ten minutes from the automatic meteorological stations (AMS) located throughout the country [14]. Hence, the green water footprint is calculated as the minimum of total crop evapotranspiration under standard conditions (ETc) and effective rainfall with a time step of 30 days. The total green component is divided by the wheat yield.
The FAO56 Penman-Monteith equation is stated as follows:

$$\mathrm{Eto} = \frac{0.408\,\Delta\,(\mathrm{Rn} - \mathrm{G}) + \gamma\,\frac{900}{\mathrm{T}+273}\,\mathrm{u}_2\,(\mathrm{e_s} - \mathrm{e_a})}{\Delta + \gamma\,(1 + 0.34\,\mathrm{u}_2)} \tag{1}$$

where:
ETo = evapotranspiration of the culture of reference [mm/day]
Rn = net radiation in the culture surface [MJ/m² day]
G = heat flow from the soil [MJ/m² day]
T = medium temperature of air at 2m height [ºC]
u_2 = wind velocity at 2m height [m/s]
e_s = saturation vapour pressure [kPa]
e_a = real vapour pressure [kPa]
(e_s - e_a) vapour pressure deficit [kPa]
Δ = vapour pressure slope [kPa/ºC]
γ = psychrometric constant [kPa/ºC]

It is possible that the FAO56 Penman-Monteith equation results are occasionally deviated from ETo of grass real measurements; however, the hypothetical culture of reference definition is used - in which the FAO56 Penman-Monteith equation is based - as an homogenized comparison value, in such a manner that the data of different zones are comparable among them [13].

2.3 Calculation of grey water

According to the ISO 14044 standard [11], the LCA has four stages: 1) Goal and scope; 2) Life cycle inventory (LCI); 3) Life cycle impact assessment (LCIA); and

4) Interpretation of results. The quantification of the grey component is proposed within the following LCA framework:

1) Life cycle inventory (LCI): consists in quantifying the volumes of polluted water, phosphatised fertilisers, nitrogenated fertilisers, pesticides, land use and agricultural machinery needed to produce one ton of wheat.
2) Life cycle impact assessment (LCIA): The LCIA's objective is to improve the understanding of the individual emissions of the life cycle inventories [11]. This is done using a weighted average of the sum of the pollutant emissions of a product system, with the help of characterisation factors. In the case of the grey wheat water footprint in Mexico, a midpoint impact category is assessed: aquatic ecotoxicity chronic (etwc), with the EDIP method [15] and the Simapro 7.2 software.

Toxic effects which are not acutely lethal and which first appear after repeated or long-term exposure to the substance are called chronic ecotoxicity. The ecotoxicity potentials (EP) are determined as the product of the quantity of substance Q emitted and the equivalency factor EF for the emission. To calculate the equivalency factors it is therefore first necessary to determine the fraction (fwc) of the emission which reaches the environment after dispersion. Also to calculate ecotoxicity factors (ETFwc) representing the substance's potential ecotoxicity. And finally, determine the biodegradability factor (BIO) for the substance [15]:

$$EP\ (etwc) = Q \bullet EF(etwc) = Q \bullet fwc \bullet ETFwc \bullet BIO \tag{2}$$

This category aggregates all toxic emissions potentially impacting the environment into water cubic meters. It corresponds to the volume to which the emission should be diluted in order to obtain a concentration of substance so low that no ecotoxic effects would be expected from the emission. This model is compatible with the grey water definition of the WFN.

Even if the theoretical volume of pollution is not the most suitable method for assessing the grey water footprint (because there is no critical dilution for persistent substances), the advantage of reporting in cubic meters gives the possibility to have the three components of the water footprint with the same intuitive units.

3) Interpretation of results: the results obtained are analyzed according to the theoretical yield, the efficiency in the use of the resource and the water stress of the area. Besides, the results are compared with other studies that have estimated the water footprint of wheat in Mexico with other techniques.

2.4 Normalisation

Normalisation of the grey component is done in order to reflect the contribution of the process to the depletion of the resource. The water stress index of the region from which the resource is extracted is used in order to normalise the impact category. Water stress is defined by the ratio of total annual freshwater withdrawals to the total renewable water resources. Severe water stress occurs above a threshold of 40% [16].

3 Results

Approximately 80% of wheat in Mexico is cultivated in four irrigation districts located in the north and central part of the country. The impacts of the low efficiency in the water use can be clearly seen, as the water stress of the zone in which such districts are located is severe (up to 91.4%). Its average yield of 5.2-6.6 ton/ha is considered low since the theoretical yield is 6-9 ton/ha. The blue component is at the upper limit of the theoretical efficiency (625-1250 m^3/ton). The green component is low due to the climate conditions under which wheat is grown (Table 1). The grey water footprint integrates the theoretical water volume that would be necessary to dilute all pollutants emitted from the extraction of raw materials, production and the use of fertilisers, pesticides, land use and agricultural machinery. In order to reflect the contribution of the process to the depletion of the resource, it is normalised with the water stress index of the region. It is thus that we obtain that the grey water footprint of wheat in Mexico for the 2004-2009 period is 19,364 m^3/ton (Table 1). This value represents the water that is necessary to avoid the ecotoxicity impacts and it also reflects the shortage caused by the agricultural process. In this way we obtain a complete and understandable methodology that assesses not only the use of the resource but also the impacts and shortage that such use causes. A methodology with these characteristics can be used both for scientific investigations and by the general public, supplying more relevant information to make decisions at the social, government and corporate levels.

Tab. 1: Water footprint of irrigated wheat in Mexico (period: 2004-2009)

Country	Blue water footprint (m^3/ton)	Green water footprint (m^3/ton)	Grey water footprint (m^3/ton)	Water stress (%)	Normalised grey water footprint (m^3/ton)	Coefficient of variation (%)
México	1,140	72	10,311	87.8	19,364	18.95

4 Discussion

Different studies on water footprint of agriculture have been published but the results are not consistent (Table 2). A full comparison of these studies is not possible because the periods of analysis, methods and boundaries are different. Therefore the need and importance of standardising a methodology that allows a clear comparison between studies.

For example, in the present study only the irrigated wheat in Mexico is assessed. For the blue component, real statistics of the volume used in the growth of wheat in Mexico are used. As a result of the faulty hydro agricultural infrastructure of the country with an efficiency of 36%, water usage exceeds the theoretical volumes of irrigation. In addition, due to a problem of salinisation of soils it is necessary to over irrigate the fields. The green component is assessed with local data and it is low due to the arid climate conditions under which the irrigated wheat is grown. The grey water is the volume to which all agriculture emissions should be diluted in order to obtain a concentration of substances so low that no ecotoxic effects would be expected from the emissions (Table 2).

A recent study by Mekonen M.M. et al. [17], quantifies the green, blue and grey water footprint of global crop production using a grid-based dynamic water balance model. The assessment was made for irrigation and rain fed crops. The blue water footprint of wheat in Mexico was estimated as the required irrigation volume. The green water footprint was estimated with national evapotranspiration averages. The grey water footprint quantification was related to nitrogen use only (Table 2). Because of these differences in the assessments of the water footprint of wheat in Mexico, the results are not consistent and it is not possible to compare these two studies.

Tab. 2: Studies on water footprint of wheat in Mexico

Study	**Country**	**Crop**	**Blue water footprint (m^3/tonne)**	**Green water footprint (m^3/tonne)**	**Grey water footprint (m^3/tonne)**	**Total water footprint (m^3/tonne)**
Mekonen and Hoekstra [17]	México (1996-2005)	Wheat	558	333	185	1,076
Chapagain and Hoekstra [18]	México (1997-2001)	Wheat	-	-	0	1,066
Farell, Turpin and Suppen	México (2004-2009)	Wheat	1,140	72	19,364	Non aggregation

On the other hand, in the study of Chapagain et al. [18], the grey water is not evaluated and the blue and green water are aggregated as a total water footprint. Adding the blue and green components and report them as a single metric can be misleading because a low value is not necessarily the best option, for example, a cubic meter of water from Mexico and the same volume from another part of the world cannot be evaluated in the same way. It is important to evaluate each component individually as they have different meanings and applications.
By integrating the environmental impacts caused by the use of water to the grey component we obtain a better perspective. To solely report water volumes hides important aspects of the environmental assessment: for example, water from an over-exploited aquifer generates bigger impacts on the environment than the same volume extracted from a lake. On the other hand, in order to combat the world water shortage, it is necessary to assess the depletion of the resource in different processes. This is not reflected when assessing water volumes only and, is clearly visible when it is normalised with the water stress index.

5 Conclusion

The assessment of the water footprint with a life cycle approach is a tool that facilitates the management of the hydric resource in a comprehensive way. It is important to assess not only the volume of water used, but also the impacts on the environment and the depletion that different activities cause when using this resource. The water footprint of wheat in Mexico has been calculated in this way emphasizing that its main problems are the low efficiency of the hydro agricultural infrastructure, forcing agricultural activity in arid and impoverished soils, as well as performing a high water-consumption activity in areas with severe hydric stress.
The assessment of a water footprint must be made locally and each component should be evaluated individually avoiding its aggregation as a final metric of water footprint. The intention of comparing water footprints is to make any necessary changes in order to diminish its effects. Hence, it is important to establish a standardized methodology that makes possible a comprehensive evaluation, comparison and the resulting decrease of a water footprint.

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References

[1] Rees WE (1992) Ecological footprints and appropriated carrying capacity: what urban economics leaves out. Environ Urban 4(2):121-130.

[2] Hoekstra AY, Chapagain AK, Aldaya MM, Mekonnen MM (2009) Water footprint manual: State of the art 2009. Water Footprint Network, Enschede, <http://www.waterfootprint.org> (accessed 21.02.2010)

[3] Ndeleble (2002) Get water neutral! Brochure distributed among delegates at the 2002 World Summit on Sustainable Development in Johannesburg. The Water Neutral Foundation, Johannesburg

[4] Hoekstra AY (2008) Towards water neutrality: reducing and offsetting the impacts of water footprints. Value of water research report series; no. 28. UNESCO-IHE Institute for Water Education, Delft http://www.waterfootprint.org.

[5] Environmental Management ISO/WD2 14046 Water footprint—Requirements and guidelines prepared by Technical Committee ISO/TC 207ISO/TC 207/SC 5 N 010/WG8. Date: 2010-10-18.

[6] Mila i Canals L, Chenoweth J, Chapagain A, Orr S, Anton A, Clift R (2008) Assessing freshwater use in LCA: Part I—inventory modelling and characterisation factors for the main impact pathways. Int J Life Cycle Assess 14:28–42.

[7] Motoshita M, Itsubo N, Inaba A (2009) Health Damage Assessment Modeling on Agricultural Water Scarcity based on Regression Analysis of Statistical Data. Presented at Life Cycle Assessment IX, 2009, Boston

[8] Motoshita M, Itsubo N, Inaba A, Aoustin E (2009) Development of damage assessment model for infectious diseases arising from domestic water consumption. In: Proceedings of the SETAC Europe: 19th Annual Meeting, 31 May–4 June 2009, Gothenburgborg

[9] Pfister S, Koehler A, Hellweg S (2009) Assessing the environmental impacts of freshwater consumption in LCA. Environ Sci Technol 43:4098-4104.

[10] Margni M, Arena P, Aoustin E, Bayart JB, Berger M, Boulay AM, Bulle C, Deschênes L, Humbert S, Juraske R, Monserrat N, Pfister S, De Schryver A, van Zelm R, Vince F, Koehler A (2009) A framework for assessing freshwater use within LCA: Results and working progress from the UNEP/SETAC Life Cycle Initiative Project on Water Use in life LCA.

[11] Environmental Management—Life Cycle Assessment—Principles and Framework (ISO 14044:2006); European Committee for Standardisation: Brussels
[12] CNA. Estadísticas Agrícolas de los Distritos de Riego. Comisión Nacional del Agua, Edición 2004 al 2009. www.conagua.gob.mx.
[13] Allen RG, Pereira LS, Raes D, Smith M (2006) Evapotranspiración del cultivo Guías para la determinación de los requerimientos de agua de los cultivos. Organización de las Naciones Unidas para la Agricultura y la Alimentación. Estudio FAO Riego Y Drenaje #56
[14] Comisión Nacional del Agua. Estaciones Meteorológicas Automáticas (EMAs). <http://smn.cna.gob.mx/emas/> (accessed 10.03.2010)
[15] Hauschild M, Wenzel H (1998) Environmental Assessment of Products. Vol 2. Chapman & Hall, Thomson Science: New York, NY
[16] Comisión Nacional del Agua. Estadísticas del Agua en México 2010. (2010) Secretaría del Medio Ambiente y Recursos Naturales www.conagua.gob.mx.
[17] Mekonnen MM, Hoekstra AY (2010) A global and high-resolution assessment of the green, blue and grey water footprint of wheat. Hydrol Earth Syst Sci 14(7):1259–1276.
[18] Chapagain AK, Hoekstra AY (2004) Water footprints of nations. Volume 1: Main Report, Volume 2: Appendices'. Water Research Report Series' No. 16 of UNESCO-IHE

Water Footprints in Four Selected Breweries in Nigeria

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Abstract The water footprint (water source, use, and cleaner production (CP) efforts) in four major breweries in different hydrological settings in Nigeria were studied. Areas of wastage of raw materials in each brewery were determined using the ABREW software. All the breweries studied were in excess water usage category of their best practice rating by 2.9 -5.08 hl/hl of product. The average cost of operating losses from non-observance of CP practice from energy and material wastage was between $0.81-$2.42 per hl of beverage produced. The annual wastewater runoff averaged about 35-61 million hectolitres. The four breweries are still far from the accepted best practice benchmark level of 6.5 hl/hl, let alone the best technology level of 4 hl/hl. The BOD and COD loadings in the waste stream were below the maximum contaminant level except in one of the multinational breweries with a potential BOD reduction of 0.99kg/hl of product.

1 Introduction

There is very high water stress in Nigeria, especially in the urban areas where the water poverty index (WPI) which in a time analysis approach, is the ratio of time spent (in minutes) in collecting a given volume of water (in litres) for domestic use in a day. WPI could be as high as 5.0 when the value should be well below zero [1,2]. Although most industries in Nigeria depend on ground water source for their production, some still draw from priority line of public water source for most of the domestic uses in the industries [3]. Africa is endowed with abundant water resources although its distribution and availability for use varies widely, with quite a number of countries facing water shortage and water stress. Regional and national water figures often conceal the dramatic effects of local water scarcity, limited or polluted supplies and inadequate distribution systems. Access to fresh

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water has been identified repeatedly as a key condition for development. National water policies and conservation efforts often tend to focus on the supply-side for domestic and agricultural use, and less commonly on industrial needs. Under these circumstances the uncontrolled use of a limited resource by water intensive industries takes on a special significance [4].

Brewing involves five major processes which are malting, wort production, fermentation and maturation, filtration and bottling of the beer [5]. High consumption of good-quality water is characteristic of beer brewing. More than 90% of beer is water and an efficient brewery will typically use between 4 and 6 litres of water to produce one litre of beer. Older technologies that are inefficiently operated especially small breweries, can easily double or triple this consumption, to the detriment of neighbouring communities and at additional cost to the company itself. In addition to water used for beer production (mashing, boiling, filtration, and packaging) breweries also use water for heating and cooling as well as cleaning and sanitation of equipment and process areas. Each uses requires a somewhat different water quality too [6]. High water consumption also means higher energy use, as much of the excess water has to be heated in the brewing and cleaning processes [7].

Untreated effluents typically contain suspended solids in the range 10–60 milligrams per litre (mg/l), biochemical oxygen demand (BOD) in the range 1,000–1,500 mg/l, chemical oxygen demand (COD) in the range 1,800–3,000 mg/l, and nitrogen in the range 30–100 mg/l. Phosphorus can also be present at concentrations of the order of 10–30 mg/l [8]. Effluents from individual process steps are variable. For example, bottle washing produces a large volume of effluent that, however, contains only a minor part of the total organics discharged from the brewery. Effluents from fermentation and filtering are high in organics and BOD but low in volume, accounting for about 3% of total wastewater volume but 97% of BOD. Effluent pH averages about 7 for the combined effluent but can fluctuate from 3 to 12 depending on the use of acid and alkaline cleaning agents. Effluent temperatures average about 30°C.

The filtration residue from the beer producing process is composted. Bottles and part of the waste glass, plastic packages and used cardboard are recycled. It was assumed that all used bottle caps; labels {removed from recycled bottles on a line before reusing the bottles), glues, plastic strips and stretch film end up in the landfill as solid waste [9]. The largest part of the dumped solid waste from the production unit is the waste glass.

The water footprint is an indicator of water use that includes both direct and indirect water use of a consumer or producer [10]. The water footprint of an individual, community or business is defined as the total volume of freshwater that is used to produce the goods and services consumed by the individual or community or produced by the business. Water use is measured in water volume consumed (evaporated) and/or polluted per unit of time. A water footprint can be calculated for any well-defined group of consumers (e.g. an individual, family, village, city, province, state or nation) or producers (e.g. a public organisation, private enterprise or economic sector). The water footprint is a geographically explicit indicator, not only showing volumes of water use and pollution, but also the locations. However, the water footprint does not provide information on how the embedded water is contributing to water stress or environmental impacts.

A water footprint consists of three components: the blue, green and grey water footprint. The blue water footprint is the volume of freshwater that evaporated from the global blue water resources (surface water and ground water) to produce the goods and services consumed by the individual or community. The green water footprint is the volume of water evaporated from the global green water resources (rainwater stored in the soil as soil moisture). The grey water footprint is the volume of polluted water that associates with the production of all goods and services for the individual or community. The latter can be estimated as the volume of water that is required to dilute pollutants to such an extent that the quality of the water remains at or above agreed water quality standards.

Life cycle assessment (LCA) is increasingly becoming an important tool for ecological evaluation of products or processes. Food production and consumption systems consume both energy and natural resources and an LCA analysis of any brewery should therefore include both energy and natural resource needs and effects on the environment [11]. For establishing a sustainable environment management system (EMS) at a brewery, the United Nations Environment Program has produced useful guidelines that are readily available.

Cleaner production is an approach to improving industrial process efficiency. Adoption of cleaner production principles reduces waste and this in turn results in lower environmental impact and occupational risks. Cleaner production is the continuous application of an integrated preventive environmental strategy to processes and products to reduce risks to human health and the environment [4]. For production processes, cleaner production includes conserving raw materials and energy, eliminating toxic raw materials, and reducing the quantity and toxicity of all emissions and wastes before they leave a process. For products, the strategy focuses on reducing impacts throughout the life cycle of the product, from raw material extraction to ultimate disposal [4]. The purpose of cleaner production is to reduce resource consumption and emissions from a production preferably by reduction of waste at source.

The long experience of national cleaner production centres (NCPCs) and the Pan-African network of cleaner production experts ensure that realistic and effective remedial measures can be devised in most cases. Cleaner production also reduces the cost of wastewater disposal by reducing the volume and strength of effluents that need to be treated. Cleaner production is thus a useful preliminary stage to designing treatment works [4,12].

The *African **BRE**wery Sector **W**ater Saving Initiative (*ABREW) study funded by UNEP looked at breweries in four African countries, namely Ethiopia, Ghana, Morocco and Uganda, in an attempt to isolate the technical, management and policy elements that could lead to greater efficiency in water use. Two breweries in Uganda were studied in more depth to identify specific technological factors that require improvement, on the assumption that these factors might be similar across the continent. The study served as a baseline from which the current study in Nigeria was based. Solid wastes and related emissions were not covered in the present study.

2 Materials and method

Four breweries were purposively selected for assessment of their water use and wastewater management based on received consent for access in reply to requests sent to several Plants. Three of these breweries are owned by the Nigerian Breweries Plc, a conglomerate that started operation in Lagos Nigeria since 1949 and is now jointly owned by Heinekens BV of Holland and Nigerian shareholders. The Ibadan Plant, Oyo State was established in 1982 while the Ama Greenfield plant in Umuezeani Village, Enugu State was completed in 2003, with the first production run finishing on April 24. Ama brewery is the biggest brewery in Nigeria, and boasts state of the art manufacturing equipment. It is designed to have a production capacity of three million hectolitres, six times larger than that of the Enugu brewery (not covered in this study). The International Breweries Plc, Ilesa in Osun State started as a privately owned brewery before it went public recently.

The focus of the study is on evaluation of water use and wastewater management systems in the industries and a cost/benefit analysis of the best practice production method used in each brewery in terms of possible savings when ABREW software is used in estimating areas and quantity of wastes. Primary data of water source(s), water use and raw materials used in production and quantity of electricity used in production were sourced through structured questionnaires to those involved with the production and purchases. This was followed by evaluation of facilities for wastewater treatment and disposal and laboratory analysis of both influent and effluent wastewaters from each brewery using standard methods [13]. The collected data were analysed using the ABREW software templates.

3 Results and Discussion

The beer industry is presently about the only most viable non-oil product in Nigeria with typical lull in sales from June to August each year. All the breweries studied were in excess water usage category of their best practice rating by 2.9-5.08 hl/hl of product even though they may be categorised as medium consumer of water (Tables 1-2).

Tab. 1: Water footprint in Lagos and Ilesa breweries

Water balance	**Lagos NB Plc**			**International Breweries Plc**		
Water use (hl/hl of product)	Actual	Optimum	Potential savings	Actual	Optimum	Potential savings
Utilities	0.67	0.25	0.42	0.70	0.52	0.18
Brewhouse	2.85	1.25	1.60	2.86	1.59	1.27
Beer processing	0.73	1.45	0.00	0.84	1.32	0.00
Packaging	3.82	2.00	1.82	3.81	2.25	1.56
Warehouse	0.45	0.05	0.40	0.54	0.05	0.49
Total water intake	8.52	5.00	**4.24**	8.75	5.73	**3.02**
Total waste-water (hl/hl)	7.45	3.5	3.95	8.05	3.85	4.20
BOD load (kg/hl)	0.77	0.80	0.00	0.77	0.90	0.00
COD load (kg/hl)	1.49	1.00	0.49	1.59	1.25	0.34

Tab. 2: Water footprint in Ama and Ibadan Plants of Nigeria Breweries Plc

Water balance	**Ama Plant of NB Plc**			**Ibadan Plant of NB Plc**		
Water use (hl/hl of product)	Actual	Optimum	Potential Savings	Actual	Optimum	Potential Savings
Utilities	0.65	0.25	0.40	0.80	0.30	0.50
Brewhouse	2.35	1.25	1.10	3.42	1.50	1.92
Beer processing	0.91	1.45	0.00	0.88	1.74	0.00
Packaging	3.01	2.00	1.01	4.58	2.40	2.18
Ware-house	0.45	0.05	0.40	0.54	0.06	0.48
Total water intake	7.37	5.00	**2.91**	10.22	6.00	**5.08**
Total wastewater (hl/hl)	-	-	-	8.94	4.20	4.74
BOD load (kg/hl)	0.59	0.88	0.00	1.79	1.20	0.59
COD load (kg/hl)	0.78	1.00	0.00	0.92	0.96	0.00

Although the breweries at Lagos, Ibadan and Umezeani belong to the same multinational, there were observed variations in the capacities, CP practice and effluent discharges based on the variation in the facilities in terms of age and

technology of equipment and management character. Both Ibadan and Lagos plants nearly doubled their optimum water use and must take concerted efforts to reduce wastage in their packaging and brewhouse processes, with possible saving of 5.08 hl/hl of product from the current 10.22hl/hl of product at the Ibadan plant.
The Ama Brewery, with its modern brewing facilities had the least water use. Data of its wastewater flow was not available as at the time of all the visits to the plant.
The variation in the optimum water use in each plant is related to the CP target set by the production engineer or management and water access conditions within the location of the plants. All the four plants have access to both groundwater (boreholes) and raw surface water from the waterworks in each community. The use of treated water from the municipal line is very negligible since the chlorine commonly used in processing the water supply will affect the taste and quality of the product. The wastewater runoff averaged about 35-61 million hectoliters. Treating this volume of wastewater to acceptable standard before discharge into a receiving stream will provide useable water, at least for irrigation by host communities. The BOD and COD loadings in the waste stream were below the maximum range except in one of the multinational breweries with a potential BOD reduction of 0.99kg/hl of product.
Apart from wastage from energy and water, the product losses are significant as shown in Tables 3 and 4. This will be good money down the drain. Tables 5 and 6 show the average cost of operating losses from non-observance of CP practice in terms of energy and material wastage ranged between \$0.82-\$2.42 per hl of beverage produced.

Tab. 3: Beer losses during production in Lagos NB Plc and Ilesa Breweries Plc

	Lagos NB Plc			**Ilesa Breweries Pl**		
Beer loss (hl/hl of product)	Actual	Optimum	Potential Savings	Actual	Optimum	Potential Savings
Utilities						
Brewhouse	0.018	0.010	0.085	0.0192	0.0120	0.0072
Beer processsing	0.056	0.025	0.031	0.0675	0.0260	0.0415
Packaging	0.021	0.015	0.006	0.0211	0.0150	0.0061
Warehouse	0.000	0.0005	0.000	0.0000	0.0006	0.0000
Total beer loss	0.096	0.051	**0.046**	0.1078	0.0536	**0.0550**

Tab. 4: Beer losses during production in Ama and Ibadan NB Breweries Plc

	Ama NB Plc			**Ibadan NB Plc**		
Beer loss (hl/hl of product)	Actual	Optimum	Potential Savings	Actual	Optimum	Potential Savings
Utilities						
Brewhouse	0.013	0.012	0.001	0.022	0.012	0.010
Beer processsing	0.04	0.027	0.013	0.067	0.030	0.037
Packaging	0.032	0.017	0.015	0.025	0.018	0.007
Warehouse	0.000	0.0004	0.000	0.000	0.0006	0.0000
Total beer loss	0.085	0.0564	**0.029**	0.114	0.0606	**0.054**

Tab. 5: Potential economic savings at Lagos and Ilesa breweries

	Lagos NB Plc			**International Breweries Plc**		
Economic savings (per hl of product)	Actual	Optimum	Potential Savings	Actual	Optimum	Potential Savings
Malt adjunct (kg)	0.11	0.10	0.01	0.11	0.106	0.004
Energy (MJ)	0.005	1	0.00	0.67	0.60	0.074
Electricity (in kWh)	0.08	0.06	0.02	0.10	0.09	0.016
Spent grain (kg)	0.00	0.00	0.00	0.00	0.00	0.00
Excess yeast (kg)	0.00	0.00	0.00	0.00	0.00	0.00
Total cost of input, ($/hl of product)	12.65	11.83	**0.82**	14.75	12.33	**2.42**

Tab. 6: Potential economic savings at Ama and Ibadan breweries

	Ama NB Plc			**Ibadan NB Plc**		
Economic savings (per hl of product)	Actual	Optimum	Potential Savings	Actual	Optimum	Potential Savings
Malt adjunct (kg)	0.12	0.10	0.02	0.12	0.11	0.01
Energy (MJ)	1.24	1.00	0.24	0.74	0.74	0.00
Electricity (in kWh)	0.10	0.07	0.03	0.10	0.06	0.04
Spent grain (kg)	0.12	0.10	0.02	0.00	0.00	0.00
Excess yeast (kg)	0.33	0.02	0.11	0.00	0.00	0.00
Total cost of input, ($/hl of product)	-	-	**2.57**	13.02	11.17	**1.86**

The government-owned waterworks in Nigeria charges an average of \$0.2 per m^3 of raw water and \$0.6 per m^3 of treated water. The cost of the 35-61 million hl which are equivalent to $3.5\text{-}6.1 \cdot 10^6$ m^3 wasted annually is about \$1,400,000 (₦210 million) to \$2,440,000 (₦366 million) using the median of \$0.4 per m^3 for both water sources. Apart from this loss, the release of pollutants into the receiving stream may lead to litigation. All the facilities have provision for wastewater treatment but there are restricted accesses to these facilities. The data given was based on report from the questionnaire.

According to the sectoral study and framework analysis from that pioneer study, water consumption and specific use (hl water/hl beer) varies greatly between breweries in the studied countries and ranges from 7.2 hl/hl in Uganda to 22 hl/hl in Ethiopia. Most breweries are still far from the accepted international best practice benchmark of 6.5 hl/hl, let alone the best technology level of 4 hl/hl.

Ogbiye [3] noted this in his recent work and recommended the need for environmental monitoring agencies to enforce free access by officials and researchers to industries. Some industries studied fill effluent tanks with unused treated water or will allow entry only on days when there will be no production. Materials recovery apart from the improvement of boiler units in the brewhouses at Ibadan and Lagos depots will result in improvement of the CP practice of the two plants.

Reducing the amount of water used not only reduces supply costs, but also the volume of trade effluent produced. The mean energy consumption in heating water for the boilers, the volume of water use and black oil, etc are beyond the standard for optimum best practice breweries (Table 7). However, reducing the strength of a brewery's trade effluent is equally important. Taking action to minimize the discharge of wastes with a high COD, e.g. residual wort and sullage, will significantly reduce trade effluent charges. Optimizing cleaning procedures, identifying leaks and stopping overflows are other areas where significant savings can be achieved [4, 9]. Water conservation and recycling will allow water consumption to be kept to a minimum. A brewery should target on achieving an effluent range of 3–5 m^3/m^3 beer produced with less than 0.3 kilograms (kg) of BOD/m^3 beer produced and 0.3 kg of suspended solids/m^3 beer produced (assuming discharge to receiving waters)[4]. Provision for recycling liquors and reusing wash waters will help reduce the total volume of liquid effluent.

Tab. 7: The average water, electricity and black oil consumption in the breweries

	Electricity	**Water**	**Black oil**
Week	Standard value 13.5 kw/hl	Standard value 0.8 m^3/hl	Standard value 5.0 l/hl
1	16.0	2.74	6.28
2	15.8	3.37	8.27
3	16.3	3.40	6.60
4	16.0	3.38	6.28
5	23.5	5.33	10.3
6	23.7	3.18	9.63
7	19.8	3.71	8.12
8	22.7	4.53	9.54
9	24.5	4.83	6.75
10	25.9	4.69	7.43
11	24.7	6.58	8.98
12	20.2	4.9	9.41
13	20.1	4.65	8.8
14	31.7	6.51	7.3
15	47.0	9.84	15.13
16	21.6	4.45	8.65
17	76.8	15.5	23.7
18	33.2	6.1	13.7
19	16.9	3.44	7.1
20	19.4	4.0	8.3
21	16.5	3.15	5.17
22	19.7	4.1	7.23
23	18.9	3.29	7.81
24	69.3	12.7	29.7
25	19.1	3.25	7.4
26	22.3	4.17	9.1
27	23.4	4.35	8.7
28	21.9	3.58	8.54
29	17.1	2.98	7.1
30	21.2	3.2	9.95
31	20.8	3.4	9.55
32	19.4	3.3	8.7
33	21.8	3.3	9.6
34	22.8	3.9	7.4
35	33.3	5.78	10.4
Mean	**23.9**	**4.84**	**9.03**

Since barley and malt used in brewing in Nigeria are imported, the need for local content in production led to a government policy for scientists to research into and recommend alternatives to barley. Their efforts were complemented by the independent research endeavour at the Federal Institute of Industrial Research, Oshodi (FIIRO), which, through some of its research report series, demonstrated that lager beer could be produced using sorghum exclusively.
Today, most of the more successful firms use maize and sorghum in their beer production process. However, the changeover in input mix has necessitated the use of expensive imported enzymes in the production process. Beer and breweries in Nigeria are among the fastest growing industries in Nigeria. Beer in Nigeria is the most popular of all alcoholic beverages consumed. In Nigeria beer constitutes 96% of all alcoholic drinks sold. Breweries in Nigeria are located throughout the country.
The effluent qualities of the four breweries are within acceptable values, except in a single instance where the COD was in excess of approved limit due to change of raw materials. The Breweries should have a Consultant to ensure compliance with regulations and advice on engineering solutions to production and process problems in line with Nigerian Standards. The use of hydrogen peroxide in an electrochemical process followed by adsorption in a fluidised bed or upflow unit will guaranty acceptable effluent quality and peace of mind on compliance with local regulations.

4 Conclusion

The efficiency levels of Nigerian breweries on water use can at best be described as medium, with rather wide variations in and between geographical locations and breweries. The four breweries studied are still far from the accepted international best practice benchmark level of 6.5 hl/hl, let alone the best technology level of 4 hl/hl. The water audit analysis, using the ABREW software showed that most significant saving is possible within each plant's brew house and packaging units. More wastewater is discharged in the processing unit as indicated by the ABREW software used. Corruption is still impacting on the enforcement of regulatory and monitoring provisions in the laws and statutes of Nigeria or the states. Rather than each plant setting its own optimum quality level, there should be a national standard that will guide all operators in providing a sustainable production for sustainable consumption and management of their business environment.

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References

[1] Sullivance C (2003) Calculating Water Poverty Index. World Dev 30(7):1195-1210
[2] Alatise M (2007) State of Water Delivery and Water Assessment in Nigeria. Hydrol Sci Bul 4(12):581-591
[3] Ogbiye AS (2011) Pollution problems and an engineering approach to the management and control of industrial effluents in Otta, Nigeria. Unpublished PhD Thesis, Covenant University, Otta
[4] United Nations Environment Programme (Industry and Environment) (2006) ABREW: Sustainable Water Utilization in African Breweries: Current Practices and Prospects, UNEP, Nairobi
[5] Koren H (1980) Handbook of Environmental Health and Safety. Pergamon Press, Paris
[6] Fix GJ (1989) Principles of Brewing Science. Boulder, Colorado: Brewers. Chapman-Hall, Ltd. University Press, London
[7] Taylor TG (2006) The Impact of Water Quality on Beer Quality. In: Priest FG, Stewart GG (eds). Boca Raton, Florida: CRC Press, New York
[8] Benient PB, Rifed NS, Navell CJ (1999) Groundwater Contamination and Remediation. Prentice Hall, London
[9] Hammer MJ, Hammer Jnr. MJ (1996) Water and Waste Water Technology. 3rd ed. Prentice Hall, London
[10] Ayoade LO (1975) Footprints of water supply and development in Nigeria. Sci Bull 4(12):81-591
[11] Feachem R, McGarry M, Mara, D (2007) Water, Wastes and Health in Hot Climate. John Wiley and Sons, Chichester
[12] United Nations Environment Programme (Industry and Environment) (1996), Cleaner. UNEP FI website for CP Financing (accessed 10 October 2007) Online:http://www.unepfi.org/
[13] APHA (1998) Standard Methods for the Examination of Water and Wastewater. APHA Inc, Washington D.C. Online: http://www.cwr.uwa.edu.au/~alexandb/solidsprotocol.html
[14] Walsh MP (1992) Transport and Environment: Local, regional and global challenges in automotive technology saving society 1(1):1-17. Available at http://www.ilpi.com

Development and Application of a Water Footprint Metric for Agricultural Products and the Food Industry

Bradley Ridoutt

Abstract The agriculture and food industries, which account for around 70% of global freshwater withdrawals and are an important source of chemical emissions to freshwater, are central to the issue of addressing global water stress. While water use efficiency is a longstanding and familiar concept, especially where water is a growth limiting factor for agricultural production, LCA-based water footprinting, which includes water use impact assessment, has recently emerged as an important parameter in LCM and the debate about sustainable food systems. This paper summarises recent case study evidence, noting that agriculture is not homogeneous and that it is dangerous to make generalisations about the water footprints of broad categories of food products and production regions. Issues of special significance to the water footprint of agriculture and food products, such as unmeasured flows, seasonal variations and rainwater flows, are discussed.

1 Introduction

To understand and address the challenges of global water stress, the agriculture and food industries are critical as globally they account for around 70% of freshwater withdrawals [1] and are a major source of emissions responsible for freshwater quality degradation. The situation is made even more serious by the expanding world population, the consequent need to increase food production, and the additional demands this may place on freshwater resources through increases in irrigation and land-use intensification. This has led to questions about the freshwater use that supports the complex and variable global supply chains that underpin food consumption in developed nations [2]. It has also stimulated debate about the larger question of the shape and form of a sustainable global food system [3,4]. The result is a rapid rise in interest in water footprint metrics by the agriculture and food industries to assist with life cycle management (LCM). This paper summarises recent case study evidence arising from the application of life

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cycle assessment (LCA)-based water footprinting in the agriculture and food industries, highlighting lessons learned and development needs.

2 Water use efficiency and water footprint

Water use efficiency (WUE) is a familiar concept in the agriculture and food industries. In countries such as Australia, where much farming is practiced in situations of low and variable rainfall and where water is a growth limiting factor, WUE is a longstanding goal, closely linked to gains in agricultural productivity. In Australia, increasing water costs and constrained water allocations have also meant that irrigation WUE (yield per unit irrigation water use) has become a business imperative in irrigated farming districts. In food product manufacturing, WUE programs are also common, to meet local regulatory requirements and as part of ongoing business efficiency improvement. For larger companies, water use and water use efficiency are now routinely reported to stakeholders, for example in annual sustainability reports.

What is new is the assessment of water use from a product life cycle perspective including impact assessment. This is an important development in LCM because WUE alone will not necessarily lead to sustainable freshwater use. The current situation is that humanity's consumption of freshwater is skewed considerably toward highly stressed watersheds (Figure 1). While WUE is desirable in a narrow sense, and is useful for benchmarking, the greater challenge is to reduce pressure on those highly stressed watersheds where environmental harm and resource depletion are already manifest. LCA-based water footprint metrics, which use regionalised characterisation factors to express the environmental relevance of water-use in a product life cycle, are therefore necessary. Case study evidence has repeatedly shown that water use inventory results are not necessarily correlated with water use impact category results (Figure 2). Other than the special situation of comparing locally-adjacent systems using the same water resource, water use is not a reliable measure of potential environmental harm. Obviously, water use in a region of water abundance does not have the same potential for environmental harm as water use in a region of scarcity. LCA-based water footprint metrics are also required to assess trade-offs between water use impacts and other environmental impacts, for example climate impacts [5].

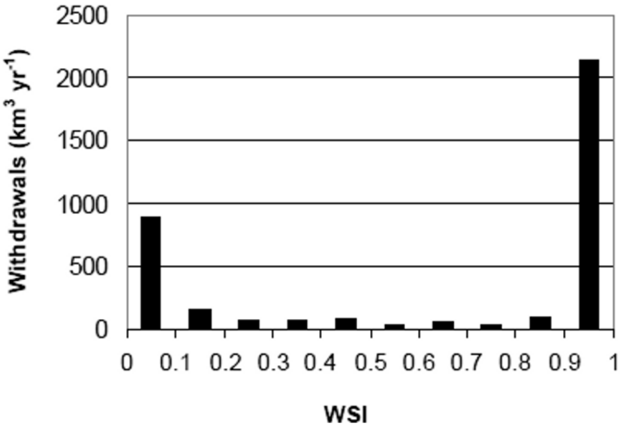

Fig. 1: Distribution of global freshwater withdrawals by local water stress index (WSI) [6]

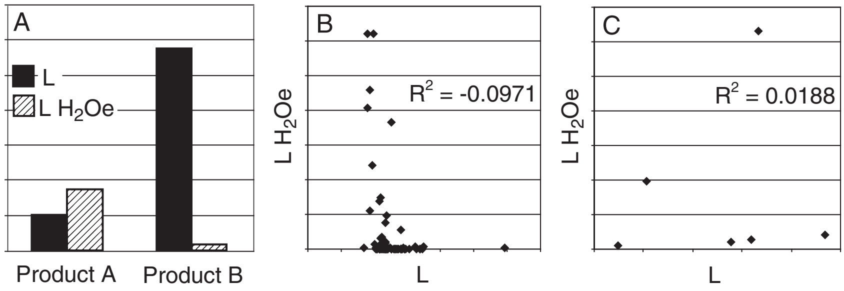

Fig. 2: Comparison of water use inventory (L) and water use impact (L H_2O-e) results for A) Two manufactured food products [7], B) Wheat grown in the Australian state of New South Wales, n = 73 [8] and C) Beef cattle production in New South Wales, n = 6 [9]

3 Case study evidence

LCA-based water footprints have now been calculated for a wide range of agricultural and food products using the local characterisation factors for freshwater consumption taken from the water stress index (WSI) of Pfister et al. [10]. This includes cereal, horticulture and dairy products as well as lamb cuts, beef cattle and manufactured (multi-ingredient) food and beverage products [e.g. 7-9,11-14]. The clear lesson is that it is dangerous to make generalisations about the water footprint of broad categories of food products and production regions. For example, various authors have raised attention to the water use associated with

livestock production, directly linking meat consumption and water scarcity [3,15,16]. However, it must be highlighted that agriculture is not homogeneous and sub-sectors vary widely in both farm practice and geography. Many low input, non-irrigated, pasture and rangeland-based livestock production systems make little impact on freshwater systems from consumptive water use. In the Australian state of New South Wales, the water footprint of beef cattle (per kg at farm gate) was found to have a range similar to that of wheat, barley and oats [9]. These results point to a need to apply LCA-based water footprinting to specific industry sub-sectors and value chains to understand where improvement efforts would be most effective in relieving pressure on freshwater systems.
Regardless of product, the highest water footprints are found where irrigated production occurs in high water stress environments. However, this should not be interpreted as an overall bias against irrigated forms of agriculture. Irrigated production systems contribute substantially to global food production. Wise use of supplementary irrigation can raise overall WUE and reduce the risk of debilitating crop failures. Irrigated agriculture can also be a very land use efficient form of production and offer other benefits. It is therefore important to note that a water footprint is not an indicator of overall environmental sustainability and that major strategic change should be undertaken only after considering the full range of relevant environmental impacts as well as other social and economic factors.

4 Issues of special significance to agriculture and food products

4.1 Unmeasured flows

A life cycle inventory of freshwater aims to quantify changes in freshwater availability through consumptive and degradative water use [17]. In the case of agricultural unit processes, which often take place over large land areas, three kinds of elementary flows are relevant. Firstly, there are elementary flows from surface and groundwater resources across the system boundary into the farming system. The most important flow of this kind is irrigation water, which is sometimes, but not always, measured. In the absence of direct measurement, there is usually local knowledge about typical irrigation water use. Secondly, land-based production systems may change the availability of freshwater by reducing the flows from precipitation which would naturally cross the system boundary to surface and groundwater resources. Here, the system boundary is defined horizontally as the production system boundary and vertically as the soil depth accessible to crop or pasture roots. An example would be the construction and operation of farm dams which intercept local runoff. Another example would be the conversion of pasture to industrial forestry which may reduce runoff and deep drainage. In some parts of Australia, to protect catchment water flows, there are restrictions on the construction of farm dams and the establishment of new forestry plantations. Thirdly, there are diffuse emissions to freshwater of fertilisers and other agricultural chemicals used in farm operations. The issue for life cycle inventory is that for the many non-irrigated farming systems it is the latter two which are critical in determining the inventory result. However, these flows are rarely measured at the farm level and as a result farming system models (such as APSIM [18]) become important. Again, it is emphasised that agriculture is not homogeneous, as shown in Table 1 where the drinking water requirements for a one year-old steer vary from 7.7 to 40.7 kg per day between two contrasting locations and months.

Tab. 1: Water balance (kg per day) for a 1 year-old steer in Bathurst (33°25' S, 149°34' E) in July (winter) compared to Walgett (30°1' S, 148°7' E) in January (summer) (for modelling details see [9])

	Bathurst (July)	**Walgett (Jan)**
Total water input	30.0	54.7
Water in feed	*18.8*	*10.9*
Metabolic water	*3.4*	*3.2*
Free water drunk	*7.7*	*40.7*
Total water output	30.0	54.7
Water in faeces	*7.8*	*9.4*
Water in urine	*18.2*	*16.3*
Water in weight gain	*0.1*	*0.1*
Evaporative loss	*3.9*	*28.9*

4.2 Seasonal variations

Local characterisation factors for freshwater consumption have been published, based on long-term withdrawal-to-availability ratios for over 10,000 watersheds globally, and taking into account monthly and annual variation in precipitation and level of storage capacity [10]. Other sets of regionalised characterisation factors are under development based on consumption rather than withdrawals and which take into account surface and groundwater, water quality, and local capacity to adapt to water stress. Characterisation factors which take into account the seasonal timing of water use are also desirable since impacts related to water use during months of low flow may differ from months of high flow. This factor is especially relevant to agricultural products as many crops are grown annually during a particular season. In addition, in countries where governance of water resources is well developed, water allocations for irrigation may vary depending upon the level of inflows to the watershed. When watershed inflows and storage levels are high, farmers may receive their full water allocations. However, when inflows and storage levels are lower, farmers may receive only a fraction of their water allocation, or no allocation at all. This is the situation for irrigated rice and cotton farmers in Australia and is the main factor determining the area of rice and cotton cultivated annually (Table 2). What this means is that irrigation occurs when the local water stress is lower than the long-term average. For agricultural sectors where irrigation water allocations vary, use of characterisation factors based on the long-term average water stress could lead to significant overstatement of potential damages from water use.

Tab. 2: Variation in area of rice and cotton under irrigation in Australia ('000 ha) [19]

Year	Rice	Cotton
2008/09	7	142
2007/08	2	58
2006/07	20	134
2005/06	102	270
2004/05	51	270
2003/04	65	185
2002/03	44	234

4.3 Rainwater flows

Natural rainfall over agricultural lands, stored in the local soil profile and accessible to plant roots (so-called green water), is by far the most important water resource for agriculture globally. Estimates of water use for food production range from 16,950 to 18,600 km^3 per year, of which around 90% is green water consumption by rainfed crops and pastures and only about 10% supplementary irrigation from surface and groundwater [20,21]. As such, the efficient use of green water, by minimising unproductive evaporative losses, is a critical part of any strategy to increase global food production. However, the use of green water in agriculture does not necessarily lead to a reduction in available surface and groundwater. This is because much of the precipitation that falls on the land surface is returned to the atmosphere through evaporation and plant transpiration processes regardless of whether agricultural production is occurring. Indeed, most agricultural systems consume a smaller proportion of the precipitation than the complex natural ecosystems they once replaced [22] or the potential natural vegetation that would regenerate if agricultural production were to cease. Therefore, the relevant issue in life cycle inventory modelling of freshwater is the change in surface and groundwater production from the land base arising from human activities linked to production.

In the absence of detailed local hydrological assessment, which would probably be rare in LCM, the generalised equations of Zhang et al. [23], relating evapotranspiration (ET, mm/year) to precipitation (P, mm/year) for grassed (Equation 1) and forested catchments can be used to estimate the flows to surface and groundwater. For the example of a 211 ha grazing property in Bathurst, Australia (P = 635 mm/year), the flows to catchment water resources are estimated as 24% of precipitation, or 317 ML/year from the farm. In the situation where precipitation is intercepted into farm dams and used for livestock drinking water (Figure 3), the flows to catchment water resources are reduced, albeit only marginally. In the example shown in Figure 3 the difference is 0.689 ML/year which would be attributed to the livestock production system.

$$ET = \left(\frac{1 + 0.5\dfrac{1100}{P}}{1 + 0.5\dfrac{1100}{P} + \dfrac{P}{1100}} \right) P \tag{1}$$

Rainfall is most critical in the context of land use as it is one of the primary factors determining the productive capacity of land, and productive land is itself a scarce resource. Agriculture and food production systems have substantial land resource requirements which need to be assessed in any full consideration of sustainability. Recently there has been much interest in the land use impact category in LCA, stimulated in part by unintended environmental consequences arising from land use change associated with some forms of biofuel production. There has also been a project group working under the auspices of the UNEP/SETAC Life Cycle Initiative to develop operational characterisation factors for land use impacts on biodiversity and ecosystem services [24]. The climate impacts of land use arising from CO_2 transfers between vegetation, soil and the atmosphere have recently been described [25]. In the context of global food production, an additional land-use related impact pathway for human health must be considered: lack of productive land for agriculture leading to malnutrition (Figure 4). The nexus between water and land is indeed critical to agriculture and the food industry.

However, it is equally critical that the two are not confounded, which would happen if green water (accessible only through land occupation) is included in an inventory of water use designed to assess the change in water available in surface and groundwater resources.

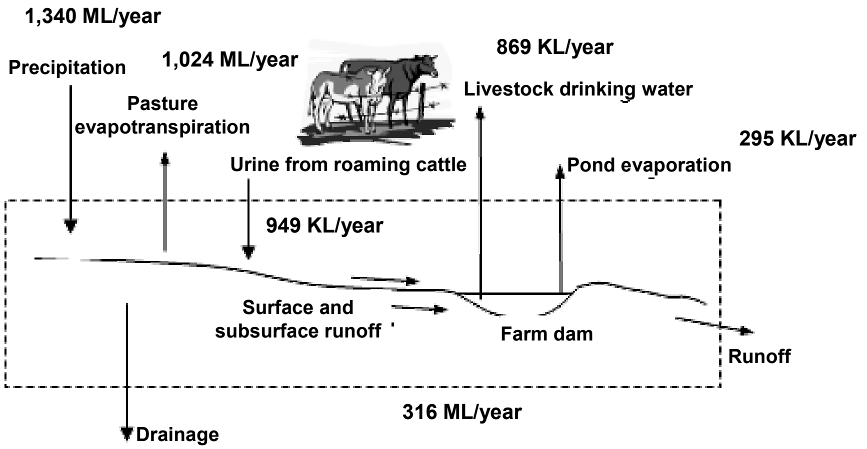

Fig. 3: Example of farm water balance model

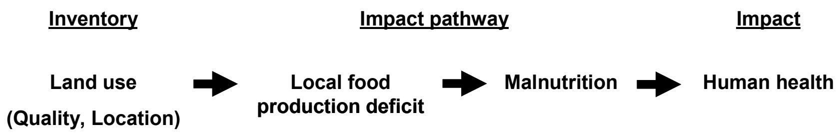

Fig. 4: Proposed impact pathway relating land use to human health

5 Conclusion

In order to transition toward sustainable freshwater use, deep cuts in product water footprints are required, somewhere in the order of 40-50% [6]. To achieve this goal, the agriculture and food industries are critical because this is where the majority of global freshwater use occurs. Water use efficiency programs are to be encouraged and will make an important contribution. However, more is required than to make existing patterns of food production and consumption more water use efficient. LCA-based water footprinting, which includes water use impact assessment, has the potential to guide structural change that reduces the food system's overall burden on freshwater resources.

References

[1] UNESCO-WWAP (2009) The United Nations world water development report 3: Water in a changing world. The United Nations Educational, Scientific and Cultural Organization, Paris

[2] Zygmunt J (2007) Hidden waters. Waterwise, London

[3] Marlow HJ, Hayes WK, Soret S, Carter RL, Schwab ER, Sabaté J (2009) Diet and environment: Does what you eat matter? Am J of Clin Nutr. 89(5):1699S-1703S.

[4] Nellemann C, MacDevette M, Manders T, Eickhout B, Svihus B, Prins AG, Kaltenborn BP (eds) (2009) The environmental food crisis: The environment's role in averting future food crises. UNEP/GRIP-Arendal

[5] Finkbeiner M (2009) Carbon footprinting – opportunities and threats. Int J Life Cycle Assess 14(2):91-94.

[6] Ridoutt BG, Pfister S, Reducing humanity's water footprint. Environ Sci Technol 44(16):6019-6021

[7] Ridoutt BG, Pfister S (2010) A revised approach to water footprinting to make transparent the impacts of production and consumption on global freshwater scarcity. Global Environ Change 20(1):113-120.

[8] Ridoutt BG, Poulton P (2010) Dryland and irrigated cropping systems: comparing the impacts of consumptive water use. 7th International Conference on Life Cycle Assessment in the Agri-Food Sector, Bari, Vol 1:153-158.

[9] Ridoutt BG, Sanguansri P, Freer M, Harper G Water footprint of livestock: comparison of six geographically-defined beef production systems. Int J Life Cycle Assess (submitted).

[10] Pfister S, Koehler A, Hellweg S (2009) Assessing the environmental impacts of freshwater consumption in LCA Environ Sci Technol 43(11):4098-4104.

[11] Ridoutt BG, Juliano P, Sanguansri P, Sellahewa J (2010) The water footprint of food waste: case study of fresh mango in Australia. J Cleaner Prod 18(16-17):1714-1721.

[12] Page G, Bellotti W, Ridoutt BG (2010) Using life cycle assessment (LCA) to assess water use in tomato production, 15th Australian Society of Agronomy Conference, Lincoln, New Zealand <http://www.regional.org.au/au/asa/2010/farming-systems/energy-balance/>

[13] Ridoutt BG, Williams SRO, Baud S, Fraval S, Marks N, The water footprint of dairy products: case study involving skim milk powder. J Dairy Sci 93(11):5114-5117.

[14] Ridoutt BG, Sanguansri P, Nolan M, Marks N Meat consumption and water scarcity: beware of generalisations. J Cleaner Prod (submitted)

[15] Pearce F (1997) Thirsty meals that suck the world dry. New Sci 153(2067):7.

[16] Liu JG, Yang H, Savenije HHG (2008) China's move to higher-meat diet hits water security. Nat 454(7203):397-397.

[17] Bayart JB, Bulle C, Deschênes L, Margni M, Pfister S, Vince F, Koehler A (2010) A framework for assessing off-stream freshwater use in LCA. Int J Life Cycle Assess 15(5):439-453.

[18] Keating BA, Carberry PS, Hammer GL, Probert ME, Robertson MJ, Holzworth D, Huth NI, Hargreaves JNG, Meinke H, Hochman Z, McLean G, Verburg K, Snow V, Dimes JP, Silburn M, Wang E, Brown S, Bristow KL, Asseng S, Chapman S, McCown RL, Freebairn DM, Smith CJ (2003) An overview of APSIM, a model designed for farming systems simulation. Eur J Agron 18(3-4):267-288.

[19] 4618: Water use on Australian farms<http://www.abs.gov.au> (accessed 24.03.2011)

[20] Rockström J, Lannerstad M, Falkenmark M (2007) Assessing the water challenge of a new green revolution in developing countries. Proceedings of the National Academy of Sciences USA, Vol. 104(15):6253-6260

[21] Rockström J, Falkenmark M, Karlberg L, Hoff H, Rost S, Gerten D (2009) Future water availability for global food production: the potential of green water for increasing resiliance to global change. Water Resour Res 45 W00A12

[22] Rost S, Gerten D, Heyder U (2008) Human alterations of the terrestrial water cycle through land management, Adv Geosci 18:43-50

[23] Zhang L, Dawes WR, Walker GR (2001) Response of mean annual evapotranspiration to vegetation changes at catchment scale. Water Resours Res 37(3):701-708.

[24] <http://www.estis.net/sites/lcinit/default.asp?site=lcinit&page_id=337831BE-0C0A-4DC9-AEE5-9DECD1F082D8>, (accessed 24.03.2011)

[25] Müller-Wenk R, Brandão M (2010) Climate impact of land use in LCA: carbon transfers between vegetation/soil and air. Int J Life Cycle Assess 15(2):172-182.

LCA Characterisation of Freshwater Use on Human Health and Through Compensation

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Abstract Impacts from water unavailability are not yet fully quantified in LCA. Water displacement from the original water body (consumption) or quality degradation of released water reduces water availability to human users. This can potentially affect human health through diseases or malnutrition or, if financial resources are available, adaptation can occur, which may generate indirect environmental impacts through the use of backup technologies such as water treatment, desalination, import of water or agricultural goods, etc. This paper proposes an inventory and impact assessment model to evaluate these potential impacts in an LCA context. Results are presented in DALY for impacts on human health and/or as a quantified water inventory to be compensated by users adapting to a situation in which water is scarce or unavailable. A fictional example on board production illustrates the full applicability of the methodology.

1 Introduction

Vital to life, water is a unique natural resource. While it cannot disappear, it can be made unavailable to specific users either by displacement or quality degradation. While potential environmental impacts from pollutant emissions into water are characterized in LCA, impacts from water unavailability are not yet fully quantified. This change in availability can lead to environmental impacts. Based on a review of existing methods to characterise water use impacts in LCA, Bayart et al. [1] suggested a general framework that considers three main impact pathways leading to water deficits for human uses, ecosystems and future generations (freshwater depletion). This paper focuses solely on human uses and proposes a method that assesses the consequences of decreased water availability for human needs, which can lead to impacts on human health. If there is sufficient economic wealth in the area, users will adapt to the lack of water by compensating with a backup technology (e.g. desalination, import of water or goods that can no

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longer be produced locally). The impacts of these compensation processes can be assessed through a traditional LCA and included in the results of the product system for which water use is being studied. This paper presents a method from inventory to midpoint and to endpoint level for the characterisation of impacts from water uses on human users. This approach is based on the loss of functionality, either quantitatively or qualitatively, of the water resource resulting from a water usage.

2 Methodology

2.1 Inventory modelling by water categories

In order to assess functionality loss, the quality of the water entering and exiting the process (or product system) should be assessed along with its associated functionalities. Different water categories based on functionalities are developed for this purpose, each one representing an elementary flow, as proposed in Boulay et al. [2]. Each water category is defined by the source (i.e. its origin, being surface, ground or rain water) and its quality, based on a combination of parameter thresholds taken from national and international water quality standards. Thresholds for an extensive number of parameters (138) are proposed, including general parameters (suspended solids, fecal coliforms, pH, etc.) organics and inorganics. However, this doesn't imply that all this information needs to be collected at the inventory phase; it only ensures that maximal guidance is provided when the information is available. Seventeen (17) water categories are defined in Boulay et al. as shown in Table 1 below. The category of the water entering the process is defined by its origin (surface water or groundwater) and quality. The latter is by default assumed to correspond to the average quality of surface and groundwater available in the relevant watershed. Water category data are provided by Boulay et al. [3] for most watersheds worldwide. The category of the water exiting the process is determined by its quality, which can be determined by combining the information already available in existing LCI databases (emissions to water) and the volume of discharged water (missing information in LCI databases to be collected). The categorised water influent and effluent allow the assessment of impacts from water degradation or consumption as further described below.

Tab. 1: Water category sample (adapted from Boulay and colleagues [2])

Water category	**1**	**2a**	**2b**	**2c**	**2d**	**3**	**4**	**5**
Source	Surface or ground							
Quality descrip-tion	Low mi-crobial low tox	Low mi-crobial medium tox	Medium microbial medium tox	Low mi-crobial high tox	High mi-crobial low tox	High mi-crobial medium tox	High mi-crobial high tox	Other
Param. 1	Thres-hold 1	Thres-hold 2a	Threshold 2b	Thres-hold 2c	Thres-hold 2d	Thres-hold 3	Threshold 4	Thres-hold 5
Param. 2	Thres-hold 1	Thres-hold 2a	Thres-hold 2b	Thres-hold 2c	Thres-hold 2d	Thres-hold 3	Threshold 4	Thres-hold 5
…	…	…	…	…	…	…	…	…
Param. 138	Thres-hold 1	Thres-hold 2a	Thres-hold 2b	Thres-hold 2c	Thres-hold 2d	Thres-hold 3	Threshold 4	Thres-hold 5

2.2 Impact assessment modelling

The model proposed in the sections below includes a midpoint, endpoint and a compensation assessment. The latter is to be used as a complementary assessment in addition to the endpoint model. For the midpoint and endpoint modelling, Figure1 below illustrates how the water categories presented above are used in the impact assessment.

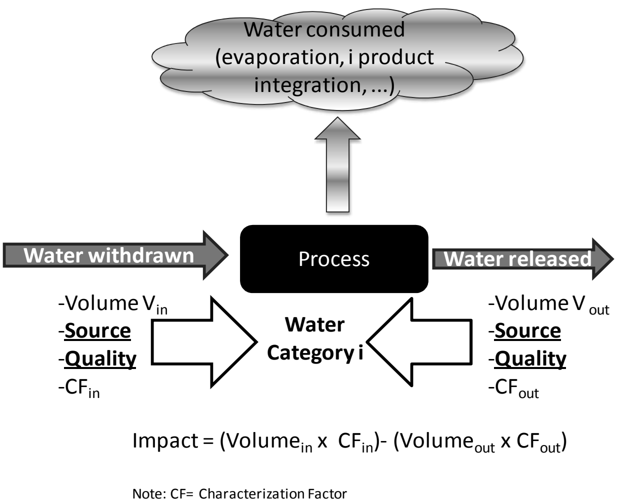

Fig. 1: Impact assessment of water use - for midpoint level (c.f. Equation 1) and endpoint level (c.f. Equation 4)

2.2.1 Midpoint assessment modelling

At the midpoint level, impacts should be characterised considering the local water scarcity, the quality and the type of resource. The impact assessment is then performed by evaluating the difference in stress between the withdrawn and released resource. From Figure 1, it is necessary to know the volume, the source and the quality of the water entering and exiting the system or process, in order to identify the relevant water categories, each associated to a stress index. Once the water categories have been identified by their respective source and quality, the water stress indicator is calculated as per Equation 1:

$$WSI = \sum_i (\alpha_i \times V_{i,in}) - \sum_i (\alpha_i \times V_{i,out}) \tag{1}$$

Where, WSI (water stress indicator) expresses the impact score at the midpoint level - representing the equivalent amount of water (m^3-eq) generating competition between users, α_i the stress index of water category i (in m^3-eq of water per m^3 of water of category i withdrawn/released) and V_i (in and out) the volumes of water category i entering and exiting the process or product system, namely elementary flows (in m^3).

Water stress index (α_i) – In Equation 1, the stress index α_i represents the characterisation factor at the midpoint level, expressing the level of competition among users due to the physical stress of the resource. It addresses quality, seasonal variation and distinguishes between surface and groundwater, as these two types of resources often do not present the same level of scarcity in a region. First, the scarcity parameter α^*_i for surface water is calculated based on the CU/Q90 ratio proposed by Döll [4]. The consumed water (CU [m^3/yr]) in the numerator represents the volume of water consumed by human uses in a region and is calculated using data from the WaterGap model [5]. While no seasonal effects are taken into account for the renewable groundwater resource availability (GWR), they are considered in the denominator for surface water by the Q90 [m^3/yr] parameter. This parameter, called the “statistical low flow”, represents the flow that is exceeded 9 months out of 10. It is therefore a lower value than the average or median flow and allows the exclusion of effect from very high flows, e.g. during monsoon periods, as this water is rarely fully available unless extensive storage facilities are available [6].
The scarcity parameter α^*_i for surface and groundwater is described in Equations 2 and 3 taken from Boulay et al. [7].

$$\alpha *_{surface,i} = \frac{CU \times (1 - f_g)}{Q90} \times P_i \tag{2}$$

$$\alpha *_{GW,i} = \frac{CU \times f_g}{GWR} \times P_i \tag{3}$$

Where CU represents the consumptive use in km³/yr in a given watershed, Q90 the statistical low flow, in km³/yr, fg the fraction of usage dependent on groundwater (obtained from WaterGap), GWR the renewable groundwater resource available in km^3/yr, P_i the inverse of the fraction of available water that is of category i.

The stress index (α_i) is then modelled in order to obtain an indicator ranging from 0 to 1, based on accepted water stress thresholds. There is an agreement in the literature associating different water stress levels (low, moderate, high and very high) with fractions of available water withdrawn (10%, 20%, 40% and 80%, respectively [6,8,9]). However, the water stress index proposed here relies on consumption-to-availability ratios, instead of withdrawal-to-availability ratios, to better capture the physical stress of the resource. Correlations were found to adapt these values for a consumptive-based water stress index [7] and the data was then fitted to an S-curve passing by a 50% scarcity when a high stress threshold is reached as proposed by Pfister and colleagues [9]. Stress indexes for all water categories were calculated and are presented in Boulay et al. [7].

2.2.2 Endpoint assessment modelling

Similarly as for the midpoint level, at the endpoint, the model characterises potential impacts on human health based on the difference between water resource extraction and emission into the environment, as per Equation 4.

$$HH_{impact} = \sum_{i=1}^{17} (CF_i \times V_{i,in}) - \sum_{i=1}^{17} (CF_i \times V_{i,out}) \tag{4}$$

Where, HH_{impact} expresses the human health impacts in DALY, CF_i is the characterisation factor of water category i for the human health impact category (in DALY/m^3 of water category i) and V_i (in and out) is the volume of water category i entering and exiting the process or product system: the elementary flows (in m^3).

Characterisation factors CF_i include three main components that can be compared to the three factors traditionally used to define emission-related impact categories

[10]: 1) fate, 2) exposure and 3) effect. As described in Equation 5, they respectively represent: 1) local water stress, 2) the extent to which user(s) will be affected by a change in water availability, and 3) the human health impacts of a water deficit for user j.

$$CF_i = \sum_{j=1}^{10} (\underbrace{\alpha_i}_{\text{FATE}} \times \underbrace{U_{i,j}(1-AC)}_{\text{EXPOSURE}} \times \underbrace{E_j}_{\text{EFFECT}}) \tag{5}$$

Where α_i expresses the water stress index of category i (dimensionless), $U_{i,j}$ the user(s) j that will be affected by the change in water category i availability (dimensionless), AC the adaptation capacity (dimensionless) and E_j the effect factor for user j (DALY/m^3).

The parameter $U_{i,j}$ is based on the functionality of water i for the specific user, as defined by the categories, and the identification of the marginal off-stream user. This latter represents the one with the lowest willingness to pay, however in the current version of the model, the distribution of withdrawals among off-stream users in a region was used as a proxy for this parameter. For in-stream users, the intensity of the activity used is that estimated by Boulay et al. [7].

The adaptation capacity (AC) defines whether the change in water availability will create deficit or compensation scenarios. The World Bank gross national income (GNI) classification [11] was chosen as the socioeconomic parameter to indicate a country's adaptation capacity (AC). It is proposed that low-income countries (GNI< \$936/cap.yr) will not be able to adapt to a change in water availability and will therefore suffer water deficits, whereas high-income countries (GNI> \$11 455/cap.yr) will have the means to fully compensate for this type of change. Middle-income countries (\$936/cap.yr < GNI <\$11 455/cap.yr) are attributed an adaptation capacity proportional to their incomes, meaning that, in these countries, both compensation and deficit partially occur.

The effect factor E_j assesses the importance of human health impacts caused by a water deficit for domestic, agriculture and aquaculture users. If a water deficit occurs for the remaining users (transport, hydro, industry, cooling and recreation), impacts will only be generated through a compensation process when occurring in countries able to compensate. This is reflected by the E_j zero value for these users. For agriculture and aquaculture, the effect factors (DALY/m^3) were determined by first assessing the damage generated by malnutrition in DALY/kcal and dividing this value by the amount of water needed to produce one kcal, either from agriculture or fisheries. For domestic use, the effect factor (DALY/m^3) relates the human health impacts associated with a lack of hygiene and sanitation when water is scarce to the water deficit for domestic use. It is calculated by dividing the ratio of health burdens from water-related hygiene and sanitation issues by the actual volume of water in deficit for domestic uses (based on a value of 50 l/cap/day to ensure low health concerns and cover most basic needs [12]). The resulting effect factors are 6.53 $\cdot 10^{-5}$, 2.02 $\cdot 10^{-5}$ and 3.11 $\cdot 10^{-3}$ DALY/m³ for agriculture, fisheries and domestic, respectively. A domestic use deficit is therefore critical, since it shows health impacts that are two orders of magnitude greater than those for agriculture or fisheries. The details on how these parameters were obtained are presented in Boulay et al. [7].

2.3 Water compensation volume modelling

Compensation here refers to the use of backup technologies by water-deprived human users to meet their needs. It only occurs in high- and middle-income countries (along with human health impacts). Impacts associated with these alternatives should be modelled with a traditional system expansion, but not all of the water used will be compensated, since compensation also depends on scarcity and adaptation capacity. Equation 6 serves to calculate the amount of water to be compensated by the single user j in m^3, $W_{comp,j}$.

$$W_{comp,j} = \sum_{i=1}^{13}(V_{i,in} \times U_{i,j} \times \alpha_i \times AC) - \sum_{i=1}^{13}(V_{i,out} \times U_{i,j} \times \alpha_i \times AC) \quad (6)$$

Where, all parameters are as described in previous Equations. Parameters $U_{i,j}$ and α_i must be adjusted from the ones presented above, as explained for the example below. This then becomes an inventory input in a system expansion, similarly to the mineral resource depletion assessment through the supplementary energy needed for subsequent abstraction [13,14]. Each compensation scenario is unique and specific to each user for whom water availability is decreased (e.g. water import, desalinisation, etc. for domestic use; food import for agriculture and aquaculture) and have to be specifically modelled by a system expansion, resulting in damages that can then be added to all of the impact categories, including human health impacts.

3 Application

Using Equation 1, some sample assessment of the midpoint indicator for a hypothetical process that withdraws 100 m^3 of water type S2a (low microbial, medium tox) and releases 80m^3 of water S3 (high microbial, medium tox) is shown for several geographical locations to illustrate the variability of potential impacts due to regionalization.

For the endpoint assessment, the complete methodology was applied to a fictitious board producing plant located in the region of Cape Town in South Africa. This region was chosen as it represents a middle-income region with therefore both impacts on human health from water deprivation and on all categories from compensation scenarios. The ecoinvent process "Corrugated board base paper, kraftliner, at plant/RER" was used, along with all water data already included in the process. The volume of water released was estimated based on the hypothesis that one cubic meter of water is evaporated per ton of board produced [7]. The quality of the influent was taken to be the locally available water [2] and was thus identified to be of category 2d (high microbial, low tox). The quality of the released water was evaluated based on the emissions to water and the volume released and resulted in water category 5 (unusable) due to the high BOD content (93 mg O_2/l). Cooling water was treated separately and assumed to be both withdrawn and released at the same quality level.

Compensation was modelled based on the following hypothesis: 1) Agriculture is the off-stream user with the lowest willingness to pay and will therefore be the one affected by 100% of the change in water availability. 2) Compensation in agriculture is assumed to be achieved through wastewater reuse and was modelled with the ecoinvent process "Water, ultrapure, at plant" adapted with the South African electricity mix. 3) Hydropower compensation was modelled with the South African electricity grid mix, as it was difficult to identify whether South Africa is moving towards nuclear, as it states to be, while it is constructing new coal fired plants. 4) The aquaculture compensation would result in about 2.2 kcal to be compensated for from the loss of fish production, and was therefore neglected. 5) Transport and recreation were not modelled.

When modelling the volume of water to be compensated for each user, Equation 6 was used along with the data provided by Boulay et al. [7]. However, the scarcity term was adjusted for hydro as this user is not affected by the quality of the available water, and the general surface water scarcity of the region was used.

4 Results

Results are presented i) at the midpoint level for several regional assessments to show the importance of regionalization and simplicity of the indicator and ii) at the endpoint level including impacts on human health from water deprivation as well as on all impact categories from compensation scenarios.

4.1 Regionalised midpoint assessment

Results for a regionalised midpoint assessment are presented in Table 2. These include the stress indexes α for water flows in and out of a hypothetical process, namely water categories S2a (low microbial, medium tox) and S3 (high microbial, medium tox). The resulting water stress indicator (WSI) in m^3 equivalent of water is calculated as per Equation 1 with inventory in and out of $100m^3$ and $80m^3$ respectively. This indicator quantifies the extent to which competition will result from the assessed water use (consumption and degradation). Results show that the local stress indicator of water quality for both the influent and the effluent is important and that considering the local water quality is therefore relevant.

Tab. 2: Midpoint indexes (m^3-eq./m^3 water withdrawn/released) and resulting water stress indicators (WSI, in m^3-eq) for a process withdrawing 100 m^3 of water type S2a and releasing $80m^3$ of water S3, in different regions

Country	Watershed	Stress index S2a	Stress index S3	WSI
France	Meuse	0.500	2.69 E-05	50
Spain	Mino	0.653	0.046	61.62
Canada	St. Lawrence	0	0	0
China	Liao	1	0.995	20.4
China	Yalu Jiang	0.272	0	27.2

4.2 Complete endpoint assessment

Results from the application of the methodology to the ecoinvent process named "Corrugated board base paper, kraftliner, at plant" are presented in this section using the four damage categories described in Impact 2002+. Detailed results for regional endpoint CF in DALY/m^3 and for the fraction of water to compensate were presented in Boulay et al. [7].

Figure 2 shows that the human health impact category is dominated by impacts from water deprivation for agriculture and domestic uses, resulting in malnutrition and diseases. A small additional impact (<3% in comparison to process) can be found in all categories from the water compensation scenarios for hydropower production and agriculture. These results are for a region with an adaptation capacity of 0.46, meaning that almost half the unavailable water will generate impacts on human health directly and the rest will be compensated. This region also presents a high stress index (0.88) for the influent water and a null index for the effluent which implies that the released water does not return a valuable functionality as this water quality is not stressed. Impacts from water use in a similar hydrological context but in a region with full adaptation capacity (1 instead of 0.46, e.g. Europe, North America, etc.), would show no impact on human health occurring from deprivation, but close to twice the compensation impacts as compared to those shown here, hence < 6%. Conversely, in a less developed region, human health impacts from compensation could almost double, becoming the largely dominating source of human health impacts from board production.

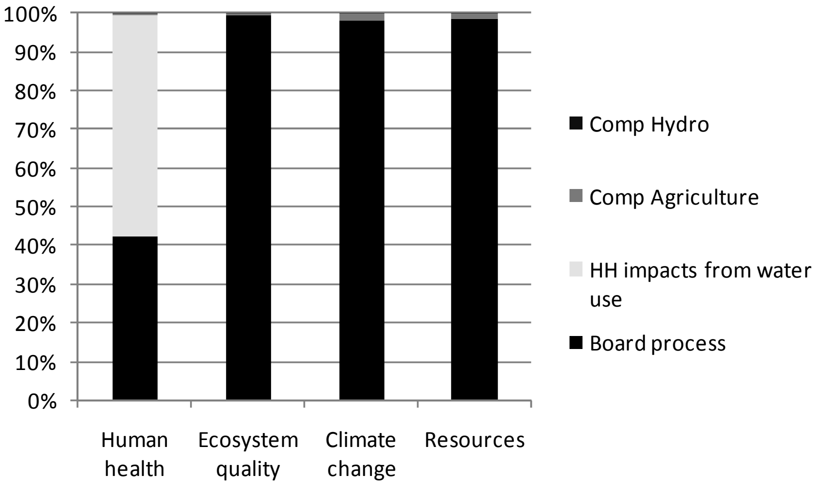

Fig. 2: Process and water use impacts from compensation and on human health from deprivation for the production of 1 ton of corrugated board in the region of Cape Town in South Africa.

5 Discussion

This methodology covers both the inventory modelling and the impact assessment of water use in a consistent framework.

The application of this method to a straightforward example illustrates the relevance of considering water use impacts in an LCA, especially from a human health perspective. This is the only method that is functionality-oriented and uses a consumptive-based scarcity ratio instead of the traditional, but misleading withdrawal-to-availability ratio. The inventory modelling takes into account the quality and the volume of water entering and exiting the process. Default water quality data are provided by this method in case no primary data on local water quality are available. To ensure the operationalization of this method within daily LCA practices, life cycle inventory databases must be expanded to account for released water volumes and therefore support the calculation of the quality of water exiting the process and thus, the water categories. To facilitate the use of these CFs, a generic dataset of effluent water quality by industry type could be generated. Moreover, a water mix similar to a grid mix could be set out based on the local surface/groundwater consumption data and local water quality data that could be used when actual inventory input data is not known. Still, at this point the methodology can be applied with data already available in most ecoinvent processes (volume and source of water influent and emissions to water in the effluent) and a hypothesis regarding the fraction of water evaporated.

References

[1] Bayart J-B et al. (2010) Framework for assessment of off-stream freshwater use within LCA. Int J Life Cycle Assess 15(5):439

[2] Boulay A-M et al. (2011) Categorizing water for LCA inventory. Int J Life Cycle Assess. In review.

[3] UNEP Global Environment Monitoring System (GEMS) Water Programme (2009) GEMStat

[4] Döll P (2009) Vulnerability to the impact of climate change on renewable groundwater resources: a global-scale assessment. Environ Res Lett 4 (035006):12.

[5] Alcamo J et al (2003) Development and testing of the WaterGAP 2 global model of water use and availability. Hydro Sci J 48(3):317-337

[6] Alcamo J, Henrichs T, Rosch T (2000) World water in 2025 - Global modeling and scenario analysis for the World Commission on Water for the 21st century, in Kassel World Water series 2000. Center in environnemental Systems Research University of Kassel

[7] Boulay A-M et al. (2011) LCA characterisation of freshwater use on human health and through compensation. Environ Sci Technol. In review.

[8] Vorosmarty CJ et al. (2000) Global Water Resources: Vulnerability from Climate Change and Population Growth. Sci 289(5477):284-288

[9] Pfister S, Koehler A, Hellweg S (2009) Assessing the Environmental Impacts of Freshwater Consumption in LCA. Environ Sci Technol 43(11): 4098-4104.

[10] de Haes UH et al. (2002) Life-cycle impact assesement: striving towards best practice, Pensacola: SETAC

[11] World Bank (2008) Data and Statistics - Country Classification [cited; Available from: http://go.worldbank.org/K2CKM78CC0.

[12] Howard G, Bartram, J (2003) Domestic Water Quantity, Service Level and Health. W.H. Organization (ed) World Health Organization, Genf, pp. 39

[13] Jolliet O et al. (2003) IMPACT 2002+: A new life cycle impact assessment methodology. Int J Life Cycle Assess 8(6):324-330.

[14] Goedkoop M, Spriensma R (2001) The Eco-Indicator 99 A damage oriented method for Life Cycle Impact Assessment. Methodology report, second edition. Pré Consultant, Amersfort

PART IV: LCM of Processes and Organisations

How to Measure and Manage the Life Cycle Greenhouse Gas Impact of a Global Multinational Company

Nicole Unger, Henry King and Siri Calvert

Abstract Unilever has recently launched its Sustainable Living Plan. It includes a target to halve the average per consumer use greenhouse gas (GHG) impact of the business by 2020. To set the GHG impact baseline for this target approximately 1600 representative products across 14 countries were measured, representing about 70 % of Unilever's sales. The results showed that less than 5 % of the product life cycle impacts occur in Unilever's own operations; the main contributors occur either with raw material suppliers or in the consumer phase. Thus, whilst programmes to reduce GHG emissions from manufacturing will continue, the largest reduction opportunity exists across the value chain. This paper will describe the approach taken to measure the GHG baseline and the challenges encountered as well as how the information and insights gained has helped guide Unilever's actions on GHG management across the value chain.

1 Introduction

Unilever is a multinational fast moving consumer goods company with a wide ranging portfolio in foods, household and personal care products; with well-known brands such as Knorr, Ben & Jerry's, Cif, Dove, Axe or Signal. Unilever has a long standing reputation on sustainability, which goes back to the two founding parties of the business, the Dutch margarine manufacturer (Margarine Unie) who aimed at providing cheaper nutrition with margarine in the late 19th century and Lever brothers in the UK who brought the first packaged soap (Sunlight) to Victorian Britain and by that significantly contributed to increase in hygiene. Following these first, mainly socially aimed activities, followed many more, including the Marine Steward Council which was founded in partnership of Unilever with the World Wildlife Fund and establishment of the Round Table for Sustainable Palm Oil.

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2 Greenhouse gases within the Unilever Sustainable Living Plan

Building on this previous work, Unilever launched its Sustainable Living Plan in November 2010 [1]. Key features of this plan are its relevance to all of Unilever's brands and products, a life cycle based approach and focus on all three pillars of sustainable development, i.e. social, economic and environmental. Furthermore, it articulates Unilever's business strategy of doubling the size of the company while reducing the environmental impact.

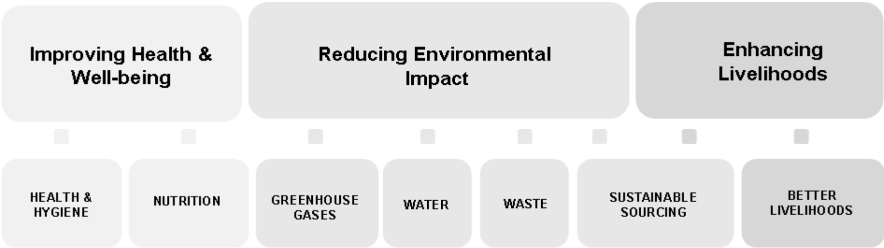

Fig. 1: The main pillars and structure of the Unilever Sustainable Living Plan [1]

An overview of the plan's structure can be seen in Figure 1. Each of these pillars has a number of targets aligned to it and in total there are 50 public, time bound goals specified in the plan.

Greenhouse Gases (GHG) is one of the four themes within the environmental pillar. The themes were chosen because of:

- Scientific relevance and scale of impact for our portfolio of products as a whole rather a specific product format or category. Unilever has conducted a number of assessments over the past 10-15 years to understand the scale of our environmental emissions and impacts starting with the Overall Business Impact Assessment Approach in the late 1990s. In addition we have also conducted many Life Cycle Assessments on our products and this has helped identify the major emissions and hotspots across the value chain.
- Relevance to external stakeholder expectations of the key environmental issues of the company
- The company's ability to measure/quantify the metrics in a practical, resource efficient and repeatable way based on life cycle thinking.

A key goal of the Sustainability Plan is to half the environmental impact (GHG, water and waste) of the business (on a per consumer use basis) by 2020 based on the footprint in 2008. This paper focuses on the Greenhouse Gas theme, describe the approach taken to establish the greenhouse gas baseline, the starting point of the target and challenges encountered. Moreover, the paper will discuss how the information and insights gained from this exercise has helped guide Unilever's actions on GHG management across the value chain.

3 Establishment of the measurement approach

3.1 The approach and baseline for greenhouse gases

In order to manage GHG it is necessary to establish a measurement process and targets. The basic requirements of the GHG measurement approach were:

- The establishment of a baseline greenhouse gas footprint based on 2008 sales,
- The measurement process is repeatable in order to track progress,
- The insights can be used to guide product development and other business decisions,
- The information can be used to guide and assess future category innovation plans.

A key challenge for a business such as Unilever is its diversity and size. Unilever sells products in over 170 countries; half of those sales are in developing and emerging countries (D&E). It was not feasible to measure every single product globally and therefore it was necessary to streamline the measurement process. This was achieved by defining a representative set of countries and products.
The factors for selection of countries included: coverage of all product categories, geographies, cultural differences in products and consumer habits, and current and future size. On this basis 14 countries were selected namely, Brazil, China, France, Germany, India, Indonesia, Italy, Mexico, Netherlands, Russia, South Africa, Turkey, UK and USA.
In order to identify representative products within these 14 countries the major products (e.g. pack size and format) were grouped clusters. From each cluster a representative product was selected for subsequent measurement. A key challenge in this clustering exercise was to strike a balance between creating enough product clusters to be representative of the product portfolio, and at the resource demands associated with data collection. In addition the product clusters needed to be suitable not only for the greenhouse gases assessment, but the same grouping was also used to estimate the water and waste baseline assessments. In total

approximately 1600 representative products were identified in the 14 countries representing about 70 % of Unilever's sales.

3.2 GHG measurement

Within Unilever life cycle assessment and life cycle thinking has been promoted and used for more than 15 years. There is an excellent level of life environmental knowledge within Unilever and this was used to guide the science behind this GHG baseline measurement. Up-scaling from individual product assessments to a portfolio assessment can be challenging as a balance had to be found between the level of detail of the results versus the input demand (e.g. data, time and resources). Previous product portfolio assessments of Unilever brands, e.g. on Ben and Jerry's Europe ([2] and [3]) or the global Knorr brand [2] provided important insights and understanding into this process.
The methodological approach for modelling and measuring GHG emissions was as follows:

- Full life-cycle approach based on ISO14040 standards.
- A 100 year time horizon for the CO_2 equivalent impacts was used.
- GHG emissions from land use associated with agricultural ingredients were partially included where data was available.
- GHG emissions from the biodegradation of petrochemical ingredients were excluded as it was very difficult to differentiate the source of a specific ingredient (e.g. oleochemical compared to petrochemical) from the existing product specification data and retrieval process. The aim is to include this area in future assessments.
- Published GHG data was used where possible but if there was no specific data (e.g. for an ingredient or process) expert choice was applied based on estimates or surrogates as necessary.
- Generic life cycle was modelled with commercial life-cycle assessment software provided by PE International (GaBi software and i-report).
- Fixed assumptions were used for phases of the life-cycle that contribute a small proportion to the product life-cycle (e.g. a single distance / mode of transport was used to describe product distribution to the retailer).

A simplified flow diagram of the representative product selection and GHG measurement process is shown in Figure 2. After product clustering and selection of representative products the necessary specification data and other relevant life cycle data (e.g. consumer habits data) were extracted from the various business IT

systems. Two generic life cycle models were then built in GaBi software (one for foods, see example in Figure 3 and one for household and personal care products) containing all possible ingredient inventories. This modelling approach helped to ensure consistency in the measurement process.
Based on the detailed models created in the GaBi expert tool, simpler tools (i-report) were created which allowed for the non-expert user to automate representative product data entry. GHG calculations were performed on a per consumer use basis and the results scaled up with the underlying sales figures of the cluster to give the total impact of all clusters in each country. The average Unilever GHG impact (per consumer use) was then calculated and used for company baseline and GHG reduction target setting.

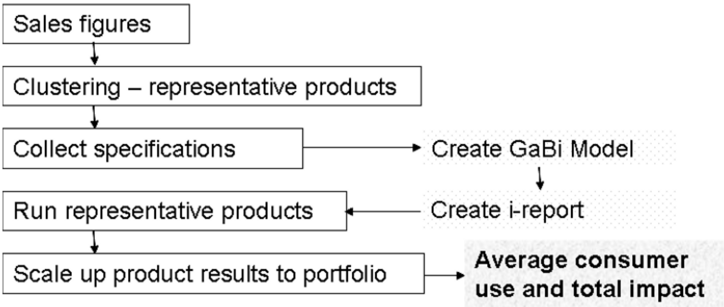

Fig. 2: Schematic flow showing the step of greenhouse gas baseline measurement process

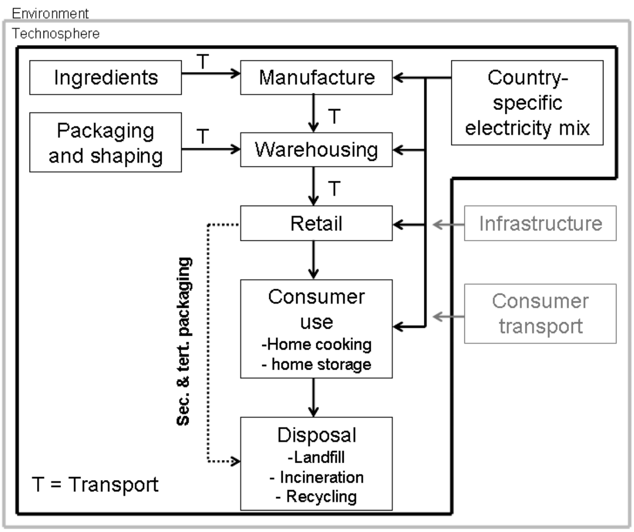

Fig. 3: Example of schematic outline of system boundaries for the food model

3.3 The results

The 2008 baseline results (Figure 4 and 5) confirmed and refined earlier GHG estimations of the business. They show that less than 5 % of the product life cycle GHG impacts occur in Unilever's own operations – the main contributions occur either with the suppliers of our raw materials or in the consumer use phase. Thus, whilst programmes to reduce GHG emissions from manufacturing will be continued and enhanced, the largest reduction opportunities exist across the value chain, in particular in the consumer use stage (68 % of the impact).

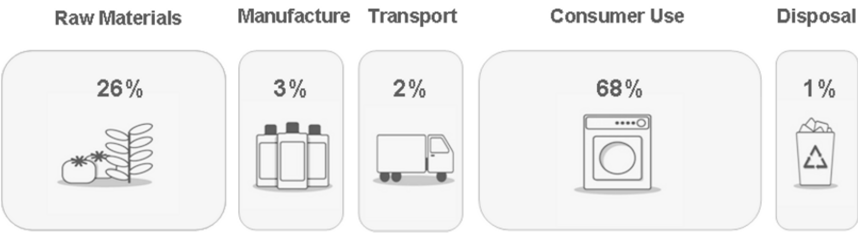

Fig. 4: Greenhouse gas footprint breakdown according to life cycle stages for the Unilever portfolio for 2008 measured in 14 countries, based on approximately 1,600 representative products [1]

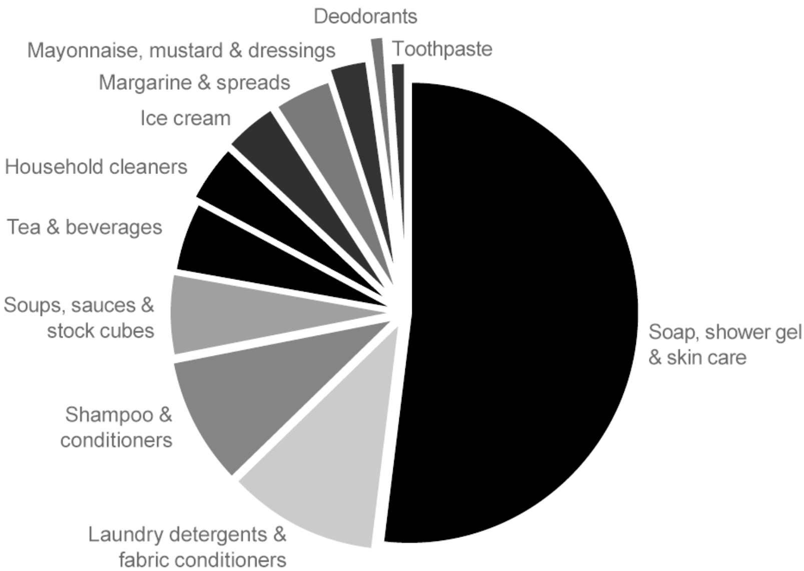

Fig. 5: Greenhouse gas footprint breakdown according to product category for the Unilever portfolio for 2008 measured in 14 countries, based on approximately 1,600 representative products [1]

When reviewing the results by business category additional insights can be obtained. Figure 5 shows the contribution of individual product categories to the overall Unilever GHG impact profile. It can be seen that soaps, shower gels and skin care products are the major contributor to Unilever's GHG footprint (52 %). This reflects the contribution from sales in countries such as the US where the consumer habit is to use hot water. In contrast in many developing countries washing is often performed with water at ambient temperature. In Unilever's case the contribution from laundry is relatively low due to the majority of sales being in countries with hand wash or ambient washing conditions. In the case of ice-cream whilst the impact per consumer use is relatively high the total number of servings is much smaller that the individual uses of say shower or shampoo products.

4 Management plan for GHG footprint

While understanding the impact of the company's GHG portfolio is important it is only when there is a management reduction plan that there will be benefits to the environment. The baseline has enabled Unilever to understand its key greenhouse gas impacts by category, life cycle phase and business. The baseline has been invaluable in getting buy-in from senior business leaders and guided the development and enhancement of reduction programs and targets which have been communicated internal and external in the Sustainable Living Plan. In the communication of the Sustainable Living Plan it has been made clear that the individual product GHG data is not appropriate for use in external claims or labelling due to the methodological simplifications and some of the underlying assumptions in the modelling.

Reducing the contribution from consumer phase is a significant challenge since this life cycle phase is not under the direct control of the company. However, it is acknowledged we that we must take a life cycle perspective and therefore all opportunities need to be explored. These include innovation-led reductions, consumer habits change, and advocacy on relevant public policy areas such as low carbon energy, machine efficiency and building standards.

In addition Unilever will continue to take actions to reduce GHG impact upstream of its manufacture. For instance Unilever has committed to purchase all palm oil from certified sources by 2015 and thereby avoid further deforestation of rainforests and the associated land use change emissions. There is also work in progress to understand the GHG emissions of key crops through the roll-out of a GHG management tool for farmers (Cool Farm tool). For details on the agricultural programme see [5].

Additionally the baseline assessment has been used as the basis of a new approach to assessing the environmental impact of internal product renovations and innovations within the innovation process management systems. It is planned to use this, combined with the project management process, to systematically challenge and reduce the environmental impacts of our future product launches.

Metrics and associated targets are often very abstract by nature and in order to make them more tangible they need to be broken down into roadmaps with actionable steps using non-expert friendly terms. Consequently in addition to the headline target of halving the GHG footprint on a per consumer basis the following commitments have been identified so far:

- To reach 200 million consumers by 2015 (400 million by 2020) with products and tools that will help them to reduce their GHG emissions while washing and showering
- To concentrate and compact laundry products
- Reformulate laundry products to reduce GHG emissions by 15 % by 2012
- Encourage consumers to wash cloths at lower temperatures and at the correct dosage in 70 % of machine washes by 2020

Finally, to ensure Unilever is progressing towards its target of halving the average per consumer use GHG impact, regular updating of the footprint is planned.

5 Recommendations, insights and outlook

Sustainability is important for many businesses but managing impacts can be challenging with today's global supply chains with often many stakeholders involved. In this paper the approach developed by Unilever to measure its Greenhouse Gas impact is described. It is important to remember that the GHG impact is only one aspect within the wider goals of the Unilever Sustainable Living Plan which addresses all three pillars (social, environmental and economic). Only by considering a wider range of aspects can potential trade-offs and synergies can be recognised and addressed. Key to the successful implementation of the Plan has been senior management commitment and the fact that sustainability is integrated into the business objectives rather than being a separate activity.

Some of the key challenges to implementation of the GHG measurement process were:

- The need to access multiple and different IT systems in the business
- Varying data quality and specificity
- Thesaurus and technical issues
- GHG measurement and data tracking is not a mainstream business process

These challenges, and the breadth of data types required, meant that it required considerable amount of effort from across the business to compile all relevant data. The project team included members from marketing, R&D, supply chain, packaging, IT, finance as well as environmental experts. Key to completing the baseline was training and awareness raising and this was often made more challenging due to the different levels of expertise and career backgrounds in each category, and also the global (and hence cultural) spread of the project team.
This is only the start of the journey and there are many challenges which will need to be addressed in the coming years. For instance the science of greenhouse gas accounting will continue to evolve (e.g. on land use change, integration of biodegradation of chemicals in the environment). Such changes will need to be integrated into the measurement if it is to remain credible.

References

[1] <http://www.sustainable-living.unilever.com/> (accessed 11.04.2011)
[2] Garcia-Suarez T, Sim S, Mauser A, Marshall P (2008) Greenhouse Gas Assessment of Ben & Jerry's ice-cream: communicating their 'Climate Hoofprint'. In: Proceedings of the 6th International Conference on LCA in the Agri-Food Sector November 12–14, Zurich
[3] <http://www.benjerry.co.uk/climateneutral/> (accessed 11.04.2011)
[4] Milà i Canals L, Sim S, García-Suárez T, Neuer G, Herstein K, Kerr C, Rigarlsford G, King H (2011) Estimating the greenhouse gas footprint of Knorr. Int J Life Cycle Assess 16(1):50-58.
[5] <http://www.growingforthefuture.com/> (accessed 11.04.2011)

Best Practice Application of LCM by Retailers to Improve Product Supply Chain Sustainability

David Styles, Harald Schoenberger and José Luis Galvez-Martos

Abstract Retailers are strategically positioned to leverage environmental improvement across product supply chains. This paper condenses retailer best practice into a proposed framework for systematic supply chain improvement based on eight best environmental management practice (BEMP) techniques. Third party product environmental certification is the preferred mechanism of improvement owing to transparency and credibility advantages, followed by use of retailer-defined environmental requirements, and implementation of supplier improvement programmes based on benchmarking and dissemination of better management practices. A BEMP to encourage consumption of front-runner ecological products is defined based on use of front-runner ecolabels. The performance of front-runner retailers is used to derive benchmarks of excellence for each technique, primarily expressed as sales shares of improved products within priority product groups. Life cycle management underpins best practice.

1 Introduction

Market prices for energy and commodities do not fully reflect scarcity concerns or environmental damage, leading to unsustainable supply chains in terms of resource depletion and environmental degradation. Product availability (supplier capacity and reliability), price and quality have traditionally been the dominant criteria used by retailers to select suppliers. In recent years there has been a trend towards the development of more collaborative relationships with suppliers in order to address sustainability criteria [1]. There are many driving forces behind this trend, but essentially retailers realise that future business success depends upon securing sustainable supply chains. A risk oriented perspective of supply chain management reflects rational management for large retail businesses, and greater control over supply chains has been linked with stronger long-term business growth [2]. Many products sold by retailers are sourced in economically

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less developed countries with poor environmental standards (weak regulation and poor management practices) and/or from small suppliers that fall below the size threshold for enforced environmental regulation. Retailers and large branded product manufactures are often uniquely positioned to influence the environmental performance of suppliers, and may be able to leverage considerable market power. However, realising this potential represents a massive logistical challenge owing to the large product assortments, large numbers of often geographically distant suppliers, and complex product value chains. This paper presents some main findings underpinning the development of a best environmental management practice (BEMP) reference document for retailers that will shortly be made available by the European Commission [3].

2 Methodology

Information on retailer actions to improve the environmental performance of product supply chains was collated from the following sources and procedures:

- Retailer sustainability reports and other relevant literature
- Communication with retailer corporate social responsibility (CSR) personnel
- meetings with retailer management in company headquarters
- Participation in the Retail Forum (http://ec.europa.eu/environment/industry/retail/index_en.htm)
- Participation in the Food Sustainable Consumption and Production Round Table (http://www.food-scp.eu/)
- Communication with supply chain consultants including the UK's Carbon Trust and the World Wildlife Fund (WWF)
- Communication and meetings with members of Technical Working Group (TWG) set up to steer development of the reference document [4]

Based on the collected information and environmental pressures for priority product groups documented in reports such as [5], eight BEMPs relevant to retailer improvement of product supply chains were described (see section 3). The primary criteria for BEMP selection were: (i) relevance to product supply chain sustainability improvement; (ii) potential to achieve significant environmental benefit; (iii) widespread applicability across retailers and product groups; (iv) measurability and transparency. BEMPs were described with an emphasis on their main environmental benefits and any environmental trade-offs, economic implications of implementation and specific applicability restrictions. The list of

BEMPs was finalised during a TWG meeting involving various retail stakeholders [4]. The eight BEMPs were ordered into a proposed sequence that can be applied by retailers to drive widespread environmental improvement of their product supply chains. BEMPs were categorised as either prerequisites for supply chain improvement, or key elements of either of two alternative but complementary strategies: (i) retailers driving supply chain improvement; (ii) retailers facilitating consumers to drive supply chain improvement.

Based on the most damaging product groups across eight environmental pressures (abiotic depletion, acidification, ecotoxicity, eutrophication, global warming, human toxicity, ozone depletion, photochemical pollution), as identified in [5], ten priority food product groups and five priority non-food product groups were taken to provide a relevant and consistent framework to measure retailer performance. Where possible, performance was measured for each priority product group based on: (i) the proportion of sales represented by a particular action (e.g. certification according to a particular third party standard); (ii) the level of environmental performance associated with that action, as classified in [3]. Percentage sales shares were calculated by value within product groups. The highest sales shares, or highest credible targets, were taken to represent 'benchmarks of excellence' for BEMP implementation.

3 Results

Product standards play an important role in demonstrating environmental performance improvement within some supply chains, but differ considerably in the level of environmental performance they represent. The retail BEMP document [3] classifies commonly used product environmental standards according to the environmental rigour of the core criteria contained within them (Table 1). This classification is approximate, with the primary aim of enabling the performance of different retailers to be compared.

Improving product supply chain sustainability involves decisions that are pertinent to core retail strategy (market positioning), and requires action throughout the retail organisation – involving top level management, procurement, quality control and marketing divisions. Priority products, improvement options and mechanisms of retailer influence must be identified.

Tab. 1: Proposed classification of widely recognised third party environment-related standards commonly applied to products

Level	Widely used standards	Product groups
BASIC	GG: Global Good Agricultural Practice	Crops and livestock
	GRLF: Greenpeace red-list fish (deselection)	Fish
	OT: Oeko-Tex 1000	Textiles
	NPC: National/regional production certification (e.g. Red Tractor British origin certification)	All food
IMPR-OVED	BCI: Better Cotton Initiative	Cotton products
	BCRSP: Basel Criteria on Responsible Soy Production	Soy (feed supporting dairy, egg and meat)
	BSI: Better Sugarcane Initiative	Sugar products
	4C: Common Code for the Coffee Community Association	Coffee
	FLO: Fairtrade Labelling Organisation (an exemplary social standard)	Agricultural products from developing regions
	RA: Rainforest Alliance (products from the tropics)	Agricultural products from the tropics
	RSPO: Round Table on Sustainable Palm Oil	Palm oil products
	PEFC: Programme for the Endorsement of Forestry Certification	Wood and paper
	RTRS: Round Table on Responsible Soy	Soy products
	UTZ	Cocoa, coffee, palm oil, tea
EXEMP-LARY	FSC: Forest Stewardship Council	Wood and paper
	MSC: Marine Stewardship Council	Wild-catch seafood
FRONT-RUNNER (NICHE)	EL: Ecolabel (Blue Angel, EU Flower, Nordic Swan)	Non-food products
	OC: Organic (EC standard, GOTS, KRAV, Soil Association, etc)	Food and natural fibre products

Three prerequisite BEMP techniques are defined:

BEMP 1. Integrate sustainable sourcing into business strategy and operations. 'Develop supply chain environmental improvement objectives and specific targets that are communicated to stakeholders and integrated into the retail organisation through a high-level unit with responsibility for implementation'. Coop CH and M&S both convey a comprehensive understanding of environmental pressures within their supply chains through sustainability reporting, and through implementation of a range of best practices [6,7].

BEMP 2. Product supply chain assessment. 'Collate scientific information on environmental hotspots for core product supply chains in order to identify priority product groups for improvement and priority improvement options'. Best practice is defined as efficiently identifying groups and options using the most readily available data first, in the order: available literature, generic life cycle assessment (LCA) data, expert consultation, and finally detailed supplier-specific data.

Detailed LCA is only required for ecodesign and for supply chains where it is necessary to benchmark the performance of individual suppliers to drive improvement (e.g. dairy production). Coop CH and M&S again demonstrate best practice for this technique owing to a pragmatic focus on the identification of improvement options and collaboration with experts.

BEMP 3. Identify effective supply chain improvement mechanisms. 'Identify chains of custody and control points to improve priority product groups and suppliers'. Best practice in this technique involves the identification of the most effective control points within supply chains of priority product groups. Best practice is demonstrated by H&M owing to the wide range of improvement mechanisms they implement at different points in the supply chain - including an audited code of conduct and a restricted chemical list for first-tier suppliers that includes requirements for second-tier suppliers, a requirement for compliance with Business for Social Responsibility wastewater quality guidelines, and a voluntary Cleaner Production Programme aimed at second-tier textile finishers (a hotspot for water pollution in the supply chain, as indicated in Figure 2).

Following implementation of prerequisite techniques, retailer actions to improve the sustainability of their product supply chains tend to focus on one of two distinct but complementary strategies. The first of these is to *drive* widespread environmental performance improvement across suppliers of priority products by establishing universally applicable requirements or improvement schemes, and comprises four BEMP techniques, listed in order of preference:

BEMP 4. Choice editing and green procurement. 'Exclude worst performing products, and require widespread certification according to environmental standards for priority product groups'. Best practice examples for can be seen in Tables 2 and 3, and include high penetration of Fairtrade certification within applicable product groups (Sainsbury's), high penetration of FSC certification within wood and paper products (B&Q), universal GlobalGAP certification of imported fruit and vegetables (Coop CH, Migros, Rewe), and high penetration of MSC certification within wild-catch seafood (M&S, Waitrose).

BEMP 5. Establish environmental criteria for products and suppliers. 'Define and enforce environmental requirements for suppliers of priority product groups, targeting environmental hotspots'. H&M and IKEA demonstrate best practice with extensive auditing to enforce code of conduct requirements across first-tier suppliers. In addition, H&M enforces chemical restrictions and wastewater quality criteria (see BEMP 3), whilst IKEA enforces a Forestry Standard (FS) throughout non-FSC wood supply chains (Table 3). Retailer requirements are classified as per third party standards in [3], to enable performance comparisons (Tables 2 and 3).

Tab. 2: Front-runner retailer performance across priority food product groups (% private label sales within each product group improved following implementation of BEMPs 4, 5, 6 or 8).

Products	Standard	BEMP	Best performers	
			2010	**Target (year)**
Coffee, tea	Improved	4	100% FT[SS]; 20% 4C[CS]	100% 4C (2012)[CS]
	Front-runner	8	3% OC[CS]	-
Dairy	Basic	4	100% NPC	
	Improved	6	100% SDG[SS]	100% SAP (2015)[M&S]
	Front-runner	8	12% OCCS	-
Farmed fish	Front-runner	8	29% OC[CS]	-
Fats and oils	Improved	4	100% RSPO[M&S,MG]; 60% BCRSP[CS]	100% RSPO (2012)[WE]; 100% RSPO (2015)[CR,CS,M&S,SS,TO]; 95% BCRSP/RTRS (2014)[CS,MG]
	Front-runner	8	15% OC[CS]	-
Fruit and vegetables	Basic	4	100% GG[CS,ICA,MG,RW]	-
	Basic	5	100% CR[CS,ICA,MG,RW]	-
	Improved	4	100% FLO (bananas)[SS]	-
	Improved	6	-	100% SAP (2015)[M&S]
	Front-runner	8	11% OC[CS]	-
Grain products	Improved	6	-	100% SDG[SS]; 100% SAP (2015)[M&S]
	Front-runner	8	20% OC[CS]	
Poultry, eggs	Improved	6	-	100% SDG[SS], 100% SAP (2015)[M&S]
	Front-runner	8	23% OC[CS]	-
Red meat	Basic		-	-
	Improved	6	-	100% SAP (2015)[M&S], 100% SDG[SS]
	Front-runner	8	10% OC[CS]	-
Seafood (wild catch)	Basic	5	100% GRLF[AD,CS,ICA,MG,M&S,SS,WE]	-
	Improved	5	100% SFS[ICA,MG,M&S,WE]	-
	Front-runner	4	62% MSC[M&S]	100% MSC (2012)[M&S]
Sugar	Improved	4	100 % FLO[M&S;SS] (bagged)	-
	Front-runner	8	8% OC[CS]	-

[AD]Axfood, [CR]Carrefour, [CS]Coop Switzerland (CH), [IA]IKEA, [KR]Kingfisher, [MG]Migros, [M&S]Marks & Spencer, [OT]Otto, [RW]REWE, [SS]Sainsburys, [TO]Tesco, [WE]Waitrose.
Standard abbreviations explained in Table 1.

BEMP 6. Intervene to encourage supplier improvement. 'Implement a data exchange system to benchmark supplier environmental performance and disseminate best practice, or assist suppliers to achieve third party certification'. This is a resource intensive technique applicable where third party standards cannot be used, and requires dissemination of better management practices (BMP) across many small suppliers (e.g. dairy or cotton farmers) to improve overall environmental performance. Best practice is demonstrated within the BCI (Table 1) and by Sainsbury's Development Groups (SDG) - established to improve the security of milk supply, these groups now use carbon foot-printing, benchmarking and BMP dissemination to drive eco-efficiency improvement. M&S is establishing a similar Sustainable Agriculture Programme (SAP) (Table 2).

BEMP 7. Strategic collaboration on product and standard development. 'Participate in research to drive supply chain improvement, including collaboration to develop international product standards and to drive innovation in products or production methods'. BEMP 7 is distinct from the supply chain impact assessment (BEMP 2) in that research is orientated towards product or supplier innovation and collaboration to develop global international third-party standards that can be used to certify supply chain environmental improvement. Coop CH provides a best practice example through active participation in the development of standards such as the BCRSP (Table 1) and through coordination and funding of strategic research aimed at supply chain improvement (e.g. breeding wheat varieties that achieve high yields under organic management).

The second strategy pursued by retailers to improve the sustainability of their product supply chains is to promote ecological consumption among consumers. Best practice has been confined to one technique:

BEMP 8. Promote front-runner ecological products. 'Use awareness campaigns, positioning, pricing and own-brand ecological ranges to promote front-runner ecological products associated with a significant price premium'. This technique is distinct from green procurement based on certification (BEMP 4) because it requires consumers to pay a significant price premium for labels that, by definition, represent front-runner products and are not universally applicable within relevant product groups. The technique is therefore based on retailers influencing consumer choice through promotional pricing, in-store product placement and advertising programmes. Best practice is demonstrated by retailers who consolidate and market ecological products within clearly defined ecological ranges. Examples include Coop CH's Naturaplan (organic food) and Naturaline (organic non-food) ranges (see Coop CH organic sales shares in Table 2), Colruyt's dedicated organic Bio Planet stores, and KF Sweden's Änglamark range (EC Flower and Nordic Swan ecolabels, organic and Fairtrade certification). Best practice retailers also intervene to increase the availability, and reduce the price, of ecological products through BMP dissemination (BEMP 6) and supply chain coordination to facilitate front-runner certification (e.g. C&A with organic cotton).

Tab. 3: Front-runner retailer performance across priority non-food product groups (% private label sales within each product group improved following implementation of BEMPs 4, 5, 6 or 8).

Products	Standard	BEMP	Best performers	
			2010	**Target (year)**
Electronic goods	Basic	4	100% A+[CS,MG]	
	Front-runner	8	EL (NA)	
Household chemicals	Front-runner	8	5% EL[KFS]	
Household furniture	Basic	5	100% CoC[IA]	
	Improved	6	92% FS[IA]	
	Exemplary	4	22% FSC[IA]	100% FSC[IA]
Textiles	Basic	5	27% OT-100[C&A;] 57% CoC[H&M]; 100 % CR[H&M]	
	Improved	4	12% BCI[IA]	100% BCI (2015)[IA]
	Front-runner	8	60% OC[CS]	100% OC 'and other sustainable' (2020)[C&A]
Wood and paper	Improved	5	92% FS[IA]; TPSS[B&Q]	
	Exemplary	4	76% FSC[B&Q]	100% FSC[IA]

[CS]Coop Switzerland (CH), [IA]IKEA, [KFS]Kooperativa Förbundet Sweden, [KR]Kingfisher, [MG]Migros, [OT]Otto Group, standard abbreviations explained in Table 1.

Based on front-runner retailer performance summarised in Tables 2 and 3, and the European Commission's target for EU Flower ecolabel penetration, two widely applicable benchmarks of excellence were established:

- A target for 100% of priority products sold to be improved through certification, environmental criteria or supplier participation in improvement programmes
- A target for 10% of priority product group sales to be comprised of third-party certified front-runner products

In addition, for wild-catch seafood and wood and paper products, the target is 100% certification according to 'exemplary' standards (e.g. MSC and FSC). Figure 1 provides a hierarchical sequence of BEMP implementation for the systematic environmental improvement of retail assortments.

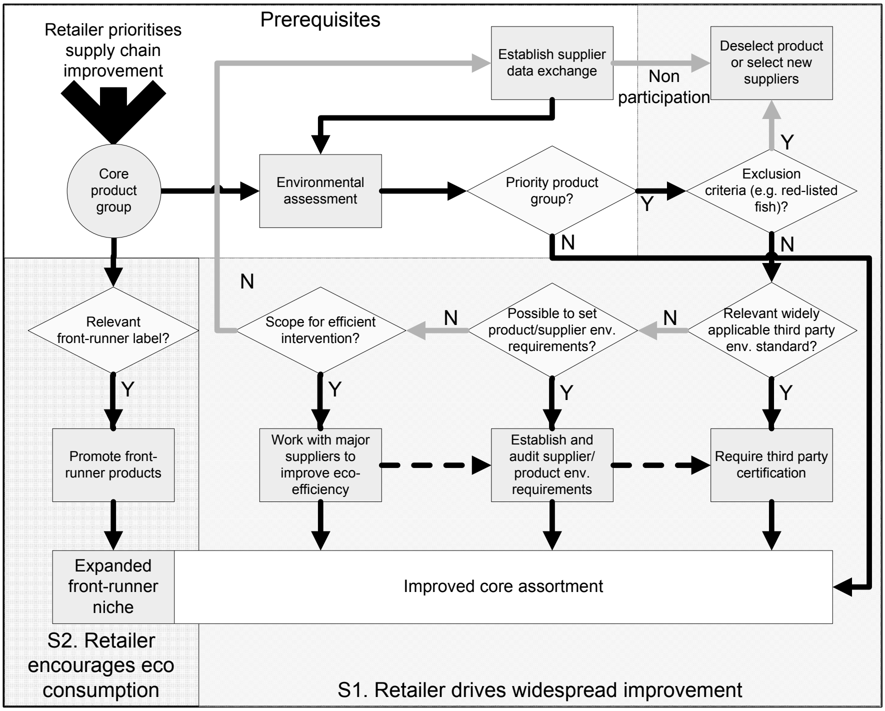

Fig. 1: Proposed sequence of key questions and actions (shaded rectangular boxes) representing best practice for systematic supply chain improvement, classified as prerequisites or one of two strategies (S1 and S2)

4 Discussion

Retailers are at an early stage of supply chain environmental performance improvement, and are pursuing a range of different actions under different strategies. The study summarised here, and the BEMP reference document it underpins [3], aims to: (i) identify best practice from an independent and scientific perspective; (ii) guide retailers to implement best practice; (iii) enable quantitative assessment of retailer performance compared with front-runners.

Three key messages arise from this study:

- (i) Detailed LCA is usually not required to identify product supply chain hotspots and improvement options, but life cycle *thinking* is.
- (ii) Third party certification is preferable to retailer-defined environmental requirements or improvement programmes, for reasons of

transparency, credibility, comparability and efficiency (suppliers comply with common standards).

- (iii) Consumer choice is not an appropriate mechanism to drive sustainability improvement across thousands of product supply chains with different environmental profiles and challenges [8,9,10].

Consequently, retailer initiatives to influence consumer choice through product lifecycle performance labelling (e.g. carbon footprints) are not considered best practice. Retailers also have a role to play in encouraging more sustainable consumption patterns (e.g. reduced meat consumption) and in reducing food waste, but these important issues are outside the scope of this paper.

Lifecycle thinking is integral to supply chain assessment and the identification of effective improvement options (BEMP 2 and 3), whilst implementation of the BEMPs to improve supply chains represents effective LCM. The value chain for cotton textiles (Figure 2) provides a good example of the need for lifecycle thinking to effectively target supply chain environmental improvement action. Cotton cultivation and textile finishing are key hotspots in the cotton-textile supply chain that can be addressed directly by retailers through green procurement of certified cotton (BEMP 4), establishment of environmental requirements (BEMP 5) or intervention to disseminate BMP (BEMP 6), and indirectly by encouraging consumers to pay a price premium for front-runner textile products (BEMP 8). Over the lifecycle of a textile product, washing is a hotspot for energy consumption that retailers can influence through; (i) ecodesign of own-brand washing powder to enable lower washing temperatures (BEMP 7); (ii) ecodesign of textiles to reduce ironing washing and requirements [11] (BEMP 7); (iii) encouraging consumers to buy ecolabelled washing powder (BEMP 8). Actions that reduce *lifecycle* impacts may increase *supply chain* impacts (e.g. including enzymes in washing powder: [12]). Front-runner retailers demonstrate effective LCM through targeted application of BEMPs across priority supply chains.

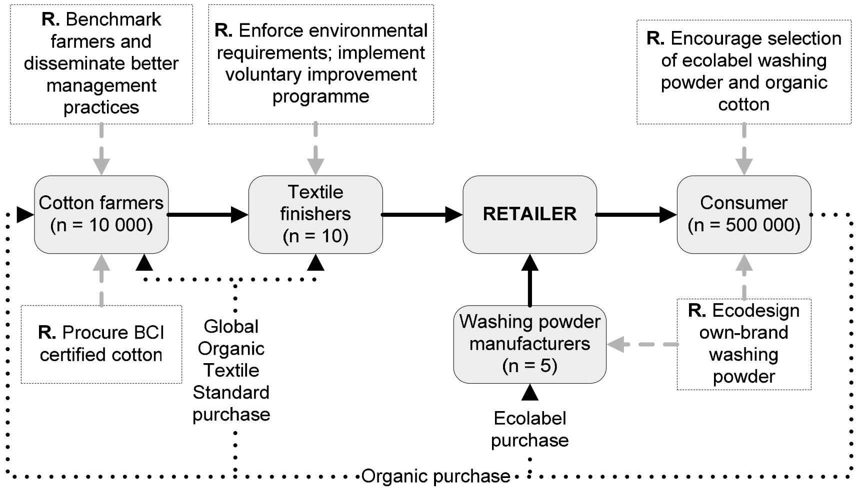

Fig. 2: Retailer control points (R) **for hotspots within the value chain of cotton textiles (n = indicative number of actors at key stages)**

Definition of BEMPs based on existing best practice assures their commercial viability. Where green procurement based on certification necessitates higher retail prices (e.g. MSC [13]), it should be accompanied by a marketing strategy that emphasises the environmental value added (e.g. Coop CH, M&S, Sainsbury's). However, large retailers can leverage their market power to demand certification as an order *qualifying* (rather than an order *winning*) criteria (e.g. FSC [14]), in which case no price premium is paid. Meanwhile, the premium commanded by some certified commodities, such as RSPO-certified palm oil, may translate into negligible increases in final product prices [15]. Any costs incurred by retailers should be balanced against marketing opportunities and the considerable business risks of inaction.

5 Conclusions

Front-runner retailers are leveraging their strategic position and market power to improve environmental performance across unsustainable product supply chains. This involves life cycle management, but not necessarily detailed LCA, to identify priority product groups and improvement options. Choice editing and green procurement based on product environmental standards are powerful tools for retailers to improve supply chains, although additional techniques are required where appropriate standards are not available. Consumer choice (and retailer action to influence it) has a more limited role to play in driving improvement across the complex range of relevant supply chains and environmental pressures.

References

[1] Ganesan S, Morris G, Jap S, Palmatier RW, Weitz B (2009) Supply Chain Management and Retailer Performance: Emerging Trends, Issues, and Implications for Research and Practice. J Retailing 85:84-94.

[2] Accenture (2009) The Sustainable Supply Chain: Balancing cost, customer service and sustainability to achieve a high-performance supply chain. Accenture

[3] European Commission (2011) Pilot reference document on best environmental management practice in the retail trade sector. (Draft January 2011), IPTS Seville. <http://susproc.jrc.ec.europa.eu/activities/intro.htm>

[4] TWG (2010) Technical Working Group for the Retail Trade sector. Workshop 18-19th November 2010, Seville

[5] EC (2006) Environmental Impact of Products (EIPRO): Analysis of the life cycle environmental impacts related to the final consumption of the EU-25. IPTS Seville

[6] Coop CH (2010) Coop Group Sustainability Report 2009. Coop CH Basel

[7] M&S (2010) How we do business report 2010: Doing the right thing. M&S, London

[8] Tukker A, Emmert S, Charter M, Vezzoli C, Sto E, Andersen M, Geerken T, Tischer U, Lahlou S (2008) Fostering change to sustainable consumption and production: an evidence based view. J Cleaner Prod 16(11):1218-1225.

[9] Horne R (2009) Limits to labels: The role of eco-labels in the assessment of product sustainability and routes to sustainable consumption. Int J Consumer Stud 33(2):175-182.

[10] Upham P, Dendler L, Bleda M (2011) Carbon labelling of grocery products: public perceptions and potential emissions reductions. J Cleaner Prod 19(4):348-355.

[11] BASF (2008) Non-iron bed linen by BASF SE. BASF Ludwigshafen

[12] Henkel (2009) Case study Persil Megaperils by Henkel Ag & Co. KGAA

[13] MSC (2009) Net benefits. The first ten years of MSC certified sustainable fisheries. MSC, London

[14] Morris M, Dunne N (2004) Driving environmental certification: its impact on the furniture and timber products value chain in South Africa. Geoforum 35(2):251-266.

[15] Paoli GD, Yaap B, Wells PL, Sileuw A (2010) CSR, Oil Palm and the RSPO: Translating boardroom philosophy into conservation action on the ground. Trop Conserv Sci 3:438-446.

Life Cycle Management Approach to the Design of Large-Scale Resorts

Kristin Lee Brown, Daniel Clayton Greer and Ben Schwegler

Abstract The Walt Disney Company has been dedicated to understanding and reducing its environmental impacts for years. As a large and complex corporation, quantifying and understanding the most beneficial and cost-effective ways to reduce these impacts can be a challenge. Life cycle costing (LCC) has already played an important role in the evaluation of design decisions at Walt Disney Imagineering (WDI). Recently, WDI has started to incorporate environmental life cycle assessment (LCA) alongside existing LCC efforts to move towards life cycle management (LCM). In order to be a practical tool for large-scale developments, LCM must be scalable and effective at a wide level of detail. Evaluating the large footprint of a major resort is highly complex and involves a wide variety of components and processes, at a scale shared by small to mid-size cities and communities. To deliver a sustainable design for a new Disney resort, our research is focused on determining what level of detail and scope will be sufficient to support design decisions on this scale, without expending more resources than necessary. Furthermore, the development of internal tools and processes will be necessary to integrate environmental impact information into the design process. Providing life cycle information to the broad spectrum of park designers and planners in a useful and understandable format is critical to fully pursuing life cycle management. Integrating these tools into the design process will help establish sustainability as a core consideration in the design of our resorts.

1 Introduction

The Walt Disney Company (TWDC) is one of the leading providers of entertainment. The company is comprised of four primary segments: 1) media networks, 2) studio segment which includes films, animated TV products, music, and live stage productions, 3) consumer products, and 4) parks and resorts – the broadest segment with the most diverse products. Walt Disney Parks and Resorts (WDPR) is the focus of the work presented in this paper.

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Since its founding, TWDC has pursued initiatives to protect and preserve the environment. Examples include the use of bio-fuels to power the steam trains at Disneyland Resort, the use of integrated pest management in Disneyland Resort to control eugenia psyllid, and advanced process control of wastewater treatment facilities at Walt Disney World Resort [1-3].

In recent years, increased awareness of issues such as global warming has launched the company to take a more quantitative approach to pursuing environmental initiatives. In 2006, TWDC conducted the first inventory of its direct and indirect greenhouse gas emissions (Scope 1 and 2); and in 2008, TWDC announced its corporate environmental goals. These goals include zero net direct greenhouse gas emissions, reduction in indirect greenhouse gas emissions, water use minimisation, zero waste to landfill, product footprint minimisation, net positive impact on ecosystems, and to inform, empower, and activate positive action for the environment [4]. Specifically, WDPR has established short-term targets to move towards fulfilling the larger corporate goals.

Within WDPR, Walt Disney Imagineering (WDI) is the master planning, creative development, design, engineering, production, project management, and research and development arm. WDI is responsible for the creation and installation of all Disney resorts (theme parks, hotels, and amenities) and cruise ships [5]. Imagineering has a responsibility to help WDPR achieve the environmental goals through sustainable design practices. Achieving these goals in the most cost-effective manner requires the implementation of life cycle management thinking.

2 Life cycle approaches to resort design

TWDC has been using life cycle cost (LCC) evaluations to balance up-front investment and long-term operations and maintenance cost decisions. The practice of life cycle assessment (LCA) has recently been introduced as an evaluation tool to help realise the environmental goals. Because WDPR is responsible for the design, construction, and operations of the resort as opposed to only one phase of the life cycle, it can benefit greatly from a focus on lowering life cycle costs and environmental impacts. On the other hand, implementing life cycle management practices within WDPR is challenging due to the scale, complexity and uniqueness of the product. Each of the major Disney resorts occupies a large land area and is comparable in size and complexity to a small- to mid-sized municipality.

In the design, construction, and operation of each resort, thousands of decisions need to be made concerning building design, materials selection, landscape practices, waste stream management, and other elements. All of these decisions have direct and/or indirect environmental implications. In order to fully implement life cycle management thinking, the active participation of a broad spectrum of designers and engineers will be required. This is difficult given LCA is relatively new and limited in its use at WDI. To build broader support and participation of the WDI design community, life cycle approaches will need to be made more comprehensible and useful.

2.1 Scale, complexity, and uniqueness of Disney resorts

Many companies using life cycle methods manufacture one major product or several lines of similar products, which allows them to control the entire process through careful adjustments made to processes, material selection, and equipment. Disney's resorts are of a much greater order of magnitude. Walt Disney World Resort in Orlando, Florida, as an example, covers over 100 km^2 of land area with over 25 km^2 of developed areas, contains more than 2,300 buildings, 23 hotels, employs over 60,000 cast members, and welcomes tens of millions of guests each year [6-8]. It encompasses all components that comprise a small city—utilities, infrastructure, food and beverage, merchandise, entertainment facilities, attractions, and media. Traditional life cycle assessment and life cycle costing methods are difficult to apply at this scale and complexity—it would be impossible to conduct a detailed assessment on every single decision made during the course of designing, building, and operating the resort.

In addition, most elements created in parks and resorts are unique, meaning that they are conceived, designed, and built as a single occurrence (as opposed to mass produced). The investment in a comprehensive life cycle evaluation on a product that is only produced once has a much greater relative cost impact. Therefore, deciding whether or not to pursue a life cycle evaluation and the granularity of any evaluation pursued for a complex one-time decision, needs to be carefully considered to not negate potential returns.

2.2 Level of environmental knowledge and experience

Designing resorts involves a blending of technical expertise and creative vision. The varieties and types of planning that are done and the decisions that are made across the many divisions at WDI represent a larger spectrum than what a traditional LCA approach encompasses. WDI spans a broad range of professional disciplines and encompasses traditional roles of an architecture firm, a technology

company, an entertainment company, a construction management company, a business planning organisation, and many others all under one roof.
Designers at WDI can be thought of as falling into one of three general groups based on their overall familiarity with environmentally sensitive design:

1) The first group consists of designers such as architects and mechanical engineers who work in fields for which "best sustainability practices" already exist (e.g. green building design). These designers are already accustomed to considering broadly adopted frameworks such as LEED, ASHRAE, or China's 3-Star system.
2) The second set of designers may be making decisions which are beneficial to life cycle impacts, such as those choosing materials which will last for the greatest amount of time with the lowest amount of required maintenance. This set of designers is already applying limited life cycle thinking without formally and explicitly evaluating those decisions from a life cycle perspective.
3) The third set of designers are interested in safeguarding the environment while adding the greatest value to Disney's product, but are not as familiar with environmental assessment and do not have either the capacity or the experience to perform it.

Because of the variety of backgrounds involved in designing resorts, the ability for every imagineer to evaluate decisions from a life cycle perspective is limited. In addition, because of the number of design decisions that need to be made, the iterative aspect of the creative design process, and the time required for making these assessments, it is impractical to have an LCA and LCC practitioner involved in every decision throughout the design process. Therefore, we must provide a feasible means for every designer to perform some level of evaluation and to determine when further expertise is required.

2.3 Technical boundaries of current design process and tools

Performing life cycle assessment is a time consuming and expensive process. At WDI, designs can change drastically, evolve rapidly, and require decisions to be made quickly. Therefore, providing environmental and financial information for every design iteration across all project components using traditional methods is costly and improbable. Understanding the stage of design at which environmental assessments make the greatest impact is critical in approaching this issue. If the life cycle assessment is conducted too early within the design process, the information may be completely irrelevant by the time the design is completed. On the other hand, if the assessment information is provided too late, the information cannot impact the decision.

It is necessary to define a process for empowering designers to incorporate sustainable solutions and for identifying the top candidates of the project components that will benefit most from a focused life cycle evaluation. This process is outlined in the following section. It is also necessary to provide designers with tools that offer real-time guidance on their design choices so they can independently evaluate less critical project components.

3 Development of a framework to implement life cycle thinking

In order to drive the use of life cycle management forward, WDI has been in the process of developing and implementing several tools and methods for the integration of life cycle information into the design process. To accomplish this, there are two main areas of work. The first focuses on carefully defining the scope and depth of analysis required to find meaningful results. The second is the development of a process and database to identify and evaluate the wide variety of potential sustainability approaches which may benefit projects.

3.1 Technical boundaries of current design process and tools

A key to moving forward on life cycle management thinking is to better define the scope and depth of analysis. For certain applications, such as environmental product declarations (EPD's), there are established and well-documented methods and standards for scope and depth of analysis. For external communication of results, EPD's or ISO 14040 certified LCA's are effective, but also expensive and time consuming to complete. On the other hand, most of the information that WDI needs for making decisions is directional—which option is likely the better choice given many assumptions—rather than explicit—exactly how many tons of carbon dioxide will this option produce over that option.

Because the majority of WDI's projects are highly conceptual, there are not ideal methods to accurately predict use-phase data. One recent assessment analysed several chilled water systems for cooling at the Shanghai resort. In parallel to the cost analysis performed for the proposed alternatives, a screening-level LCA was conducted to study the environmental impacts of each configuration. In this particular assessment, assumptions were made about attendance, average weather conditions, park expansion plans, and cooling loads for a period of twenty years. Any changes in these assumptions had the potential to shift the analysis. Because of the speculative nature of the designed systems, it is not possible to evaluate the system with accuracy, and is imperative to conduct rigorous sensitivity analyses to understand how assumptions shift the conclusions of assessments.

In this context, it is critical to carefully define the scope of studies and determine what is to be included or excluded in the assessment. For most decisions, it is sufficient to complete life cycle assessments at the screening level and only examine components likely to have an impact larger than the margin of error in the analysis. In terms of depth, modelling generic components is preferable to pursuing an assessment on specific models or vendors, as they are likely to change in the intervening years between initial environmental assessment and final installation. For example, the chiller systems study was performed in 2010, but Shanghai Disney Resort is not scheduled to open for at least five years [9]. Therefore, it is important that conclusions are general and directional, rather than absolute, with qualifying statements and assumptions called out explicitly and stated simply.

In another assessment, a design team needed to determine whether to pursue an electrically-powered or fossil fuel-powered boat, and if fuel-powered, which type of fuel was best. The LCC and LCA were performed at a general level. Because the design was still undergoing major changes, the LCA team did not study the entire ride system, only the manufacture and combustion of fuel and the production and consumption of electricity. Recommendations were given for which option would likely be the best in the long run. This provided the team with a general direction to move into the next phase of their design. During the iterative design process, the team can use this single early analysis as directional information in planning the ride propulsion system without requiring a continuous assessment.

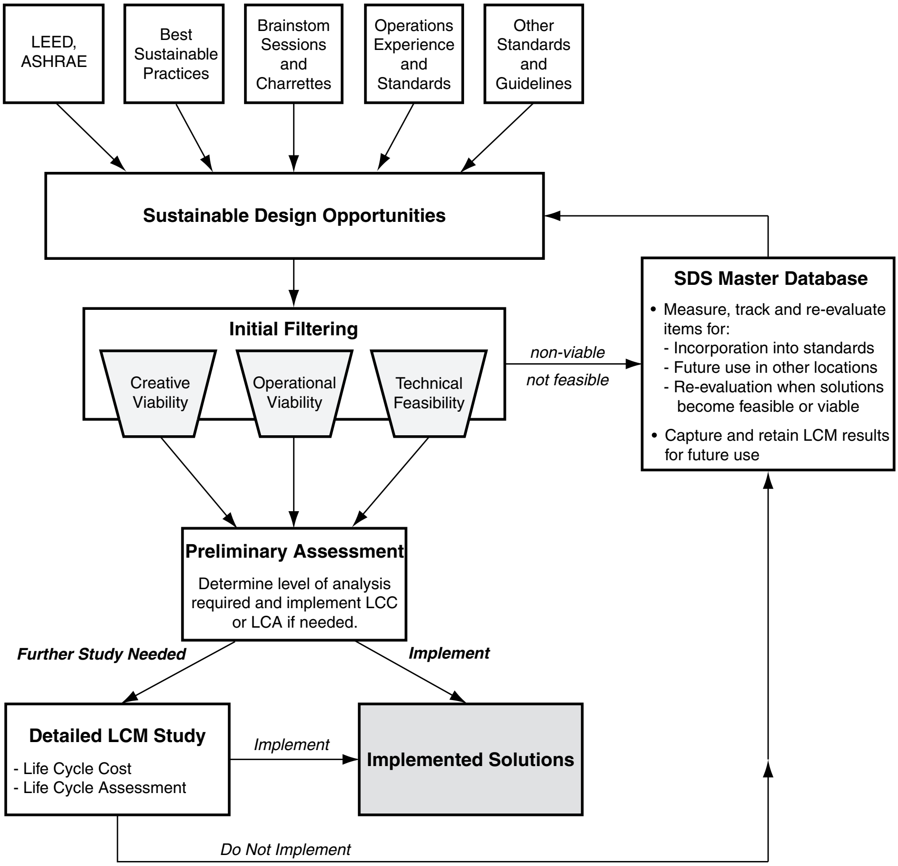

Fig. 1: Conceptual process map for evaluating and implementing sustainable design solutions (SDS) within WDPR

3.2 Encouraging the adoption of sustainable design solutions

The development of general design guidelines, checklists, and standards will help designers who do not have the capacity to make direct use of raw life cycle information. WDI has developed a process and database for evaluating sustainable design solutions (SDS) which is likely to reduce overall environmental impact (e.g. Figure 1). This process has been rolled out and the database was distributed to design teams for consideration and evaluation.

The database was first generated from various other design guidelines such as LEED, ASHRAE, China's 3-Star, Disney's own design and operations standards, and other common approaches to sustainability. The solutions within the database are given to design teams for initial filtering in terms of creative viability, operational viability, and technical feasibility. If a solution will not meet the creative intent, cannot be reliably operated and maintained, or is not technically feasible, the solution is rejected.

After this initial screening, the potential solutions which remain need to be either accepted for implementation or examined more rigorously. For some solutions, such as integrated pest management for Disney's landscaping, there are both environmental and cost benefits in the long run, with virtually no additional initial investment. The effort that would be expended by conducting an LCA and LCC for the idea would likely cost more than the returns generated from the solution itself. For any ideas which are very obviously beneficial, a detailed assessment is not necessary and the idea is implemented.

For larger investments and solutions whose benefits are unclear, some level of LCA and LCC needs to be performed. However, this analysis does not necessarily need to be completed in great detail. As discussed earlier, conducting these assessments at the appropriate level of scope and depth is most effective. Based on the results of these more rigorous assessments, solutions are either accepted for implementation or deferred for this specific application and returned to the larger database.

All of the ideas in the initial database remain within the database whether they are accepted or rejected. Information is maintained on whether each idea was adopted as a new design standard, implemented on a case-by-case basis, or rejected, and the criteria for which it was rejected. This allows designers to understand which design solutions are more viable and why certain approaches have been used or not used in the past.

Furthermore, WDI's operating partners within the resorts are establishing new design standards and specifications for which they have found environmental and economic paybacks. Integrating these findings into the master database reduces the assessment load for designers.

4 Next steps and future work

Future work will centre around two main areas. The first is the development of more comprehensible methods for quantifying and communicating environmental impact information. The second is the development of tools to provide designers with better access to environmental impact information.

Improving the accessibility to and usability of life cycle cost and life cycle assessment information will help TWDC drive for solutions to provide the greatest environmental benefits for the lowest cost. Future work includes developing consensus on a framework and set of impact characterisations to be used consistently across all assessments. These metrics will align closely with TWDC's corporate goals and WDPR's short-term targets, and provide decision makers a more consistent basis for evaluating and funding projects. In addition, having a clear mechanism to provide feedback to designers on the environmental contributions of their decisions will further involve them in the sustainable design process.

In order to make the SDS database more accessible and to solicit input from designers, several technologies are being assessed. In its most basic form, the database is delivered as a spreadsheet detailing the resort areas, corporate goals, and design standards to which a design solution may be relevant. The next version of this database must allow broader access, fast updating, and the immediate integration of designer feedback. A platform being tested for this purpose is Semantic MediaWiki, which allows semantic linkages to be stored within the concepts themselves [10]. Such a platform can enable collaboration amongst designers and facilitate knowledge transfer between projects.

Integrating environmental and cost impacts into design tools, such as architectural design software, can help drive forward life cycle approaches. To date, the design tools used at WDI do not include these components, although limited software add-ons are becoming available which help designers estimate manufacturing impacts. The development of in-house tools and training programs will broaden the knowledge and use of life cycle management across TWDC. Creative design charrettes focused on sustainability have encouraged imaginers to bring the same level of creativity to environmental thinking that they bring to their designs, resulting in new and innovative solutions. Supporting collaboration between life cycle management thinking and creative design will further promote resort designers to make responsible choices and develop products to inspire our guests.

References

[1] The Walt Disney Company Environmentality <http://corporate.disney.go.com/environmentality/press_releases/2007/2007_0416.html> (accessed 14.04.2011)

[2] Dahlsten DL et al. (1995) Parasitoid shows potential for biocontrol of eugenia psyllid. California Agric 49(4):36-40.

[3] Schwegler BR et al. (1995) Biological Activity Monitoring for Real Time Control of Biological Nutrient Removal: A Case Study, Automating to Improve Water Quality. Minneapolis, Minnesota

[4] The Walt Disney Company Announces Significant Long-Term Environmental Goals. <http://corporate.disney. go.com/news/corporate/2009/2009_0309_cr_release.html> (accessed 13.04.2011)

[5] Walt Disney Imagineering Fact Sheet. <http://corporate.disney.go.com /media/news/Fact_WDI_08_10.pdf>, (accessed 14.04.2011)

[6] Reedy Creek Improvement District Comprehensive Plan (2008) Future Land Use Element. <http://www.rcid.org/2008Comp_main.cfm> (accessed 13.04.2011)

[7] Aecom (2009) Themed Entertainment Association Attraction Attendance Report. <http://www.aecom.com/deployedfiles/Internet/Capabilities/2009 Theme Index Final 042710_for screen.pdf> (accessed 13.04.2011)

[8] Walt Disney World Resort Fact Sheet. <http://corporate.disney.go.com/media/news/Fact_WDW_08_16.pdf> (accessed 13.04.2011).

[9] Shanghai Disney Resort. <http://en.shanghaidisneyresort.com.cn/en/> (accessed 14.04.2011)

[10] Semantic MediaWiki. <http://semantic-mediawiki.org> (accessed 14.04.2011)

Greening Events: Waste Reduction Through the Integration of Life Cycle Management into Event Organisation at ESCi

Marta Anglada Roig, Sonia Bautista Ortiz and Pere Fullana i Palmer

Abstract This paper explains the strategy and procedures followed by the 'Greening Events' project to spread green practices in the event organisation at ESCi. The academic years 2008-2009, 2009-2010 and 2010-2011 have been analysed in order to establish a relationship between the kg of waste type produced by person per day. The outcomes will serve as a basis to identify the areas to focus on and the greening principles that need to be implemented before, during and after each event, following a life cycle perspective.

1 Introduction

The organisation of conferences, small and medium-size meetings, workshops and other types of events has become an increasingly important contributor to environmental impact. Universities are no exception, being responsible for holding conferences, lectures, graduation ceremonies and other tutorial activities addressed at both the students and the academic community.

In 1993, ESCi was created as a joint initiative between the *Generalitat* (Government) of Catalonia and Universitat Pompeu Fabra. The founding objective was to prepare students, from within the university environment, for the necessary internationalisation of the Catalan economic and productive fabric. The School is now hosting more than 400 national and international students each year that are coursing studies on International business and marketing studies, and four different kinds of master degrees. The internalisation of the school has also been achieved by hosting the Environmental Management Research Group (GiGa) which has recently been awarded with a UNESCO Chair in Life Cycle and Climate Change, and by hosting different activities organised by private and public companies and organisations.

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ESCi is currently working towards implementing greening principles in the organisation of all its events, following a life cycle perspective. Specifically, this paper presents and discusses the first phase of the 'Greening Events' project, which ESCi is undertaking in collaboration with the UNESCO Chair in Life Cycle and Climate Change.

2 Project description (aim and scope)

The 'Greening Events' project, funded by the Catalan Recycling Centre of the Catalan Waste Agency (ARC), aims at implementing a waste prevention strategy during the preparation and celebration of all the events that take place at ESCi's premises, focusing in all those areas that have a potential impact on waste generation (documentation, publicity, registration, catering, exhibitions and lecturing among others). It is an on-going project that started in September 2010 and will finish in June 2011.

To develop the waste prevention strategy for event organisation, different activities have been carried out with the aim of identifying the critical areas that produce an environmental impact (and therefore the generation of waste), and that will need to have the greening principles implemented. The following sections present in detail the different activities.

2.1 Data collection

A data inventory of all the events held at ESCi between 2008 and 2011 has been analysed. Data compilation has been done using questionnaires that have been developed and adapted according to the type of event, the number of participants attending each event and the type and amount of waste that can be generated at each event type. The final aim of the project is to establish a relationship between the kg of waste type produced by person per day.

The outcomes of this first analysis are key to classifying the different events held, and establishing the areas to focus on and the greening principles that need to be implemented before, during and after each event, following a life cycle perspective.

2.2 Methodology

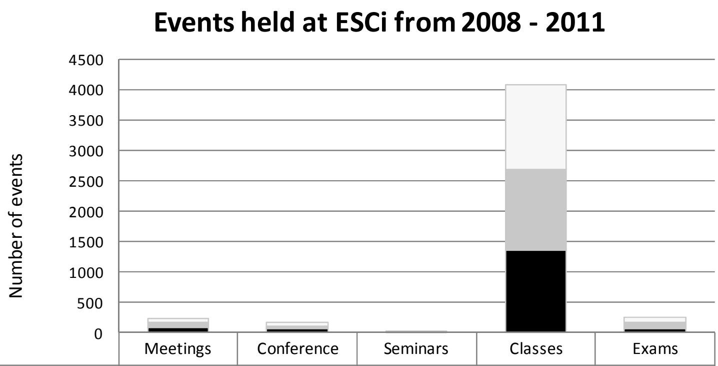
To achieve the objectives of the project, a case study approach was used. The study is structured in three phases: (1) a case study analysis of the 'FENIX: 1st International Conference on Life Cycle Thinking' held in September 2011 with 208 participants, the results of which were used as a basis to extrapolate and calculate the amount of waste that can be produced in the different event types; (2) a cross-case analysis, selecting the most relevant variables, in order to attain general rules and better understand the relation between event type, event participants and type of waste generated; (3) a compilation of the barriers encountered during the preparation of the cases and the way ESCi has tried to overcome them with the aim extracting some greening recommendations that will be included in the final guide. The main event types held at ESCi may be categorised as follows:

- **Meeting:** small- to medium-sized meeting of 2-25 people that can last from 1 hour to one full day. Depending on the duration of the meeting, this can include a coffee break.
- **Conference:** one full-day event of more than 80 participants including a coffee break and a catering.
- **Seminars:** short event of 2-3 hours addressed at either a small or a large group (more than 80 people), but which does not include any coffee break and is usually held in a conference hall or large room.
- **Classes:** bachelor degree classes and master classes.
- **Exams:** short event of 1-2 hours where paper waste is produced.

In order to calculate the amount of waste produced per person in each one of the event categories (waste generation indicator), empirical data obtained from the FENIX conference were used and combined with suitable assumptions in order to extrapolate the results and obtain a participant's 'waste fingerprint' for each event.

3 Preliminary results

The detailed analysis of all the events held at ESCi since 2008 shows that events are organised both by the school itself, and by external actors (mainly private companies and public and private organisations) that hire ESCi's premises for a specific amount of time. Figure 1 presents a breakdown of the different types of events held at ESCi during the academic years 2008-2009, 2009-2010 and 2010-2011.



	Meetings	Conference	Seminars	Classes	Exams
Events academic year 2010-11	66	54	3	1389	73
Events academic year 2009-10	82	57	8	1332	101
Events academic year 2008-09	87	61	3	1349	74

Fig. 1: Number and type of one-day events held at ESCi during the academic years 2008-2009, 2009-2010 and 2010-2011

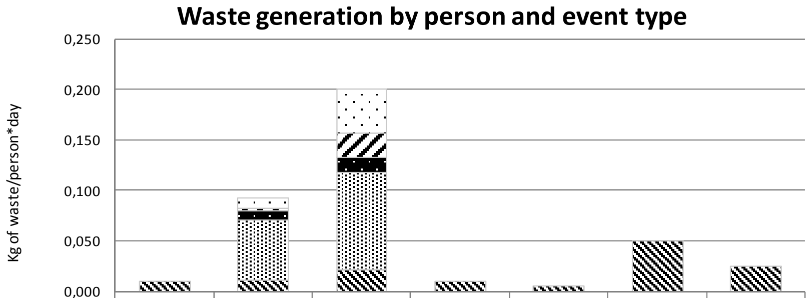
Table 1 presents the assumptions and empirical data used to calculate the waste generation indicator for each event type.

Tab. 1: Assumptions and empirical data collected per event type

	Assumption				g waste/ (pers.*day)
Regular meetings	It is assumed that one person uses two sheets of paper per meeting of A4 format, considering that one A4 sheet weighs 4.98 g.				4.98*2 = 9.96
Exams	It is assumed that an average of 5 A4 sheets is used in each exam.				4.98*5 =24.9
Bachelor degree class	Classes are mainly given with electronic material, and information is exchanged online. It is therefore assumed that only one A4 sheet is used per person per class.				4.98*1 = 4.98
Master class	Master classes produce a bigger amount of paper waste and it is assumed that ten A4 sheets are used per person per class.				4.98*10 = 49.8
Coffee break [1]	g paper/ pers.	g glass/pers.	g plastic/ pers.	g organic waste/pers.	g general waste/pers.
	0.962	9.96	8.65	2.40	9.62
Catering [2]	g paper/ pers.	g glass/pers.	g plastic/pers.	g organic waste/ pers.	g general waste/ pers.
	10.1	37.0	5.80	21.2	34.62

[1, 2] Based on the results obtained in the case study of the FENIX: 1st International Conference on Life Cycle Thinking held at ESCi in September 2010.

On the basis of the data in Table 1, the waste generation indicator [kg/(person*day)] was calculated for each type of event. Figure 2 presents such results.



	Meeting without coffee break	Meeting with coffee break	1-day conference	Seminars	Bachelor Degree class	Master class	Exams
Kg general waste/pers.	0	0,010	0,044	0	0	0	0
Kg organic waste/pers.	0	0,002	0,024	0	0	0	0
Kg plastic/pers.	0	0,009	0,014	0	0	0	0
Kg glass/pers.	0	0,060	0,097	0	0	0	0
Kg paper/pers.	0,010	0,011	0,021	0,010	0,005	0,050	0,025

Fig. 2: Waste generation by person and event type

Figure 2 shows how the different events have as a common denominator, namely the generation of paper waste as the most relevant waste type. Other kinds of waste are mainly produced during the coffee breaks and luncheons organised in larger meetings such as seminars and conferences. As a result, actions must be taken towards:

- Minimising the paper waste produced per person in those events that do not include either a coffee break nor a catering (meetings without coffee breaks, seminars, bachelor degree classes, master classes and exams), and
- Minimising the generated waste in the coffee breaks and caterings (meetings with coffee break and 1-day conference), with a special focus on reducing the amount of glass and general waste produced per person.

There is a significant difference between the paper waste generated in the Bachelor degree class and the Master class, where the amount of paper is ten times bigger. This is because the material used in the Bachelor degree classes is mainly given in electronic format in comparison to the Master classes. It is therefore

important to change practices and walk towards the teaching methods used in the Bachelor degree classes.
Regarding the glass waste generated as part of the coffee break and catering, it is mainly caused because of the use of single 330 mL bottles for juices and soft drinks. The use of draft drinks served in pitchers or larger bottles are to be recommended, as a prevention method to minimise waste generation.

4 Next steps and expected results

The next steps to be undertaken before the project ends are the integration of the life cycle perspective into event organisation, with the aim of minimising waste generation. The life cycle approach seeks to identify possible improvements to goods and services in the form of lower environmental impacts and reduced use of resources across all life cycle stages.
The life cycle approach will serve as a decision tool to establish the most suitable solution to reduce waste generation according to the event type, type of participant and type of service offered. A life cycle analysis will be performed during the second phase of the project, with the GaBi software package (GaBi 2007a), and the bundled professional database (GaBi 2007b) as the principal source of background data. This analysis will help the event organisers decide which is the best option to be used during the catering for and communication of the event, in order to minimise amount of waste generated. For example, the use of biodegradable cups and reusable cups for the catering will be compared.
The conclusions of the project will be presented in the form of a practical guide with recommendations and examples of best practice cases studies to help reduce the environmental impacts that are generated during the celebration of an event at ESCi. These recommendations, applicable for all the event types described here, will focus on minimising the main environmental impacts caused by the generation of waste. The guide will give practical guidance on which products should be purchased to reduce the impact, and which practices should be followed before, during and after the organisation of an event. Other small-scale events, such as teaching classes, will also be addressed specifically, as these are seldom treated in the available literature on greening events.
To complement this practical guide, communication and training activities on how to introduce the life cycle perspective into event organisation are also foreseen in the project. These activities will be specifically addressed at ESCi's staff and the school alumni.

5 Conclusions

According to the results obtained in this first part of the study it cannot be said that a waste prevention strategy in event organisation exists yet at ESCi. However, actions will be implemented in the second phase of this project to develop and implement greening event principles, according to the event types held at ESCi.
In order to succeed in minimising the generation of waste, detailed actions will need to be implemented. Some examples are the printing practices in the Master classes, and the minimisation of packaging waste used during the coffee breaks and the catering served.

References

[1] Anglada M, Clement S, Schraffl F, Zimmermann M (2009) Green Meeting Guide 2009, 1st Ed. United Nations Environment Programme

[2] Guinée J et al. (2001) Life Cycle Asessment. An operational guide to the ISO standards". Centre of Environmental Science – Leiden University (CML)

[3] UNE-EN ISO 14040:2006. Environmental management. Life cycle assessment -Principles and framework.

[4] UNE EN ISO 14043:2001. Gestión medioambiental. Análisis del ciclo de vida. Requisitos y directrices. AENOR, Madrid

Challenges for LCAs of Complex Systems: The Case of a Large-Scale Precious Metal Refinery Plant

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Abstract Umicore Precious Metal Refining (UPMR) runs a high-tech industrial metal refinery which recovers 17 different metals from end-of-life consumer products and from by-products of the non-ferrous industry. We present an approach for an attributive gate-to-gate LCA study of this system, which is characterised by multi-input/multi-output processes, changing feed compositions and time lags. We propose five assumptions to reduce the complexity of the highly dynamic system. We compiled inventory data for over thirty sub-processes and allocated it over the metals passing the sub-process by either a mass-based or metal revenue based allocation. The exemplary results for rhodium, platinum, tellurium and copper (impact assessment method: global warming potential) show a high dependence of allocation choice and different patterns of the metals for metal revenue based allocation due to the high volatility of prices.

1 Introduction

Fostering the use of more sustainable products can help to reduce the environmental footprint of our daily living. Life cycle assessment (LCA) studies can support the development of such products by quantifying environmental impacts of a product over its full life cycle. The life cycle includes resource provision, manufacturing and use, as well as end-of-life treatment, which can either comprise waste management or reuse respectively recycling. The supply of metals as part of the resource provision in a product's life cycle includes the refining of either primary material or secondary materials (production scrap and end-of-life products). In this contribution we present an approach for performing an LCA study of the process of a large metal refining company, Umicore Precious

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Metals Refining (UPMR), which produces - amongst others - precious metals (e.g. platinum group metals) and special metals (e.g. indium and tellurium) from end-of-life consumer products and from by-products of the non-ferrous industry. A major challenge of this project was to master the fact that the life cycles of these metals are characterised by strong interlinkages of metal streams. This holds true for the primary production, as these metals often are by-products of major metals, but also for secondary production since these metals typically occur in complex mixtures in end-of-life products [1]. Hence, metal refining mostly involves several metals at a time and requires efficient separation and refining processes to both close the resource cycle and minimise environmental impacts early in the supply chain.

2 Methodology

2.1 Goal and scope

2.1.1 Objective of the study

We present an approach to quantify the environmental impacts of 17 metals refined at the facility of UPMR in Hoboken, Belgium, from a gate-to-gate perspective. The Hoboken system consists of strongly interlinked sub-processes (in particular smelter, blast furnace and refinery steps, see Figure 1), which have a multi-input/multi-output character, internal loops, changing feed compositions and time lags. This paper aims at i) explaining the overall approach, which led to an Excel tool that can be linked to the existing monitoring system of the company, and ii) highlighting how methodological choices, such as allocation rules, affect the results. We do not focus on individual life cycle inventories of specific metals, but show exemplary results for the metals platinum, rhodium, tellurium and copper for one impact assessment method. Further results will be shown in a forthcoming publication. The approach developed in this project can serve as a first step towards a comparative study with other primary or secondary producers.

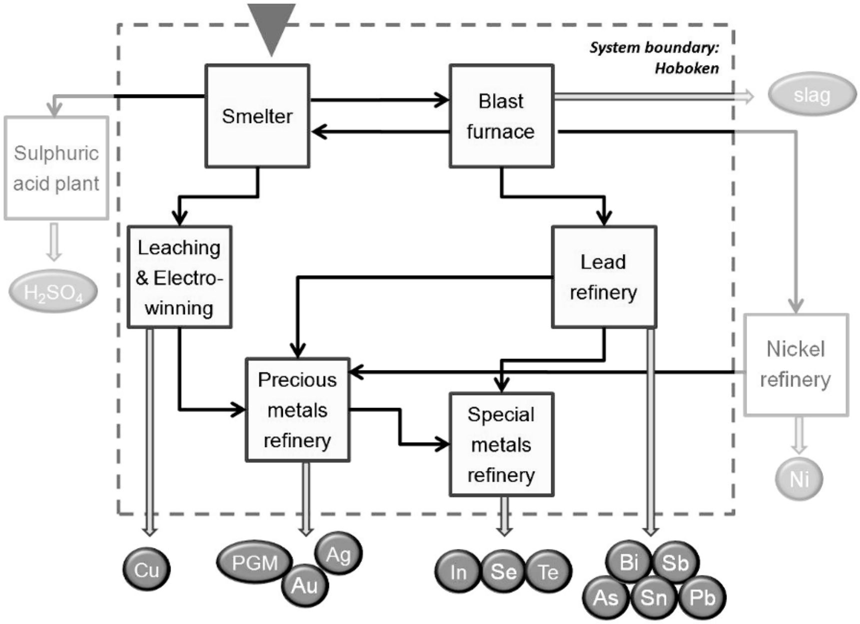

Fig. 1: The Hoboken metal refinery system (adapted from [2]). The triangle indicates the main entry point for feed material. Note that the number of sub-processes and arrows are not complete.

2.1.2 System boundaries and data sources

System boundaries for the foreground system modelled in this study were defined by the Hoboken site, i.e. all process steps that are related to metal recovery at the Hoboken plant were included, except sampling and assaying (the analytics necessary before a feed material is accepted by UPMR). The two non-metal products of the system, sulfuric acid and slag (sold to cement producers), are excluded due to their minor importance (cut-off). The process flow chart (see Figure 1) was broken down into over thirty sub-processes for which data on material and energy inputs was available from the internal reporting system of the company. Data on outputs, that is emissions and waste streams, was only available on a coarser scale, therefore these measurements had to be reallocated to the sub-processes based on the knowledge of the responsible process engineers. The annual metal output per sub-process is monitored by UPMR and internal material loops were calculated based on transfer coefficients available from UPMR's metal flow model. The gate-to-gate standpoint of this study implies that the feed material, which could be secondary material as well as valuable by-products of primary metal production, was not accounted for. In other words: the feed was assumed to be "free" of environmental impacts. We accounted for the metal product quality as it leaves the Hoboken site in a marketable product. As Table 1 shows, this can either be a pure refined metal or an intermediate product that can be further refined elsewhere. The background system (e.g. the supply chains for materials and energy provision) was modelled by using the Ecoinvent database v.2.2 [3]. For some internal processes taking place at the Hoboken site, which provide auxiliary process inputs, we compiled own inventories.

Tab. 1: Metals processed at Hoboken and grade of the product leaving the plant (+intermediate products, either further refined elsewhere or directly sold on the market; *produced as technical grade and high purity products, for the analysis it was theoretically assumed that all is technical grade; **leaves the plant as refined product and as an alloy with Bi; ++not available)

Metal	Grade [%]	Metal	Grade [%]
Precious metals			
1. Platinum (Pt)	99.95	5. Ruthenium (Ru)	+
2. Palladium (Pd)	99.95	6.Gold (Au)	99.99
3. Rhodium (Rh)	99.95	7. Silver (Ag)	99.99
4. Iridium (Ir)	+		
Special metals			
8. Antimony (Sb) - sodium heyhydroxo antimonite	+	11. Selenium (Se)*	99.50
9. Bismuth (Bi) - in Pb alloy	+	12.Tellurium (Te)	99.50
10. Indium (In)	99.99		
Other metals			
13. Lead (Pb)**	99.97	15: Tin (Sn) - calcium stannate	+
14. Copper (Cu) – cathode	NA++	16. Nickel (Ni)	+
17. Arsenic (As)	+		

2.1.3 Functional units

The functional units (FU) of this study refer to 1 kilogram specific *metal content* leaving the Hoboken system, in a grade and specification as indicated in Table 1. Our approach only accounts for metals in the system, while other material flows, such as those forming compounds, are neglected.

2.1.4 Allocation rules

Inside the Hoboken system, allocation rules need to be defined since the metals take different routes through the system and occur in different combinations in each sub-process. We present results for different allocation rationales that are based on:

- Mass (kilogram), i.e. physical allocation
- Value (US Dollar) of fully refined metal, i.e. revenue/price allocation, with
    - average prices between 2000 and 2010
    - average prices in 2000
    - average prices in 2010

For the revenue allocation ("revenue" = price of the metal times its amount), we used the prices of the "pure" metal, even though some metals leave the plant as intermediate product (see Table 1), as the prices of the intermediates were not possible to determine. Since metal prices showed high volatility in recent years, we tested the robustness of revenue allocation by calculating it with i) a ten year average, ii) with the price before the metal boom and iii) with the price during/after the metal boom . The results can be further validated by calculating all allocations for two years; with mass flow data from 2008 and 2009.

2.2 Life cycle inventories (LCI)

2.2.1 Quantification of inventory data per sub-process

In this first step, an inventory for each sub-process in the foreground system was established. *Inventory data* includes the use of materials and energy and the discharge of waste and emissions.

The internal reporting system of UPMR accounts for the use of energy and auxiliary materials per process unit. In our study we calculated with over thirty sub-processes. Data on emissions and waste streams was available on a coarser scale, for instance when several processes use the same chimney, and had to be allocated to the sub-processes using a combination of judgments by engineers and cost allocation models.

For instance, emissions to water were allocated to waste water streams that are piped to the waste water treatment plant (WWTP) based on judgments of the

process engineers. The waste water streams, in turn, were allocated to sub-processes based on an internal cost allocation model. This cost allocation model was also the basis to distribute material and energy inputs as well as outputs (emissions and waste) of the WWTP to the sub-processes.
For some internally produced materials (different process water qualities, pressurised air and steam) we compiled inventories for producing one unit based on data from the plants providing the material. These plants take advantage from waste heat produced elsewhere in the Hoboken system, making the production process more economic. For sulphuric acid that is produced from SO_2 in the smelter off gas, we did not provide an own inventory, since i) only a small fraction is used internally and most sulphuric acid is sold on the market and ii) the production process does not deviate from industrial standards.

2.2.2 Distribution of inventory data over metals in sub-process

In the second step, the inventory data of the sub-processes was allocated to the metals passing that sub-process. The number of metals using a sub-process can differ: While the smelter is part of every recovery process as all metals pass through it, sub-processes in the precious metals refinery for instance can be part of only one or few metal recovery processes. Since the metals' retention times in the system and their annual share in the feed vary strongly, one major challenge of this project was to answer the question, how annually reported inventory data can be allocated to single metals in a sub-process.
In our approach we reduced the complexity of the highly dynamic system by applying the following three key assumptions:

1) The use of energy and auxiliary material depends on the *quantity* of metal handled in the sub-process, while differences in the *type* of metal (their *"quality"*), are neglected (for instance could some metals be more demanding to be treated in a process than others);
2) The amount of material and energy needed per sub-process and ton of metal handled stays constant over time;
3) Each process is short in time.

As a consequence, we first looked at each sub-process *n* separately and distributed the inventory data *m* over the metals *i* handled in that sub-process based on their share on the total metal stream (Equation 1, see Figure 2 for explanation of terms).

$$Inv_{n,m,i} = \frac{In-flow_{n,i}}{In-flow_{n,tot}} \cdot Inv_{n,m} \quad (1)$$

The annual input flow of metal *i* in sub-process *n* (*In-flow_n,i*) is calculated by dividing the annual output of metal *i*, which is regularly reported for each sub-process *n* (*Out-flow_n,i*), by its yield of the whole process (Equation 2). We use data on inputs instead of outputs in order to account for internal losses.

$$In-flow_{n,i} = \frac{Out-flow_{n,i}}{yield_{n,i}} \tag{2}$$

The total metal input flow for the sub-process *n* (*In-flow_n,tot*) is the sum of the flows of each metal (Equation 3)

$$In-flow_{n,tot} = \sum_{i=1}^{17} In-flow_{n,i} \tag{3}$$

For the metal revenue based allocation, the physical metal flows were translated into a money flow by multiplying them with their price per mass unit. The metal stream includes internal loops (the metals can pass a sub-process several times).

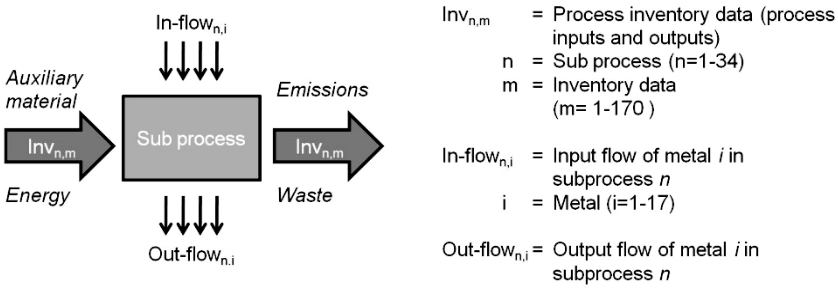

Fig. 2: Schematic view on one sub-process. A sub-process handles one to all metals onto which its use of auxiliary materials and energy and its generation of waste and emissions need to be allocated.

2.2.3 Compilation of life cycle inventories per metal

In the third step, an inventory per metal for the whole Hoboken plant (*Inv_Hoboken,m,i*) was calculated.

The inventory data per sub-process (*Inv_n,m,i*) was divided over the metals going through that sub-process according to their share on total metal stream, which included internal loops (see Equation 1). In order to compile a complete inventory for each metal in the whole Hoboken plant for the functional unit of one kilogram specific metal content in the product, the reference of each sub-process needed to be changed to the metal input *excluding* internal loops. The resulting inventories for metal *i* per sub-process could then be summed up and form the complete inventory of metal *i* in the Hoboken plant.

A metal's throughput generated in sub-process *n* by the input of 1 kilogram of the metal in the smelter was known from UPMR's "metal studies", which are internally prepared models on metal flows that are based on measurements of how the outflow of a metal from a sub-process divides over further sub-processes. With these "split factors" (e.g. 90% of metal *i* leaves the smelter to sub-process *n1* and 10% to sub-process *n2*) for every metal and every sub-process it was possible to model how many times a metal in the annual feed mix entering at the smelter circulates in the plant before it leaves Hoboken. The calculation of the total Life Cycle Inventory of 1 kilogram metal *i* in the Hoboken plant is shown in Equation 4.

$$Inv_{Hoboken,m,i} = \sum_{n=1}^{34} inv_{n,m,i} \times \frac{In-flow_{n,i}}{In-flow_{n=smelter,i}} \qquad (4)$$

This proceeding implies that two more assumptions have to be added to the three mentioned in chapter 2.2.2:

4) When calculating the number internal loops for one metal in one sub-process, we used a theoretical model that is based on the flow of the annual feed composition through the Hoboken system, assuming that all input material enters at the smelter. In reality, feed material that is already similar to the material mix of another sub-process can enter later in the process, which is not possible to account for in our approach. Assuming the smelter to be the single entry point is close to reality, but neglects that some feed is treated more efficiently.
5) We used the average annual feed composition when calculating the number of internal loops; however, since the retention time in the system varies per metal as well as their annual share in feed material, metals that enter the system in the last month of year y will leave the system at some moment in year y + 1. This implies that we assume that the share of a metal in the feed (which can vary between years) does not affect the numbers of its internal loops.

2.3 Integration with life cycle impact assessment (LCIA)

By following the steps outlined before, we obtained a list of inputs (material and energy) and outputs (emissions and waste), which were related to one kilogram of metal *i* refined in the Hoboken plant.

The environmental impacts of these background materials and services were quantified by using the Ecoinvent database (v2.2) as implemented in Simapro.

We chose different impact assessment methods but focus here on the global warming potential, applying a time frame of 100 years (GWP) [4]. Results for other impact assessment methods will be published in due time.

3 Results

The choice of the allocation rationale determines how much environmental impact is owed to one kilogram of a specific metal present in the Hoboken plant. This is illustrated in Figure 3 for the impact assessment method GWP and four selected metals:

- Platinum, the probably best known and most used platinum group metal;
- Rhodium, the most expensive of all metals refined in Hoboken;
- Tellurium, used in thin-film photovoltaic;
- Copper, one of the carrier metals, with a relatively low price, present in high quantities while it is not the main focus of UPMR's business.

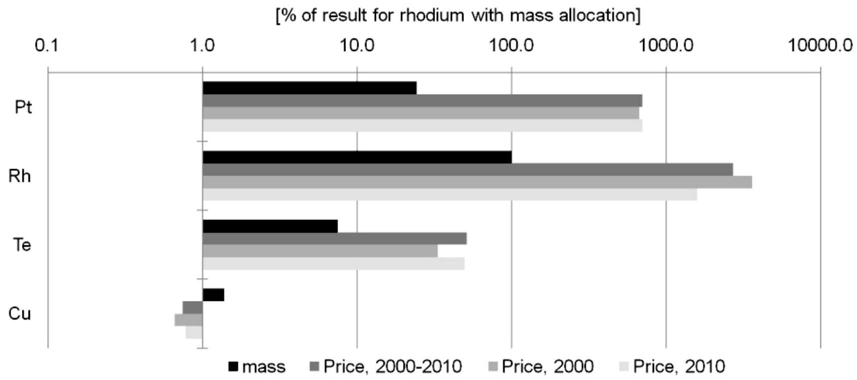

Fig. 3: Results for material flow data from 2009. The bars refer to different allocation methods (black: mass allocation; grey tones: revenue allocation, with average price of metal between 2000-2010, average price of metal in year 2000 and average price of metal in year 2010). The values are given as percentage of the results for rhodium, mass allocation (in logarithmic scale). Note that the results presented here are preliminary, since they do not include solid waste and a last check of some numbers is lacking.

The decision on allocation rationales has a significant impact on the results of each metal, as Figure 3 shows. Depending on a metal's price, the impacts can either be significantly higher (e.g. by more than an order of magnitude for rhodium) or lower (e.g. for copper) compared to a purely mass based allocation. Rhodium is the most expensive precious metal with an average price of about 85'000$ per kilogram between 2000 and 2010, which is by several orders of magnitude higher than the price for a kilogram of copper (average price between 2000 and 2010: 4.25$ per kilogram). Platinum and tellurium are in-between with an average price of about 32'000$ and 140$ per kilogram, respectively. The three metal revenue based allocations depicted in Figure 3 show different patterns for each metal, while the ranking remains stable for the four metals chosen. In the first decade of the 21st century, the metal market experienced high volatility, which, however, did not affect all metals similarly. Rhodium, for instance, had a price of about 17'000$ per kilogram in 2003 and of 211'000$ in 2008, which dropped down to 79'000$ in 2010. The price curve for platinum is smoother, which is reflected by the smaller variations between the revenue allocations. However, the sensitivity towards choosing a reference year for metal prices can not only be explained by the volatility of the price of the respective metal. In addition, the share of the value of a specific metal in the total value of metals using the same process is relevant. Therefore, the effect of considerable price differences diminishes later in the refinery steps, where only few metals are still involved in the same process.

4 Discussion

We listed five assumptions underlying our approach in section 2.2.2 and 2.2.3. The first assumption referring to differences in a metal's quality is probably the most relevant. Including rather than omitting this aspect would imply to look at thermodynamic characteristics of the metals and their compounds and probably also to distinguish their composition in the feed. However, due to the high resolution of more than thirty sub-processes considered in our approach, we can capture the different investments for refining specific metals by i) only allocating process impacts to the metals actually demanding it, and, related to that, ii) accounting separately for the last refining steps of most metals (which are in general the most "expensive") and iii) including the amount of internal loops of a metal. The second assumption implies that there is no "base load" of the sub-processes; however it is reasonable to assume that UPMR is trying to keep the load constant for a high efficiency, which can be validated by cross-checking with different years (provided that no installation of new sub-processes has changed the process flow chart). The second assumption together with the third is the basis for our approach that avoids a detailed representation of the highly dynamic character of the process by focusing on sub-processes, where the factor time is less relevant (the factor time includes different retention times from several weeks to a number of months and changes in annual feed composition).

The exemplary results for four metals and one impact assessment method showed that choosing between a purely mass-based allocation and a metal revenue-based allocation has a strong effect on how the impacts of the large refinery plant of UPMR are distributed over the metals. The rationale of the mass-based allocation is that all metals refined at Hoboken are considered equally important as drivers for the process. For the metal revenue allocation, the rationale is that the process is driven by the more economically valuable elements, which is affected by the volatility of metal prices. However, a short term reaction on fluctuations in metal prices is hardly possible for UPMR, since i) the feed spends a certain time in the system, ii) feed material is secured for longer time in the future, iii) metal prices are hedged (i.e. fixed when the material arrives at the plant, for payment in the future), and iv) the large infrastructure can only be adapted in the medium and long term.

When considering UPMR's business model in more detail, it appears that both allocation rationales cannot fully account for its specific incentives:

In metals refining different metals have to be handled at the same time because they are fed to the plant as metals mixtures in the different feed materials. This means that not only the metals as such drive the process, but also the type of feed, which ranges from by-products of the non-ferrous industry to e-scrap and automotive catalysts. The business model of UPMR is to offer clients to refine metals from feed they provide. The client receives the refined metal and pays Umicore a treatment fee for this service. From this point of view, all metals can in theory equally contribute to the company's revenue. In that sense, treatment fees are more relevant as driver of the process than metal prices. Furthermore, some interdependencies in the process are not considered by the allocation rationales, as some metals fulfil a function as carrier metal in the process, which makes their presence relevant for recovery processes of other metals.

5 Conclusion and outlook

Our study provides insights into environmental impacts early in the product supply chain by focusing on the process of a large scale, industrialised metal refinery. In particular, it shows how 17 metals entering the same refinery plant differ in their environmental impacts, and how sensitive the analysis is towards mass-based versus metal revenue based allocation. When interpreting the results of this study, it has to be kept in mind that they are only valid for the specific process design and product portfolio of UPMR, that is: the process is optimised for a feed mix and not for a specific metal. Furthermore, the study only refers to a specific average annual feed composition at UPMR and cannot be generalised for primary and secondary sources (as both types of feed enter the process).

The current work shows a large number of academic and practical challenges that are associated with obtaining reliable LCA data for large, complex metallurgical operations. Using LCA as a tool to quantify the performance of recycling processes is important and necessary to move towards a sustainable society. Nevertheless, it needs to be kept in mind that the quality of the inventory data has to be good, and the limitations of the model have to be recognised to make well-thought decisions in the operational and policy area.

For future work it is necessary to include further specific characteristics and dynamics of the complex metallurgical processes in the approach. More specifically, it would be helpful to include a differentiation of input material (and e.g. take into account different heating values), as well as distinguishing between the metals produced from a specific feed. One option would be to change the research question to which environmental burdens and benefits occur from treating a certain feed, with e.g. CO_2 being a burden and platinum recovery being a benefit. In that case it would also be possible to allocate environmental burdens of the treatment process to the life cycle of producing the feed material.

References

[1] Hagelüken C, Meskers CEM (2010) Complex life cycles of precious and special metals. In: Graedel T, Van der Voet E (eds) Linkages of Sustainability, 1st edn. MIT Press

[2] <http://www.preciousmetals.umicore.com/PMR/Process/> (accessed 14.04.2010)

[3] <http://www.ecoinvent.ch/> (accessed 14.04.2010)

[4] IPCC (2007) IPPC Climate Change 2007: Synthesis Report, Cambridge University Press

Life Cycle Inventory of Pine and Eucalyptus Cellulose Production in Chile: Effect of Process Modifications

Patricia González, Mabel Vega and Claudio Zaror

Abstract This work reports the life cycle inventory (LCI) of cellulose production in Chile, following a cradle-to-gate approach. Primary data have been used in this study, and cover 100% of pine and eucalyptus cellulose production capacity in Chile. Results show that pine based cellulose presents greater chemical and environmental loads than eucalyptus. Most fossil energy consumption takes place in raw materials transport, chemical manufacturing and limestone kilns in cellulose plants. Effluent discharges are associated with bleaching operations. New forestry practices and pulping and bleaching processes, introduced during the last decade, has led to significant reductions in water consumption, pollutant discharges, and air emissions. However, despite the sharp reduction in chemicals consumption due to process improvements and new technology, greater associated GHG emissions have been recorded as a result of the significant increase in the share of thermoelectric generation experienced during the last decade in Chile.

1 Introduction

Chile is a major exporter of bleached kraft cellulose, with an annual production around 5,000,000 ton per year, supported by more than 2 million hectare of pine and eucalyptus plantations. Forestry and industrial wood processing features large amounts of water and chemicals consumption, as well as significant effluent discharges and air emissions, and occupational risks. Since early 1990s, Chilean cellulose production capacity has doubled, leading to growing concerns about potential environmental impacts. Thus, industry has been under increasing pressure to improve process performance, according to modern standards. As a result, thorough technological upgrading has taken place in the last decade, and new mills feature state of the art processes. Indeed, all plants now feature elementary chlorine free (ECF) bleaching sequences, secondary effluent treatment and air emission abatement technologies.

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Over 90% of cellulose plants are located within the Bio Bio region in southern Chile, where most forestry plantations are located [1]. Climate change has significantly affected rainfall, reducing water availability for industrial purposes. Moreover, nowadays water is a key issue for future development of cellulose production in the country. Additionally, reduction in hydroelectric capacity has led to massive introduction of thermoelectric power generation, with a considerable increase in greenhouse gases (GHG) emissions associated with electricity production. Within this context, different initiatives to reduce water and carbon footprints, effluent toxicity, and other environmental and occupational hazards along the cellulose value chain have been developed. This work presents the effect of process modifications on the environmental loads associated with cellulose production in Chile, following a life cycle approach.

2 Methodology

This works has been conducted within the ISO 14040-2006 methodological framework [2]. This LCI followed a cradle-to-gate approach, with the system boundaries set at the nursery where selected seedlings are grown, and at Talcahuano port (Bio Bio Region), where Chilean cellulose is shipped to international markets.
Primary data from 7 bleached kraft cellulose plants were obtained for the period 2008-2009, accounting for 100% of beached cellulose production in Chile. Data included raw materials, energy and chemicals consumption, air emissions, water discharges and solid waste.
Electricity LCI (central interconnected system, SIC) was considered taking 2008 as a reference year. Data from the National Electricity Distribution System (CEDEC), National Energy Commission, and yearly reports from main electricity generation and distribution companies were used to determine the electricity LCI. Additionally, data from mills featuring chorine based bleaching sequences, typical of Chilean mills during the early 1990's, was also collected from historical records corresponding to three representative local plants [3]. The corresponding Chilean electricity LCI was also estimated for that reference year.

3 Results

The main processes included in the production system are described below.

3.1 Process description

3.1.1 Forestry subsystem

Nursing stage: Good seedling quality is a fundamental requirement to achieve acceptable forestry productivity. Current nursing practices in Chile include careful site selection, optimal nutrient addition, genetic improvement, disease control, controlled irrigation, and careful handling. Since *Eucalyptus globulus* is sensitive to low temperature, covered root methods are used, with 300 plants/m^2. In the case of *Pinus radiata*, bare-root procedures are also found, with 200 plants/m^2.
Establishment of plantation: This stage is highly critical, since plants are transported from the protected nursing environment to the plantation site, where they will be subjected to environmental stress and competition from other species. The prospective site has to be thoroughly cleaned using manual, mechanical and/or chemical means to remove existing vegetation. In some cases, controlled burning is used to eliminate those residues. Deep ripping on compacted soils is usually carried out in order to improve the soil physical properties, and water retention capacity. Planting is carried out during winter, and beginning of spring. Plant density is selected according to the type of final product, and site quality. In Chile, the average initial stocking is around 1,600 trees/ha, both for pine and eucalyptus plantations. In some cases, preventive fertilisation is conducted when required. Post-plantation chemical weed control is carried out in spring. During the first few years, competition by other species, and pests should be kept under control, since small tress may not be able to survive under those conditions.
Forest management: This includes all activities carried out between the establishments of plantation and harvesting. Management practices depend on product objectives [4]. In Chile, pine plantations are managed to obtain knot-free wood, timber, and pulpwood. On the other hand, most eucalyptus plantations are aimed at producing pulpwood, with negligible uses in plywood, and board making. Pruning is necessary to produce knot-free timber. Thinning is carried out to stimulate growth by regulating stand density. In the case of pine plantations, non-commercial thinning is practiced at about 4-5 years, whereas commercial thinning is carried out at 7-14 years, mostly for pulp making, reaching a final density around 300-500 trees/ha. Depending on management, soil quality, and climate, 30-80 m^3/ha pulpwood could be obtained from commercial thinning.
Harvesting: Chilean *Pinus radiata* and *Eucalyptus globulus* harvest cycles are 20 and 10 years, respectively. Harvesting comprises felling, delimbing, bucking, transport to roadside, loading, and hauling. A wide range of techniques, and equipments are currently used. Intermediate processing operations, such as

delimbing, bucking, or chipping, can take place at the stump, at roadside, or at the mill. Transport to roadside could be accomplished using animals or tractors, transported in the bunk of an off-road vehicle, or moved by cable. Depending on forest management practices, and soil quality, around 200-500 m^3/ha and 200-300 m^3/ha are harvested, for pinewood and eucalyptus, respectively. All major forestry companies maintain active reforestation programs, to keep a reliable biomass stock for long term industrial processing, and sustainability. Reforestation of harvested plots is conducted after appropriate site preparation.

3.1.2 Transportation subsystem

Trucks (10-25 ton) are used for woody raw materials transport, whereas most chemicals and cellulose products are transported by rail. The distance between plantations and industrial sites ranges within 50-150 km, whereas most bulk chemicals (eg. sodium hydroxide and chlorate) are produced at industrial states in Talcahuano (Bio Bio region), located at 100-150 km from mills.

3.1.3 Pulp production

All leading Chilean companies feature a close integration between forestry and industrial processing. Around 50% harvested pinewood goes directly as a raw material for pulping, while the rest is sent to sawmills. In turn, sawmills generate pulping residues that represent 30% of total input. On the other hand, most harvested eucalyptus wood is used for pulping, with a negligible amount destined to board manufacturing.

Wood preparation: This includes wood debarking, and chipping. Chips are then screened, and stored. These operations involve considerable electrical power consumption. In modern plants, bark and other woody residues are used as fuel for steam and electricity production.

Digesting: Chips are treated at high temperature and pressure (160-180°C), with a NaOH and Na^2S solution at high pH (white liquor). This chemical attack leads to the dissolution of around 52-54 % and 46-48 % of initial wood, for pine and eucalyptus, respectively. Traditional cooking for pine achieved a kappa number around 30 and 25, for pine and eucalyptus pulp, respectively. Modern cellulose mills included in this study feature extended cooking, obtaining a crude pulp with kappa values around 24 and 14, for pine and eucalyptus, respectively. After cooking, crude fibres are separated from the black liquor, and washed to remove residual chemicals.

Oxygen delignification: This represents one of the major technological improvements. Modern plants incorporate an oxygen delignification stage, under alkaline conditions, which reduces lignin content by around 50-60%, to reach a kappa value in the range 11-13 and 6-8, for pine and eucalyptus, respectively.

Energy and chemicals recovery systems: The black liquor has a high calorific potential and contains most of the digestion chemicals. The black liquor is concentrated using evaporators, and burnt in the recovery boiler, where all organic matter is oxidised to CO_2. The solid residue (Na_2CO_3 and Na_2S) is dissolved and mixed with CaO to regenerate the "white liquor" for digestion, leaving Na_2CO_3 as a residue. In turn, the CaO is regenerated by thermal treatment of Na_2CO_3 in a lime kiln.

Bleaching: Chileans plants move from chlorine-based sequences in the early 90's, to elementary chlorine free (ECF) bleaching sequences. These feature the use of ClO_2 as oxidation agent (D), and oxidative alkaline extraction, with O_2 and H_2O_2 (EOP). In the case of pine pulp, a $D_0E_{OP}D_1D_2$ sequence is normally used, whereas a shorter sequence ($D_0E_{OP}D_1$) is used for eucalyptus pulp. Liquid waste from bleaching contains chlorine compounds that are not suitable for burning in the recovery boiler due to potential corrosive effects. These effluents account for most of the contaminant load of final waste waters.

ClO_2 in-plant generation: All surveyed plants generate ClO_2 by reduction of $NaClO_3$ under acid conditions (i.e. in the presence of H_2SO_4), using CH_3OH as the reducing agent. It is interesting to note that $NaClO_3$ is produced by a local electro-chemical plant, that also generates NaOH, using NaCl as a common raw material.

Residual gas treatment and disposal systems: Gas emissions mainly come from the recovery boiler, the lime furnace, the power boiler, and vents. These are composed of particular matter, SO_2, mercaptans, and other volatile hazardous organic and inorganic pollutants (e.g. chloroform, methanol, chlorine dioxide). In this study, gas emissions are characterised on the basis of CO_2, SO_2, TRS (total reduced sulphur), and HAP (hazardous airborne pollutants). TRS is a generic parameter to account for mercaptans, and H_2S. HAP takes into account all other volatile hazardous waste. Due to legal pressures, abatement measures, such as, electrostatic precipitators, gas filters, scrubbers, and incinerators, have been introduced in all plants. The latter is used to incinerate non condensable gases. Gases from boilers are released to the atmosphere through 50-60 m chimneys.

Residual liquid treatment and disposal systems: Liquid wastes are generated from bleaching, and from general washing and cleaning operations. Bleaching effluents are highly coloured and contain dissolved organic and chlorinated organic compounds, and residual bleaching chemicals. Washing effluents may contain suspended solids, and other components from accidental spills. Environmental loads from final effluents are expressed in terms of generic parameters: COD, and

AOX, to account for the total organic load and organo-chlorinated compounds, respectively. Effluent treatment systems feature pH neutralisation, cooling, primary sedimentation, and activated sludge.

Residual solid treatment and disposal systems: All woody residues from processing (e.g. bark, knots) and effluent treatment sludge, are burnt in power boilers, to generate steam and electricity. Solid residues (e.g. boiler ash, dregs, grits, sand, stones, spent oils, dirty chips, dust, etc) are disposed in in-plant controlled landfill.

3.2 Inventory results

Results on chemical and environmental loads associated to bleached kraft cellulose in Chile are summarised in Tables 1 and 2.

Tab. 1: Main chemical loads associated to BK cellulose production in Chile

	Eucalyptus BK pulp		Pine BK pulp	
	1992	**2008**	**1992**	**2008**
	kg /bone dry tonne cellulose			
Sodium hydroxide	22	25	47	35
Sodium chlorate	17	24	28	34
Chlorine	27	0	55	0
Calcium carbonate	16	15	30	18
Oxygen	0	23	0	30
Sulphuric acid	16	21	23	33
Hydrogen peroxide	2	3	1	4
Methanol	1	3	3	4
Forestry biocides (various)	$2\ 10^{-3}$	$1\ 10^{-3}$	$4\ 10^{-3}$	$3\ 10^{-3}$
Fuel oil+diesel	90	85	104	93

In general, chemical loads associated with eucalyptus cellulose are much lower than pine based cellulose. This is explained by the lower lignin content and greater density of eucalyptus wood, as compared with pine. Additionally, technological improvements have significantly reduced bulk chemicals requirements in most cases. Sodium chlorate appears to increase due to the complete substitution of chlorine by chlorine dioxide as bleaching agent. This is in agreement with published LCI cellulose reports [5].

Tab. 2: Main environmental loads of BK cellulose production in Chile

	Eucalyptus BK pulp		Pine BK pulp	
	1992	2008	1992	2008
	Kg /bone dry tonne cellulose			
Emissions to air				
CO_2 (fosil)	366	395	458	462
N_2O	0.002	0.004	0.004	0.005
CH_4	0.21	0.23	0.28	0.28
GHG	372	401	466	471
TRS	6.1	0.2	8.2	0.4
MP	0.05	0.05	0.08	0.07
Discharges to water				
COD	890	114	1190	146
AOX	4.2	0.1	7.6	0.1
NO_3^-	2	2	4	3
$PO_4^=$	1	1	2	1
$SO_4^=$	10	14	16	29
Natural resources depletion				
Crude oil, in ground	101	108	119	123
Natural gas, in ground (m^3)	6	8	9	10
Hard coal, in ground	19	20	34	27
Water from surface sources	81	32	98	34

Pine cellulose presents greater environmental loads than eucalyptus pulp. Liquid effluents are one of the main environmental aspects, and might seriously affect the quality of recipient waters. Since most cellulose production is located within the Bio Bio river basin, pollutants could seriously affect drinking and irrigation water. Finally, it can be seen that process improvements since 1992 led to a significant increase in greenhouse gases. Manufacturing of bulk chemicals such as sodium hydroxide and chlorate heavily rely on electrochemical processing, with considerable electricity consumption. Although overall bulk chemical consumption decreased since 1992, the Chilean electricity matrix experienced a drastic increase in the share of coal based thermoelectric plants, leading to a greater carbon footprint of cellulose. Only fossil GHG emissions are shown here.

4 Conclusions

This paper shows that process modifications have resulted in overall improvements in environmental performance. On the other hand, it shows the importance of the electricity matrix in the cellulose carbon footprint.

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References

[1] González P, Parra, O, Zaror CA (1999) Modelling Wastewater Discharges from the Bleached Kraft Cellulose Industry in the Bio Bio River Basin. Lat Am Appl Res 29(1):21-26.
[2] ISO 14040:2006. Life Cycle Assessment: principles and framework. ISO Publications
[3] González P, Parra, O, Zaror CA, Vesovic V (1998) Life cycle inventory analysis of bleached kraft cellulose production from pinewood and eucalyptus in Chile". Environmental Engineering and Management Conference. Barcelona. WIT Press - Computational Mechanics Publications, Southampton
[4] Gerding V (1991) Manejo de las Plantaciones de Pinus radiata D.Don en Chile. Bosque 12:3-10.
[5] Ryynänen H, Nelson P (1996) Environmental Life Cycle Assessment of Some New Methods for Producing Bleached Pulps from Australian Eucalyptus Woods. Appita 49:167-172.

Life Cycle Assessment of Integrated Solid Waste Management System of Delhi

Amitabh Kumar Srivastava and Arvind Kumar Nema

Abstract In this study quantity and composition of solid waste of Delhi is predicted till the year 2024 and feasibility of different options for long term management are evaluated. Then the life cycle assessment (LCA) is carried out to examine the environmental impact caused by the each option. The input for LCA is considered as quantity and composition MSW and energy whereas the output is taken as air emission, water emissions and energy recovery. The result of LCA indicates that recycling has least environmental impact. Moreover landfills produce less environmental impact than the incinerators during initial years and as the years passes the landfills produce more environmental impact than incinerators due to waste accumulation in landfill.

1 Introduction

With increasing population of urban areas, the environmental and sanitation work has been a challenging task for municipal or civic authorities. The inappropriate resources and budget has ushered complex constraints in solid waste management planning particularly in developing countries. The system modelling and operation research techniques have been used by many researchers to solve the solid waste management problems under the multiple constraints.

Shekdar et al., [1] and Mohan [2] discussed the long term planning of compost plants and dump sites for solid waste management of Indian cities using linear programming. Sudhir et al., [3] presented the nonlinear goal programming for solid waste management of Chennai city of India. Badran and El-Haggar [4] have presented a mixed integer programming model for solid waste management of Egypt. Rathi [5] has evaluated the solid waste management models for Mumbai, India involving Municipal Corporation, public private partnership and community participation. Huang et al., [6] and Huang et al., [7] have developed a long term planning model for integrated solid waste management (ISWM). Cheng et al., [8]

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applied linear programming model for site selection of solid waste disposal techniques under multiple criterion including cost, public acceptance and political concerns. The solid waste management facilities are considered as obnoxious facilities and their siting has negative impact on the ambient environment [9].

The consideration for environmental pollution while planning for solid waste facilities has been widely addressed by several researchers [10,11,12]. Chang and Wang [13] presented the linear programming model for solid waste management under multi-objectives including cost, air pollution, noise impact and traffic congestion. Chang et al., [14] included the contamination potential due to landfill leachate.

But the Existing solid waste management (SWM) planning model provides only limited assistance to decision makers struggling to find strategies that address their multifarious concerns of environmental burden. Life cycle assessment (LCA) is an effective tool to evaluate the environmental burdens associated with a product, process, or activity by identifying, quantifying and assessing the impact of the utilised energy, materials and the wastes released to the environment. The environmental impact of MSW has been extensively studied using LCA method [15-20]. A recent review shows that most of the LCAs are unclear regarding whether or not life cycle emissions from energy inputs or capital equipment are included in the calculation of results [21].

2 Integrated solid waste management of Delhi

The capital city of Delhi is one of the fastest growing cities of the India due to huge investment to improve infrastructure of city. The population density of city was 274 persons/ km^2 in 1901, increased to 1,176 persons/km^2 in 1951 and 9,294 persons/km^2 in 2001 [22]. The urbanisation of city has adversely impacted its resources and environment. Due to increasing quantity and limited resources, the waste management of Delhi has now become exigent task for the authorities. Recent judicial and government interventions, the concerned authorities are devising new plans and system to sort out this problem of city.

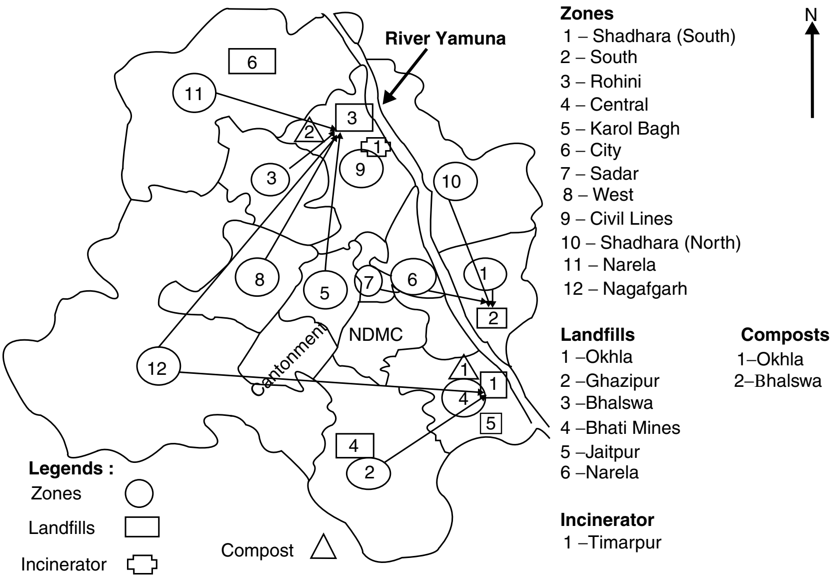

Fig. 1: Present waste flow in the study area

The solid waste management of city is being carried out by three local bodies of city namely, Municipal Corporation of Delhi (MCD), New Delhi Municipal Corporation (NDMC) and Delhi Cantonment Board (DCB). MCD is the largest among all these local bodies, covering almost 96% of the area; the rest of the area is almost equally divided between NDMC and DCB. MCD covers approximately 1390 km^2, with about 43% of the area urban. MCD has divided the area of Delhi under its administration in 12 zones. In addition to these other agencies like central pollution control board (CPCB), Delhi Pollution Control Committee (DPCC), and Ministry of Non-Conventional Energy Resources (MNES) are also making efforts to suggest efficient technology and system.

2.1 Generation and composition of solid waste

As per an estimate resident of Delhi generates 7000 tons of solid waste daily and projected to be increase 17000 to 21000 tons/day [23]. The data collected from MCD for the period September 2002 to August 2006 is plotted as shown in Figure 2. It can be seen that there is significant seasonal fluctuations in the quantity of waste being collected. Delhi has various heterogonous sources which contribute their waste into their municipal solid waste (Table 1). The average composition of solid waste is shown in the Table 2, which shows the major components are organic waste and others (mainly inert). The composition of recyclables such as metals, paper, plastic and glass is comparatively less.

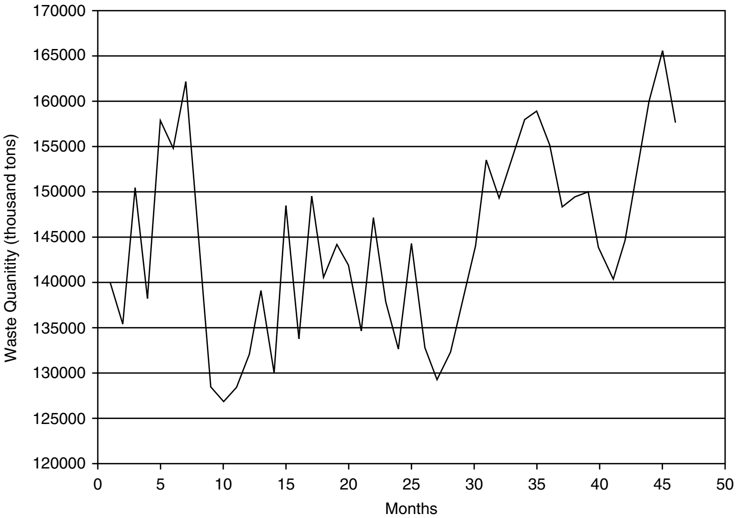

Fig. 2: Fluctuation of waste quantities in Delhi

Tab. 1: Source wise generation of the MSW (tonnes/day) in Delhi

Residential waste	Main shopping centres	Vegetable and fruit markets	Construction waste	Hospital waste	Industrial waste
3,010	1,017	538	382	107	502

Source: [24]

Tab. 2: Solid waste composition of Delhi (in %)

Waste Component					
Paper	Plastic	Organic	Metal	Glass	Others
5.60	6.00	38.60	2.00	1.00	46.80

Source: [25]

2.2 Current practices for MSW management in the study area

The solid waste of Delhi is collected from the road side community bins by MCD and disposed off into dump sites. The components involved in the current system are described in the following sections.

2.2.1 Reuse and recycling

Reuse and recycling of waste component is always seen as one the most important part of solid waste management system. In Delhi, like the urban centres of other developing countries, recycling of MSW is widely carried out by informal sectors. The informal recycling sector refers to the waste recycling activities of waste pickers and waste collectors, paid mainly by the sale of collected materials. Agarwal et al [27] estimated that there are around 80,000–100,000 waste pickers and 17,857 waste collectors involved in the recycling industry in Delhi. The annual turnover of this informal sector is about INR 180 million [28].

2.2.2 Composting

The organic component of the waste can be processed for compost, if it is properly segregated. Therefore, municipalities of some Indian cities have started two bins system to segregate the organic waste at source. Taking view on the percent composition of organic fraction in the solid waste, first engineered composting plant was started at Okhla by MCD in the year 1980. Later this plant was expanded with some additional capacity in 1985. However, this plant was not operational during 1991 -1995 due to low quantity of waste material and higher operational cost. In addition to these plants, one more composting plants were established in Bhlaswa during 1998. The details of all three composting plants are given in Table 3.

Tab. 3: Details of existing composting plants in Delhi

Locations	Starting year	Area (ha)	Capacity (tons/day)
Okhla	1980	3.24	150
Okhla (Expansion)	1985	3.44	200
Bhalswa	1998	4.9	500

Source: [24]

2.2.3 Incineration

The organic component of the waste can be processed for compost, if it is properly segregated. Therefore, municipalities of some Indian cities have started two bins system to segregate the organic waste at source. Taking view on the percent composition of organic fraction in the solid waste, first engineered composting plant was started at Okhla by MCD in the year 1980. Later this plant was expanded with some additional capacity in 1985. However, this plant was not operational during 1991 -1995 due to low quantity of waste material and higher operational cost. In addition to these plants, one more composting plants were established in Bhlaswa during 1998. The details of all three composting plants are given in Table 3. However, these composting plants do not function at the anticipated level and along with the process residue; a major fraction of MSW is diverted to the landfills due to cheaper alternative.

2.2.4 Disposal

The present rate waste diversion towards waste dumps/ landfills is about 91% [24] This clearly shows the violation of MSW rules; because, only inert must be diverted to landfills, which accounts to only around 40% of the total waste (Table 2). The disposal of waste components other than inert pace is not only creating environmental problems but also depleting valuable land resource of capital city of Delhi. A total of eight dumpsites have been exhausted so far by dumping of MSW in Delhi and three dumpsites are in operation [29]. The area covered by these dumpsites is at least 1% (14.83 km^2) of Delhi's total area [24]. It was estimated that the active and closed dumpsites produce approximately 81.5 million litres of leachate annually [23]. In addition to presently operating three dumpsites, three engineered landfills are in the initial phase of development.

3 System boundaries for LCA

The system boundary for the present study is depicted in Figure 3. The functional unit used in the scenarios has been defined as the amount of municipal solid waste generated in the different zones of Delhi. The system boundaries selected for the life cycle of solid waste was defined from the source and up to disposal site i.e. landfills. Collection and Transportation of waste is omitted from the present study as the emissions are common for all the scenarios considered in the study.
The various waste management facilities shown in Figure 3 is used in the ISWM model. For optimal waste allocation to the various facilities, a linear optimisation is used and solved by commercially available software LINGO 11.0. The details of the optimal waste allocation can be referred at [30].

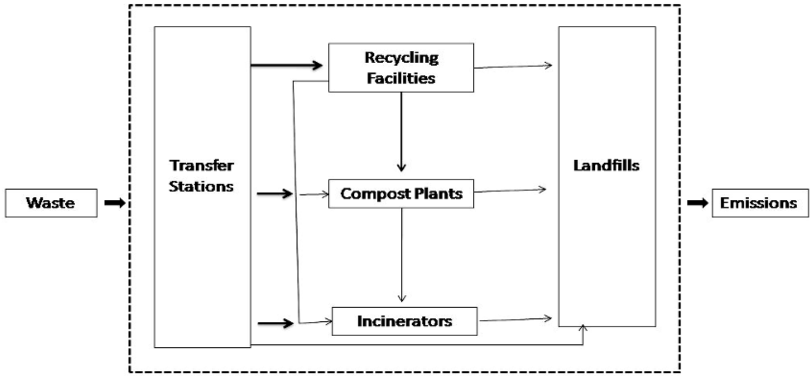

Fig. 3: System boundary for the present study

4 Results and discussion

The ISWM model was run for optimal waste allocation for cost minimisation as the objective of the model. For waste allocation to each of the facilities, the emissions are calculated by a life cycle model developed on spread sheet. The inventory used in the study was obtained from the Municipal Corporation of Delhi. The emissions for minimum cost strategy is given in Table 4. In ISWM of Delhi, the waste allocations for the landfills in all the planning periods are very high. Therefore the all type of emissions from landfills are very high when compared to other waste processing facilities. However emissions like SO_x, NO_x and energy consumption for per unit waste quantity from incinerators are high in the initial planning periods than the same for other facilities like landfills but emissions of Green House Gases from landfills are very high in later stage of planning periods because of accumulation of waste in landfills. According to the processes in the landfills there are several reactions which occurs during initial years of landfilling and steady-state seems are obtained after degradation of waste even after closure of landfills. It can be noted that the potential emissions of methane (CH_4) from landfilling during the survey time-period will be approximately 10 times higher than same from compost plants. As compost plants also involves the biological processes.

The emissions from the recycling and reuse facilities are negligible except energy consumption. The energy consumptions for the recycling and reuse facilities is comparatively less when compared with same for developed countries because most of the segregation and recycling is done manually.

Tab. 4: Emissions for ISWM

Facility / process	Planning Periods	Energy (103BTU/Period)	Particulate Matter (103kg/period)	NO_x (103kg/period)	SO_x (103kg/period)	CO (103kg/period)	CO_2 (103kg/period)	CH_4 (103kg/period)
Reuse/Recycling (5112)*	2007-09	128075	-	-	-	-	-	-
Compost (912)		17328	11856	-	-	-	-	82992
Incineration (219)		25342	15987	46866	2847	12483	3285	-
Landfilling(3173)		418245	526917	30008110	5418650	85951	85951	861976
Reuse/Recycling (7123)	2009-14	178075	-	-	-	-	-	-
Compost(2173)		45087	30849	-	-	-	-	215943
Incineration (547)		63298	39931	117058	7111	31179	8205	-
Landfilling (5918)		662342	834438	47521540	8581100	136114	136114	1365046
Reuse/Recycling (7149)	2014-19	178725	-	-	-	-	-	-
Compost (2550)		48450	33150	0	0	0	0	232050
Incineration(547)		63298	39931	117058	7111	31179	8205	-
Landfilling(6625)		741470	934125	53198750	9606250	152375	152375	1528123
Reuse/Recycling (7149)	2019-24	178725	-	-	-	-	-	-
Compost(2749)		48450	33150	-	-	-	-	232050
Incineration (590)		63298	39931	117058	7111	31179	8205	-
Landfilling (7125)		741470	934125	53198750	9606250	152375	152375	1528123

*Values shown in parenthesis is the quantities of waste allocated (in 1,000 tonnes)

5 Conclusion

This paper evaluates the environmental emissions from ISWM model developed for Delhi. A life-cycle-based methodology was used to calculate emissions of a set of pollutants, including CO, CO2, NO_x, SO_x, particulate matter, and GHE, and energy consumption. The model can be used to develop optimal strategies and decision support system for representing and examining ISWM policies in a systematic manner while considering cost and environmental implications.

References

[1] Shekdar AV, Krishnaswamyt KN et al. Long term planning for solid waste management in India.Waste Management and Research, 9, 1991, 511-523.
[2] Gupta S, Mohan K, Prasad RK et al (1998) Solid waste management in India: options and opportunities, Resour Conserv Recycl 24:137-154.
[3] Sudhir V, Muraleedharan VR, Srinivasan G (1996) Integrated solid waste management in urban India: A critical operational research framework. Socio-Econ Plann Sci 30(3):163-181.
[4] Badran MF, El-Haggar SM (2006) Optimization of municipal solid waste management in Port Said–Egypt. Waste Manage 26:534–545.
[5] Rathi S (2006) Alternative approaches for better municipal solid waste management in Mumbai, India. Waste Manage, 26, 2006, 1192–1200.
[6] Huang GH, Chi GF, Li YP (2005) Long-term planning of an integrated solid waste management system under uncertainty - I. Model development. Environ Eng Sci 22(6):823-834.
[7] Huang GH, Sae-Lim N, Chen Z, Liu L (2001) Long-term planning of waste management system in the City of Regina - An integrated inexact optimization approach. Environ Modell Assess 6(4):285-296.
[8] Cheng S, Chan CW, Huang GH (2003) An integrated multi-criteria decision analysis and inexact mixed integer linear programming approach for solid waste management. Eng Appl Artif Intell 16(5-6):543-554.
[9] Chang NB, Wang SF (1996) Managerial fuzzy optimal planning for solid waste management systems. J Environ Eng 122(7):649-658.
[10] Chang NB, Wei YL (2000) Siting recycling drop-off stations in an urban area by genetic algorithm-based fuzzy multi-objective nonlinear programming modeling. Fuzzy Sets Syst 114(1):133–149.
[11] Nian TK, Zheng DF, Luan MT (2004) Fuzzy sets theory for multi-objective system and its application to site selection of municipal solid wastes landfill. Rock Soil Mech 25(4):574-578.
[12] Gautam AK, Kumar S (2005) Strategic planning of recycling options by multi-objective programming in a GIS environment. Clean Technol Environ Polic*y* 7(4):306-316.

[13] Chang NB, Wang SF (1996) Comparative risk analysis of solid waste management alternatives in a metropolitan region. Environ Manage 20(1):65–80.
[14] Chang NB, Shoemaker CA, Schuler RE (1996) Solid waste management system analysis with air pollution and leachate impact limitations. Waste Manage Res 14(5):463-481.
[15] Björklund A, Finnveden G (2005) Recycling revisited – life cycle comparisons of global warming impact and total energy use of waste management strategies. Resour Conser Recycl 44:309–317.
[16] Rodriguez-Iglesias J, Maranon E, Catrillon L, Riestra P, Sastre H (2003) Life cycle analysis of municipal solid waste management possibilities in Asturias, Spain. Waste Manage Res, 21:535–548.
[17] Emery A, Davies A, Griffiths A, Williams K (2007) Environmental and economic modeling: a case study of municipal solid waste management scenarios in Wales. Resour Conserv Recycle 49:244–263.
[18] Banar M, Cokaygil Z, Ozkan A (2009) Life cycle assessment of solid waste management options for Eskisehir. Waste Manage 29:54–62.
[19] Den Boer J, Den Boer E, Jager J (2007) LCA–IWM: a decision support tool for sustainability assessment of waste management systems. Waste Manage 27(8):1032–1045.
[20] Bjarnadóttir HJ, Friöriksson G, Johnsen T, Sletsen H (2002) Guidelines for the Use of LCA in the Waste Management Sector. Nordtest TR 517, 2002.Nordtest, Espoo, Finland
[21] Cleary J (2009) Life cycle assessments of municipal solid waste management systems: a comparative analysis of selected peer-reviewed literature. Environ Int 35:1256–1266.
[22] <http://www.environment.delhigovt.nic.in/> (accessed on 31/05/2007)
[23] <http://www.adb.org /Documents/Events/2005/Sanitation -Wastewater -Management/paper-kumar.pdf.> (accessed 12/06/2006)
[24] Talyan V, Dahiya RP, Anand S Sreekrishnan TR (2007) Quantification of methane emission from municipal solid waste disposal in Delhi. Resour Conserv Recycl 50(3):240-259.
[25] Sharholy M, Ahmad K et al. (2007) Municipal solid waste characteristics and management in Allahabad, India. Waste Manage 7(4):490-496.
[26] CPHEEO (2000) Manual on Municipal Solid Waste Management. Ministry of Urban development, Government of India, New Delhi
[27] Agarwal A, Singhmar A, Kulshrestha M, Mittal AK (2005) Municipal solid waste recycling and associated markets in Delhi, India. Resour Conserv Recycl 44:73–90.
[28] <http://www.cpheeo.nic.in/> (accessed on 24/03/2006)
[29] <http://www.cpcb.nic.in/pamsweb/statusmsw.htm> (accessed on 18.6.2007)
[30] Srivastava AK, Nema AKA (2000) Multi-Objective and Multi-Period Location-Allocation Model for Solid Waste Disposal Facilities: A Case Study of Delhi. Int J Environ Health 2(2):184-197.

LCM of Rainwater Harvesting Systems in Emerging Neighbourhoods in Colombia

Tito Morales-Pinzón, Sara Angrill, Joan Rieradevall, Xavier Gabarrell, Carles M. Gasol and Alejandro Josa

Abstract Potential environmental impacts of water harvesting systems for rain to emerging neighbourhoods in Colombia were studied. Two tools were integrated into a simulation model (life cycle analysis and system dynamics). This was performed as an application case study in two urban areas of Colombia (Bogota and Pereira). We modelled a standard neighbourhood with 10 residential 5-storey buildings of 24 apartments. The results show that it is possible to avoid in every neighbourhood 150,729 kg CO_2e and 44,857 kg CO_2e, respectively.

1 Introduction

South American countries and particularly in Colombia, experiences in LCM are limited and can be considered a new subject, particularly in the environmental impacts study of systems of rainwater harvesting (RWH) for urban domestic use in buildings.

There is great pressure on water resources and water supply networks, due to the growing increase in the construction of new neighbourhoods in developing countries, especially Colombia. However, Colombia is one of the top 20 countries of the world's water supply with an average rainfall greater than 2,612 $L·m^{-2}$ per year [1].

In Colombia, there are dwelling projects already executed and poorly planned with respect to the mains water supply available, usually with difficulty to grow and low efficiency of service delivery [2]. The application of environmental criteria in urban design of new neighbourhoods in Colombia, has not yet considered the benefits of rainwater harvesting (RWH) in the context of a sustainable

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management of this resource. These benefits could include free access to water supply, mitigation of pressure on aquifers and surface courses; prevention of floods caused by soil sealing attributable to urbanisation. Besides, the use of rainwater on a large scale is perceived as an adaptation strategy to climate change against the reduction of water availability [3].

The planning of water supply with stormwater systems is very important in Colombia because we expect more than 1,000,000 new houses according to policies of urban growth for the next 4 years [4]. The life cycle analysis can be a tool for evaluating environmental RWH systems dynamics, contributing to identify the more environmentally friendly strategy. This research helps to identify the environmental impacts attributable to these systems in two urban areas of Colombia.

2 Objectives

This research responds to the following objectives: a) to assess the behaviour of potential environmental impacts in life cycle of rainwater harvesting systems of emerging neighbourhoods of Colombia. Neighbourhoods with different pluviometry ranks and similar constructive density; b) to propose a dynamic approach as a tool for life cycle assessment of this systems from the perspective of LCM and dynamics systems in developing countries.

3 Methodology

For this research, we regard the emerging buildings as sustainable construction as economic, social and environmental aspects. In this sense, our contribution to emerging neighbourhoods is focused from the environmental dimension. We have chosen as pilot study of neighbourhoods located in two important cities of Colombia: Bogotá and Pereira, each one with average rainfall of 794 and 2,258 $L \cdot m^{-2} \cdot year^{-1}$ to represent the conditions of urban growth leading the country (Table 1). We propose the construction of a new neighbourhood built with identical characteristics in each city.

According to estimates made by Ballén, Galarza and Ortiz [8], approximately 56% of average domestic water consumption in Colombia is spent on activities that do not require drinking water quality and 47% is used in laundry and toilet. This data was used as an estimate of the demand for rainwater in the neighbourhoods studied. The RWH system was divided into 2 subsystems: infrastructure and energy use. We analysed the location of the underground tank to be the most appropriate for a country with high earthquake risk.

Tab. 1: Population, domestic water consumption and general climatic data for the selected urban areas

Urban area	Population [a]	Domestic water consumption (Lcd)[b]	Altitude (MSL)[c]	Temperature (ºC) [c]	Rainfall ($L•m^{-2}•year^{-1}$) [c]	Days of rainfall per year[c]
Bogotá	7,347,795	116	2,547	13.4	794	186
Pereira	383,623	118	1,367	21.3	2,258	230

Note: (a) Projections based on census population [5]; (b) Average of water consumption in Colombia [6]; (c) Average characteristics from urban weather stations for monthly series from 1970 to 2008 period [7].

3.1 Simulation model

We created a simulation model using system dynamics methodology. This model was developed at the Stella software [9]. The model considers RWH potential and tap water as input flows to the system. Water consumption has been divided into two flows, potable water demand and rainwater demand. Each one water flows, has been assigned the environmental impacts calculated for the equivalent of the functional unit considered throughout the system (Equation.1, 2 and 3).

$$I_{(t)} = I_{(t-dt)} + (I_{Dr} + I_{Dp}) \cdot dt \quad (1)$$

$$I_{Dr} = I_i + I_u \quad (2)$$

I: Matrix of potential environmental impact of the system

I_{Dr}: Matrix of potential environmental impact of RWH

I_{Dp}: Matrix of potential environmental impact of tap water

I_i: Matrix of potential environmental impact by infrastructure

I_u: Matrix of potential environmental impact by energy use

$$D = D_r + D_p \quad (3)$$

D: water system demand

Dr: rainwater demand

Dp: potable water demand

3.2 Environmental calculation tools

The LCA methodology considers the entire life cycle of the RWH infrastructures for each scenario. However, the impact of the recycling process of materials at the end of its life cycle is outside the boundaries of the system, as there is much uncertainty in the recycling process 50 years hence.
The aim of the system is the maximum uptake of rainwater with the lowest environmental impact infrastructure. The functional unit has been defined as the collection, storage and supply of $1m^3$ of rainwater provided per person and day to be used as non-potable water for a constant demand of laundry and toilet.

3.2.1 Description of the system under study

The system consists of a standard neighbourhood of $100x100m^2$ with 10 residential five-storey buildings of 24 apartments ($700m^2 \cdot built^{-1}$). The average density of people per household has been assumed on 4 inhabitants per dwelling [10]. We have focused on the analysis of a underground tank by apartment building, leaving environmental analysis of the deposit to the neighbourhood level for another study.

3.2.2 Environmental modelling tools

Only the classification and characterisation stages [11] have been considered. The method 2001 baseline v2.04 CML has been used [12] and the impact selected categories were: abiotic depletion potential (ADP, kg Sb-e), acidification potential (AP, kg SO_2e), global warming potential (GWP, kg CO_2e), human toxicity potential (HTP, kg 1,4-DB-e), ozone layer depletion potential (ODP kg CFC-11e) and photochemical ozone creation potential (POCP, kg C_2H_4e). The ecoinvent 2.0 database [13] has been used, associated to the software SimaPro7.2.0 [14].
The data have been adjusted to the context of Colombia. We have estimated the impact associated with consumption of tap water from the average consumption of inputs of water treatment plants in Colombia [15]. The impact of energy consumption was calculated on the basis of references [16]. The life span of the rainwater storage tank, pipes and pumps is of 50, 25 and 15 years, respectively [17].
The size of the tank has been determined using the model, which allows to size the volume of a tank through a continuous daily balance of supply and demand along the year. Data from the weather station of airport El Dorado (Bogotá) and the

airport Matecaña (Pereira) has been used, which are within the average rainfall of Colombian cities.

4 Results and discussion

Under current conditions mean monthly rainfall in a standard neighbourhood, the model results shows in Bogota city a potential consume up 3,900m^3 of rain water year. The same case in the city of Pereira can potential consume up 15,500 m^3.

4.1 Model of environmental impacts

This potential consume would be possible in neighbourhoods with rainwater storage tanks big enough. However, the potential environmental impacts could increase exponentially and therefore the environmental efficiency of the system is quickly diminished. At each impact category studied, we found a functional relationship between harvesting rainwater (used within the system) and the potential environmental impact. The results showed a potential limit of rainwater consumption in each urban area. Potential consume in Pereira neighbourhood is significantly higher than Bogota. All potential impacts of the system show a similar behaviour as presented in the category GWP (Figure 1).
The model reveals a functional relationship between potential impacts and the supply of rainwater. The general model is exponential (Equation 4).

$$I = ke^{RWH/a} \tag{4}$$

I: Potential Impact
RWH: Rainwater Harvesting (m^3·year^{-1})
k, a: constants

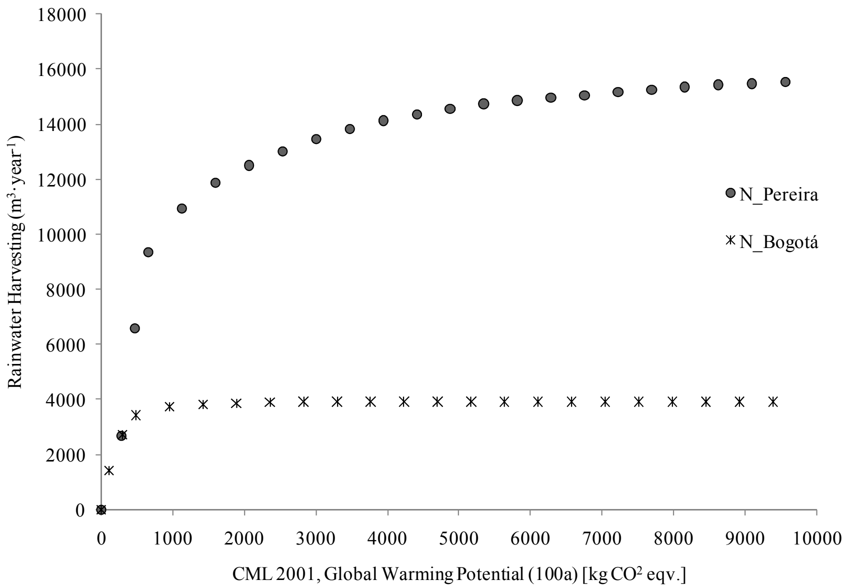

Fig. 1: Global warming potential behaviour based on RWH

The models have higher fit in Pereira (R^2> 0.96) than Bogotá (R^2> 0.67) (Table 2). These can be used to estimate the potential environmental impacts of RWH systems in new neighbourhoods. Knowing the demand for rainwater, we could estimate a minimum potential environmental impact of system.

Tab. 2: Estimated parameters of exponential model for each urban area

CML 2001, potential impact (I)		Urban area	k	a	R^2
Abiotic depletion potential	kg Sb-e	Pereira	4.58E-04	8.06E-02	0.987
		Bogotá	8.35E-03	4.09E-13	0.677
Acidification potential	kg SO_2e	Pereira	3.68E-04	1.12E-01	0.963
		Bogotá	8.23E-03	2.15E-13	0.673
Global warming potential (100a)	kg CO_2e	Pereira	4.43E-04	1.18E-01	0.984
		Bogotá	8.34E-03	3.61E-11	0.677
Human toxicity potential	kg 1,4-DB-e	Pereira	4.53E-04	2.65E-01	0.986
		Bogotá	8.35E-03	1.80E-11	0.678
Ozone layer depletion potential	kg CFC-11e	Pereira	4.83E-04	7.92E-05	0.991
		Bogotá	8.37E-03	5.71E-16	0.680

We found a functional relationship between the storage volume of rainwater and RWH. This harvesting tends to a limit as shown in Figure 2. The adjusted model is logarithmic. The model shows best fit for Pereira (R^2=0.97) than Bogotá

(R^2=0.72). The limit of rainwater harvesting in the system is 15,522m^3 and 3,921m^3 for Pereira and Bogota.

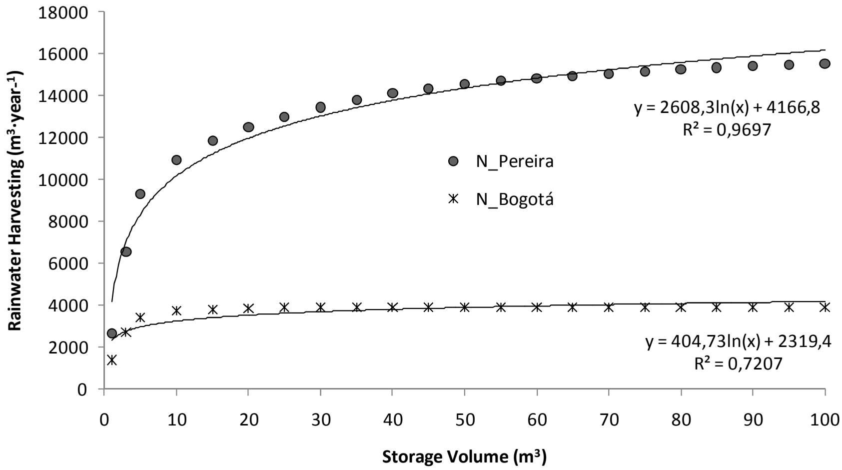

Fig. 2: Relationship between storage volume and RWH.

4.2 Potential environmental impacts

Using the model, the greatest potential environmental impact is found associated with infrastructure in all categories. Since the potential rainwater supply, optimal volumes of 15 and 85m^3·built^{-1} (Pereira and Bogota) were found. Higher impacts of both volumes were found in the 85m^3 deposit. Except in Pereira for ADP impact category, in all of them over 95% of the total potential impact is in infrastructure (Table 3).

For 85m^3 of storage volume, we found global warming potential of 7,783 kg CO_2e per year in Pereira neighbourhood and 7,960 in Bogotá (Table 3). However, for each case, we would be leaving to deliver to the environment near to 199,242 kg of CO_2e in Pereira and 46,210 kg of CO_2e in Bogotá if implemented RWH systems from the start of the new work (Table 4).

The avoided environmental impacts do not decrease the total water demand. This continues to grow as a result of the dynamics of urban growth of cities in Colombia. However, the consequences are positive as to reduce pressure on tap water and this influences the required infrastructure and inputs required for purification.

Tab. 3: Potential environmental impacts of the RWH system in each urban area and storage volume of 85m^3

Potential impacts	Total		Infrastructure		Energy use	
	Pereira	Bogotá	Pereira	Bogotá	Pereira	Bogotá
ADP kg Sb-e	9.77E+01	9.65E+01	9.52E+01	9.64E+01	2.50E+00	1.00E-01
AD kg SO_2e.	3.43E+01	3.16E+01	2.85E+01	3.12E+01	5.80E+00	4.00E-01
GWP kg CO_2e	8.16E+03	7.98E+03	7.78E+03	7.96E+03	3.77E+02	2.46E+01
HTP kg 1,4-DB-e	4.22E+03	4.15E+03	4.08E+03	4.14E+03	1.41E+02	9.20E+00
ODP kg CFC-11e	1.38E-01	1.38E-01	1.38E-01	1.38E-01	1.84E-05	5.46E-07

Tab. 4: Tap water potential environmental impacts avoid

Potential impacts	Potential impacts of 1m^3 of tap water		storage volume of 15m^3		storage volume of 85m^3	
	Pereira	Bogotá	Pereira	Bogotá	Pereira	Bogotá
ADP kg Sb-e	8.05E-02	7.27E-02	954	277	1261	285
AD kg SO_2e	8.71E-02	8.30E-02	1,033	316	1365	326
GWP kg CO_2e	1.27E+01	1.18E+01	150,729	44,857	199,242	46,210
HTP kg 1,4-DB-e	8.83E+00	8.12E+00	104,706	30,899	138,406	31,831
ODP kg CFC-11e	1.29E-05	1.28E-05	1.53E-01	4.88E-02	2.02E-01	5.02E-02

The analysis shows the contribution percentages impacts of each subsystem, but also allows to show the efficiency in the use of the deposit. Under the conditions of Pereira (P) and Bogotá (B), 15m^3 tanks have higher rates of use. This implied greater use of pump and therefore a higher energy consumption (Figure 3). The total supply of rainwater is 11,856 (P) and 3,806 m^3·year^{-1} (B).

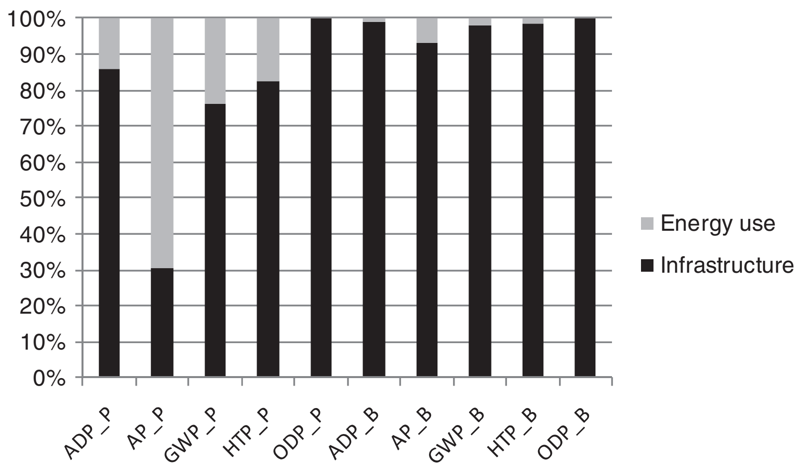

Fig. 3: Proportion of total environmental impacts and contribution of the systems urban for 15m^3·built^{-1} storage volume and 11,856 m^3·year^{-1} of RWH potential

Under the conditions of a standard neighbourhood in Pereira (P) and Bogotá (B), $85m^3$ tanks have lower rates of use. This implied lower use of pump and therefore a lower energy consumption than $15m^3$ tanks (Figure 4). The total supply of rainwater is 15,251 (P) and $3{,}921m^3\cdot year^{-1}$ (B). In a standard neighbourhood, $85m^3$ storage volume would be optimal for Pereira and $15m^3$ in Bogota.

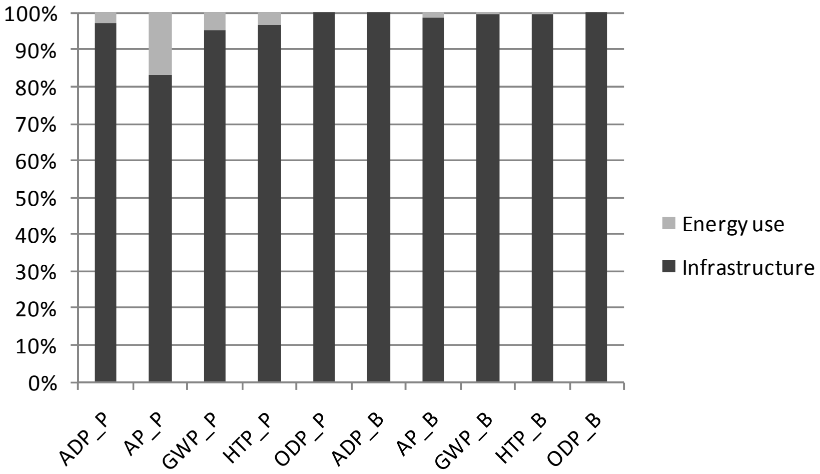

Fig. 4: Proportion of total environmental impacts and contribution of the systems urban for $85m^3\cdot built^{-1}$ storage volume and $3{,}806\ m^3\cdot year^{-1}$ of RWH potential

The model allowed to estimate a function relating the Storage Volume with GWP of all household water consumption (Figure 5). The model obtained is quadratic (Equation 3). From the equations for Bogota and Pereira, we can deduce that the equilibrium point for the storage volume is given by the next expression (Equation 5):

$$GWP = aV^2 + bV + c \tag{5}$$

$$V_e = -\frac{b}{2a} \tag{6}$$

Using Equation 5 we find that the equilibrium storage volume values are 75.4 and $45.8m^3$ to Pereira and Bogotá, respectively. These values change depending on the impact category analysed and the model fit.

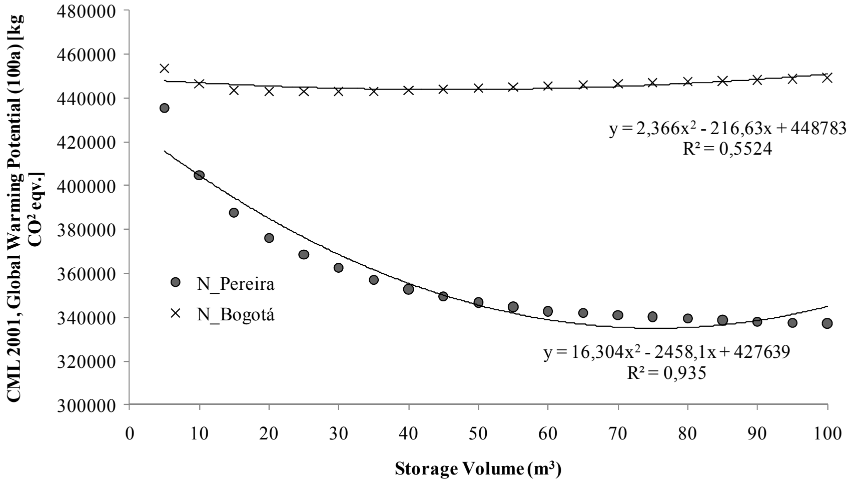

Fig. 5: Relationship between storage volume and GWP (100a) of household water consumption

5 Conclusions

We found functional relationships between the rainwater harvesting system required and the potential environmental impacts. This relationship can be expressed as an exponential model, where the impacts are calculated in terms of harvesting rainwater volume. These models have the higher coefficient of determination (R^2) in the urban area of Pereira, in principle for the rainwater potential.

In the context of Pereira and $15m^3$ storage volume, the percentage of abiotic depletion in energy use is higher than Infrastructure. In all other impact categories analysed the greatest potential impact is associated with infrastructure.

Although the potential impacts increases with the size of the deposit, avoided impacts justify larger volume. This is related to the decrease in the consumption of mains water. We have developed a methodology that helps to decide the best system of rainwater harvesting using two tools of life cycle analysis and system dynamics.

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References

[1] FAO-Aquastat. <http://www.fao.org/nr/aquastat/> (accessed August 2010)

[2] Ministerio de Ambiente (2008) Vivienda y Desarrollo Territorial-MAVDT, Colombia. Política de Gestión Ambiental Urbana

[3] Urbana. Revista de la Construcción Sostenible. 2010. Cámara Colombiana de la Construcción (Camacol) Edición #0046, Colombia

[4] Angrill S, Morales T, Cerón I, Gabarrell X, Josa A, Rieradevall J (2011) Environmental impact of rainwater harvesting integration in new construction compared with renovated buildings. Application to urban planning for emerging neighbourhoods in Bogotá. IV International Life Cycle Assessment Conference in Lati-America. Coatzacoalcos, Mexico

[5] Departamento Nacional de Planeación-DANE-, Colombia. Proyecciones de Población departamentales y municipales por área 2005 – 2020. In <http//www.dane.gov.co> (accessed July 2010)

[6] Instituto de Hidrología, Meteorología y Estudios Ambientales -IDEAM-Colombia (2001) El Medio Ambiente en Colombia. Segunda Edición

[7] Instituto de Hidrología, Meteorología y Estudios Ambientales -IDEAM-Colombia (2008). Informe anual sobre el estado de ambiente y los recursos naturales renovables en Colombia. Estudio nacional del agua. Relaciones de demanda de agua y oferta hídricap.

[8] Ballén SJA, Galarza G MA, Ortiz MRO (2006b) Sistemas de aprovechamiento de agua lluvia para vivienda urbana. [Rainwater Harvesting Systems for urban housing]. Int. Symp. on VI SEREA, Seminário Iberoamericano sobre Sistemas de Abastecimento Urbano de Água, João Pessoa, Brazil

[9] High Performance Systems (2011) Stella 9.1.4. Lebanon, USA

[10] Defensoría del Pueblo de Colombia-DPC (2009). Diagnóstico del cumplimiento del derecho humano al agua en Colombia

[11] ISO 14042:2000. Environmental Management –life cycle assessment- life cyle impact assessment. International Organisation for Standardisation ISO. International Standard. 14042. Geneva, Switzerland

[12] Guinée JB (ed.), Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, van Oers L, Wegener Sleeswijk A, Suh S, de Haes HU, de Bruijn, AH, van Duin R, Huijbregts MAJ, Lindeijer E, Roorda AAH, Weidema BP (2001)

Life cycle assessment: an operational guide to the ISO standards. Parts 1 and 2. Ministry of Housing, Spatial Planning and Environment (VROM) and Centre of Environmental Science (CML), Den Haag and Leiden, The Netherlands (Guinée JB, final editor)

[13] Ecoinvent (2009) Swiss Centre for Life Cycle Inventories. Ecoinvent database v3.0. Technical report. http://www.ecoinvent.ch/ (accessed 10 May 2011)

[14] PRé Consultants (2010) SimaPro 7.2.0. Amersfoort, the Netherlands

[15] <http://www.cra.gov.co>, Comisión de Regulación de Agua Potable y Saneamiento Básico (Colombia) (accessed 25.03.2011)

[16] Ortiz-Rodríguez O, Castells F, Sonnemann G (2010) Life cycle assessment of two dwellings: One in Spain, a developed country, and one in Colombia, a country under development. Sci Total Environ 408(12):2435-2443

[17] Roebuck RM, Oltean-Dumbrava C, Tait S (2010) Whole life cost performance of domestic RWH systems in the United Kingdom. Water and Envir. J. ISSN 1747-6585.

PART V:
LCM in the Agriculture and Food Sectors

Environmental Profiles of Farm Types in Switzerland Based on LCA

Daniel U. Baumgartner, Johanna Mieleitner, Martina Alig and Gérard Gaillard

Abstract The goal of the study was to assess the environmental impacts of a network of Swiss farms consisting of farms of different types, production regions and farming systems in order to analyse the environmental profile per farm type. Agronomic, technical and economic data of 105 Swiss farms have been collected for the year 2008. We identified three types of environmental profiles subject to the analysed functional units: i) a rather favourable environmental profile, e.g. 'combined suckler cows'; ii) a profile with favourable impacts regarding the function of land cultivation, but unfavourable for the productive and financial function, e.g. 'other cattle'; and iii) a profile with favourable impacts for the productive function, but unfavourable for the financial function and the function of land management, e.g. 'combined pigs/poultry'. In conclusion, we found that an environmental optimisation over three functions is challenging as areas of conflicts are present, especially between conserving land management and high productive output.

1 Introduction

Within the food supply chain the agricultural phase has an important share of the environmental impacts, particularly for the production of meat and milk [1]. Therefore, in order to reduce the environmental burden of food production it is important to identify optimisation measures on the farm level. At the same time it has to be considered that multifunctional agriculture fulfils per se different functions, e.g. land management, food and feed production, and provision of income. On a regional or national level, farms differ according to their structure, which implies different environmental strengths and weaknesses.
Our objective was to assess the environmental profiles of different farm types using a network of Swiss farms consisting of farms of different types, production regions and farming systems in order to identify the need of action according to the respective farm type. Based on this identification of environmental hotspots

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improvement measures should be found. The analysed influencing factor is the farm type (e.g. 'arable crops', 'dairying', 'suckler cows', 'combined pigs/poultry').

2 Material and methods

2.1 Data collection and farm classification

Agronomic, technical, and economic data of 105 Swiss farms have been collected for the year 2008 in the framework of the multi-year project Life Cycle Assessment – Farm Accountancy Data Network (LCA-FADN) which was financed by the Swiss Federal Office for Agriculture (FOAG) and Agroscope Reckenholz-Taenikon Research Station ART [2]. Farms were contacted according to a sampling plan. The participation in the project was voluntary. The participating farms were classified according to the FAT99-typology [3]. The classification is done by using ratios of land utilisation, e.g. ha arable land per ha total farm land, and ratios of different animal categories, e.g. livestock unit of dairy cows per livestock unit of all cattle.

2.2 LCA calculation, system boundary, assessed functions and impact categories

LCA calculation were performed using the SALCA farm model developed by Agroscope ART [4]. The applied functional units were *ha utilised agricultural area (UAA)* for the function of land management, *MJ digestible energy (DE)* for the productive function, and *CHF gross revenue (GR)* for the financial function.
The system boundary was set at the farm gate. The system comprises the farm as an economic unit. Included are the total UAA, the infrastructure and the employed production means, e.g. diesel, mineral fertilisers, purchased animals. Not included are buildings for living, forest, farm shop, processing of agricultural goods, tourism or working for third parties. The temporal boundary is set from the harvest of the preceding main crop to the harvest of the actual main crop for arable cropping. For all other activities the boundary is a calendar year.
Four environmental impacts, i.e. non-renewable energy resources (hereafter: energy demand) [5], global warming potential (GWP, 100a) [6], eutrophication potential (EutP; EDIP) [7], and terrestrial ecotoxicity (TTox; CML) [8] were assessed representing the three management axes according to [10].

2.3 Data assessment, input groups

The data assessment was performed on the farm level using descriptive statistics and the statistics software R [9]. In order to allow a more detailed analysis of the environmental impacts, the infrastructure and the production means a farms needs to produce food or feed were grouped into so-called input groups (Table 1) [2].

Tab. 1: Overview of the input groups and the emissions related to them.

Input group	Emission sources
Buildings, equipment	The production of the buildings and equipment
Machines	The fabrication of machines
Energy carriers	The production and use of energy carriers, e.g. diesel, electricity, fuel oil, natural gas, etc.
Fertilisers/nutrients	The production of mineral fertilisers as well as direct field emissions from spreading mineral fertilisers and farmyard manure
Pesticides	The production and application of pesticides
Purchase of seeds	The cultivation of purchased seeds
Purchase of feedstuffs	The cultivation of purchased feedstuffs
Purchase of animals	The rearing of purchased animals
Animal emissions	Digestion, as well as emissions occurring in the stable, at the manure storage, as well as while grazing
Other inputs	Inputs such as silage film, bird control, fleece, lubricating grease, etc.

2.4 Environmental profiles for different farm types

In the attempt to perform a comprehensive as possible analysis of the environmental performance, we need to combine the results from different environmental impacts, as well as in respect of the different functions of agriculture. This leads us to the concept of environmental profiles for the assessed farm types. Based on the management triangle described in [10] impact categories can be grouped in three main axes: resource use-driven impacts, nutrient-driven impacts, and pollutant-driven impacts. In the environmental profile each corner represents one axis: GWP stands for the resource use-driven impacts, eutrophication potential (EutP) represents the nutrient-driven impacts, and terrestrial ecotoxicity (TTox) typifies the pollutant-driven impacts. The closer the corner to the centre of the environmental profile triangle, the smaller is the environmental impact of the respective category. The environmental profile triangle is normalised, meaning the maximum value per impact category over all farm types is set at 1. All other values (for the remaining farm types) are a fraction of 1.

3 Results

3.1 Variability of environmental impacts per ha UAA

For the four examined impact categories, i.e. energy demand, GWP, eutrophication potential, and terrestrial ecotoxicity an important variability per ha UAA was observed. For the energy demand the range between the farm with the lowest and the highest demand comprised a factor of 22. Regarding GWP and eutrophication potential the range was a factor of 19 and 13, respectively. For terrestrial ecotoxicity a factor of 106 was obtained. This large variability suggests that there is room for improvement among the Swiss farms.
The example of the energy demand per ha UAA for the different farm types shows an important difference between the medians of the different farm types (Figure 1). E.g. the median of 'combined pigs/poultry' is 2.6 times higher than the median of 'combined suckler cows'. This considerable difference expresses the differing energy requirements subject to varied production modes. Still, the variability between the medians of the different farm types is smaller than within some of the farm types. The above described variability between the medians and within the farm type can be found as well for the three other environmental impacts per ha UAA. This again highlights the optimisation potential for the individual farm.

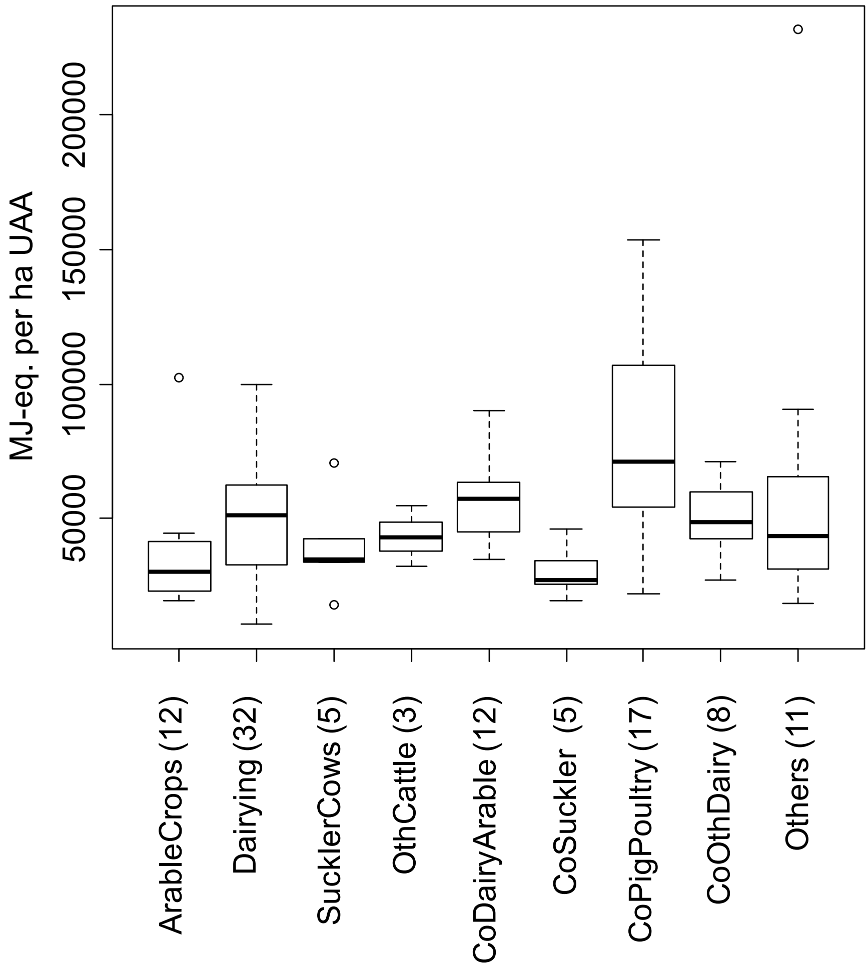

Fig. 1: Energy demand per ha UAA (utilised agricultural area) for different farm types (according to [3]) - 'ArableCrops' = Arable crops; 'Dairying' = Dairying; 'SucklerCows' = Suckler cows; 'OthCattle' = Other cattle; 'CoDairyArable' = Combined dairying/arable; 'CoSuckler' = Combined suckler cows; 'CoPigPoultry' = Combined pigs/poultry; 'CoOthDairy' = Combined others/dairying; 'Others' = other types. In brackets: number of farms per type

3.2 Environmental profiles for different farm types

By combining the LCA results for three different impact categories representing each one of the management axes resource use, nutrient, and pollutant management subject to the three assessed functions, i.e. land management, food and feed production, and provision of income we obtained the environmental profiles of the examined farm types (see chapter 2.4).

We identified four types of environmental profiles (Figure 2):
i) A rather favourable environmental profile, i.e. low environmental impacts were achieved for all three functions. This is the case for 'combined suckler cow'. The combination of low input meat production with arable cropping seems to be comparatively favourable regarding the environmental performance.
ii) A profile with favourable impacts regarding the function of land management, but unfavourable for the productive and financial function, as for 'suckler cows' and 'other cattle'. Here the areas of conflicts are revealed between low input farming and a comparatively low food production as well as a low gross revenue.
iii) A profile with favourable impacts for the productive function, but unfavourable for the financial function and the function of land management. This profile was found for the types 'combined dairying/arable', 'combined pigs/poultry', and 'combined others/dairying'. This is the opposite to the second type of profile. However, here as well the areas of conflict between the three examined functions are disclosed. These types have a comparatively high productive output, but at the same time their environmental impact per gross revenue or per UAA is also high. Finally, a subtype of this profile is the case for 'arable crops' and 'dairying': Both showed a need for action in two out of three management axes calling for targeted measures.

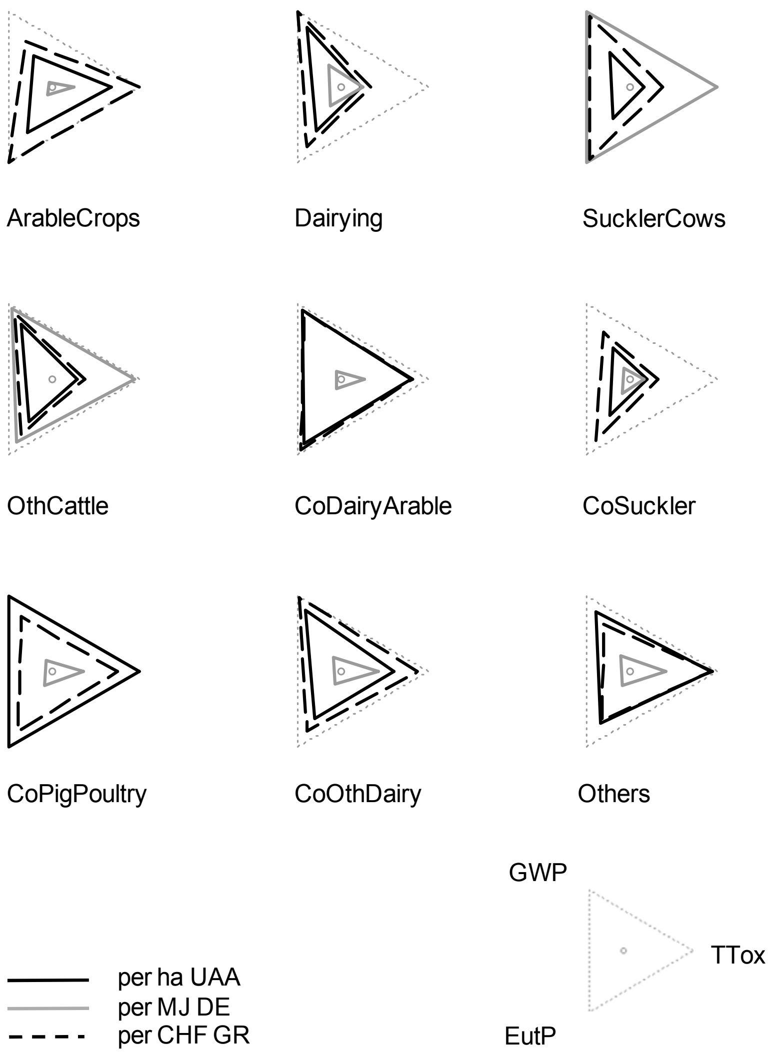

Fig. 2: Environmental profiles for the 3 functions land management, productive function and financial function for different farm types - 'ArableCrops' = Arable crops; 'Dairying' = Dairying; 'SucklerCows' = Suckler cows; 'OthCattle' = Other cattle; 'CoDairyArable' = Combined dairying/arable; 'CoSuckler' = Combined suckler cows; 'CoPigPoultry' = Combined pigs/poultry; 'CoOthDairy' = Combined others/dairying; 'Others' = other types. ha UAA = hectares of utilised agricultural area; MJ DE = mega joule digestive energy; CHF GR = gross revenue in Swiss francs. GWP = global warming potential; EutP = eutrophication potential; TTox = terrestrial ecotoxicity

3.3 Need for action according to farm type

On the basis of the environmental profiles different spheres of action can be identified for each single farm type (Table 2).
Exemplarily the case of the combined farm types (with the exception of 'Combined suckler cows') shall be discussed. Their weaknesses in the environmental analysis were found regarding the function of land management and the financial function. In order to overcome these weaknesses two lines of action can be taken: i) increasing the productive output with an unchanged environmental impact or ii) reducing the environmental impact while keeping the productive output stable. Both strategies should also have an impact on the financial function as either the income should increase or the costs should diminish.

Tab. 2: Identified spheres of action for the different farm types

Farm type	Spheres of action
Arable crops	Nutrient and pollutant management
Dairying	Resource use and nutrient management
Suckler cows Other cattle	Resource use and nutrient management, including an increase of the production performance
Combined dairying/arable crops Combined pigs/poultry Combined others/dairying	All environmental impacts, especially regarding the function of land management as well as the financial function
Combined suckler cows	Nutrient management and to a lesser extent resource-use management

Improvement measures for the example of the type 'combined pigs/poultry'
Within the farm type 'combined pigs/poultry' the sources of the environmental impacts were assessed. Over the 17 farms belonging to this type, the input groups *purchase of feedstuffs* and *purchase of animals* proved to have an important share in all four assessed impact categories, i.e. energy demand, GWP, eutrophication potential, and terrestrial ecotoxicity. Furthermore the standard deviations of those two input groups were high in all four impact categories. The combination of these two characteristics, i.e. importance and variability of certain inputs, indicates that there is room for improvement in those input groups, as some farms specialised in pig or poultry fattening were able to produce more environmentally friendly than others. To a lesser extent, this is also valid for the input groups *energy carriers*, *fertilisers/nutrients*, as well as *animal emissions*.
This analysis was also performed for the other farm types with more than 10 participating farms, i.e. 'arable crops', dairying', and 'combined dairying/arable

crops'. The overall assessment showed that the input groups *energy carriers* and *fertilisers/nutrients* are of importance with a high variability. Furthermore, for all farm types with animal production the input group *purchase of feedstuffs* as well as *animal emissions* were important.
So in order to enhance the environmental performance of the assessed farm types, improvement measures in the area of the identified input groups have to be taken.

4 Conclusions

There is an important variability of the environmental impacts between the farm types and within the farms of the same type. This indicates that there is room for improvement of the environmental performance. However, an environmental optimisation over all three assessed functions, i.e. land management, food and feed production, and provision of income is difficult to achieve. Areas of conflict are present especially between a conserving land management and a high productive output (e.g. farm types 'suckler cows' or in an opposite way 'combined pigs/poultry'). The developed environmental profiles allow to identify the need of action within the different farm types in order to obtain an environmental optimisation. These environmental hotspots are a key leading us to a directed analysis on the influencing factors for the environmental performance. The assessment of the most common farm types has revealed that the use of energy carriers, fertilisers/nutrients, and feedstuffs, as well as direct animal emissions are major influencing factors regarding the environmental performance of Swiss farms. Still, as there is a considerable variability of the production mode between farms of the same farm type, it remains difficult to express commonly applicable recommendations. Therefore the most promising way of reducing the environmental burden of farms is through an individual analysis leading to targeted improvements.

References

[1] Foster C, Green K, Bleda M, Dewick P, Evans B, Flynn A, Mylan J (2006) Environmental impacts of food production and consumption: A report to the Department for Environment, Food and Rural Affairs. Manchester Business School, Defra, London

[2] Hersener J-L (2011) Zentrale Auswertung von Ökobilanzen landwirtschaftlicher Betriebe. Forschungsanstalt Agroscope Reckenholz-Tänikon ART, Zürich, Schweiz. (in preparation)

[3] Meier B (2000) Neue Methodik für die Zentrale Auswertung der FAT, Methodische Grundlagen der Zentralen Auswertung von Buchhaltungsdaten. Forschungsanstalt Agroscope Reckenholz-Tänikon ART, <http://www.agroscope.admin.ch/betriebs-wirtschaft/04362/04363/04379/index.html?lang=de> (accessed 15.04.2011).

[4] Nemecek T, Freiermuth Knuchel R, Alig M, Gaillard G (2010) The advantages of generic LCA tools for agriculture: examples SALCAcrop and SALCAfarm. Proceedings of the 7th Int. conference on life cycle assessment in the agri-food sector, Bari, Italy.,22-24 September 2010: 433-438.

[5] Frischknecht R, Jungbluth N, Althaus H-J, Doka G, Hellweg S, Hischier R, Nemecek T, Margni M, Spielmann M (2004) Implementation of life cycle assessment methods - ecoinvent data v1.1. Swiss Centre for Life Cycle Inventories (ecoinvent), Dübendorf,

[6] IPCC (2007) Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, USA

[7] Hauschild M, Wenzel H (1998) Environmental assessment of products - Vol. 2: Scientific back-ground. 2nd edn. Chapman & Hall, London, UK

[8] Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, van Oers L, Wegener Sleeswijk A, Suh S, Udo de Haes HA, de Bruijn H, van Duin R, Huijbregts MAJ, Lindeijer E, Roorda AAH, Weidema BP (2001) Life cycle assessment - An operational guide to the ISO standards. Ministry of Housing, Spatial Planning and Environment (VROM) and Centre of Environmental Science (CML), Den Haag and Leiden, Netherlands

[9] <http://www.r-project.org/> (accessed 15.04.2011).

[10] Nemecek T, Huguenin-Elie O, Dubois D, Gaillard G (2005) Ökobilanzierung von Anbausystemen im Schweizerischen Acker- und Futterbau. Agroscope FAL Reckenholz, Zurich, Switzerland

The Use of Models to Account for the Variability of Agricultural Data

Brigitte Langevin, Laurent Lardon and Claudine Basset-Mens

Abstract LCA outputs are often presented as point estimates measuring potential impacts although average impacts values may be misleading to rank different options, especially in the case of agricultural products. In an LCA study comparing different slurry application techniques, NH_3 and N_2O emissions have been estimated through two approaches, experimental data collected from the literature and mathematical simulations over different soil and climate conditions. Both approaches lead to similar ranges of emissions; however the simulation-based approach allows us to construct a probability distribution of emissions whereas the limited number of experimental studies leads only to the definition of a range of emissions. A better knowledge of the variability of emissions helps the practitioner to sort alternatives and to detect situations where they are not discernable. Moreover the knowledge of the distribution and of its most impacting sources of variability leads to the definition of more informative and significant typologies.

1 Introduction

Uncertainty is pointed out as one of the major challenges of life cycle assessment [1] although being generally ignored. However the precision and the representativeness of inventory data condition the credibility of a single value used to estimate an emission to the environment. Moreover, reasoning with point estimates only may be misleading to rank different options with LCA [2]. This is particularly true for unit functions where an agricultural phase is involved and represents a major source of impact. Agricultural production is indeed a major source of uncertainty since most emissions are diffuse and often difficult to measure directly or

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are too delayed. Moreover the high variability of soil and climate conditions but also of agricultural practices induces a corresponding variability of emissions.
This paper focuses on the environmental assessment of slurry application. Four spreading techniques are compared regarding the application of 100 kg of nitrogen from slurry over 1 ha, which is the functional unit of the study. The four techniques are broadcast spreading, e.g. with a splash plate, band spreading, e.g. with trailing hoses, surface application followed by an incorporation of the slurry with a harrow and direct injection. The former two are surface applications, while in the latter two, a part of the slurry is applied below the soil surface. A former analysis [3] has demonstrated that most of the impact is associated with the nitrogen emissions, whereas the building and operating of the spreading machines have a relatively negligible impact. When spread on the soil, a significant part of the slurry ammonium (NH_4^+) in contact with the atmosphere volatilised while the remaining ammonium is nitrified or directly assimilated by plants. Nitrate (NO_3^-) can be taken up by plants, denitrified into N_2, or leached into ground-water and rivers. Nitrous oxide (N_2O) is an intermediate product of the denitrification sequence and a by-product of nitrification, and may leak to the atmosphere. Ammonia (NH_3) volatilisation is responsible for acid rains. N_2O is a potent greenhouse gas with also a deleterious effect on the ozone layer. Moreover, all reactive species of nitrogen contribute to ecosystems eutrophication [4].

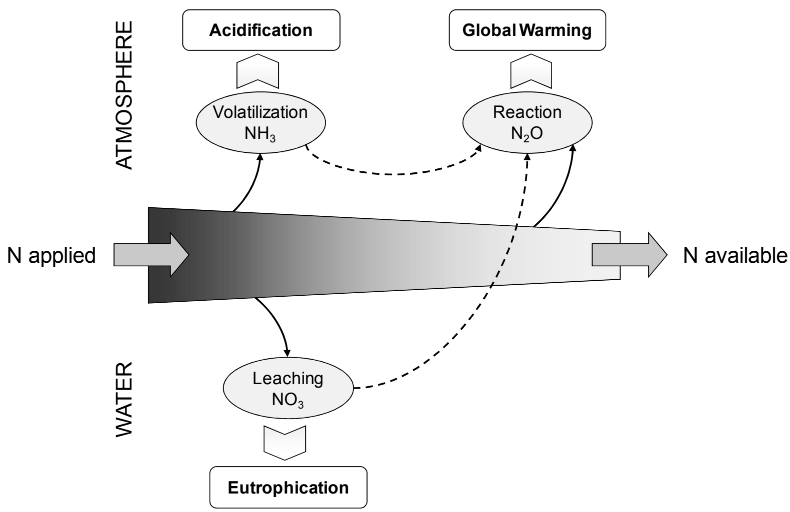

Fig. 1: Nitrogen losses and associated environmental impacts (plain lines: direct emissions, dashed lines: indirect emissions)

This simplified description of nitrogen emissions in soils (Figure 1) shows how emissions of NH_3, NO_3^- and N_2O are the result of several interconnected phenomena that depend on a large number of parameters such as the exchange surface between the slurry and the atmosphere, soil pH, the occurrence of anoxic conditions, air moisture or temperature. Several of these parameters are related to the soil and climate conditions only, whereas others mostly depend on the application technique.

This study aims to compare two approaches to perform an LCA for agriculture, either based on emission data collected in the literature or based on emissions predicted with simulation models.

2 Estimation of emissions based on a literature review

A literature review allows one to collect field measurements of ammonia, nitrous oxide, and nitrate emissions following manure application. However as these emissions are highly sensitive to soil and weather conditions, a simple collection of results might lead to unfair comparison between techniques. Hence only studies comparing several application techniques in the same conditions have been selected and emissions are expressed here in terms of a relative factor compared to a nominal technology (band spreading) [3]. This data collection exhibited a variability of emissions for different techniques which is the result of both the inner variability of the measurements (i.e. two identical observations will give two different measurements) and the diversity of environmental parameters controlling the emissions (e.g. weather, soil texture…). Figure 2 shows that the variability for each technique is often higher than the differences between their average values. We had not enough information to assess the representativeness of the studied sites in order to derive a probability distribution. Hence, all measured data were considered equally probable and interpreted in terms of intervals. Yet, when comparing minimal, maximal and average values, it seems still possible to rank application techniques according to their NH_3 emissions (Figure 3); however the ranking of techniques regarding N_2O emissions is more debatable due to their extreme and average values. This example demonstrates that the variability of an emission can be as important as its average value and that without a proper analysis of the structure of this variability, it could be impossible to rank different options. Such *extrema* able to affect the credibility of a ranking can correspond to a specific case; hence a meaningful typology could be achieved by excluding this case from the average class (and by creating a new class for it). A solution to overcome this issue is to use simulation models to assess the origins of this variability.

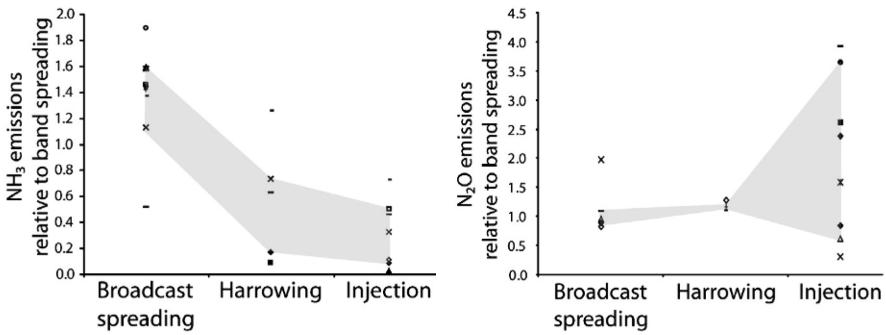

Fig. 2: Distribution of NH_3 (left) and N_2O (right) emissions expressed as relative factors of band spreading emissions (each point represents one experiment, grey areas are the corresponding likely emission ranges)

3 Modelling and simulation of field emissions

3.1 Model structure

When using a simulation model to forecast emissions related to different scenarios, one has to make sure the model is of course the most accurate but also that it is able to show differentiated answers to the different scenarios one wants to assess. We used the model OSEEP [5], which combines models from soil physics, agroclimatology and agronomy to simulate the effect of slurry application techniques on nitrogen emissions at the field scale. Figure 3 shows an overview of the combination between these different models. STICS [6] is a model able to simulate the growth of a crop and the fate of nitrogen in the soil. Indeed nitrogen emissions cannot be properly described independently from the continuous uptake by plants and the turnover of soil organic matter. Moreover in order to provide a description of NH_3 volatilisation accounting for the effect of slurry incorporation, the volatilisation module of STICS was replaced by the Volt'Air model [7,8]. Both models use parameters relative to hydraulic transfers. Those parameters being directly affected by engine traffic, it was necessary to add the CompSoil model [9] describing soil compaction. The effect of soil compaction on hydraulic transfer parameters and soil compaction was modelled through pedotransfer functions [10,11]. The developed model provides simulations of several nitrogen flows to the environment and the growth of the culture. Nitrate emissions are simulated as well by OSEEP but the delay of its emission to the environment makes difficult its attribution to a specific activity in the cultural practices. Hence for the sake of clarity and concision, we will focus here on NH_3 and N_2O emissions only.

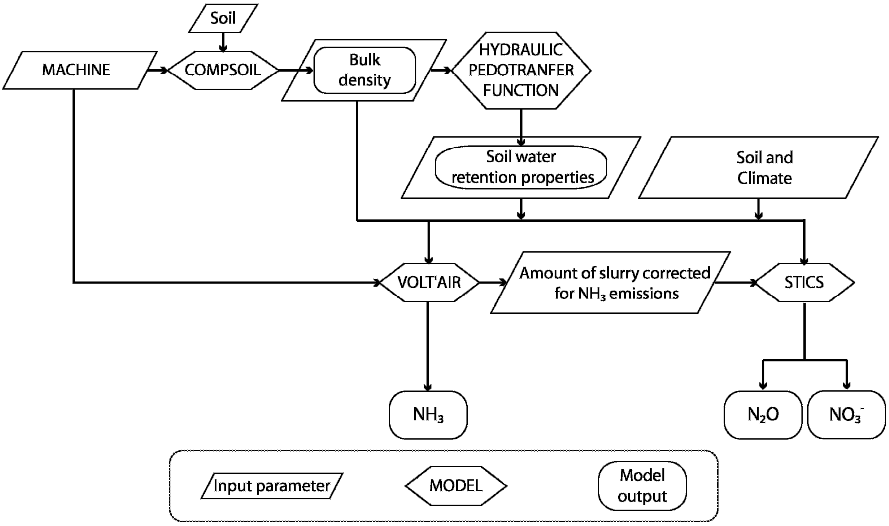

Fig. 3: Architecture of the simulation model OSEEP

3.2 Comparison of emissions ranges

The simulated emissions were calculated for 5 sites in France for 7 years. The same four slurry application techniques studied in the literature-based approach were compared: broad spreading, band spreading, harrowing after surface application and direct injection. Figure 4 compares the distribution of NH_3 emissions of each technique over the whole set of climate and soil conditions. It can be seen that variability of soil and climate conditions is able to create a large variability of emissions. However, the shapes of the distribution make it possible to differentiate technical options independently of local situations, and to explain it by the effect of incorporation depth. Moreover comparison between emissions intervals determined by the literature analysis and distributions determined by simulations (Figure 5) shows consistent results. This tends to demonstrate that the simulation models are properly tuned and that the variability covered by the literature was of the same order to the one used for the simulations.

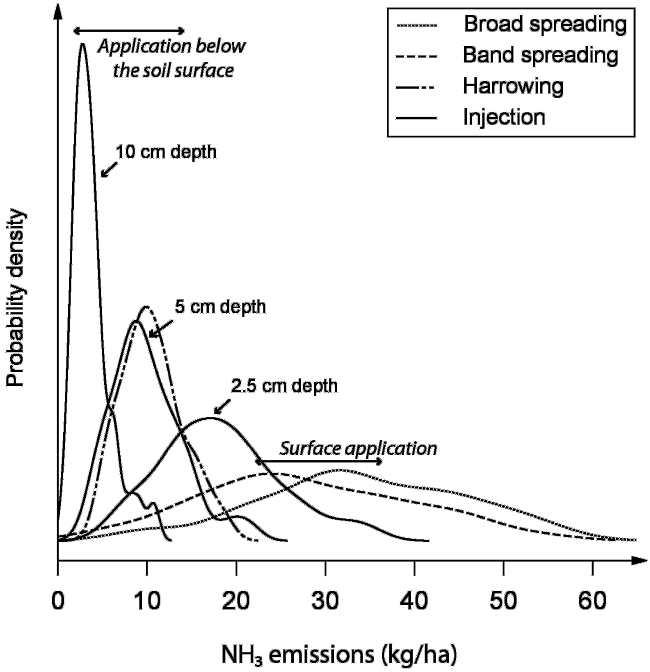

Fig. 4: Distribution of NH_3 emissions for the slurry application techniques

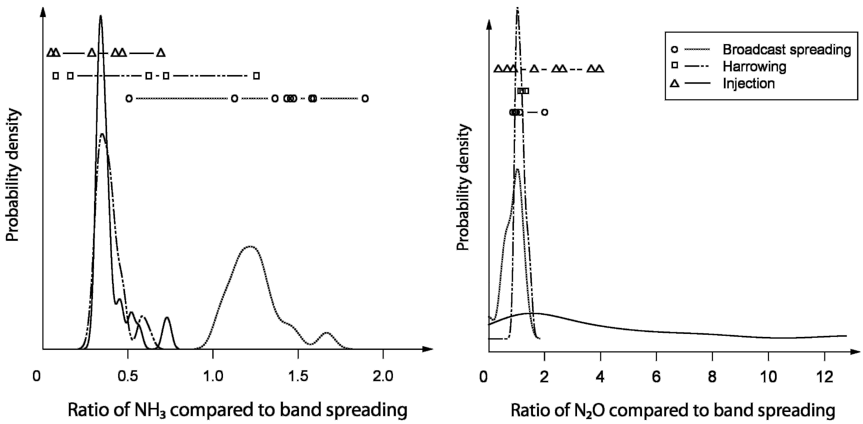

Fig. 5: Simulated density functions (curved lines) and ranges of relative factors calculated from the literature review (symbols and straight lines on top) for NH_3 (left) and N_2O (right)

4 Discussion

The ranking based on simulations is more conclusive than that obtained with literature data only. Indeed, we assumed uniform distributions for literature data, which is a conservative approach but that renders the options less discernable. Regarding the uncertainty analysis of the results, models are more instructive in allowing the calculation of more specific distributions and typologies of situations in which the studied alternatives can be reliably compared. Yet, uncertainty obtained by simulation is constrained by the variability of the situations selected. The collection of relevant and representative soil and climate parameters can require a substantial effort. Though as each input parameter is known, it is easier to compare results, conversely to data collected from publications where some details can be missing or where too many parameters are simply unknown.

Both approaches have their specific advantages and drawbacks. Regarding the time allowed to the life cycle inventory, the literature-based approach is easier to perform whereas the use of models can be very time-consuming to be properly handled. Regarding the calculation of average values, both approaches entail risks of error and sampling issues. On one hand, literature data may lead to false results due to sampling bias, especially in the case of agricultural data, which are highly variable. On the other hand, simulated data may also lead to false average values due to the simplification of the model or the errors attached to the determination of its input parameters.

As a conclusion, the use of literature data is more simple to handle and may be sufficient to draw robust conclusions if the whole range of values is propagated into discernable LCA outputs intervals; if there is a need to refine the comparison between the studied alternatives, the use of advanced simulation models is then required.

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References

[1] Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, Hellweg S, Koehler A, Pennington D, Suh S (2009) Recent developments in Life Cycle Assessment. J Environ Manage 91(1):1-21.

[2] Björklund AE (2002) Survey of approaches to improve reliability in LCA. Int J Life Cycle Assess 7(2):64-72.

[3] Langevin B, Basset-Mens C, Lardon L (2010) Inclusion of the variability of diffuse pollutions in LCA for agriculture: the case of slurry application techniques. J Cleaner Prod 18(8):747-755.

[4] Galloway JN, Cowling EB, Seitzinger SP, Socolow, RH (2002) Reactive nitrogen: Too much of a good thing ?. Ambio 31(2):60-63.

[5] Langevin B (2010) Prise en compte de la variabilité des émissions au champ dans l'Analyse de Cycle de Vie des systèmes agricoles. Application à l'épandage de lisier. Dissertation, École Nationale Supérieure d'Arts et Métiers, Paris

[6] Brisson N, Mary B, Ripoche D, Jeuffroy MH, Ruget F, Nicoullaud B, Gate P, Devienne-Barret F, Antonioletti R, Durr C, Richard G, Beaudoin N, Recous S, Tayot X, Plenet D, Cellier P, Machet JM, Meynard JM, Delécolle R (1998) STICS: A generic model for the simulation of crops and their water and nitrogen balances. I. Theory and parameterization applied to wheat and corn. Agron Sust Dev 18(5-6):311-346.

[7] Génermont S, Cellier P (1997) A mechanistic model for estimating ammonia volatilization from slurry applied to bare soil. Agricultural and Forest Meteorology 88(1-4):145-167.

[8] Le Cadre E, (2004) Modélisation de la volatilisation de l'ammoniac en interaction avec les processus chimiques et biologiques du sol: le modèle Volt'Air. Dissertation, INA-PG, Paris, France

[9] O'Sullivan MF, Henshall JK, Dickson JW (1999) A simplified method for estimating soil compaction. Soil Tillage Res 49(4):325-335.

[10] Wösten JHM, Lilly A, Nemes A, Le Bas C (1999) Development and use of a database of hydraulic properties of European soils. Geoderma 90(3-4):169-185.

[11] Schaap MG, Leij FJ, Van Genuchten MT (2001) Rosetta: A computer program for estimating soil hydraulic parameters with hierarchical pedotransfer functions. J Hydrol 251(3-4):163-176.

Modular Extrapolation Approach for Crop LCA MEXALCA: Global Warming Potential of Different Crops and its Relationship to the Yield

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Abstract MEXALCA (Modular EXtrapolation of Agricultural LCA) extrapolates crop inventory data and impacts from an original country inventory to all producing countries worldwide. This allows estimates of worldwide means weighted by production volumes and of the environmental impact distribution. In this paper, the relationship between the yield and the environmental impacts is analysed in order to test whether the yield alone can be used as an extrapolation criterion. The results show that the global warming potential (GWP) per kg decreases with increasing yields for the means of the 27 studied crops. When comparing the production of a crop in different countries, the relationship between GWP per kg and yield exists only for those crops where the contribution from basic cropping operations and tillage to the GWP is significant. Considering the yield alone therefore generally allows only a poor approximation of the GWP.

1 Introduction

Businesses wishing to analyse the environmental impacts of their products using life cycle assessment (LCA) methods increasingly require large amounts of very detailed data which are rarely readily available. This is particularly true for companies operating global and rapidly changing supply chains with a large range of products and ingredients originating from all over the world. Thus, there is an urgent need for data that are sufficiently reliable while being supplied without extensive, time consuming and costly collection. Furthermore, the variability of environmental impacts of agricultural and food products can be considerable [1]. Therefore, alternative cost-effective approaches are required to estimate the ranges

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of impacts of different crops. Milà I Canals et al. [2] presented two different approaches to filling data gaps: use of proxy data and extrapolation. Proxies are data describing an alternative product, considered to have similar impacts as the product under study. Milà I Canals et al. [2] distinguished scaled proxies, direct proxies and averaged proxies. Using proxy data means that the original values are used without transformation, beyond statistical calculation like averaging. Extrapolation means on the contrary that the original values are transformed in some way. For example, key parameters like the yield for crop production or the feed conversion ratio are used to adapt the original dataset. Another possibility is to take into account a number of influencing factors.

For both approaches - using proxies and extrapolation - we need to understand the most critical factors influencing the environmental impacts of a crop product: pedoclimatic conditions, affecting the direct field emissions and also the use of inputs like fertilisers and irrigation water, farming practice, prevalence of pest and diseases, etc. All these factors also influence the crop yield [1]. The latter is shown to be a major factor influencing the environmental impacts per product unit. In this paper we use the extrapolation method MEXALCA (Modular EXtrapolation of Agricultural LCA, [3]) aiming at extrapolating life cycle inventories and impacts between different geographies in order to analyse the relationship between the yield and the environmental impacts and to show in which cases the yield can be a sufficient criterion for selecting proxies and extrapolation.

2 Extrapolation methodology MEXALCA

Detailed data are required for at least one typical production system that is then used as the baseline for the extrapolation to all other (target) countries in the world producing the same crop [3]. Ideally this should be a representative dataset for a large producing country and extreme situations should be avoided. This base country life cycle inventory is split into nine modules corresponding to the main farming operations and inputs known to dominate the environmental impacts of crop cultivation (Table 1). With the exception of the basic cropping operations, which are assumed to be constant worldwide, each module is varied as a function of the agricultural intensity index and of the yield (except soil tillage, since it is assumed that no-till agriculture does not significantly change the yield [4]). Mathematical functions relating global statistics of crop yields and agricultural production intensity (mainly FAOSTAT) in the baseline and target countries, respectively, are applied for the geographical extrapolation for each modular farming input. The extrapolation provides estimates of environmental impacts for each producing country; the variability within a country due to regional differences, farming systems, etc. is not considered. The worldwide distribution is obtained by weighting the single country values by the respective production volumes. The first version of the extrapolation methodology and validation using data from the ecoinvent database is described in [3]. The method has been further applied to a number of other crops [5]. Moreover, the method has been adapted by using the same estimator for irrigation as for tillage, variable machinery use, fertiliser and pesticide use.

Tab. 1: Overview of the modules of MEXALCA

Category	Module	Impacts	Input parameter
General	Basic cropping operations	Constant per area unit	-
Machinery	Tillage	Per ha (area unit)	% of no-till area
Machinery	Variable operations	Per ha (area unit)	Mechanisation index
Fertilisation	N fertilisation, including N-emissions	Per kg N applied (input mass unit)	kg N applied
Fertilisation	P fertilisation, including P-emissions	Per kg P_2O_5 applied (input mass unit)	kg P_2O_5 applied
Fertilisation	K fertilisation	Per kg K_2O applied (input mass unit)	kg K_2O applied
Plant protection	Pesticide application	Per kg pesticide (active ingredient) applied (input mass unit)	kg active ingredient
Irrigation	Irrigation	Per m^3 of irrigation water supplied (input volume unit)	m^3 water used
Product drying	Product drying	Per kg of evaporated water (mass unit)	kg water evaporated

To include the effect of land use change, potential emissions from deforested land were included by applying an approximation model, where the total above ground biomass of cleared forests is calculated as an emission of CO_2, while changes in soil organic matter are not considered. The emissions of deforestation are allocated to 100% to the agricultural area (arable land + permanent crops + pastures), other driving factors for deforestation (like timber production) are neglected. The emissions in a given country were distributed to the country's total agricultural area, which means that each ha of land occupied in that country carries the same burden from deforested land, not taking into account the fact that some crops may be grown more on deforested land than others. The result therefore depends on the countries where the considered crop is grown. They indicate potential risks, but it is a rough model and the results must thus be interpreted with caution. Carbon fixation by crops was not considered, although this may arguably be relevant for plantations.

The reliability of the extrapolation results mainly depends on numerous factors like modelling assumptions, factors not considered, and the quality of the underlying data (production inventories, choice of original countries, yield statistics, statistics of agricultural input use) [3].

3 Contribution of modules to the global warming potential

The method was applied to 27 crops [5]. MEXALCA results were validated by using data from the ecoinvent database [6] and using literature data for non-renewable energy demand and global warming potential (GWP). The validation results are reported in [5]. The method performs reasonably well for the impact categories non-renewable energy demand, GWP and ozone formation potential as well as land occupation.

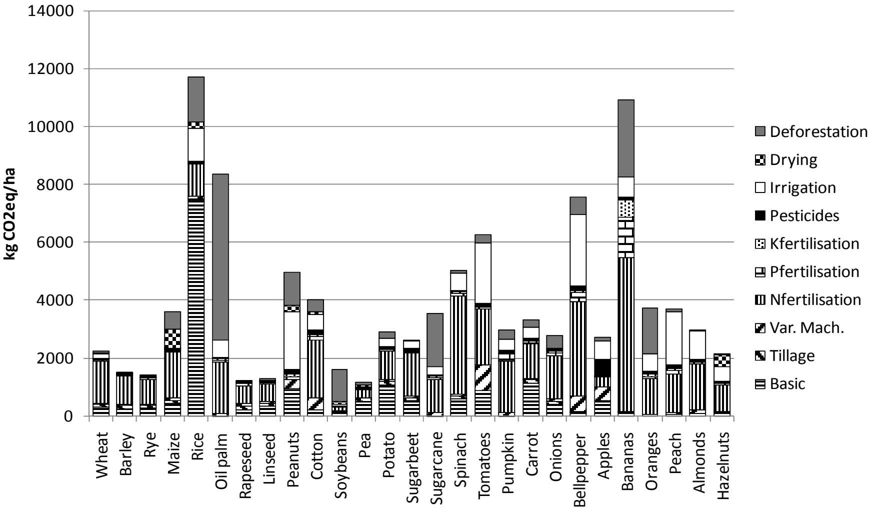

Fig. 1: Worldwide means of the global warming potential per ha and growing season of 27 crops weighted by production volumes, showing the contribution of the modules and the potential effects of deforestation.

The contribution of the nine MEXALCA modules as well as of deforestation to the GWP is shown in Figure 2 per ha and growing season (in case of permanent crops it is one year) and in Figure 3 per kg product (fresh matter). The highest

GWP per ha was found in rice (mainly due to the methane emissions from rice fields that are included in the basic cropping operations), followed by bananas, oil palm, bell peppers and tomatoes. Low GWP values per ha were found for pea, rapeseed, linseed and cereals. Soybean would have the lowest value of all crops without consideration of deforestation. On a per kg basis the pattern was different due to the very different yields. Rice had still a very high impact, but peanuts were at a similar level (mainly due to low yields). The lowest GWP values were found for the sugar crops (sugar cane and sugar beet). We have to consider that the results are given per kg of fresh mass in FAOSTAT (and this is also the unit in which the traded commodities are expressed) and that the dry matter content differs considerably between crops.

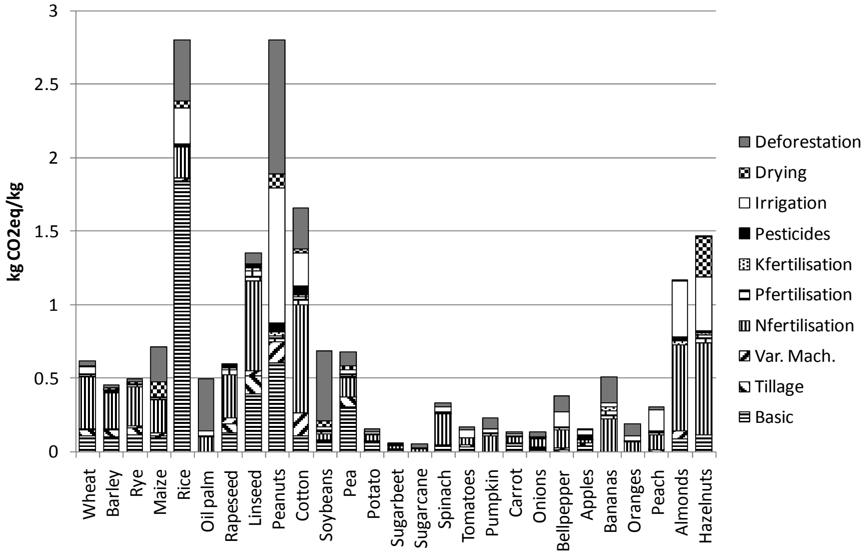

Fig. 2: Worldwide means of the global warming potential per kg product of 27 crops weighted by production volumes, showing the contribution of the modules and the potential effects of deforestation.

The contributions from deforestation varied considerably between the crops. The relative contributions from deforestation to the GWP (Figure 4) were highest for oil palm (mainly Indonesia and Malaysia) and soybeans (mainly Brazil), two crops much discussed with respect to land use change issues. Sugar cane could also have a considerable contribution, which is due to the fact that 32% of the production is located in Brazil. For oranges, the main contributions come from Brazil and Indonesia. For peanuts, Indonesia, Nigeria and Myanmar contributed most to this impact. The deforestation impact for onions was dominated by Indonesia, Myanmar and Brazil; for bell peppers Indonesia played the major role. For pumpkin this was due to Indonesia, Cameroon and Philippines; for maize to Brazil and Indonesia. Without the inclusion of land use change the N fertilisers (and related emissions) irrigation and basic cropping operations had the highest impact for most crops.

4 Relationships between yield and environmental impacts

4.1 Comparison between crops

The effects of the yield were investigated first for the worldwide means of the GWP of 27 crops weighted by production volumes (as presented in Fig. 1 and Fig. 2). All results presented in this subchapter are excluding the deforestation effects, since the latter depends only on the producing countries and by definition has no relationship with the yield.

The GWP per ha (weighted worldwide mean) was not dependent solely on the yield (Fig. 3). There was a high variability of GWP of crops with low yields on the one hand; high yielding crops on the other hand did not necessarily have high GWP impacts. Per kg there was a clear dependence on the yield: The higher the yield of a crop on average, the lower the GWP per kg. The correlation coefficient between the GWP per kg and the inverse of the yield (ha/kg) was r^2=0.39, i.e. the yield explained almost 40% of the variance. The weighted means were less variable per ha (coefficient of variation, CV=71%) than per kg (CV=107%).

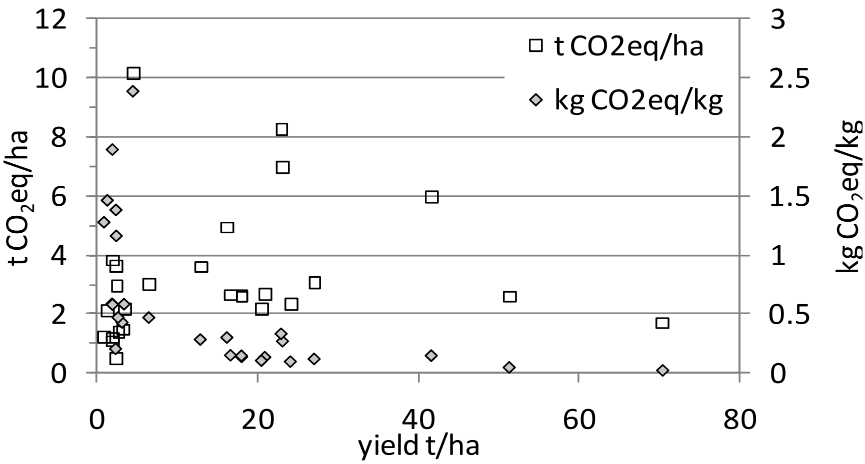

Fig. 3: Worldwide means of GWPs of 27 crops per ha weighted by production volumes and per kg as a function of the yield.

4.2 Comparison between geographies

In a second step the yield effects were investigated for all country values of the selected crops. Only countries with a contribution of at least 0.1% to the worldwide production were considered, since the yield data from countries with marginal productions appeared to be unreliable. For all investigated crops, the GWP per ha increased with the yield. A higher yield generally correlates with a more intensive production and higher use of inputs like fertilisers or pesticides. Wheat is such an example of a crop, where we can observe the increase of GWP per ha, while there was a slightly increasing trend for the GWP per kg with increasing yield (Fig. 4).

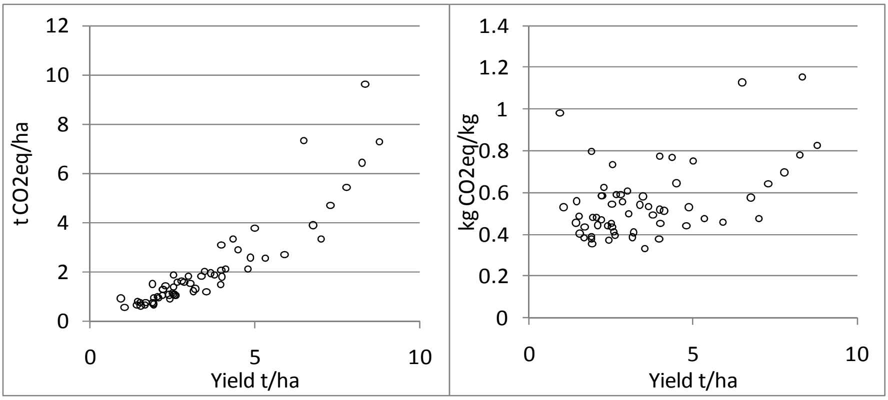

Fig. 4: Global warming potential of wheat per ha (left) and per kg (right) as a function of the yield for all producing countries with a share >0.1% of the worldwide production volume.

A different type of relationship where GHG is inversely correlated with the yield per ha was found for a second group of crops where the basic cropping operations had a major contribution. This case is illustrated by pea (Fig. 5). The basic cropping operations are assumed constant worldwide, which - together with a low yield - results in a high impact per kg.

In general the GWP was more variable per ha than per kg (variability as expressed by the coefficients of variation, weighted worldwide standard deviation divided by the weighted worldwide mean).This means that higher GWP per ha of more intensively managed crops are partly compensated by the higher yields, when calculated per kg. The results presented in this paper are model outputs and therefore dependent on the model assumptions. Indeed, the same kind of analysis results in similar findings, when applied to values derived from the literature.

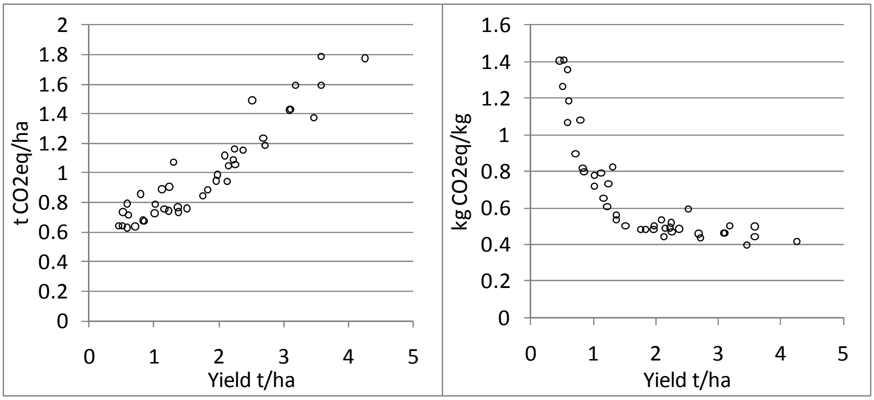

Fig. 5: Global warming potential of pea per ha (left) and per kg (right) as a function of the yield for all producing countries with a share >0.1% of the worldwide production volume.

5 Discussion and conclusions

The systematic application of MEXALCA to a range of crops has allowed us to generate efficiently impact values representing estimates for production around the world. With these estimates, relationships between impacts and production parameters may be established which may shed light in e.g. guiding the choice of proxy data or extrapolating datasets in order to bridge data gaps.
When we analyse the relationships between yield and GWP we need to distinguish the difference between crops and within crops on the one hand and the considered functional unit on the other hand, namely ha and kg. The analysis of the weighted means of different crops showed no trend per ha, but a decreasing trend per kg. This implies that when we use GWP impacts from a similar crop as a surrogate for a crop for which data are missing, the yield should be included in the extrapolation model for the environmental impacts per kg of product. However, 60% of the variance was due to other factors than the yield.
Comparing GWP of the same crop produced in different countries, we found an increase of the GWP per ha with increasing yield per ha, which is generally related to higher farming intensity. For the GWP per kg, no relationship with the yield was observed for most crops. Other factors than the yield seem to be more relevant: the ratio between inputs or emissions on the one hand and the yield on the other hand is determining the impacts together with pedoclimatic conditions, the farming system, etc. For crops, where the GWP is mainly caused by the basic cropping operations and tillage, the GWP per kg decreases with increasing yield.

For other impact categories, the situation might be different: land use related impacts for example are generally related to the inverse of the yield.
Extrapolation with the yield alone is likely to provide good estimates only in a few situations, namely when the contribution from basic cropping operations and tillage to the GWP is significant. In fact, when the impacts from a crop are to be used as a proxy for another crop, the yield should be considered. For the extrapolation between producing countries of a given crop, the GWP decreased for those crops where the area-related impacts dominate, while for the other crops either no trend or a slight increase was observed. A case by case analysis is therefore required. Low yields do not necessarily lead to high environmental impacts per kg.

References

[1] Nemecek T, Kägi T (2008) Ecoinvent-based extrapolation of crop life cycle inventories to new geographical areas. In: Proceedings of the 6th International Conference on LCA in the Agri-Food Sector – Towards a sustainable management of the Food chain, Zurich

[2] Milà i Canals L, Azapagic A, Doka G, Jefferies D, King H, Mutel C, Nemecek T, Roches A, Sim S, Stichnothe H, Thoma G, Williams A (2011) Approaches for addressing Life Cycle Assessment data gaps for bio-based products. J Ind Ecol.In press.

[3] Roches A, Nemecek T, Gaillard G, Plassmann K, Sim S, King H, Canals LMi (2010) MEXALCA: a modular method for the extrapolation of crop LCA. Int J Life Cycle Assess 15:842-854.

[4] Chervet A, Ramseier L, Sturny W, Tschannen S (2005) Direktsaat und Pflug im 10-jährigen Systemvergleich. AgrarForschung 12(5):184-189.

[5] Nemecek T, Weiler K, Plassmann K, Schnetzer J, Gaillard G, Jefferies D, García–Suárez T, King H, Milà i Canals L (2011) Globally averaged GWP and its Variability for a Multitude of Crops as estimated with a Modular Extrapolation Approach (MEXALCA).In prep.

[6] Ecoinvent Centre (2007)Ecoinvent Data - The Life Cycle Inventory Data. Swiss Centre for Life Cycle Inventories, Dübendorf

Regional Assessment of Waste Flow Eco-Synergy in Food Production: Using Compost and Polluted Ground Water in Mediterranean Horticulture Crops

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Abstract The potential eco-synergetic effects of using two waste flowsfor the substitution of mineral fertilisers is assessed from nutrient and environmental points of view. The two wastes are: composted organic municipal waste (slow release of nutrients) and nitrate polluted water (rapid nitrogen release).Catalonia is selected as a representative Mediterranean area of study. Macro-data at county level was used for the calculations, geographic information system, for the illustrations, and IPCC impact factors, for the environmental quantification. Compost and polluted water are able to supply 35-50% of the nutrient demand of Catalan horticulture production (330,000 tons of horticulture products per year), leading to reduction of 46% of the global warming potential of mineral fertiliser production.Moremineral fertilisersare saved in urban and agriculture intensive areas.

1 Introduction

Intensive horticulture has produced increasing economic and social benefits and a more efficient use of resources; nevertheless, the increase of inputs has had bad consequences for the environment. Two of the main problems derived from the high use of mineral fertilisers are the loss of nutrients and the resulting pollution of aquifers. Most intensive horticulture areas in Europehave been declared regions vulnerable to nitrate pollution [1]. Mild winter climates concentrate a major vegetable crops production, because weather is generally more favourable.

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On the other side, the European Directive 2006/12/EC [2] on waste settled that the Member States should take measures for the treatment of their waste in line with the waste hierarchy, which considers recycling as one of the priority options. Therefore it is necessary to reduce the amount of the organic fraction of the municipal solid waste (OFMSW) being dumped, in order to minimise environmental impacts, and also the loss of organic resources.

Composting is one of the most broadly used OFMSW treatments in Europe and in the world [3].However, composting plant managers usually came across with the rejection of farmers to apply the organic product in their fields. One reason is that nutrients from compost are not immediately available for the plant after compost application in soils, but they are slowly released in the long term, besides, there is a lack of training in its application. It could lead to shortage of nutrients during plant growth and, therefore, lower yields.

Nevertheless, an adequate use of composted products, together with a readily available nitrogen supplementation, have proved to reach similar yields as conventional fertilisation [4-7], saving a part of the economic and environmental costs of produce and use inorganic fertilisers. Highly nitrate polluted water may be a potential source of rapid nitrogen, apart from mineral fertilisers, and its use could mean a decrease in the concentration of contaminants.

Fertilisation decisions are extremely relevant, as fertilisation production has been reported as one of the determining factors for the environmental performance of horticulture products, particularly for those with low energy consumption [7,8].

Taking into account the principles of the industrial ecology, the aim of this paper is to study the potential eco-synergetic effects of using two waste flows, composted organic municipal waste and nitrate polluted water, which are slow and rapid sources of nutrients, respectively. We assess the maximum compost production in the area of study and state its potential use for supplying, together with the nitrogen in ground water, the nutrient demand of the horticulture sector. The results include the balance of nutrients and the global warming avoided due to mineral fertiliser savings.The area of study, Catalonia, was selected as representative of other Mediterranean regions due to the high population density, and therefore, large production of wastes; because it has firmly committed for composting as the major treatment for OFMSW [9]; because it has an annual production of almost 330,000 tons of horticulture products [10]; and because it presents relevant levels of nitrates in ground water.

This paper provides also a good example of the potentialities of the joint use of geographical information systems and life cycle assessment.

2 Methods and area of study

Area of study, data sources and considerations for the calculations and tools used for the assessment are briefly explained below. The information was processed at county level.

2.1 Area of study

Catalonia is an autonomous region located at the north-east of Spain and with a total area of about 32,000 km^2. It borders on France to the north and on Mediterranean Sea to the south. At the beginnings of 2011, it had more than 7.5 million of inhabitants. The region of study is divided into 41 counties of 145-1,785 km^2.

2.1.1 Horticultural production and nutrient demand

The productions and areas cultivated per horticulture crop (for instance, tomato, onion, lettuce, etc.) and per county are obtained for 2008 [10].
Regarding the calculation of nutrient demand (nitrogen, phosphorus and potassium), fertilisation recommendations for integrated cultivation management are used [11,12].

2.1.2 Potential nutrients from OFMSW compost

The total amount of organic waste from households, restaurants, caterers and retail premises generated (source-separated or collected with the bulky waste) in Cataloniais calculated at county level, using the following data: total municipal solid waste generation per county in 2009 [13] and average content of organic waste in the municipal solid waste for Catalonia, which is 36% [9]. We consider a 10% of OFMSW non-composted due to likely technical, geographical and social difficulties for the whole source-separated collection.
Catalan average mass reduction during composting is 78% [14]. The average nutrient supply of compost the first year was calculated using analysis from Huerta et al.[14] and considering the mineralisation rates of 14% (N), 37% (P) and 78% (K), from several compost reviews.

2.1.3 Potential nutrients from ground water

The available data from the Catalan quality controlling net is used[15]. Average nitrate content per county and for the last 5 years is calculated. In order to do calculations we have assumed the total irrigation water (6,500 m^3/ha·year) coming from ground water, which is supplemented with rainfall water. Total nitrogen applied with irrigation water is calculated multiplying nitrate content by the total amount of water used in each county. Supply of phosphorus or potassium is not taken into account as the concentrations in water are negligible.

2.2 Comparison scenarios

The nutrient savings and environmental improvement of the assessed eco-synergy scenario, which considers the use of nutrients coming from compost and polluted ground water, are calculated from the initial or current scenario. The maximum amount of compost per county depends on the potential compost production and is restricted to do not apply more than 170 kgN/ha·yearfrom organic sources [16]. The rest of nutrients are supplied by mineral fertilisers.

The initial scenario, considers that the entire nutrient demand from horticulture is supplied by mineral fertilisers and the slender compost applied to soils is not substituting inorganic fertilisation. Nowadays, less than 15% of the OFMSW generated in Catalonia is being applied to soils as compost [9,17].

2.3 Geographic information system

The spatial information generated was presented with a geographic information system (GIS) software, MiraMon$^{(R)}$ v7, allowing data processing and visualisation.

2.4 Global warming potential

For the global warming potential (GWP) calculation, impact factors from IPCC 2007 were used and only classification and characterisation steps were applied.

For the impact saving calculation only the avoided manufacture of mineral fertilisers is taken into account. Neither pumping station nor OFMSW management options nor transport of fertilisers are being included. The environmental data related to mineral fertiliser production was obtained from Boldrin et al. [3].

3 Results

3.1 Nutrient demand and potential supply

The nutrient demand from horticulture sector and potential supply from compost and ground water for Catalonia region, at county level, are presented in this section. Nitrogen demand and supply are shown in Figure 1 and 2, respectively.
The total amount of required nutrients from horticulture crops, which is calculated following section 2.1.1, is about 1,900 tons of N, 300 tons of P and 2,150 tons of K per year. Nitrogen demand per county is represented in Figure 1. Coastal counties and 4 and 5 are the ones with higher demand of N and, therefore, higher horticulture production.

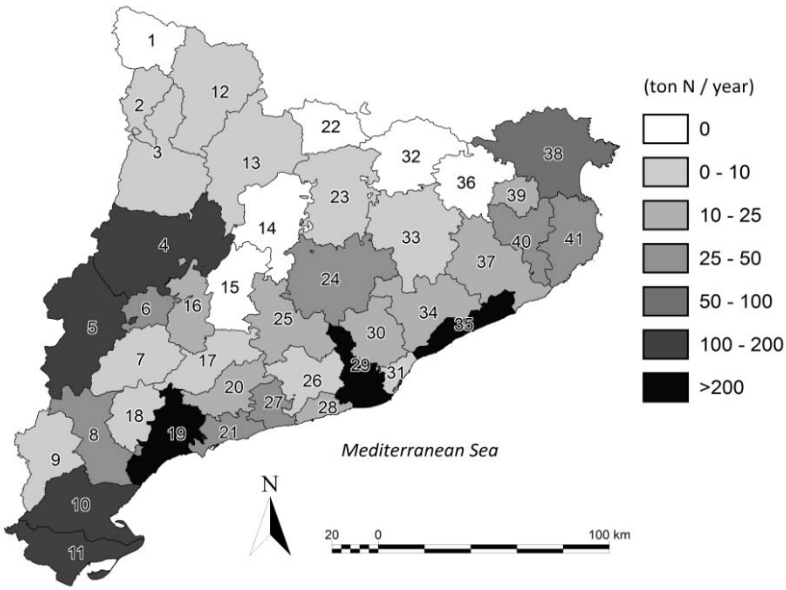

Fig. 1: Nitrogen demand from the Catalan horticulture sector

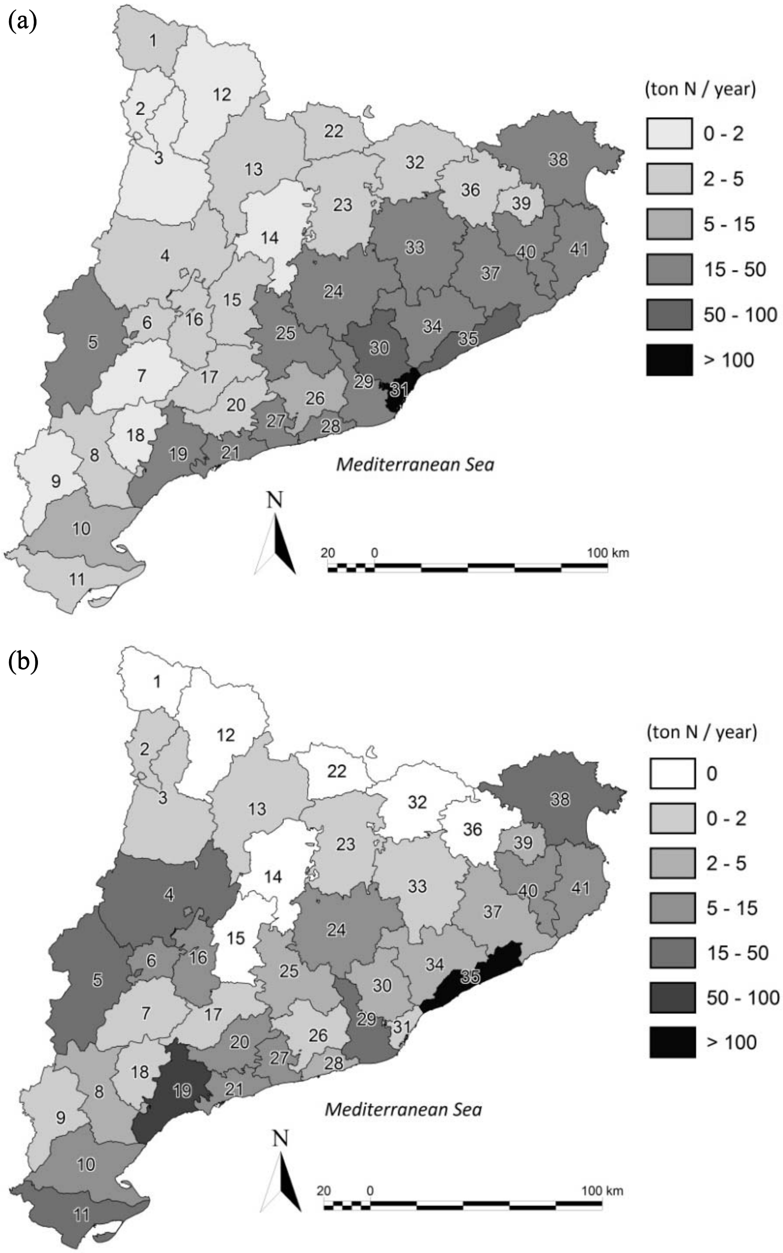

Fig. 2: Nitrogen offer from the two wastes. (a) Potential nitrogen available the first year from OFMSW composted. (b) Potential nitrogen available from ground water irrigation

Figure 2a shows the amounts of nitrogen available (in the short term) from the Catalan OFMSW generation if 90% of it is composted (see section 2.1.2). Counties with higher populations obviously have upper amounts of OFMSW to manage. Catalan population is mainly concentrated in coastal counties, being Barcelona city (county 31) and its surroundings the higher populated and the major generators of waste. County 31 has a generation of nearly 600 thousand tons of OFMSW per year, which represents almost half of the total Catalan generation (1,450 thousand tons). The potential nutrient supply from compost is about 950 tons of N, 450 tons of P and 2,050 tons of K per year

Upper nitrogen concentrations in ground water are measured in those counties with a major dedication to agriculture or stockbreeding: 4, 5, 6, 15, 19 have concentration above 50 mg NO_3/l and 33 and 35 have levels above 75 mg [15]. These areas are under regional regulation [16], in accordance with the Directive 2006/118/EC [1], for the protection of water against nitrate pollution from agricultural sources. Figure 2b shows the amount of nitrogen supplied by ground water according to the irrigation demand of each county (section 2.1.3). The potential amount of nitrogen coming from irrigation water for Catalonia is almost 500 tons per year.

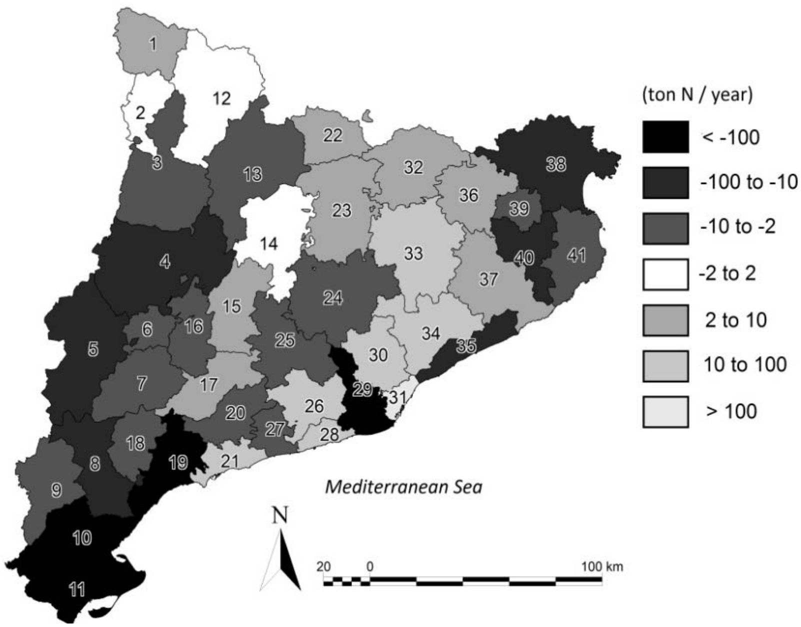

Fig. 3: Balance of nitrogen for the eco-synergy scenario (negative values show shortage, positive values show surplus)

3.2 Nutrients balance for compost scenario

According to the abovementioned amounts of nitrogen required by Catalan horticulture sector and supplied by compost and ground water (Figure 1 and 2), Figure 3 shows the balance of nitrogen at county level. Most coastal and west counties have shortage of nitrogen, whereas the rest have too much compost generation for their consumption.

Figure 4 illustrates the potential contributions from compost and ground water in the eco-synergy substitution of fertiliser. Almost 50% of mineral fertilisers are saved for N and P, half using compost and half using irrigation water, for the former, and using only compost, for the latter. Regarding potassium, compost supplies 35% of the demand.

However, the total amounts of truly saved mineral fertilisers by compost differ considerably from the potential nutrient supply for the entire Catalonia. According to section 3.1 and Figure 4, 41% of the potential nitrogen from compost is used, 30% of the P and 37% of the K. It is as a consequence of conflicts with places where compost is produced and places where nutrients are required (Figure 3), and because maximum amount of compost applied per hectare is limited [16].

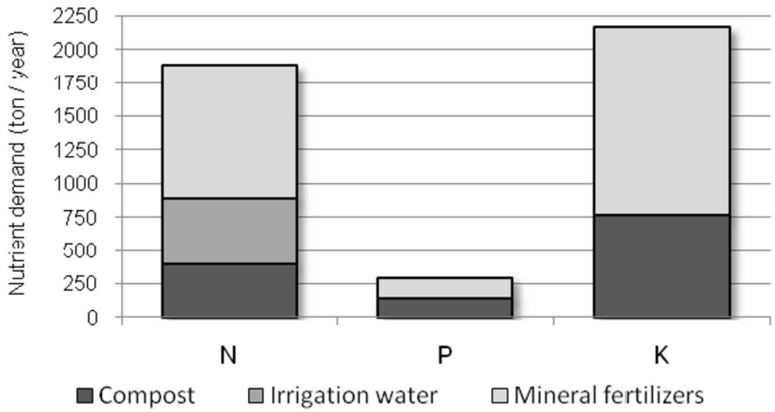

Fig. 4: Contribution of the three sources of nutrients to the total demand of nutrients of Catalan horticulture sector in the eco-synergy scenario

3.3 Environmental impact savings

The GWP of the mineral fertilisers saved, thanks to nutrient contribution of the eco-synergy compost and ground water, is presented in Figure 5. More than 9,000

tons of CO_2e, 46% of the current emissions from mineral fertilisers manufacture, are saved for the whole Catalonia.
Major GWP is avoided where there is high generation of OFMSW (i.e. population), high nitrate content in ground water and, specially, high demand of nutrients (i.e. horticulture production), for instance, county 35, 5 and 19.

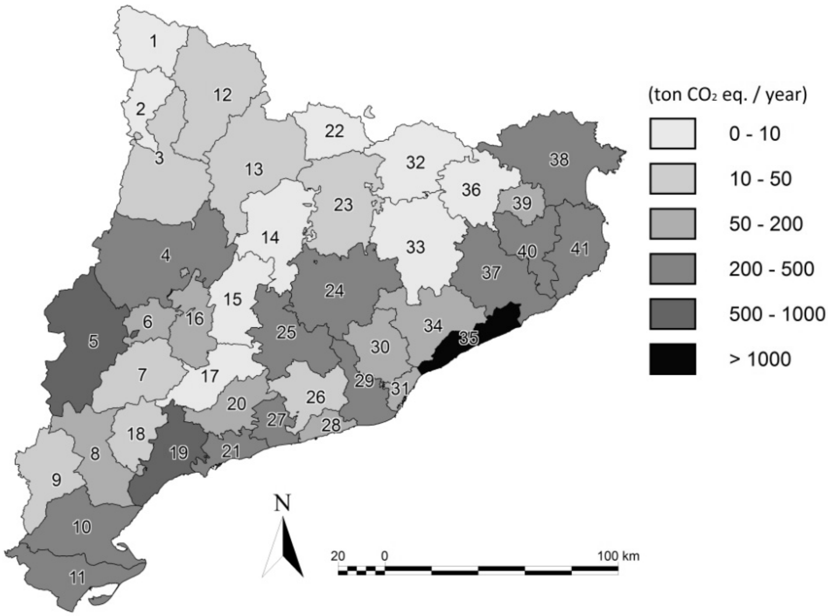

Fig. 5: Global warming savings for the eco-synergy scenario

4 Discussion and conclusions

The total amount of required nutrients from horticulture crops in Catalonia is about 1,900 tons of N, 300 tons of P and 2,150 tons of K per year. From them, the compost and polluted water eco-synergy combination is able to supply between 35-50% of the demand of nitrogen, phosphorus and potassium. Polluted water supplies rapid available nitrogen to the crops, while compost nitrogen and other nutrients are slowly released, providing a source of nutrients in the long term.
Two waste flows, that before were problematic, become a way to reduce inorganic fertiliser consumption.The potential reduction would lead to avoid 46% of the current emissions from mineral fertiliser manufacture. Furthermore, other environmental savings would take place due to reduction in the amount of

OFMSW going to landfill, decrease in the nitrate content of nitrates in ground water, thanks to bio-filtration effects, and effects of organic matter in the health of soils and plants.
Higher substitution rates, and therefore larger impact savings, with the eco-synergy of the two wastes are achieved in urban and agriculture intensive areas. It is consequence of major pollution of aquifers, major generation of organic waste and major demand of nutrients.
Taking into account the possibility of transporting compost from one county to another, it could increase even more the savings. Moreover, less than 41% of the nutrients of the potential production of compost would be used by horticulture sector; therefore other agriculture sectors could take profit of this surplus.
From the results of this paper, the joint use of the tools LCA and GIS inside the framework of industrial ecology appears a valuable strategy for territorial planning and decision taking.

5 Further research and points of concern

Aiming to build a more realistic model and to obtain more comprehensive results it is necessary to stress in several of the suppositions and data used throughout the paper. Further research would be focus on:

- Consider real values of OFMSW generation per county or municipality.
- Add other irrigated crops apart from horticulture ones
- Take into account the residual effects of nutrient release from compost in the long term
- Assess the potential higher savings if transport of compost between counties is considered
- Assess the loads of an excessive application of P and K applied with compost
- Add other impact categories apart from GWP, as eutrophication and acidification potential, relevant for the water pollution

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References

[1] European Parliament and European Council (2006)Directive 2006/118/EC, on the protection of groundwater against pollution and deterioration.OJEU

[2] European Parliament and European Council (2006)Directive 2006/12/EC, on waste.OJEU

[3] Boldrin A, Andersen JK, Moller J, Christensen TH, Favoino E (2009)Composting and compost utilization: accounting of greenhouse gases and global warming contributions. Waste Manage Res 27:800-812.

[4] Favoino E, Hogg D (2008) The potential role of compost in reducing greenhouse gases. Waste Manage Res 26:61-69.

[5] Hargreaves JC, Adl MS, Warman PR (2008) A review of the use of composted municipal solid waste in agriculture,AgricEcosyst Environ 123:1-14.

[6] Diacono M, Montemurro F (2010) Long-term effects of organic amendments on soil fertility. A review. Agron Sustainable Dev 30(2):401-422.

[7] Martínez-Blanco J, Muñoz P, Antón A, Rieradevall R (2011) Assessment of tomato Mediterranean production in open-field and in standard multi-tunnel greenhouse with compost or mineral fertilisers from an agricultural and environmental standpoint. J Cleaner Prod.doi:10.1016/j.jclepro.2010.11.018.

[8] Martínez-Blanco J, Antón A, Rieradevall J, Castellari M, Muñoz P (2011)Comparing nutritional value and yield as functional units in the environmental assessment of horticultural production with organic or mineral fertilisation. The case of Mediterranean cauliflower production. Int J Life Cycle Assess 16(1):12-26.

[9] Catalan Department of Environment and Housing, PROGREMIC: Plan of municipal waste management for Catalonia 2007-2012, ARC, 2007.

[10] Catalan Department of Agriculture, Farming, Fishing, Food and Natural Areas.<http://www20.gencat.cat/portal/site/DAR> (accessed 21.03.2011).

[11] Spanish Ministry of Environment and rural and marine areas (2010), Guia práctica de la fertilización racional de los cultivos en España, MARM

[12] Catalan Department of Agriculture, Farming, Fishing, Food and Natural Areas (2010)Norma tècnica per a la produccióintegradad'hortalisses

[13] Catalan Office of Waste, ARC<http://www.arc-cat.net>(accessed 21.03.2011).

[14] Huerta O, López M, Soliva M, Zaloña M (2008)Compostatge de residus municipals: control del procés, rendiment i qualitat del producte.ESAB

[15] Catalan quality controlling net<http://aca-web.gencat.cat/aca>(accessed 21.03.2011).

[16] Catalan Parlament (2009)Decret 136/2009, d'aprovació del programa d'actuació aplicable a les zones vulnerables en relació amb la contaminació de nitrats que procedeixen de fonts agràries i de gestió de les dejeccions ramaderes.DOGC

[17] Giménez A, Soliva M, Huerta O (2005)*El mercat del compost a Catalunya: Oferta i demanda.* ESAB and ARC

Assessing Management Influence on Environmental Impacts Under Uncertainty: A Case Study of Paddy Rice Production in Japan

Kiyotada Hayashi

Abstract The influence of management practices on the environmental impacts of paddy rice production was analysed using life cycle assessment (LCA) with a strong focus on uncertainty. The environmental impact categories that were assessed comprised greenhouse gas (GHG) emissions and energy use. Farm accountancy data about paddy rice cultivation, prepared by the prefectures of Japan for improving extension services, were utilised. Simplified LCA and uncertainty analysis, including Monte Carlo methods and resampling (nonparametric bootstrapping), were applied. The results indicate that correlation coefficients (means and 95% confidence intervals) between the area under rice cultivation and environmental impacts were negative and that the environmental impacts of rice cultivation were lower with direct seeding than with transplanting.

1 Introduction

Rice is the principal food item in Asian countries, with 90% of global rice production being in Asia. Because rice is also commonly cultivated in Europe and North America, the life cycle assessment (LCA) of rice attracts much interest beyond the boundary of Asia [1-5]. One reason for this focus is the increased global attention to environmentally sustainable agricultural production and consumption. Indeed, environmental product declarations and the carbon footprint may necessitate the introduction of life cycle thinking in rice production.

However, earlier LCAs of rice production failed to incorporate uncertainty, despite the importance of uncertainty management in agricultural production. Although sensitivity analysis was conducted to test the stability of the results [3], it is different from uncertainty analysis such as Monte Carlo simulations. Because agriculture is conceptually located at the interaction point between nature and human activities, agricultural practices are different from manufacturing

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processes. This difference means that effectively dealing with uncertainty is the key to successful agricultural management.
In addition, previous LCAs of rice production did not address the relationship between environmental impacts and management practices. However, to realise sustainable agricultural production, it is crucial to consider the influence of management practices on environmental impacts in association with uncertainty, because management practices are the driving force of sustainable agriculture.
Therefore, this study analyses the influence of management practices on environmental impacts, with a strong focus on probability distributions. Greenhouse gas (GHG) emissions and energy use were used as impact categories. Farm management data on paddy rice cultivation, prepared by the prefectures of Japan for improving extension services, were used for the analysis. Simplified LCA and uncertainty analysis, including Monte Carlo methods and resampling (nonparametric bootstrapping), were applied to obtain results for a representative rice farm in Japan, derived from a variety of representative rice farms defined by each prefecture.

2 Methods

2.1 Farm accountancy data

The data used for preparing the life cycle inventories of paddy rice cultivation were farm accountancy data that had been compiled by each prefecture in Japan. The primary purpose of data construction is to improve the quality of the agricultural extension services that are provided by each prefecture. Most of the prefectures publicize the data in the form of printed books or electronic worksheets. These documents provide details about the inputs and outputs of various agricultural materials. For example, the data include paddy yields, cultivation conditions (e.g., whether to practice midseason drainage), the use of fossil fuels and electricity, the availability of farm machinery and facilities, and the utilization of fertilizers, pesticides, seeds and seedlings, and auxiliary materials.
The data are different from those provided by the Farm Accountancy Data Network (FADN) in Europe, which are based on statistical sampling; the current data provide a basic description of the representative farm types in each region, although actual farm records are used for constructing the farm types. Although

this implies that statistical inference based on the data may not be straightforward, the data are useful if advanced statistical methods are applied, such as resampling.

2.2 Simplified agricultural LCA

This study employs simplified LCA, which means that for the current assessment, emission factors are based on the embodied energy and emission intensity data for Japan using input-output Tables (3EID) [6] and the national greenhouse gas inventory [7]. This approach was used to handle many observations, rather than developing and applying a modularised life cycle inventory database [8]. As a result, GHG emissions and energy use were selected as environmental categories.
The system boundary is defined as cradle to gate (CTG). To equalise the system boundary for each observation (accountancy data for the farm type), records in which the crop drying process or any other farm work was outsourced were excluded from the analysis. As a result, 90 samples were available. Both the functional units, area (ha) and products (kg of brown rice), were employed in this study.

2.3 Uncertainty analysis

The sources of uncertainty in the simplified LCA of rice, when using the farm accountancy data, are divided into 3 categories [9,10] (Table 1). Because system process data (emission factors), which are based on the Japanese input-output (IO) tables, are used to assess GHG emissions and energy use, the sources are associated with the foreground processes of rice production systems. Parameter uncertainty includes the temporal and spatial variability of the farm types and imprecise specifications of, for example, yields. Scenario uncertainty is related to the variability in the specifications of representative farm types, because there is a wide variety of farm types, and it is difficult to determine representative types. In addition, although system boundaries are almost identical among observations because of the exclusion of farm types that outsourced farm work, inconsistencies may exist. Simplified methods were used for estimating the emissions of methane and nitrous oxide from paddy fields; this may be the cause of model uncertainty.

Tab. 1: Example sources of uncertainty in the simplified LCA of rice

Component	Sources
Parameter (input data)	Temporal and spatial variability Imprecise specification
Scenario (normative choices)	Variability in the specifications of representative farm types Inconsistent system boundaries
Model uncertainty (mathematical relationships)	Simplification of emission models

One of the original perspectives in this study is the uncertainty analysis of the relationships among variables, in addition to the uncertainty analysis of values. The analysis includes the relationship between production size and environmental impacts, in addition to the relationship between cultivation practices and environmental impacts. Therefore, resampling (nonparametric bootstrapping) methods were primarily used in this study. Monte Carlo methods were also used for estimating model uncertainty with respect to methane emissions from paddy fields.

Because our approach employs the system process inventories for assessing background emissions, it is difficult to apply Monte Carlo methods, except for methane emissions. Thus, uncertainty propagation is beyond the scope of this study. Rather, this study presents the first assessment of the relationships between management practices and environmental impacts, with a special focus on uncertainty. Although the relationships between management practices and environmental impacts have been previously studied [11,12], the uncertainty of the relationships has not been evaluated.

In this study, model uncertainty is related to methane emissions from paddy fields. The emissions were estimated using Monte Carlo simulation under the assumptions that (1) decision makers do not have a priori knowledge about the occurrence of soil types (i.e., andosol, yellow soil, lowland soil, grey soil, and peat soil), (2) the probability distribution of methane emissions is uniformly discrete, and (3) midsummer drainage is practiced in all observations. More specifically, Latin hypercube sampling was applied.

The influence of management practices on environmental impacts was assessed by calculating fossil CO_2 emissions. The reason for this selection is that the effects of production scale and the difference between direct seeding and transplanting may be measured by fossil CO_2 emissions. Because it is assumed that midsummer drainage is practiced in all observations, the differences in methane emissions are determined by soil types, which are not related to management decisions. Uncertainty in the influence was estimated by nonparametric bootstrapping.

The software used for the estimation was @Risk (version 5.7) and S-PLUS (TIBCO Spotfire S+® 8.1J for Windows). Although the default parameter values were applied in the calculation, the number of iterations (i.e., the number of bootstrap samples) was set to 10000.

3 Results

3.1 Estimation of environmental impacts

The estimated average energy use was 8.07 MJ/kg, and the average of GHG emissions was 1.45 kg CO_2e/kg, with a 95% confidence interval of [1.40, 1.50], in which the uncertainty of methane emissions was considered. The probability distributions for both environmental categories were upper-tailed. The Gamma distribution was recognised as the best-fitting distribution for both cases, if the chi-squared statistic is applied for judging the fitness.

3.2 Effect of scale

Tab. 2: Correlation coefficients between production size and environmental impacts

	Mean	95% confidence interval
Energy input per area (MJ/ha)	–0.184	[–0.324, –0.005]
Energy input per product (MJ/kg)	–0.234	[–0.371, –0.050]
CO_2 emissions per area (kg/ha)	–0.164	[–0.310, –0.006]
CO_2 emissions per product (kg/kg)	–0.192	[–0.331, –0.041]

The correlation coefficients between the size of rice production areas and environmental impacts were negative (Table 2). For example, the coefficient between CO_2 emissions per kg of products and the production size was –0.192, with a 95% confidence interval of [–0.331, –0.041]. This result implies that the "ecology of scale" [13] may be present, although factors other than scale must also be considered [14]. In addition, it is important to note that scale is measured by the size of paddy rice production area, rather than farm business size.

3.3 Difference between direct seeding and transplanting

The environmental impacts of rice cultivation were lower with direct seeding than with transplanting (Table 3). For example, CO_2 emissions per kg of products for direct seeding were 0.52, with a 95% confidence interval of [0.49, 0.59], while those for transplanting were 0.60, with a 95% confidence interval of [0.57, 0.64]. The relationship between CO_2 emissions per kg of products for direct seeding and transplanting is illustrated in Figure 1. Although there is an intersection between the distribution for direct seeding and that for transplanting, it is possible to ascertain that CO_2 emissions per kg of products are greater for transplanting than for direct seeding.

Tab. 3: Differences between direct seeding and transplanting

		Mean	95% confidence interval
Energy input (MJ/kg)	Direct seeding	7.46	[6.92, 8.31]
	Transplanting	8.33	[7.92, 8.82]
CO_2 emissions (kg/kg)	Direct seeding	0.52	[0.49, 0.59]
	Transplanting	0.60	[0.57, 0.64]

3.4 Life cycle perspectives on CO_2 reduction

To mitigate CO_2 emissions from life cycle perspectives, it is important to calculate the percentages of the foreground and background processes. As shown in Table 4, most of the emissions are direct field emissions from farm work (33.1%), machinery production (36.2%), and fertilizer production (14.9%). The 95% confidence intervals for these processes are larger than those for the other processes.

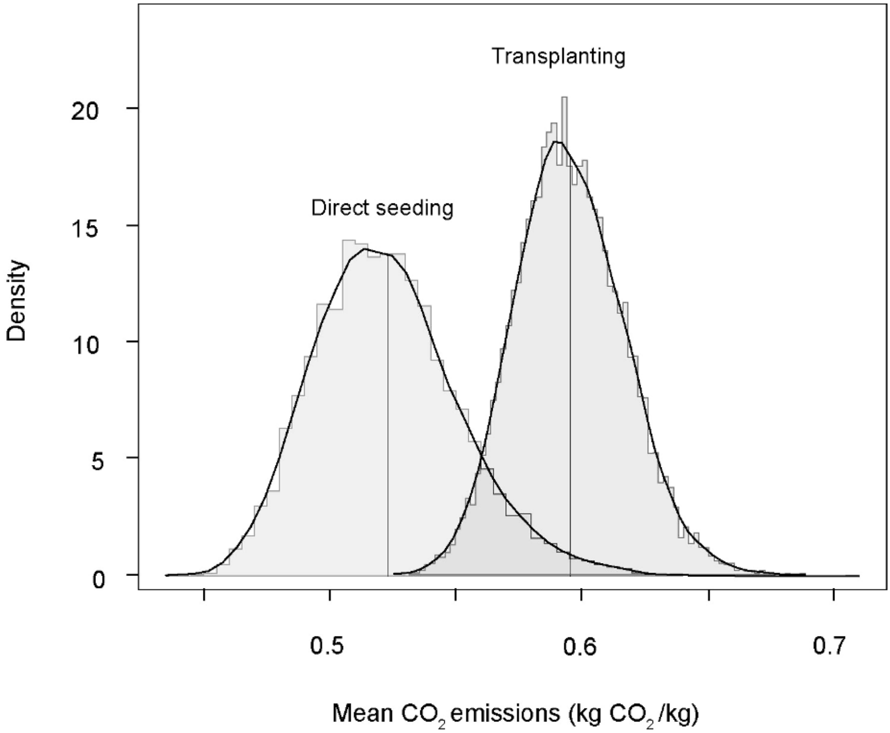

Fig. 1: Difference in mean CO_2 emissions between direct seeding and transplanting

Tab. 4: Percentage of CO_2 emissions from foreground and background processes

	Mean	95% confidence interval
Direct emissions from farm work	33.1	[31.3, 35.5]
Machinery production	36.2	[34.0, 38.4]
Seedling production	0.9	[0.8, 1.0]
Fertiliser production	14.9	[13.7, 16.0]
Pesticide production	7.9	[7.3, 8.5]
Auxiliary materials	7.0	[6.2, 7.9]

4 Discussion

The results of this study have several implications. First, the fact that the probability distributions of GHG emissions and energy use were upper-tailed means that upper side outliers exist. Hence, the use of maximum values for a policy framework, such as a carbon footprint, may not be pragmatic. In other words, outliers or extremes are not appropriate for use as standards.

Second, the effect of production scale is important in developing farm management strategies. This is because scale is one of the most important parameters for improving management performance. In addition, attention should be given to scope (diversification) and integration (internalization of the material cycles), which is important for assessing rural development performance [15].

This theory necessitates the introduction of multicriteria decision analysis, which means that the result (the rank order of decision alternatives) must be confirmed to be dependent on decision criteria. Thus, even if our focus is restricted to environmental impacts, the results on the effect of scale should change when other environmental categories, such as biodiversity, are used. Indeed, the margins of paddy fields may be important for cultivating biodiversity, although they may also represent obstacles for expanding business size.

Third, the result on the comparison between direct seeding and transplanting implies that there are further possibilities of developing direct seeding practices. However, criteria other than GHG emissions and energy use would be necessary for selecting appropriate management practices.

Fourth, in order to mitigate CO_2 emissions from paddy rice production, life cycle perspectives are important. In this case, the improvement of energy efficiency and machinery utilization, and a decrease in fertilizer application while maintaining yield levels are effective methods for decreasing emissions. The results also imply that the inclusion of capital goods in the assessment is important in conducting LCAs.

5 Conclusions

This study assessed the influence of management practices on environmental impacts in paddy rice production, with a specific focus on the uncertainty of parameters, scenarios, and models. Monte Carlo methods and nonparametric bootstrapping were applied in the analyses. An important contribution of this study is the presentation and demonstration of a method to construct representative numerical values, while directly focusing on uncertainty. The next step of this study will be the use of modularised life cycle inventories, in which uncertainty propagation in supply chains will be analysed with resampling methods.

References

[1] Harada H, Kobayashi H, Shindo H (2007) Reduction in greenhouse gas emissions by no-tilling rice cultivation in Hachirogata polder, northern Japan: Life-cycle inventory analysis. Soil Sci Plant Nutrition 53(5):668-677.

[2] Roy P, Shimizu N, Okadome H, Shiina T, Kimura T (2007) Life cycle of rice: Challenges and choices for Bangladesh. J Food Eng 79(4):1250-1255.

[3] Blengini GA, Busto M (2009) The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy). J Environ Manage 90(3):1512-1522.

[4] Hokazono S, Hayashi K, Sato M (2009) Potentialities of organic and sustainable rice production in Japan from a life cycle perspective. Agron Res 7(S1):257-262.

[5] Kasmaprapruet S, Paengjuntuek W, Saikhwan P, Phungrassami H (2009) Life cycle assessment of milled rice production: Case study in Thailand. Eur J Sci Res 30(2):195-203.

[6] <http://www.cger.nies.go.jp/publications/report/d031/index.html> (accessed 7.4.2011)

[7] Greenhouse Gas Inventory Office of Japan (GIO), Center for Global Environmental Research (CGER), National Institute for Environmental Studies (NIES) (2009) National Greenhouse Gas Inventory Report of Japan.

[8] Hayashi K, Uchida S, Hokazono S, Sato M (2010) Modeling life cycle inventories for crop production in Japan: Development of the NARO LCI database. LCA Food 2010: VII International Conference on Life Cycle Assessment in the Agri-Food Sector, Bari, Italy 1: 455-460.

[9] Huijbregts MAJ, Gilijamse W, Ragas, AMJ, Reijnders L (2003) Evaluating uncertainty in environmental life-cycle assessment: A case study comparing two insulation options for a Dutch one-family dwelling. Environ Sci Technol 37(11):2600-2608.

[10] Lloyd SM, Ries R (2007) Characterizing, propagating, and analyzing uncertainty in life-cycle assessment: A survey of quantitative approaches. J Ind Ecol 11(1):161-179.

[11] Mouron P, Nemecek T, Scholz RW, Weber O (2006) Management influence on environmental impacts in an apple production system on Swiss fruit farms: Combining life cycle assessment with statistical risk assessment. Agric Ecosyst Environ 114(2-4):311-322.

[12] Thomassen MA, Dolman MA, van Calker KJ, de Boer IJM (2009) Relating life cycle assessment indicators to gross value added for Dutch dairy farms, Ecol Econ 68(8-9):2278-2284.

[13] Schlich E, Fleissner U (2005) The ecology of scale: assessment of regional energy turnover and comparison with global food. Int J Life Cycle Assess 10(3):219-223.

[14] Jungbluth N, Demmeler M (2004) 'The ecology of scale: Assessment of regional turnover and comparison with global food' by Elmar Schlich and Ulla Fleissner. Int J Life Cycle Assess 10(3):168-170.

[15] Hayashi K (2011) A new conceptual framework for assessing rural development performance: Sustainability of scale, scope and integration. In: Behnassi M et al (ed) Global Food Insecurity: Rethinking Agricultural and Rural Development Paradigm and Policy, Springer

Assessing Environmental Sustainability of Different Apple Supply Chains in Northern Italy

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Abstract The application of environmental assessment methods in the fruit sector is conventionally divided into a field phase and a retail phase. Although there are important differences in the environmental impacts in field phase, a major part of the impacts is related to the management of the fruit and the distribution chain in the retail phase. In this paper, the environmental impact of fruit production is quantified in the production and retail phase of apple production in Piedmont in Northern Italy. Three main scenarios have been identified: (I) direct selling, (II) distribution to local markets and (III) distribution to national markets. A complete life cycle assessment (LCA) has been performed on the three apple supply chains. Results show the importance of retailing strategies for the environmental sustainability of such food item.

1 Introduction

Over the last 50 years, the advance of new technologies, improved facilities and infrastructure at all levels of the food supply chain has led to an enormous expansion of the food availability in the markets. Recently, this transformation of the food retail system has arisen concerns about the environmental impacts of transporting food increasingly long distances prior to its consumption [1]. This during the last years an ongoing debate about the environmental convenience of regionalisation versus globalisation of alternative food systems has emerged [2]. Although assessments generally show that impacts of locally produced food are smaller, the results are often controversial both in terms of methodology and concepts [3]. Several studies aim to quantify the contribution of transportation in the food sector in a given area; e.g. it is estimated that greenhouse gases (GHG)

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emissions from transporting food around the UK contribute 3.5% to total UK GHGs [4].
There are several assessment approaches available to estimate the potential environmental impacts of a product or service, and Life Cycle Assessment (LCA) is estimated to be one of the most sophisticated [1]. Although many aspects of environmental accounting methodologies in food production are already investigated, applications of LCA in the fruit sector are still rare. Particularly, most of investigations in the fruit sector focus on the orchard stage [5] or on different retailing scenarios of fresh fruits [6] or fruit products [7].
Most of LCAs on food supply chains are done from a consumer point of view [3]; particularly, evaluations are conducted for products that arrive from different supply chain at the same consumer point, e.g. apple in UK [1] or orange juice in Denmark [8]. In this study, the environmental assessment has been conducted from a producer/retailer point of view, comparing different transport strategies from the same area of production.
Thus, the objectives of this research are (I) to quantify the main environmental impacts of the apple supply chain in Piedmont (II) to evaluate the relative impact of the two investigated phases (production and retail) on the overall environmental burden of the fruit; (III) to quantify the differences in environmental impact of the investigated distribution systems, particularly the impact of transportation.

2 Methodology

This study has been performed in accordance with the guidelines and requirements of the ISO 14040 standard series and with the cradle-to-use approach as the basis for the Life Cycle Inventory (LCI) of the study. Data regarding agricultural inputs production and distribution, resources consumption and agrotechniques have been obtained directly from the growers, who filled in a questionnaire for the season 2009-2010. Collected data were weighted consulting the Italian protocols for such production. Data regarding supply chains have been obtained from retailers through interviews and field surveys.
The assessment covers the whole supply chain, including all the stages from the agricultural production up to the beginning of the consumer's phase, of apples of the cultivar Golden Delicious cultivated in Piedmont. This cultivar has been chosen because of the wide range of distribution, compared to ancient cultivars that present mainly local commercialisation.
According to previous studies [5,9], the production phase has been modelled in 6 different stages: nursery, orchard installation, low production due to young plants,

full production, low production due to old plants, and orchard destruction (Figure 1). Resource consumption and emissions from each stage has been quantified; also the production of differentiated apple farming inputs and their transport to the field are included in the system boundary of the production stage. Pesticides were accounted for both in terms of the production and in terms of emissions. Emissions were calculated using Pest-LCI tool [10]. If characterisation factors were not available for the exact pesticide, alternative pesticides with the same chemical and physical properties were chosen for the evaluation. The lack of data about the effects of pesticide residues in crops and groundwater and of sprays on 'bystanders' continue to be matters for general debate [1]. Fertilisers are accounted both for production and emissions through a nutrient balance according to average physiological requirements of the plants. Construction of all buildings and infrastructure was omitted, as is common in LCA studies [1]. Although farm machinery is often included [5] in LCA studies, in our case it was not possible to consider production and maintenance of farm machineries because of the lack of on-farm data.

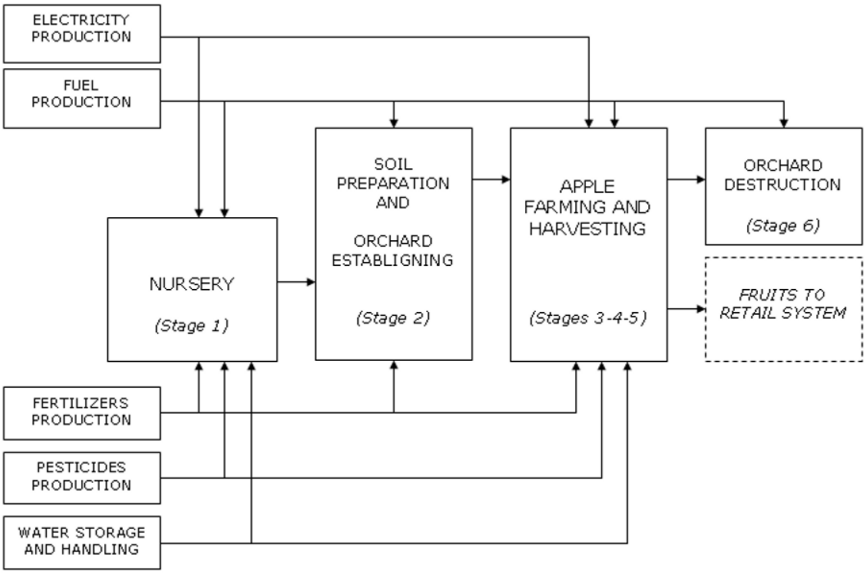

Fig. 1: System boundary and modelling of the apple production phase. Dotted box refers to processes that differ according to the three scenarios

The retailing phase has been modelled in three different scenarios according to the major supply chains that start from Piedmont orchards (Figure 2). The three supply chains consist of the following steps: (A) direct selling; after harvest, fruits are collected in the retailing deposit, stored in refrigerators, then washed and processed for the selling directly in the store without packaging, just reusable bags, bins or paper shopping bags, (B) distribution to local markets; after collection and refrigerated storage, fruits are processed in plastic bins for transportation to regional wholesale markets in local fresh markets up to 150 km from the retailer deposit, than sold in paper shopping bags; (C) distribution to national markets; after collection and refrigerated storage fruits are processed in plastic bins for transportation to national wholesale markets up to 800 km, were fruits are both packed for large-scale distribution and sold without packaging to local fresh markets in paper bags. Distances and means of transportation were obtained primarily from the farmers, the processing industry and relevant websites. The inventory data on the transport was obtained by assuming truck "specifics" as modelled by the GaBi 4 Professional Database. For all the transportation an average load of 85% was assumed and backhaul journeys were not considered because of modern logistic providers try to avoid as much as possible to move empty vehicles, and often the returning journey is utilised to move products from other systems. Plastic containers have been modelled according to another LCA case on fruit packaging [11].

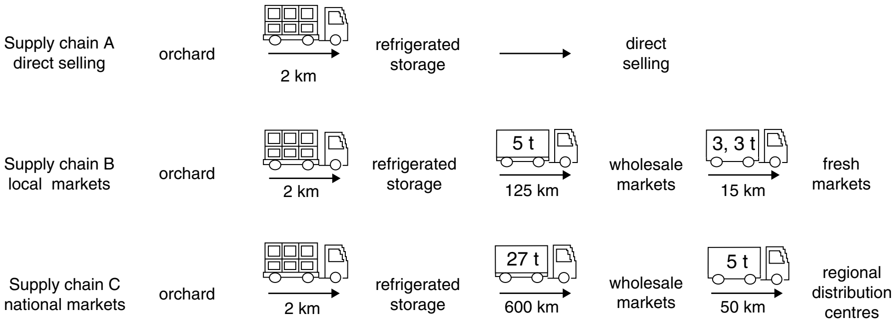

Fig. 2: Schematic description of transport channels for the considered supply chains

Storage and consumption within the consumer's house have not been included because of the high variability of possible situations in consumer behaviours [12]. The functional unit was 1 kg of 'Golden' apple delivered to the consumer. This is consistent with the general function of a supply chain from the perspective of the major Piedmont retailers, and it is also the most commonly used functional unit for such kind of studies [e.g. 1,5,6]. As the cultivar was the same for the three supply chains, it was not necessary to consider more specific functional units such as nutritional values or income based units [12].

3 Results

3.1 Characterisation

Based on the emissions estimated in the inventory analysis, the environmental impacts in the impact categories of the EDIP method was calculated. The impacts from the three supply-chains are illustrated in Figure 3. The complete production phase (including annual and whole orchard processes) results in the main impact in the categories acidification potential, nutrient enrichment potential and photochemical oxidant potential. In contrast, the contribution of the production phase to ozone depletion potential is below 35% of the total impact, in the three scenarios. The contribution of the retail phase from scenario A to C increases slightly in most of the impact categories, but dramatically in global warming potential. Most precisely in the complete supply-chain of a kilogram of Golden Delicious apples collected into an average Piedmont retailer, the global warming potential ranges from 0.0661 to 0.1221 kg CO_2e. The production phase accounts for 0.0622 kg CO_2e in all of the scenarios; on the contrary contribution of greenhouse gasses from the retail phase varies very much. Retail phase in scenario A accounts for 0.0038 kg CO_2e (5.84% of the whole global warming potential of the scenario) although it includes storage, processing and direct selling only. The retail phases of the other two scenarios include packaging and transport as well, and it accounts for 0.0919 kg CO_2e (59.65%) and 0.1221 kg CO_2e (66.25%) in scenario B and C respectively.

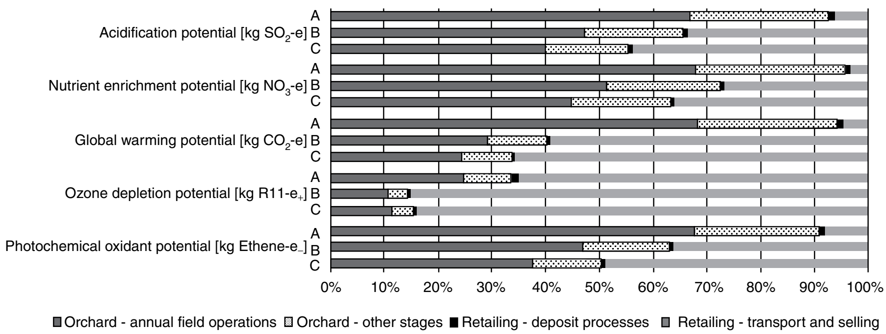

Fig. 3: Hotspot analysis for the three supply chains. "Orchard – other stages" considers: nursery stage, installation and destruction of the orchard, low production years due to young and old plants. "Retailing – deposit processes" considers: storages, processing and selling of fruit

3.2 Normalisation and weighting

In order to be able to assess the impact of the different impact categories compared to the impacts that an average person would otherwise be responsible for, the results were normalised and according to the EDIP method (1997). The results of the characterisation were normalised with reference to the total impacts of activities in Europe. The unit of the normalised results is person equivalents (PE) which corresponds to the impact one person has in a given category. The dominating impact categories are similar to those commonly identified in agricultural LCAs; global warming potential, nutrient enrichment potential and acidification potential (Figure 4). Those three categories have almost the same values in supply chain A (from 5.59E-06 to 7.59E-06 PE), but they vary in supply chain B and C with a dominant contribution of global warming potential (1.77E-05 and 2.12E-05 PE in B and C respectively).

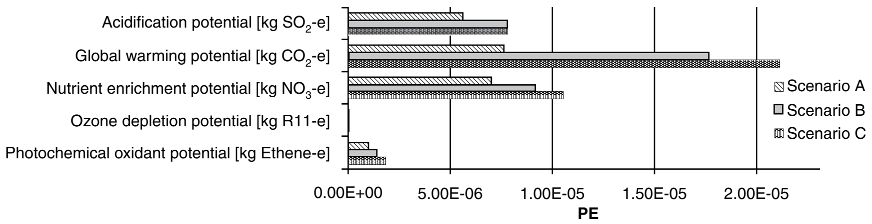

Fig. 4: Normalised impact assessment for 1 kg of Golden Delicious produced in Piedmont, at the end of three main supply chain scenarios

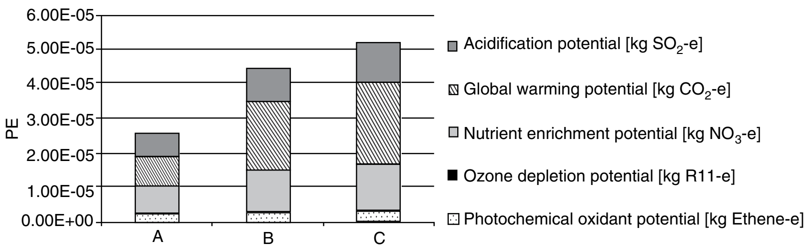

Fig. 5: Weighted results (EDIP method 1997) presented as the sum of the weighted personal equivalent (PE) for each investigated supply chain A, B and C

In order to compare the total environmental impacts of the three scenarios against each other, weighing was performed in accordance with the EDIP (1997). In this method, political targets are used to scale the importance of the different impact categories against each other. The unit of the results are person equivalents according to the target that are given for the future. The results are presented in Figure 5, and show that supply chain A results in 2.71E-05 PE; supply chain B, results in 4.60E-05 PE and supply chain C results in 5.42E-05 PE. According to this weighting method to total impact of distributing apples on national markets are this approximately twice as big as at the local market, indicating the impact of transportation is a great importance.

4 Conclusions

As scenarios were set-up considering the same production phase (as average values of the investigated orchards in Piedmont) it was obvious that the longest supply chain would present the largest environmental impact. Therefore, the purpose of the study was not to compare the three scenarios in terms of which supply chain results in the largest impacts, but more to quantify the impacts of the various parts of the supply chain.

As expected, the complete production phase contributes significantly to the environmental impacts of the direct selling scenario (supply chain A) and decreases in percentage with increasing transport distance. Considering the weighted results from all impacts categories, the production phase contributes 92% of the environmental impacts in supply chain A, 54% in supply chain B and 46% in supply chain C. That means that transportation from producers to consumers plays an important role in determining the environmental impacts of apple supply-chains in Northern Italy. Particularly, in the longest supply chain, more than a half of the environmental impacts are due to transportation.

As highlighted in previous works [5,9] the application of LCA just to full production years will underestimate the environmental impacts of the production phase. In our study, field processes out of the full production years (such as nursery, installation, low yield years etc.) contribute from 13 to 26% of the weighted values of the assessment for the whole supply chain, in scenario C and A respectively and contribute to 28% of the weighted impacts of the whole production phase.

Furthermore, from Figure 5 it can be easily seen that the contribution to the overall environmental impacts (expressed as the sum of weighted contribution per impact category) remains almost constant in the three scenarios except for global warming potential that varies from 8.51E-06 PE in scenario A to 1.98E-05 and 2.37E-05 PE in scenario B and C respectively.

References

[1] Sim S, Barry M, Clift R, Cowell SJ (2007) The relative importance of transporting in determining an appropriate sustainability strategy for food sourcing. Int J Life Cycle Assess 12(6):422-431.
[2] Clemente G, Escobar N, Sanjuan N, Anton A (2010) Orange juice, which one should I drink? VII International conference on life cycle assessment in the agri-food sector, Bari, 2010 (2):253-258
[3] Van Passel S (2010) Food miles to assess sustainability: a revision. Sust Dev. DOI: 10.1002/sd.485.
[4] Garnett T (2011) Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)?. Food policy 36:23-32.
[5] Mila i Canals L, Clemente Polo G (2003) Life cycle assessment of fruit production. In: Mattsson B, Sonesson U (ed) Environmentally friendly food processing. Woodhead Publishing Limited and CRC Press LLC, Cambridge and Boca Raton
[6] Blanke MM, Burdick B (2005) Food (miles) for thought – Energy balance for locally-grown versus imported apple fruit. Environ Sci Pollut Res Int 12(3):125-127
[7] Cholette S, Venkat K (2009) The energy and carbon intensity of wine distribution: a study of logistical options for delivering wine to consumers, J Cleaner Prod 17:1401-1413.
[8] Knudsen MT (2010) Environmental assessment of imported organic products. Focusing on orange juice from Brazil and soybeans from China. Dissertation, Aahus University
[9] Cerutti AK, Bagliani M, Beccaro G, Peano C, Bounous G (2010) Comparison of LCA and EFA for the environmental account of fruit production systems: a case study in Northern Italy. VII International conference on life cycle assessment in the agri-food sector, Bari, 2010 (2): 99-104
[10] Birkveda M, Hauschild MZ (2006) PestLCI—A model for estimating field emissions of pesticides in agricultural LCA. Ecol Modell 198:433–451.
[11] Singh PS, Chonhenchob V, Singh J (2006) Life cycle inventory and analysis of re-usable plastic containers and display-ready corrugated containers used for packaging fresh fruits and vegetables. Packag Technol Science 19:279-293.
[12] Roy P, Nei D, Orikasa T, Xu Q, Okadome H, Nakamura N, Shiina T (2009) A review of life cycle assessment (LCA) on some food products. J Food Eng 90:1–10.

The Effect of CO_2 Information Labelling for the Pork Produced with Feed Made from Food Residuals

Hideaki Kurishima, Tatsuo Hishinuma and Yutaka Genchi

Abstract In this study, we attempts to evaluate customer reaction to the labelling of food residuals recycling and CO_2 reduction for pork, using a web marketing survey and an in-store survey. The results are as follows: (1)The willingness to pay (WTP) for the pork produced with feed from food residuals was approximately an additional 19.3 yen / 100g-pork in comparison with to ordinary Japanese pork; (2) The WTP for life cycle-CO_2 (LC-CO_2) reduction was approximately an additional 0.4 yen / g-CO_2. It can be concluded that many consumers have positive feelings towards pork produced with feed made from food residuals and see added value in this process. Moreover, it is suggested that labels with information concerning resource recycling and LC-CO_2 reduction would encourage consumers to purchase the pork produced with feed from food residuals.

1 Introduction

In Japan, approximately 21 million tons of food residuals are generated every year. As most food residuals are high-moisture, they are not suitable for incineration. Moreover, landfill sites for incineration residue are limited. In recent years, food residuals from urban areas have come to be regarded as a renewable resource, and recycling is expected to lessen overall environmental burdens. The food waste recycling system has been in the centre of attention and a “Law for the Promotion of the Utilization of Recyclable Food Resources (Food Recycling Law)” has been established in 2000. The law names composting, bio-gasification, and feeding as possible ways of recycling of food waste. Currently, more than half of all food waste discarded from food-related industries is recycled while hardly any households-waste is recycled. The Food Recycling Law particularly specifies

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the use of food residuals as livestock feed as a preferred way of utilizing its components and calorie content. Furthermore, in our study it is cleared that the pork produced with feed made from food residuals emits less lifecycle CO_2 compared to ordinary Japanese pork [1].
However, for the dissemination of feed from food residuals it is essential that consumers actively evaluate such pork with respect to resource recycling and LC-CO_2 reduction, and ultimately purchase such pork. Therefore, it is important to analyse consumers' acceptance and to display relevant information and by this promote purchasing.
In this study, we attempt to evaluate customer reaction to the labelling of food-residual-recycling and CO_2 reduction for pork by using a web marketing survey and an in-store survey.

2 Methodology

2.1 Internet marketing survey

2.1.1 Conjoint analysis

First, the customer reaction to informations concerning food-residual-recycling and CO_2 reduction for pork is quantified with help of a conjoint analysis. The analysis is one of the econometric methods often adopted to evaluate goods such as the environment which are not treated on the market [2]. In this analysis, several question methods can be applied, such as ranking and pairwise valuation and choice experiments (CE). In this study, CE was adopted which is closest to the actual purchase behaviour. Items which measure utilities are called "attributes" and their levels "standards." Four alternative plans composed of the four attributes were provided within the questionnaire. Respondents were asked to select their most preferred choice. Figure 1 shows an example of a questionnaire used in this study.
The collected data are then analysed econometrically using the following conditional logit model in the background of the CE. CE is based on the random utility function shown in Equation (1):

$$U_{in} = V_{in} + \varepsilon_{in} = (\beta_1 X_1)_{in} + (\beta_2 X_2)_{in} + \cdots + \varepsilon_{in} \tag{1}$$

where U describes the total utility, V is the observable component of the total utility, ε is the unobservable component, X is the vector of the attributes, β describes parameters of attributes, i is the number of alternatives, generally called profiles, and n is the number of respondents. Parameters of the observable utility function V are estimated using the conditional logit model.

Q. Please select your preferred pork, considering the balance between each attribute.
Then, pleasecheck the plan you selected at the bottom column.

	Pork 1	Pork 2	Pork 3	Pork 4
Feeding of feed from food residuals	Used	Used	Not used	Not used
CO2 fractuatuation	50g increase	50g decrease	50g decrease	Not influctuation
Production area	Domestic	Homegrown	Foreign Countries	Domestic
Added expense	100 JPY	50 JPY	50 JPY	0 JPY
	↓	↓		↓
Check the prefered pork				

Fig. 1: Exemplary questionnaire sheet

When ε is assumed to be independently and identically distributed with a Gumbel distribution (a type 1 extreme value distribution), the probability P_{ni} that alternative i is selected within the set of all possible alternatives C = {1, 2, … ,J}is displayed in Equation (2).

$$P_{ni} = \frac{\exp(V_{ni})}{\sum_{j=1}^{J} \exp(V_{nj})} \tag{2}$$

The log likelihood function for the maximum likelihood estimate is shown in Equation (3).

$$\ln L = \sum_{n=1}^{N} \sum_{j=1}^{J} d_{ni} \ln P_{ni} \tag{3}$$

N is the number of respondentsa and d_{ni} is the dummy variable (d_{ni} = 1 when individual n selects alternative i, and d_{ni} = 0 when individual n selects any alternative other than i). Subsequently, the utility parameters maximising Equation (3) are calculated.

If conjoint analysis examines attributes including the payment, it can estimate the utility of products in monetary values. By comparing the marginal utility of attributes with the payment, the monetary values per unit of change for each attribute (marginal willingness to pay (MWTP)) can be calculated with help of Equation (4).

$$MWTP_{x1} = -\frac{\beta_{x1}}{\beta_{payment}} \tag{4}$$

2.1.2 Survey design

Table 1 shows the attributes and standards of the conjoint analysis in this study. Attribute 1 is “with or without feeding of feed from food residuals.” Attribute 2 is “fluctuation of CO_2 generation.” As there is a possibility of CO_2 generation in the drying process during feed production, the standard of CO_2 increase was set to include such information. Attribute 3 is “production area” here “homegrown” represents circulation within the region. Finally, attribute 4 is “added expenses”, employed for estimating the MWTP. From these attributes and standards, we identified alternative plans using Orthoplan of SPSS Conjoint. Sets composed of these alternatives, as shown in Figure 1, were provided to each respondent seven times (Pork 4 is ordinary Japanese pork).

Tab. 1: Attributes and standards of conjoint analysis

	Feed from food residuals	CO_2 fluctuation	Production area	Added expenses
Level 1	No use	100g increase	Domestic (not homegrown)	0 JPY
Level 2	Use	50g increase	Homegrown	10 JPY
Level 3	-	No fluctuation	Foreign countries	50 JPY
Level 4	-	50g decrease	-	100 JPY
Level 5	-	100g decrease	-	-

The Internet survey was implemented to 3,946 residents of Tokyo metropolitan area in December 2009.

2.2 In-store survey

In addition to the above mentioned internet survey, another survey was conducted at shops that actually sell pork produced by using feed made from food residuals. First, a point-of-purchase (POP) advertising tool was prepared that contained information regarding the recycling circle and CO_2 emissions (Figure 2). Our study has revealed that pork produced with liquefied feed made from food residuals can reduce CO_2 emission by about 31g-CO_2 / 100g-pork compared with ordinary Japanese pork [1]. Second, with the POP advertising displayed at the meat sales counter, the buying behaviour of the customers was observed and a survey was conducted after the customers finished their expenses was conducted. In this survey, while being shown image of the POP advertising, the customers were asked questions regarding the following items:

1) Awareness of pork produced with liquefied feed made from food residuals
2) Experience of having bought such pork
3) Recognition rate of the POP advertising displayed in the survey
4) Understandeability of the display of environmental information
5) Necessity to display environmental information
6) Influence of such information on buying decision
7) Reasons to buy/not to buy such pork and
8) Willingness to pay for such pork (compared with ordinary Japanese pork)

The survey was conducted at two shops at one day in February, 2011, and 210 sample data sets were collected.

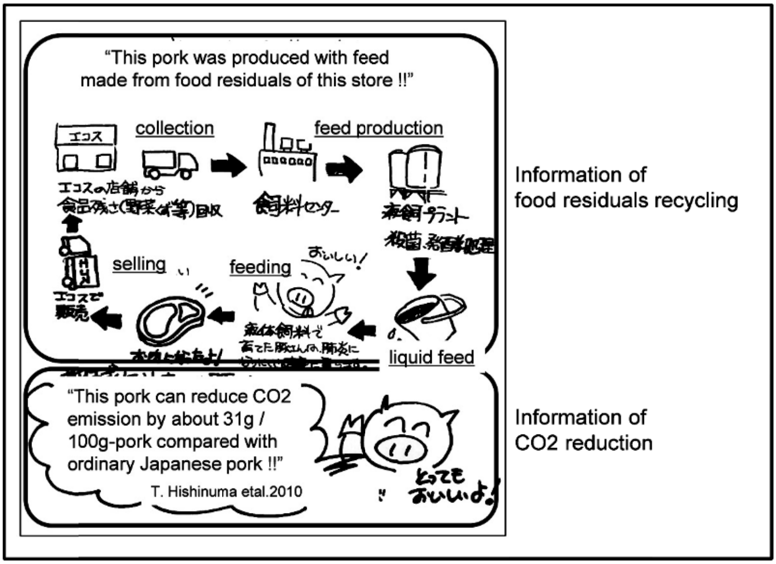

Fig. 2: Point-of-purchase (POP) advertising tool

3 Results and discussion

3.1 Internet marketing survey

We assumed the following linear model, for the analysis of the survey (Equation 5). Then we presumed the parameter β.

$$V = \sum_{d=1}^{1} \beta_{EF} EF_d + \beta_{cf} CF_d + \sum_{d=1}^{2} \beta_{AP} AP_d + \beta_{\cos t} COST \qquad (5)$$

EF is a dummy variable that indicates whether feed from food residuals was used or not, AP is a dummy variable for the production region, CF is the amount of fluctuation of CO_2 generation, and COST is the amount of added expenses. The outcome of the analysis is exhibited in Table 2.

Tab. 2: Estimation result of conjoint analysis

		β	MWTP(JPY)	
Feeding of feed from food residuals		4.38E-01	19.3	***
CO_2 fluctuation		-8.13E-03	-0.4	***
Production area		-	-	-
	Foreign country	-1.65E+00	-72.7	***
	Homegrown	3.79E-01	16.7	***
Added expenses		-2.27E-02	-	***
Likelihood ratio index (LRI)		0.223		

Per 100g pork, ***Significant at < 1%, 1 EURO= about 110 yen

The utility of the pork produced with feed from food residuals (attribute 1) is positive; and the amount of MWTP was approximately 19.3 yen/100g-pork in comparison with that of ordinary Japanese pork. This means that consumers assign additional value to food residuals recycling. It is suggested that the high acceptance is also due to the information provided about food residual-recycling. Concerning the fluctuation of CO_2 generation in the second attribute, utility to emission increase became negative (as cost to reduction was positive). The amount of WTP for LC-CO_2 reduction was approximately an additional 0.4 yen/g-CO_2. According to our research, pork produced with liquid feed made from food residuals reduces emissions of approximately 31g-CO_2/100g-pork in comparison with ordinary Japanese pork [1]. Therefore, an additional expense of approximately 12.4 yen/100g-pork can be expected. In addition, when a previous food residuals recycling is included, the additional expense is approximately 31.7 yen/100g-pork.

Based on these results, it is assumed that including the information for resource recycling and LC-CO_2 reduction on the labels would lead consumers to purchase the pork produced with feed from food residuals.

3.2 In-store survey

The in-store survey results are as follows. Combinations of responses like “knows very well” and “do not know much but heard about it” were turned out to be 44.1% of all subjects. Of those, 76.3% had bought such pork at least once. This indicates that promoting familiarity can increase the demand.

However, only 21.0% of all subjects noticed the displayed POP advertising. Only 14% of those who had not heard of such pork before took note of the display. Therefore, the influence of the POP advertising on buying behaviour could not be observed in the survey. After being shown the image of the POP advertising, however, about 60% answered that it was easy to understand and 95% answered that it could influence their buying intention. In addition, 82.6% replied that such information was useful. Thus, it can be concluded that the contents of the provided information was highly appreciated.

Moreover, as shown in Figure 3, about half of the subjects indicated their willingness to pay 20 yen /100g-pork than for ordinary Japanese pork. The results were similar to the results from internet survey.

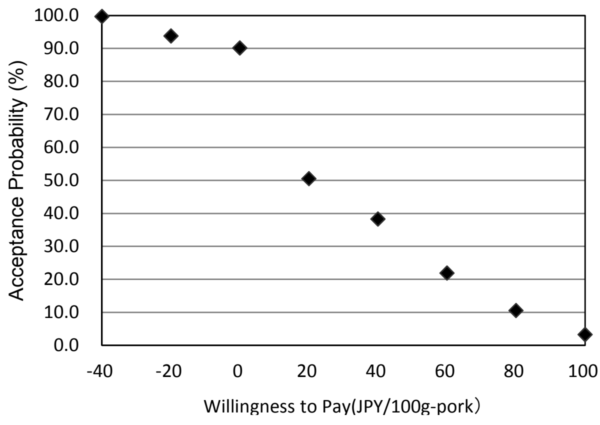

Fig. 3: Willingness to pay for the pork produced with feed made from food residuals

4 Conclusions

In this study, we attempted to evaluate customer reaction to the labelling of food residual-recycling and CO_2 reduction for pork using a web marketing survey and an in-store survey. As a result, many consumers assigned added value pork produces feed made from food residuals. Providing more information regarding food residual-recycling and CO_2 reduction for pork production will increase the awareness of consumers and acceptance will also be increased. However, because the actual changes in buying behaviour could not be examined in this survey, our next task is to review the method and period for which the information is provided so that we are able to analyse whether it actually affects buying behaviour.

References

[1] Hishinuma T, Kurishima H and Genchi Y (2010) Life cycle assessment of GHG emissions from Japanese pork production using feed from food residuals. Greenhouse Cases and Animal Agriculture Conference, Banff (Canada)

[2] Motoshita M, Sakagami M, Kudoh Y, Inaba A (2009) Consumer's acceptance of CO2 reductive choices in shopping and improvement potential of supporting information. The 4th International Conference on Life Cycle Management, Cape town

PART VI:
LCM in the Packaging Sector

Role of Packaging in LCA of Food Products

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Abstract This article presents the results from life cycle assessment case studies of packed food products made in the Futupack2010EKO Project, where environmental impacts of different food packaging options were investigated. Also environmental impact scenarios resulting from the unutilised food supply caused by food wasted in households as a function of different sizes of packaging were included. The studied environmental impacts were climate change, eutrophication and acidification. A consumer survey was carried out to determine and model the amount of food waste from consumers. The results of the LCA case studies showed that the production chain of the wasted food was usually a more significant source of environmental impacts than the packaging production chain. Packaging solutions that minimise the generation of food will lead to the lowest amount of total environmental impacts over the entire product-packaging-chain.

1 Introduction

Investigations and comparisons of the environmental impacts of packaging are often made for the packages alone, which ignores the basic functions of packaging. The role of packaging is to protect, enable distribution, inform, and help display products on the shop shelf. The assumption of the study was that a package that packaging which fails to fulfil these functions properly will lead to great wastage due to damaged products and subsequently causing bigger and totally unnecessary negative environmental impacts. It has been claimed that 15-25% of the climate impacts of consumption are caused by food and nutrition [1]. The hypothesis of the study was that without proper packaging the impacts would be even higher and that the contribution of packaging alone to climate impacts is very much lower than the impacts that would arise from possible food loss [2].

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One key objective of our project was to evaluate the role of the packaging in preventing food losses. In practice, the environmental impacts of three food products and the corresponding packaging alternatives and their recovery options were assessed by LCA in separate case studies. One unique element within these case studies was a consumer survey on the amounts of food waste of the selected product groups generated by households, as a function of different sizes of packaging. It was expected that this might be a dominating factor when comparing environmental impacts of different food packaging solutions.
Twelve Finnish companies and associations representing the food and ICT-industry, packaging and packaging material manufacturers, as well as a purchasing and logistics company of a major retailer are involved in the project together with five research institutions. The participating companies were Borealis Polymers, Environmental Register of Packaging PYR, Fazer, the Finnish Corrugated Board Association, HK Ruokatalo, Inex Partners, M-real, Nokia, Pyroll, Raisio, Stora Enso and UPM Raflatac. The study was carried out by MTT, Lappeenranta University of Technology, Aalto University School of Economics, Association of Packaging Technology and Research, and VTT.

2 Methodological issues, data sources and modelling

The environmental impacts of the three food products and the corresponding packaging alternatives and their recovery options were assessed by LCA in the case studies. The environmental impacts of food product chains were analysed holistically including also the waste management and recovery options of packaging alternatives. The studied impact classes were climate change, eutrophication and acidification and the characterisation was modelled according to the sources [3-5]. The studied food products were sliced dark rye bread, whole meat cold cults (ham), and Soygurt-drink (soy based yoghurt like product). The functional unit (FU) was defined to be 1000kg of each product consumed.
The data concerning the conversion of packaging was obtained from the producers in all three cases and the data on the production chains of basic packaging materials was mainly based on secondary data sources (Plastics Europe/APME, EAA). The data concerning liquid packaging board was obtained from the LCA-study made by the producer. For corrugated board the LCA-study of average Nordic corrugated board was obtained from the Finnish Corrugated Board Association. The packaging alternatives studied are presented in Table 1.

The data concerning the bread baking process was a weighted average of three bakeries belonging to Fazer Bakeries. The data relating to flour production in the mill was obtained from Fazer Mill and Mixes. Crop production emissions were modelled using activity data of crop cultivation from the Finnish national quality database and for imported rye the average European data from Ecoinvent was used. The data concerning fertiliser and lime production chains and the main part of the data of other agricultural raw materials were obtained from the producers. The modelling of pig farming activities and the assessment of related environmental burdens, such as GHG emissions from manure management, were based on the growth data of pigs supplied by Tike, the Information Centre of the Ministry of Agriculture and Forestry. The feed production data was obtained from the large feed producers in Finland and the impacts of grains were modelled as for rye. The data concerning the slaughtering was obtained from HK Ruokatalo's slaughterhouse. The data concerning the production process of cold ham cuts was obtained from HK Ruokatalo's production plant.

Soybeans used in the production of Soygurt were cultivated in Ohio, USA. The supplied data was specific activity data from soybean cultivators in Ohio from 170 farms. Nitrous oxide emissions were modelled by using data from American investigations of nitrous oxide emissions from soya cultivation in the USA [6]. The data of the production process of Soygurt was obtained from the production plant of Raisio. The blueberry additive production data was obtained directly from its producer and rough calculations were made of the fuel consumption of the blueberry picking including the return flights of the blueberry pickers from South-East Asia.

Packaging in the disposal phase and related waste management options differ regionally, and thus four scenarios were made for waste management modelling. Consumers' willingness to sort their packaging and food waste also differ from consumer to consumer. For instance fibre based packages may be of benefit if the opportunity for recycling is arranged. On the other hand, packages of any material may end up in land filling, if it is difficult for consumers to empty them off food residues. Waste management of the losses of the product itself was also modelled by different scenarios.

Tab. 1: The packaging alternatives of the case studies

Soygurt (soy based yoghurt like product)				
Primary package	Liquid packaging board with aluminium barrier (aseptic brick shape package)		Polypropylene cup with aluminium cover	
Product size	750ml		150ml	
Package weight	23g		6g	
Secondary package	Corrugated board		Corrugated board	
Rye bread				
Primary package	Polypropylene (PP)	Polypropylene (PP)	Polyethylene (PE)	Paper/PE with PP window
Product size	500g (9 slices)	220g (4 slices)	500g (9 slices)	500g (9 slices)
Package weight	3.6g	2.6g	5.3g	8.2g
Secondary package	PE boxes (reused)	PE boxes (reused)	PE boxes (reused)	PE boxes (reused)
Ham				
Primary package	PP/PE/PA/EVOH PP/PE/EVOH (upper lid)	Carton/PE/EVOH PP/PE/EVOH (upper lid)	APET/EVOH/PE PE/EVOH/PET (upper lid)	APET/EVOH/PE PE/EVOH/PET (upper lid)
Product size	300g and 150g	300g and 150 g	300g	150g
Package weight	5.3g	12.5g	15.5g	7.9g
Secondary package	PE boxes (reused)	PE boxes (reused)	PE boxes (reused)	PE boxes (reused)

We had two present state and two future scenarios (Table 2). The two present state cases represent a metropolitan area with average recovery rates: one scenario without the energy recovery option (scenario 1) and the other with an energy recovery option for mixed plastics (scenario 2). In both present state options a share of the waste ends up in a landfill and a part is lost to material recovery and/or to waste water treatment. These proportions were based mainly on the Finnish packaging waste statistics gathered annually by PYR, the environmental register of packaging. The other two waste management options are future scenarios, of which the first emphasises energy recovery instead of material recovery and land filling for all waste types (scenario 3). The latter scenario emphasises material recovery for the materials that can be recycled (fibres and plastics of tertiary packaging in retail level) and the rest of the materials are recovered as energy (scenario 4). The amount of packaging waste is calculated on the basis of the functional unit and the food waste amounts reflect the results of the consumer survey.

Tab. 2: Waste management scenarios of the project.

Waste management scenarios	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Name of the scenario	Present state without energy recovery	Present state with energy recovery	Energy recovery scenario	Maximum recycling scenario

The direct emissions from waste treatment and waste transportation are included in the study, as well as indirect emissions from fuel production. Avoided emissions are also taken into account: If the biodegradable material is composted, peat production for the purpose of soil enrichment is avoided. Correspondingly, the energy recovery from waste substitutes heat production from other energy sources. Natural gas was identified as the main fuel to be replaced.

When materials are recycled, some credit must be allocated to the primary product and some to the secondary product (usage). In this study, for fibre based packaging materials, this problem was solved using an open allocation procedure in an open product system as presented in ISO/TR 14049 [7]. Allocation factors are based on the number of times the same material is recycled.

The amount of food waste generated in households was estimated with a help of consumer survey and experimental data. In the case of ham and Soygurt the consumer survey was extended to cover cold cuts of all types of meat and all yoghurt-like products (milk and soy based). The internet-based consumer survey covered over 500 respondents for each product group. According to the responses received the amounts of waste appeared to be only 1-4%, for the selected food groups which is quite small amount compared to many other studies, for example according to WRAP [8]. The Soygurt producer conducted emptying experiments to investigate how much product is left in the package after the consumer empties the package in different ways.

It should be noted that waste amounts are only based on consumers' own estimations and are therefore only indicative. The majority claimed that they don't waste any food but in some households the amount of waste was as high as 10-20% of the purchased amount and based on the finding and other food waste studies the following scenarios were formulated: For dark bread 0, 5.5 and 11%

waste, for ham 0, 4, 8 and 16% waste and for Soygurt 2, 4, and 6% waste for polypropylene cup and 5, 8 and 11% waste for liquid packaging board.

3 Results

According to the results of the study the carbon footprint of the production chain of the packages was only 1-3% of the carbon footprint of the whole product-packing systems, except the polypropylene package of Soygurt, in which the package part was over 10% of the total carbon footprint of the package-product system. The other parts of the product chain such as agriculture and food processing are more significant as shown in Figure 1. In the other impact classes of the study, eutrophication and acidification, the contribution of packages to environmental impacts was also low.

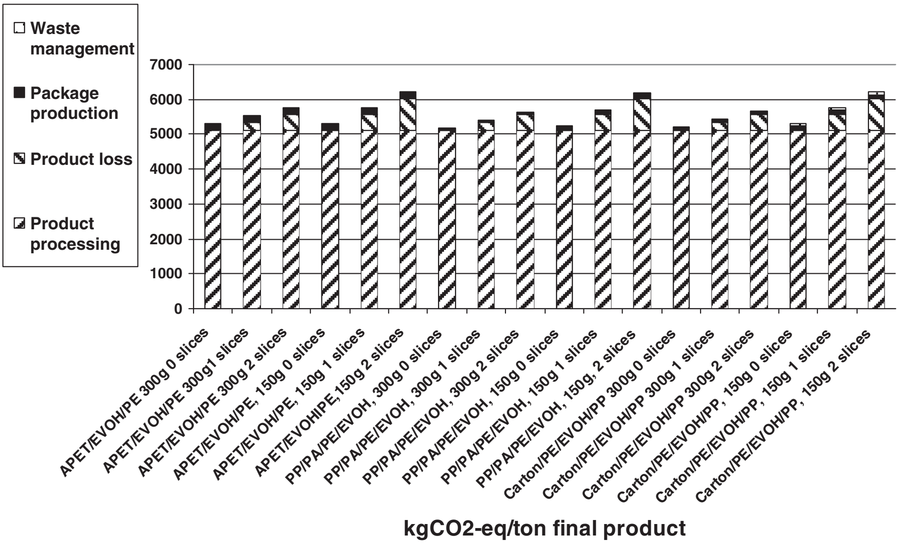

Fig. 1: The carbon footprint of ham system divided in four phases: Waste management, packaging production, production chain of ham and unnecessary production of ham due household waste. (Household product loss scenarios 0 slices, 1 slice and 2 slices of ham). Waste management scenario 1 (present state without energy recovery)

The contribution of the carbon footprints of package production, waste management and production of the food that ends up wasted by consumers (surplus food) are presented in Table 2. The production chain of the surplus food was found much more significant compared to package production and the waste management (Figure 2), even if the product loss caused by the consumers was just a few per cents. The production of even a half a slice of rye bread and one slice of ham cause more greenhouse gas emissions than the production chain of packaging according to Figures 2 and 3. The exception was the Soygurt polypropylene package case, where the package production and the waste management were more significant than producing the surplus food. The reason was the high amount of package material and high water content of the product. The environmental impacts of the product chain of the product were therefore quite low.

Eutrophying and acidifying emissions of producing the surplus food, which ends up as waste by consumers, were also higher than the corresponding emissions of production of packaging. Exceptionally the acidifying emissions of production and waste management of the Soygurt packages were relatively high. The big variation between scenarios in the case of Soygurt was caused by management of surplus product: in scenario 1 it ends up in landfill causing high methane emissions, and in the other scenarios, ending in wastewater treatment, the greenhouse emissions relatively low.

According to Figure 2 and 3, where the production chain of actual food product is excluded, the differences between household waste scenarios are more remarkable concerning the carbon footprint than differences between different packaging alternatives. The differences between the used packing alternatives in eutrophying and acidifying emissions were also not as significant as the differences between household waste scenarios in cases of rye bread and ham.

Tab. 3: The contribution of packaging production, waste management and product loss off the investigated products for the carbon footprint of system (incl. variation between different WM scenarios).

	Dark bread	**Ham**	**Soygurt, LPB**	**Soygurt, plastic**
Production of packages	2-3%	1-2.5%	5-6%	10-13%
Waste management	0-3%	0-1.5%	0.5-7%	1.5-4.5%
Production of surplus food	0 -11%	0-16%	5-11%	2-6%

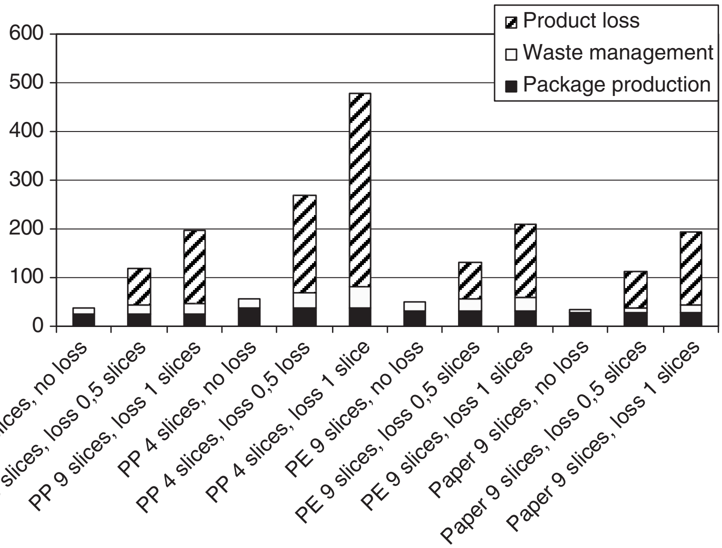

Fig. 2: The share of package production, production chain of product loss and waste management of the carbon footprint of dark bread packaging system, other parts of production chain of bread excluded. The waste management method considered is the current state with energy recovery waste management (scenario 2).

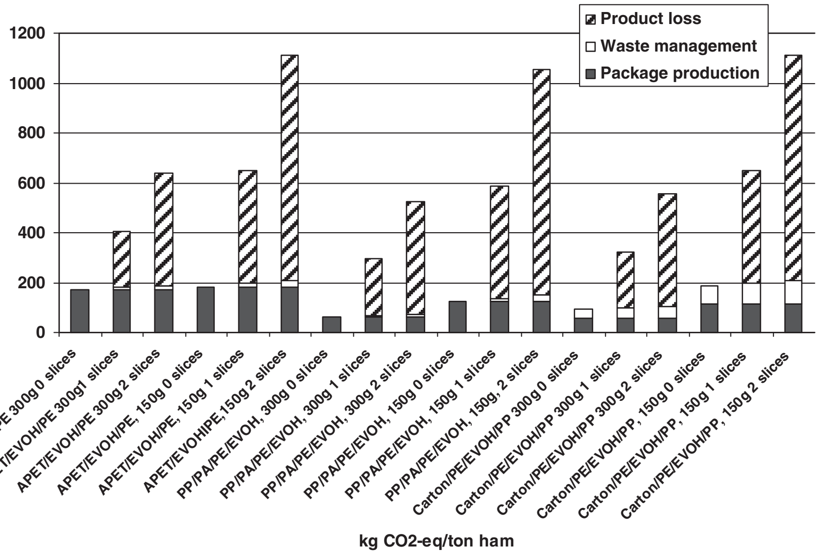

Fig. 3: The share of package production, production chain of product loss and waste management of the carbon footprint of ham packaging system, other parts of production chain of ham excluded. The waste management scenario is the current state without energy recovery option (scenario 1).

The waste management method of product loss and packages also has to be taken into account according to the results of the study. With regard to Soygurt (Figure 4), the differences between the waste management scenarios for carbon footprints were significant. In scenario 1 the surplus product was assumed to end up in landfill, which causes high methane emissions and in other scenarios the product loss ends up in wastewater treatment, which does not produce remarkable greenhouse gas emissions. In other cases the differences in the carbon footprints of waste management were lower. One reason was that lower amounts of food waste were assumed to end up to landfill in waste management scenarios.

Because of the methane emissions in landfills the carbon footprint of the fibre-based package is in scenario one much higher than in other waste management options. In the energy recovery scenarios the carbon dioxide emissions of fibre-based packages were biogenic and so the carbon footprints of waste management of fibre-based packing alternatives were low. If the product loss itself goes to landfill instead of composting or burning, this will also cause high methane emissions (Fig. 4). According to the results burning of the product loss instead of composting can be a useful alternative too. In rye bread and ham cases the significance of waste management was low in relation the whole product chain of the product-packing system concerning also eutrophication and acidification. In this picture polypropylene cup alternative has higher GHG-emissions than LPB alternative. In other studied impact classes LPB alternative had higher eutrophying emissions, but polypropylene cup had higher acidifying emissions. The most important reason of the difference is the amounts of packing material in relation to the production caused by different package sizes.

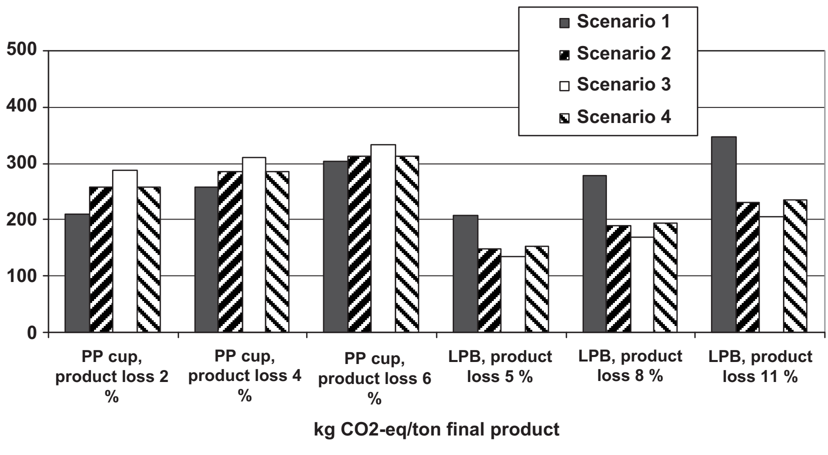

Fig. 4: The product loss, package production chain and waste management in different waste management scenarios in soygurt-case.

4 Conclusions and discussion

The share of environmental impacts of packaging is usually relatively small compared to entire product-package systems. The contribution of the entire food chain to the climate impacts, including agriculture and food processing, is dominating. The share of the production chain and the waste management of packaging is usually 5% of the environmental loads at the highest and in many cases as low as 2%. These concern greenhouse gas emissions, but are same in terms of eutrophication and acidification as well. According to the results, production of even small additional amounts of food products, which ends up to waste causes more environmental impacts than packaging production.

These results reveal for yet another group of food items, that packages are not harmful for environment as such but instead indirectly prevent environmental impacts by enabling for consumer to use the food, (rather than having to cause the spoilage of the product and thus causing the need to manufacture a new product for replacement). Packaging designers can use these results to further emphasise the importance of protectiveness of packaging in distribution chain and to enhance the quality attributes that prevent food losses caused by the consumer. The quality attributes appreciated by consumer are often such that also prevent food losses e.g. prevention of leakages, open-dating, protection, declaration of contents, hygiene, instructions, easiness to empty completely, to dose and to storage, as well as resealability and the optimal quantity of the product in the packaging [9].

Even though preferring quality attributes like these which prevent food losses, the consumer might not realise the environmental effects of their actions, if the food is not consumed as nutrition. The consumer might still think that he is making an environmentally conscious decision by choosing a bigger packaging size because it uses lesser packaging material per food unit. But this could have exactly the opposite effect, if the food is over-consumed, which is common especially in smaller household sizes and for food items that deteriorate quickly. These myths should be clearly communicated to the wide audience. Raising awareness could influence on shopping decisions and eating habits and for packaging development. Consumer is ultimately the decision maker, but the packaging designer can by his part make some sustainable choices for the consumer, by identifying and designing optimal materials and packaging sizes for different households.

It was found that the consumers have difficulties to evaluate the amount of food losses they produce as a function of different packaging solutions. The task to analyse the environmental effects caused by different packaging alternatives is thus complicated, but it would be important in research and packaging development point of view.

Based on our case studies and numerous studies it is not possible to state whether some material is causing more impacts compared to other materials, because this depends on the application of materials and demands for them concerning for example protectiveness. When the product losses were assumed to be the same in parallel packaging alternatives, the variations between different product systems were minor when comparing the whole product-packaging product systems. The package-product systems are individual and this investigation does not give universal introduction in package design concerning package size, shape, material etc.

There are, however, some food products, where the role of packaging is more significant, such as products with high water content e.g. soft drinks.

References

[1] Seppälä J, Mäenpää I, Koskela S, Mattila T, Nissinen A, Katajajuuri J-M, Härmä T, Korhonen M-R, Saarinen M, Virtanen Y (2009) Environmental impacts of material flows caused by the Finnish economy. Suomen ympäristökeskus (SYKE)

[2] Katajajuuri J-M, Virtanen Y (2007) Environmental Impacts of Product Packaging in Finnish Food Production Chains. 5th International Conference LCA In Foods, Gothenburg, Sweden

[3] Solomon S, Qin D, Manning M, Alley RB, Berntsen T, Bindoff NL, Chen Z, Chidthaisong A, Gregory JM, Hegerl GC, Heimann M, Hewitson B, Hoskins BJ, Joos F, Jouzel J, Kattsov V, Lohmann U, Matsuno T, Molina M, Nicholls N, Overpeck J, Raga G, Ramaswamy V, Ren J, Rusticucci M, Somerville R, Stocker TF, Whetton P, Wood RA, Wratt D (2007) Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge

[4] Seppälä J, Knuuttila S. Silvo K (2004) Eutrophication of aquatic ecosystems. A new method for calculating the potential contributions of nitrogen and phosphorus. Int J Life Cycle Assess 9(2):90-100.

[5] Seppälä J, Posch M, Johansson M, Hettelingh J-P (2006) Country-dependent characterisation factors for acidification and terrestrial eutrophication based on accumulated exceedence as an impact category indicator. Int J Life Cycle Assess 11(6):403-416.

[6] Ogle SM, Del Grosso J, Adler PR, Parton WJ (2008) Soil. Nitrous Oxide Emissions with Crop Production for Biofuel: Implications for Greenhouse Gas Mitigation. Conference The Lifecycle Carbon Footprint of Biofuels, Miami Beach, FL, USA

[7] International Organization for Standardization (2000) ISO/TR 14049, Environmental management - Life cycle assessment - Examples of

application of ISO 14041 to goal and scope definition in inventory analysis.

[8] WRAP Waste and Resources Action Programme (2008) The Food We Waste.

[9] Williams H, Wikström F, Löfgren M (2008) A life cycle perspective on environmental effects of customer focused packaging development. J Cleaner Prod 16(7):853-859.

Packaging Legislation and Unintended Consequences: A Case Study on the Necessity of Life Cycle Management

James Michael Martinez

Abstract This paper presents a case study of the unintended consequences associated with ignoring life cycle management (LCM) tools in enacting restrictive packaging legislation. In 1988, the city council in Portland, Oregon, USA, enacted an ordinance requiring food vendors to discontinue the use of polystyrene foam foodservice products. In the ensuing years, it became clear that the Portland ban had failed to improve environmental quality because the city council ignored LCM data on the environmental advantages of foam. LCM that considers multitudinous variables can improve decision-making and lead to effective environmental stewardship; unfortunately, policymakers sometimes ignore LCM data, relying instead on outdated, unreliable information or subscribing to conventional wisdom that produces facile, short-sighted conclusions, as the Portland case study illustrates.

1 Introduction

As life cycle management (LCM) has become a desirable means of encouraging environmental, economical, and social sustainability, policymakers have responded by enacting legislative initiatives, especially in the area of product packaging, to promote environmental stewardship. Such initiatives, when they properly employ LCM tools, are laudable. Ignoring or misusing LCM tools, however, can lead to unintended consequences. Policymakers may resolve to prohibit the use of particular products owing to widespread misperceptions, yet the ban may have an opposite effect than what the policymaker intended, ironically undermining LCM goals [1].

The case of Portland, Oregon, USA, illustrates the problem of unintended consequences. A picturesque city of more than 400,000 people situated at the confluence of the Willamette and Columbia rivers in the state of Oregon in the

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northwestern United States, Portland officials tout the city as the most “environmentally friendly” area in the 48 contiguous states. Since the 1980s, city council members have striven to enact “green” measures to promote Portland’s “liveability.” Recent initiatives include a bicycle transportation plan to reduce traffic congestion and a funding plan to improve storm water management [2].

2 The 1988 Portland ordinance

Portland’s “green” initiatives are not new. More than two decades ago, during the summer of 1988, the city council considered a measure prohibiting the use of polystyrene foam for prepared food in restaurants, grocery stores, and retail establishments. Activists contended that the use of foam was contrary to the goal of showcasing an “environmentally friendly” city. Nonetheless, the ordinance sponsor, Councilman Bob Koch, withdrew his proposal when he learned that the supposed environmental problems associated with foam did not exist and, in fact, a ban would increase rather than decrease environmental impacts as well as drive up costs for foodservice operators [3].

Another commissioner, Earl Blumenauer, anxious to ingratiate himself with environmental activists, reintroduced the ordinance despite Koch’s evidence that such a ban would be counterproductive. Blumenauer pushed his colleagues to enact the ordinance without considering LCM data or performing a credible investigation into the salient issues. He succeeded. The final ordinance read, “As of January 1, 1990, restaurants, grocery stores and other retail vendors have been prohibited from using polystyrene foam (PSF) containers for prepared food. The ban also applies to vendors who renew a lease or initially lease city space and activities that require a city permit (including for use of parks).” Following the opening paragraph, the ordinance repeated a litany of complaints about polystyrene foam that either were no longer accurate or had never been accurate. Moreover, because the complaints relied on hearsay and conventional wisdom in lieu of considering LCM data, city council members provided no mechanism for measuring whether the ordinance would damage environmental quality owing to unintended consequences [3,4].

3 Subsequent investigations

3.1 Hocking and Rathje

In the ensuing years, it became clear that when polystyrene foam foodservice products were unavailable in a marketplace, suppliers and consumers usually relied on alternative containers manufactured from polyethylene (PE) plastic-coated paperboard. Despite repeated entreaties by polystyrene manufacturers and other interested parties, members of the Portland City Council refused to consider LCM data comparing the environmental attributes of these competing foodservice products. In the absence of action by the city council, outside researchers initiated their own investigations [3,5].

In 1991, Dr. Martin Hocking, an associate professor of chemistry at the University of Victoria, British Columbia, Canada, performed a study of foam and paper disposables and found that "the environmental impact from the chemicals and energy used in making paper cups, as well as the emissions from incinerating or burying paper cups, exceeds the impact of making and disposing of cups made of plastic foam." Hocking undertook follow-up studies that validated and expanded on his initial findings [6-8].

During the late 1980s and early 1990s, Dr. William L. Rathje, who was then an archaeologist working with the Garbage Project at the University of Arizona, investigated the contents of the municipal solid waste (MSW) stream as well as the composition of sanitary landfills in the United States. He found that, compared with many other materials, polystyrene foam comprised a small percentage of MSW by weight and volume. U.S. Environmental Protection Agency data corroborated those findings when the agency determined that all polystyrene foodservice packaging accounted for less than 0.5 per cent, by weight and volume, of MSW [9,10].

Along with his writing partner, Cullen Murphy, Dr. Rathje debunked a persistent myth about polystyrene foam deposited into landfills. "The fact that plastic does not biodegrade, which is often cited as one of its great defects, may actually be one of its greatest virtues," they wrote. They explained that landfills are designed to discourage biodegradation. Landfill engineers remove sunlight, oxygen, and water- the three features essential for biodegradation to occur. Thus, although some biodegradation takes place, the goal is to retard decomposition. The fact that polystyrene foam products do not biodegrade is a benefit, not a detriment [11,12].

3.2 Franklin associates and life cycle management

Although the Hocking and Rathje & Murphy studies challenged popular assumptions about solid waste management, the first LCM study of polystyrene foam foodservice products did not appear until more than a decade after those seminal works. In 2006, Franklin Associates, a U.S.-based solid waste management consulting firm, produced a peer-reviewed life cycle inventory (LCI) report on behalf of an industry trade association, the Polystyrene Packaging Council (now called the Plastics Foodservice Packaging Group), comparing an average-weight polystyrene hot beverage cup with the alternative product most likely to be used, an average-weight polyethylene (PE) plastic-coated paperboard hot beverage cup. (A subsequent Franklin Associates study, produced in February 2011, also included polylactic acid [PLA] products in the comparison matrix.) [13,14]. The 2006 study examined more than one size of beverage cup, but it is instructive to report on 16-ounce hot cups as an example of the findings. Using LCM techniques, Franklin Associates researchers determined that:

(1) An average-weight 16oz. polystyrene hot beverage cup requires two-thirds as much energy to manufacture as an average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup with a corrugated cup sleeve (see Figure 1). The figures below compare an expandable polystyrene (EPS, or polystyrene foam) cup, a polyethylene (PE) plastic-coated paperboard hot beverage cup (PE/Paper), and a polyethylene (PE) plastic-coated paperboard hot beverage cup with a corrugated cup sleeve (PE/Paper + Sleeve) [13].

(2) An average-weight 16oz. polystyrene hot beverage cup produces slightly more than a third as many air emissions than an average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup (see Figure 2) [13].

(3) The manufacture of an average-weight 16oz. polystyrene hot beverage cup generates less than one third of the waterborne emissions generated during the manufacture of an average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup. See Figure 3 [13].

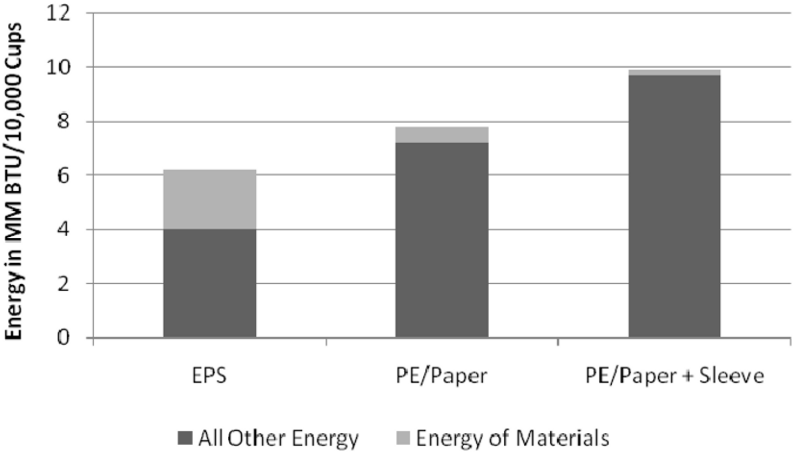

Fig. 1: Energy usage required in the manufacture of 16oz. hot cups

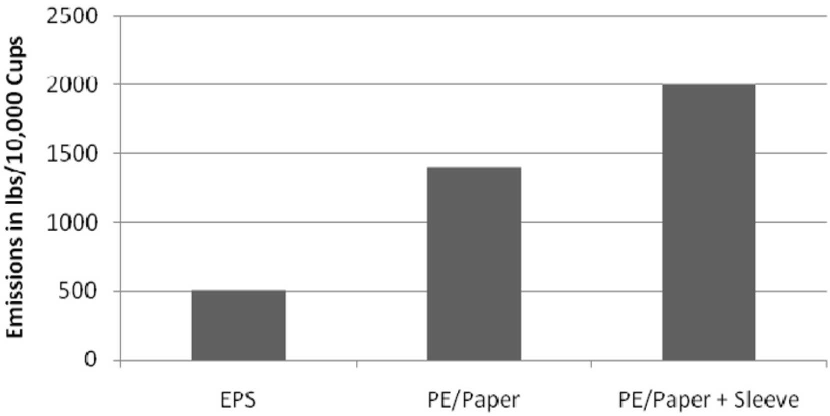

Fig. 2: Air emissions generated in the manufacture of 16oz. hot cups

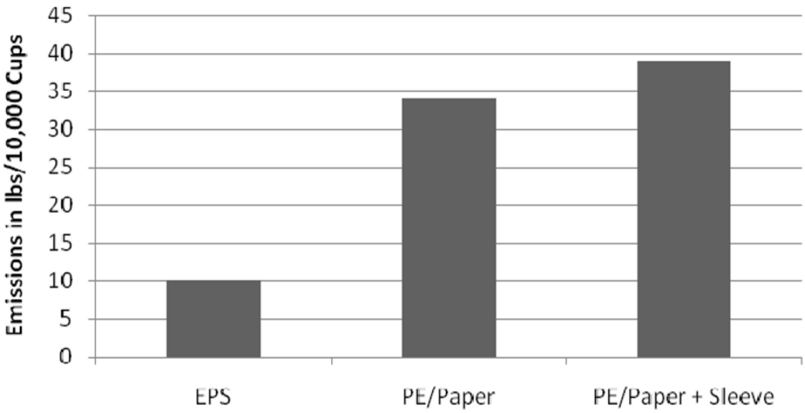

Fig. 3: Waterborne emissions generated in the manufacture of 16oz. hot cups

(4a) When a comparison is made based on greenhouse gas (GHG) emissions, the differences between the two products are much less pronounced. An average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup produces fewer GHG emissions than an average-weight 16oz. polystyrene hot beverage cup, but the differences in emissions are negligible (see Figure 4) [13].

(4b) An average-weight polystyrene 16oz. hot beverage cup produces two thirds as many GHG emissions as an average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup with a corrugated cup sleeve (see Figure 4) [13].

(5a) An average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup produces more than three times as much total waste by weight as an average-weight 16oz. polystyrene hot beverage cup (see Figure 5) [13].

(5b) An average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup with a corrugated cup sleeve produces more than five times as much total waste by weight as an average-weight 16oz. polystyrene hot beverage cup. See Figure 5 [13].

(6a) When a comparison is made based on volume, the differences between the two products are much less pronounced. An average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup produces more total waste by volume than an average-weight 16oz. polystyrene hot beverage cup, but the differences in volume are negligible (see Figure 6) [13].

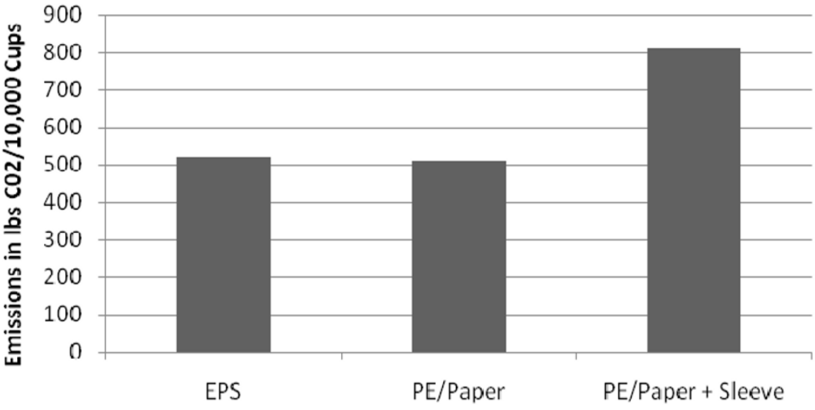

Fig. 4: Greenhouse gas emissions generated in the manufacture of 16oz. hot cups

(6b) An average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup with a corrugated cup sleeve produces approximately twice as much total waste by volume as an average-weight polystyrene 16oz. hot beverage cup (see Figure 6) [13].

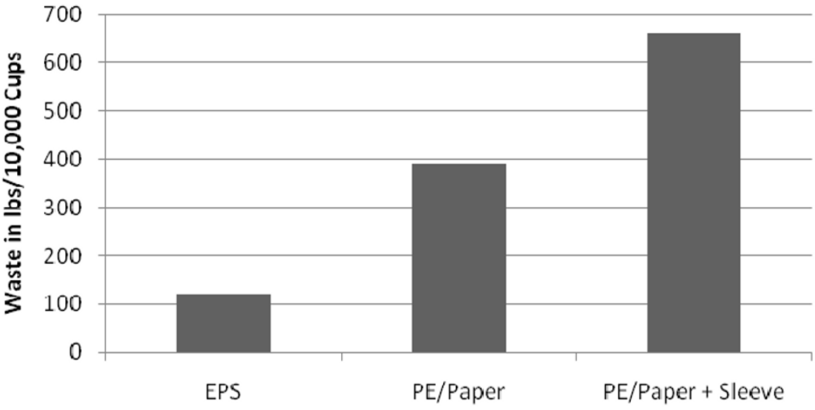

Fig. 5: Solid waste (by weight) for 16oz. hot cups

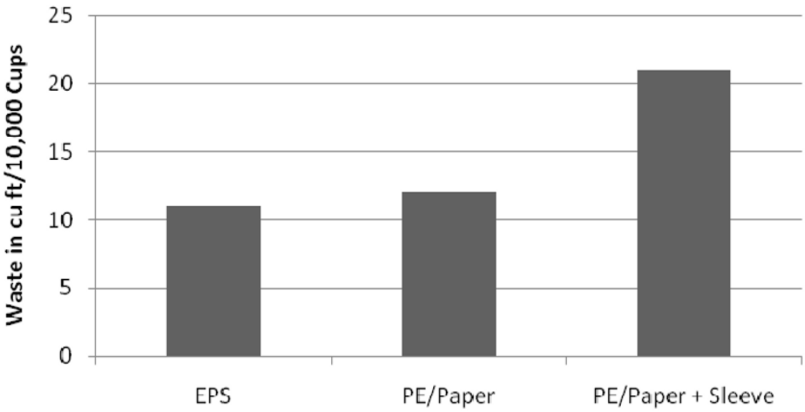

Fig. 6: Solid waste (by volume) for 16oz. hot cups

The conclusion of the Franklin Associates LCI study, as summarised in Table 1, indicated that the manufacture and disposal of an average-weight 16oz. polystyrene hot beverage cup produced fewer negative environmental consequences than did the manufacture and disposal of an average-weight 16oz. polyethylene (PE) plastic-coated paperboard hot beverage cup. When a corrugated cup sleeve was added, the differences were even more pronounced. In the table, a "+" means polystyrene foam contains an environmental advantage; an "=" means the difference is negligible; and a "-" means polystyrene foam contains a disadvantage [13].

Tab. 1: Summary comparison of environmental effects of 16oz. hot cups

Category	EPS v. PE/Paper	EPS v. PE/Paper & Sleeve
Energy usage	+	+
Air emissions	+	+
Waterborne emissions	+	+
GHG emissions	=	+
Solid waste (weight)	+	+
Solid waste (volume)	=	+

3.3 The Cascade Policy Institute

In a 2007 report following up on the Franklin LCI study, a 501(c)(3) non-profit educational organisation, the Cascade Policy Institute, considered the economic effects of Portland's polystyrene ban. According to the institute's report, the ordinance led to higher costs for restaurants, onerous enforcement costs for the city, and reliance on inferior products. Moreover, the ban, intended partially to reduce litter, merely exchanged one type of litter for another [3,15].

4 Conclusion

The lesson here is that the proper use of LCM studies requires rigorous analysis in lieu of relying on conventional wisdom, which can lead to unintended consequences. An example of the unintended consequences can be found in the Portland, Oregon, ordinance prohibiting the use of polystyrene foam foodservice products. The city council's decision to ban foam, which remains in effect as of this writing, led to the use of products that cause greater harm to the natural environment and are more expensive to purchase. Efficacious public policy requires data produced by LCM studies to promote environmental, economical, and social sustainability. Entities that ignore life cycle data and analyses do so at their peril.

References

[1] Stevens C (1994) The environmental life cycle and trade. Organisation for Economic Cooperation and Development (OECD) Observer 118(1):8-10.
[2] <http://www.portlandonline.com/shared/cfm/image.cfm?id=297213> (accessed 18.02.2011).
[3] Hardy M, Charles JA Jr (2007) Sustainable failure: why Portland's polystyrene ban should be repealed. Cascade Policy Institute, Portland, Oregon
[4] Municipal Ordinance §17.102.300, et seq. (1988), <http://www.portlandonline.com/auditor/index.cfm?c=ciiij#cid_215460 > (accessed 21.02.2011).
[5] Alexander JH (1993) In defense of garbage. Praeger
[6] Naj AK (1991) Foam cups damage environment less than paper cups, study says. The Wall Street Journal, 01.02.1991
[7] Hocking MB (1991) Is paper better than plastic? Consumers' Research 74(1):28-29.
[8] Hocking MB (1994) Reusable and disposable cups: an energy-based evaluation. J Environ Manage 18(6):889-899.
[9] Rathje W, Murphy C (1992) Rubbish! the archaeology of garbage. HarperCollins

[10] United States Environmental Protection Agency (2009) Municipal solid waste in the united states, 2008 facts and figures. <http://www.epa.gov/osw/nonhaz/municipal/msw99.htm> (accessed 22.02.2011).
[11] Rathje WL (1989) Rubbish! The Atlantic Monthly 246(6):99-109.
[12] Rathje WL, Murphy C (1992) Five major myths about garbage, and why they're wrong. Smithsonian 23(4):113-122.
[13] Franklin Associates Ltd (2006) Final peer-reviewed report: life cycle inventory of polystyrene foam, bleached paperboard, and corrugated paperboard foodservice products. Prairie Village, Kansas
[14] Franklin Associates Ltd (2011) Life cycle inventory of polystyrene foam, paper-based, and pla foodservice products. Prairie Village, Kansas
[15] The Polystyrene Packaging Council (1996) Polystyrene and its raw material styrene: manufacture and use. Washington DC

Carbon Footprint of Beverage Packaging in the United Kingdom

Haruna Gujba and Adisa Azapagic

Abstract The food and drinks sector is the major user of packaging in the UK, accounting for 70% of the total. With the consumption of packaging in the UK estimated at over ten million tonnes, the life cycle environmental impacts of packaging could be significant. This work focuses on drinks packaging and estimates the carbon footprint of packaging used for five types of beverage in the UK: fruit juice, water, milk, beer and wine. The types of packaging considered are: carton; glass, PET and HDPE bottles; and aluminium and steel cans. The results show that the carton packaging has the lowest carbon footprint ranging from 90-111kg CO_2e/1000 litres of beverage and glass bottle the highest, from 150-761kg CO_2e/1,000 litres. A significant variation has been found in the carbon footprint for the same type of packaging material, mainly influenced by the size and weight of the containers and the recycling rates. The manufacture of raw materials and the packaging are the main hot spots for all packaging.

1 Introduction

Packaging provides many vital functions including protection, storage, portioning and preservation of products as well as information to consumers. This makes it an essential part of a product. In the UK, the production of packaging is estimated at over ten million tonnes [1]. The food and drinks sector accounts for about 70% of the total packaging consumed in the UK; the drinks sector alone uses over 4 million tonnes annually [2]. The sheer production volume as well as waste management (packaging makes up one fifth of all household waste in the UK [3,4]) could potentially have significant environmental impacts along the supply chains.

This paper focuses on drinks packaging and estimates the carbon footprint of different types of packaging used in the UK for milk, fruit juice, water, beer and wine. The aim is to compare the carbon footprints of different packaging and

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identify the hot spots. The study has been carried out following the ISO 14044 LCA methodology [5] and using GaBi software [6].

2 Goal and scope of the study

The goal of the study is to analyse and compare the carbon footprints of different beverage packaging used in the UK. The scope of the study is from ‘cradle to grave’ and the functional unit is defined as ‘the amount of packaging required to deliver 1,000 litres of beverage’. Four beverage product categories are considered: milk, fruit juice, water, beer and wine. The types of packaging used for these products are: carton; glass, PET and HDPE bottles; and aluminium and steel cans.

The system boundary is depicted in Figure 1. The life cycle stages considered are production of raw materials and packaging (including tops and labels), filling, transport and post-consumer waste management. Due to limited data availability, energy used for storage and consumption of beverage as well as the secondary and tertiary packaging are not considered. The production of beverage is also excluded from the system boundary.

3 Data sources and assumptions

Primary data have been used wherever possible, including for types and weights of the packaging and energy consumption at the filling stage. The secondary (LCI) data for raw materials, energy, transport and waste management have been sourced from the Ecoinvent [7], ELCD [8] and GaBi [7] databases.

All the packaging materials and sizes for each beverage have been identified and obtained from a variety of UK retail stores; the results are summarised in Tables 1-4.

All transport is assumed to be by 22 tonne trucks and all distances are set at 100 km. The transport stages include the transport of the raw materials for the containers, tops and labels to the manufacturing site, transport of packaging to the filling stage, the transport of filled packaging to retailers and transport to landfill, incineration, recycling and re-use sites.

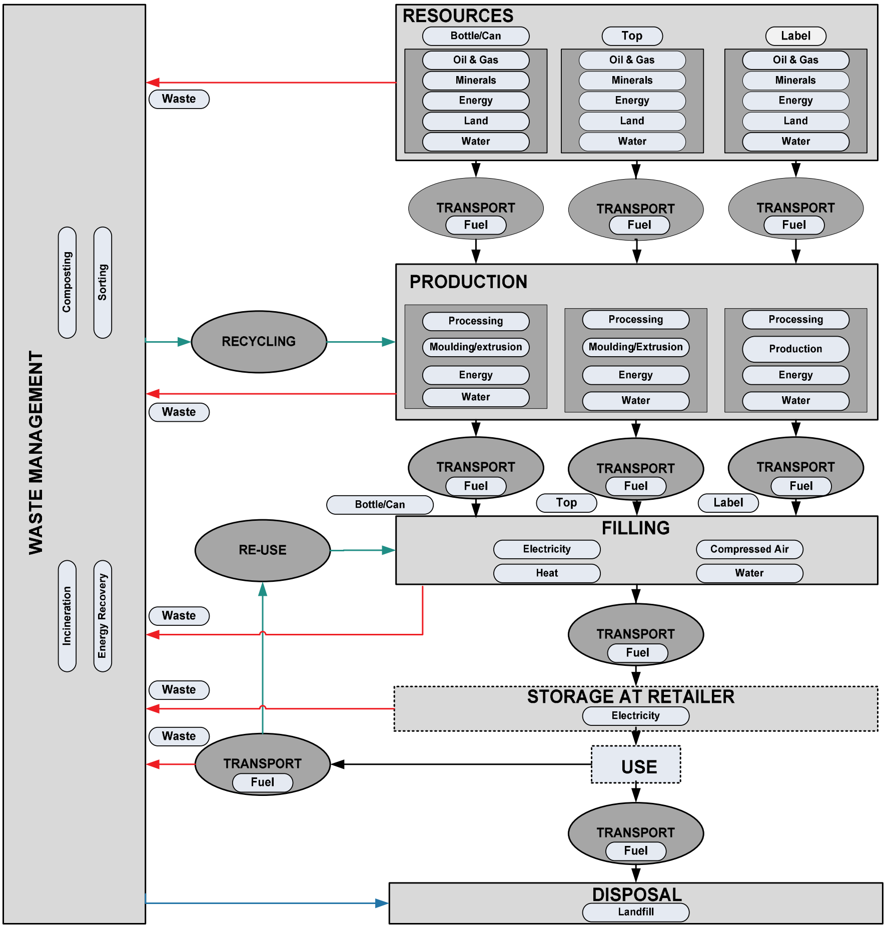


Fig. 1: System boundary for the beverage packaging

The data for end-of-life waste management for all the packaging materials are based on the current UK situation. These data were obtained from various Government and trade agencies as well as literature sources [9-12]. Other hypothetical scenarios have also been considered to study the effects on the results of different waste management options. The following end-of-life options have been considered in different proportions: re-use of packaging, virgin and recycled materials, incineration with energy recovery, incineration without energy recovery and landfilling. Following the convention in LCA, the biogenic carbon dioxide is not considered.

Tab. 1: Characteristics of milk packaging

Container type	HDPE	HDPE	Carton	White glass
Capacity (l)	0.568[a]	1.136[a]	1.000	0.568
Material for top	HDPE	HDPE	PP	Al. foil
Material for label	HDPE	HDPE	-	-
Container weight (kg per 1,000 l)	36.97	29.05	37.00	445.00
Top weight (kg per 1,000 l)	3.01	1.51	3.05	0.25
Label weight (kg per 1,000 l)	0.74	0.51	-	-
Total weight (kg per 1,000 l)	40.72	31.07	40.05	445.25

[a] 0.568 litres = 1 pint; 1.136 litres = 2 pints

Tab. 2: Characteristics of juice packaging

Container type	PET	Carton		White glass	Green glass
Capacity (l)	0.330	0.250	1.000	0.375	0.750
Material for top	PP	PP (straw)	PP	$AlMg_3$[b]	$AlMg_3$
Material for label	LDPE film	-	-	Kraft paper	Kraft paper
Container weight (kg per 1,000 l)	66.67	40.00	33.00	658.67	780.00
Top weight (kg per 1,000 l)	8.88	0.79	1.75	7.73	2.00
Label weight (kg per 1,000 l)	1.27	-	-	0.69	0.67
Total weight (kg per 1,000 l)	76.82	40.79	34.75	667.09	782.67

[b] Alloy with 3% of magnesium

Tab. 3: Characteristics of water packaging

Container type	PET			Green glass
Capacity (litres)	0.500	1.500[c]	1.500[c]	0.750
Material for top	PP	PP	PP	$AlMg_3$ & PP
Material for label	LDPE film	LDPE film	Kraft paper	Kraft paper
Container weight (kg per 1,000 l)	34.00	22.00	29.33	605.30
Top weight (kg per 1,000 l)	4.50	1.50	1.50	2.57
Label weight (kg per 1,000 l)	0.74	0.54	1.25	2.16
Total weight (kg per 1,000 l)	39.24	24.04	32.08	610.03

[c] These bottles are shown respectively as PET1 and PET2 in Figure 4

Tab. 4: Characteristics of beer and wine packaging

Container type	Aluminium	Steel	PET	Green glass	
Capacity (l)	0.500	0.500	2.000	0.750	0.750
Material for top	$AlMg_3$	$AlMg_3$	PP	Wood	$AlMg_3$
Material for label	-	-	Kraft paper	Kraft paper	Kraft paper
Container weight (kg per 1,000 l)	29.9	69.9	22.00	622.67	734.66
Top weight (kg per 1,000 l)	6.10	6.10	1.40	6.73	5.27
Label weight (kg per 1,000 l)	-	-	1.77	1.39	1.36
Total weight (kg per 1,000 l)	36	76	25.17	630.79	741.29

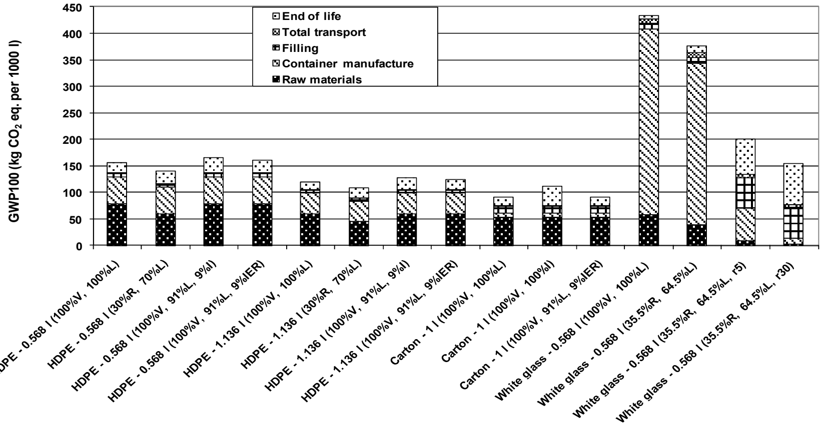
4 Results

4.1 Milk packaging

The results for the milk packaging are shown in Figure 2. As can be seen, carton has the lowest carbon footprint among the options considered, ranging from 90-111kg CO_2e per 1,000 litres of milk, depending on the end-of-life scenarios. The lowest carbon footprint is for the carton (100% virgin, 91% landfilled and 9% incineration with energy recovery) and the highest is for the carton (100% virgin and 100% incineration without energy recovery). The raw materials are the main hot spots, accounting for 48%–60% of the total carbon footprint.
The next best option is the HDPE bottle, with the carbon footprint in the range of 110-170kg CO_2e per 1,000 litres of milk. Using a 30% recycled HDPE decreases the carbon footprint by 10% compared to the virgin bottle for both bottle sizes (1 and 2 pints). Overall, the larger the bottle, the lower the impact due to the lower amount of material required per functional unit. Like cartons, the major hot spots for the HDPE bottle are the raw materials which account for 50% of the impact and the bottle manufacture which contributes 30–36%.
Finally, the carbon footprint of glass bottles ranges from 150-440kg CO_2e per 1,000 litres of milk. As the results in Figure 2 indicate, re-using (glass milk bottles are re-used in the UK; no other glass bottles are re-used) and recycling of glass can reduce the impacts considerably. For example, re-using a bottle 30 times reduces the carbon footprint by 58% compared to a single use and recycling 35.5% of glass reduces the impact by 13% compared to virgin glass. The manufacture of the bottles is the major hot spot, contributing over 80% of the total for the virgin glass; this contribution reduces to 33% when the bottle is re-used

five times and to as low as 8% when it is re-used 30 times. However, the cleaning of the bottles for re-use increases the carbon footprint, contributing about a third to the total for 30 times re-use (see Figure 2).

Overall, comparing the best options for each packaging, using carton containers (100% virgin, 91% landfilled and 9%incineration with energy recovery) saves 17 kg CO_2e over HDPE bottles (1.136 litres, 30% recycled, 70% landfilled) and 64 kg CO_2e over glass bottles (re-used 30 times, 35.5% recycled, 64.5% landfilled).



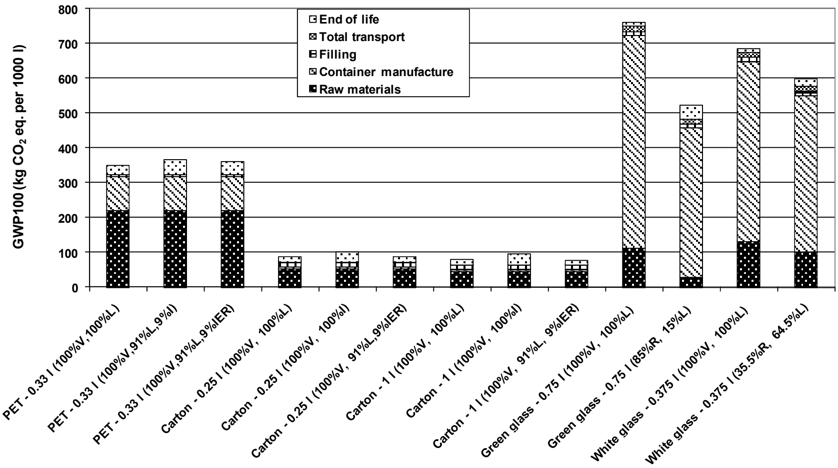
[V – virgin; L – landfill; R – recycled; r – re-use; I – incineration; IER – incineration with energy recovery]

Fig. 2: Carbon footprint of milk packaging

4.2 Juice packaging

The carbon footprints of cartons, PET and glass bottles used for packaged juice are shown in Figure 3. Overall, carton has the lowest impact ranging from 77-103kg CO_2e/1,000 litres of juice. The carbon footprint of PET bottles is around 350kg CO_2e/1,000 litres while that of glass varies from 524-761kg CO_2e/1,000 litres. At 77kg CO_2e, the 1 litre carton (100% virgin, 91% landfilled and 9% incineration with energy recovery) is the best option while the 0.75 litre glass bottle (100% virgin, 100% landfilled) is the worst option (glass milk bottles are re-used in the UK; no other glass bottles are re-used). Using an 85% recycled glass reduces the carbon footprint of glass bottles by 32% compared to virgin glass.

The raw materials are the major carbon hot spot for the cartons and PET bottles, contributing on average 48%–60% and 60%, respectively. The majority of the carbon footprint (80%) for glass is from the bottle manufacture.

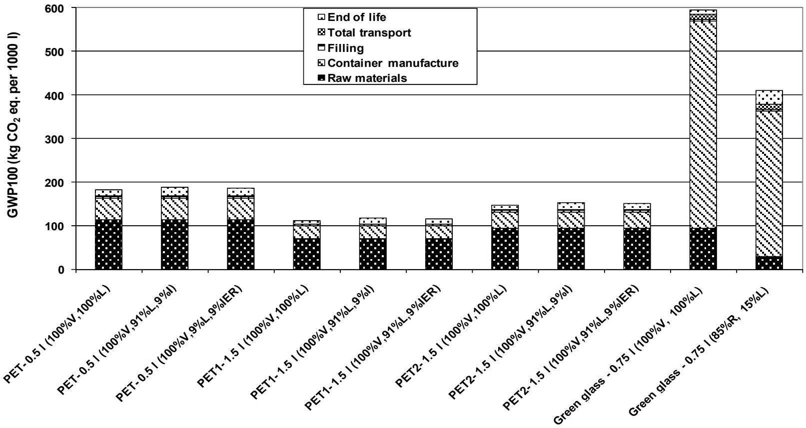


[V – virgin; L – landfill; R – recycled; I – incineration; IER – incineration with energy recovery]

Fig. 3: Carbon footprint of juice packaging

4.3 Water packaging

As shown in Figure 4, amongst the options considered, the best option for water packaging is the 1.5 litre PET1 bottle (100% virgin, 100% landfilled), with the carbon footprint equal to 112kg CO_2e /1,000 litres of water (see Table 3 for the characteristics). This is 3.6 times lower than the carbon footprint of the best glass bottle option (0.75 litre, 85% recycled, 15% landfilled). The influence of the weight of the bottles is also significant: for example, the carbon footprint of PET2 1.5 litre bottle is over 30% higher than that of PET1 1.5 litre bottle (both 100% virgin, 100% landfilled).

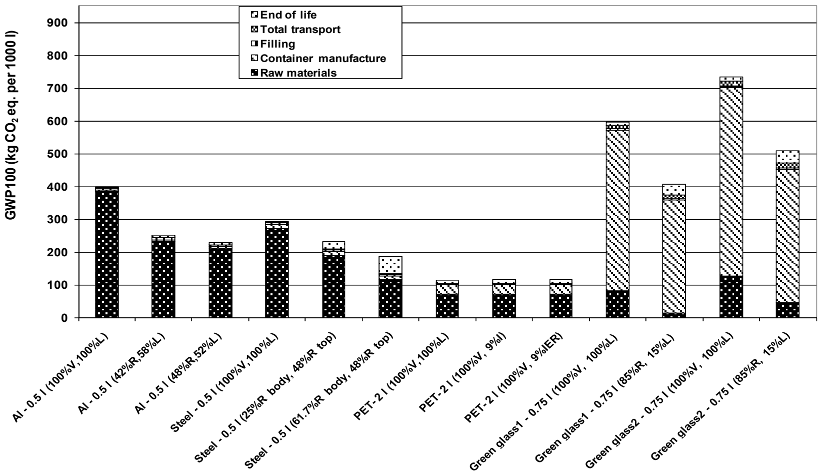


[V – virgin; L – landfill; R – recycled; I – incineration; IER – incineration with energy recovery]

Fig. 4: Carbon footprint of water packaging

4.4 Beer and wine packaging

Comparison of the beer and wine packaging is shown in Figure 5. At 113kg CO_2e/ 1,000 litres of alcohol, the 2 litre PET bottles (100% virgin, 100% landfilled) have the lowest carbon footprint. This is 1.5; 2; and 3.6 times lower than the carbon footprint for the best steel, aluminium and glass options, respectively.



[V – virgin; L – landfill; R – recycled; I – incineration; IER – incineration with energy recovery]

Fig. 5: Carbon footprint of beer and wine packaging

As before, a difference in weight of the containers can influence the carbon footprint results significantly. For example, the 15% difference in weight between Glass1 and Glass2 leads to a difference in the carbon footprint of around 20% (see Table 4 for the container weights and Figure 6 for the carbon footprint results).

5 Conclusions

The results of this study show that carton and plastic (HDPE and PET) drink packaging have lower carbon footprints than aluminium, steel and glass. However, with higher recycling rates, aluminium and steel cans are comparable to PET. Generally, re-use of glass bottles is preferred to recycling and landfill. The findings also indicate that, for the same type of material, larger containers have lower carbon footprints than the smaller ones due to the lower amounts of materials required per functional unit. However, the carbon footprints can differ considerably for the same type and size of packaging due to different weights. The main hot spots for all types of packaging are the manufacture of raw materials and packaging.

References

[1] Department of Environment, Food and Rural Affairs, DEFRA (2005) Producer responsibility obligations (packaging waste) regulations 1997 (as amended), <http://www.defra.gov.uk/environment/waste/producer/packaging/documents/package-datanote.pdf>

[2] Keynote (2003) Packaging (food & drink) industry. <http://www.keynote.co.uk/market-intelligence/view/product>

[3] Department of Environment, Food and Rural Affairs, DEFRA (2009) Making the most of packaging: a strategy for a low-carbon economy. <http://www.defra.gov.uk/environment/waste/producer/packaging/documents/full-packaging-strategy.pdf>

[4] WRAP (2008) Lightweighting the can pack, Waste and Resources Action Programme. <http://www.wrap.org.uk/downloads/Lightweighting_the_can_pack_FINAL1.50658f00.5891.pdf>

[5] ISO 14044 (2006): Environmental management - life cycle assessment - requirements and guidelines, Geneva

[6] PE International (2009) GaBi 4.3 software and database for Life Cycle Engineering, <http://www.gabi-software.com/uk-ireland/software/gabi-4/>

[7] Ecoinvent Swiss Centre for Life Cycle Inventories (2007) Ecoinvent Database v 2.0. Dübendorf

[8] EU JRC (2009), European Reference Life Cycle Data System (ELCD) v. 1.0.1. 2007.

[9] Waste and Resources Action Programme, WRAP (2007) Using recycled content in plastic packaging: the benefits. <http://www.wrap.org.uk/downloads/Using_recycled_content_in_plastic_packaging_the_benefits.996bf005.3597.pdf>

[10] Department of Environment, Food and Rural Affairs, DEFRA (2009) Packaging: achievement statistics. <http://www.defra.gov.uk/ environment/waste/producer/packaging/data.htm>

[11] Department of Environment, Food and Rural Affairs, DEFRA (2007) Incineration of municipal solid waste. <http://www.defra.gov.uk/environment/waste/wip/newtech/pdf/incineration.pdf>

[12] Waste Online (2005) Packaging recycling information sheet. <http://www.wasteonline.org.uk/resources/InformationSheets/Packaging.htm>

Enhanced Resource Efficiency with Packaging Steel

Evelyne Frauman and Norbert Hatscher

Abstract Packaging steel is used for the safe and efficient distribution of different products worldwide. In the long line of improvement of steel packaging the total volume of canned products per tonne of packaging steel has increased dramatically in the last 50 years. This result is directly linked to a better use of the resources necessary for making packaging steel. The recycling rate for packaging steel in the EU is now over 70%. Efficient recycling can be seen as a multi-use system from the material point of view. The recycling of the core material enables the industry to avoid a CO_2 burden in the production route. These above mentioned characteristics have to be taken into account when studying the life cycle of packaging steel. Resource efficiency has direct effects on other life cycle parameters such as greenhouse gas emissions or energy use. This will be shown in some examples. An outlook for future developments will be given as well.

1 Introduction

Packaging steel is used for the safe and efficient distribution of different products worldwide. The combination of a strong material with perfect barrier behaviours enables to deliver safe products to the consumer without losses [1].

The recycling rate for packaging steel in the EU is now 71%; some member states such as Germany or Belgium have reached recycling rates of more than 90% [2,3]. Efficient recycling can also be seen as a multi-use system from the material point of view. Steel recycling is typically the electric arc furnaces (EAF) process that converts steel scrap into new steel by remelting it, but steel recycling also occurs when steel scrap is added during the basic oxide furnace (BOF) process [4]. The recycling of the core material enables the industry to avoid a CO_2 burden in the production route.

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Furthermore, it can be demonstrated that in the long line of improvement of steel packaging the total volume of packaging steel per tonne of canned products has decreased dramatically in the last 50 years [2]. This result is directly linked to a better use of the resources necessary for making packaging steel.
These above mentioned characteristics have to be taken into account when studying the life cycle of packaging steel. Resource efficiency has direct effects on other life cycle parameters such as greenhouse gas emissions or energy use [5]. This will be shown in some examples.
For many years, APEAL has commissioned life cycle studies and sustainability studies to clarify the benefits of packaging steel for the environment and the society. An outlook for future developments will be given in this article as well.

2 Recycling rate and reduction of CO_2 emissions

APEAL publishes yearly updates of the steel packaging recycling rates achieved in the 27 European countries. These data are compared with the Eurostats' Environmental Data Centre on Waste [6] data, which are published about one year later, and their coherence is verified.

2.1 Increasing recycling rates in Europe

The recycling rates for steel have continuously increased since the 1970's. It is easy to explain what makes steel exceptionally recyclable: its magnetic properties make steel easy and economical to sort and recover, and well-established routes for collection and recovery of steel cans have ensured increasing recycling rates over the years.
It can be seen in Figure 1 that steel is amongst the most recycled packaging materials in Europe.

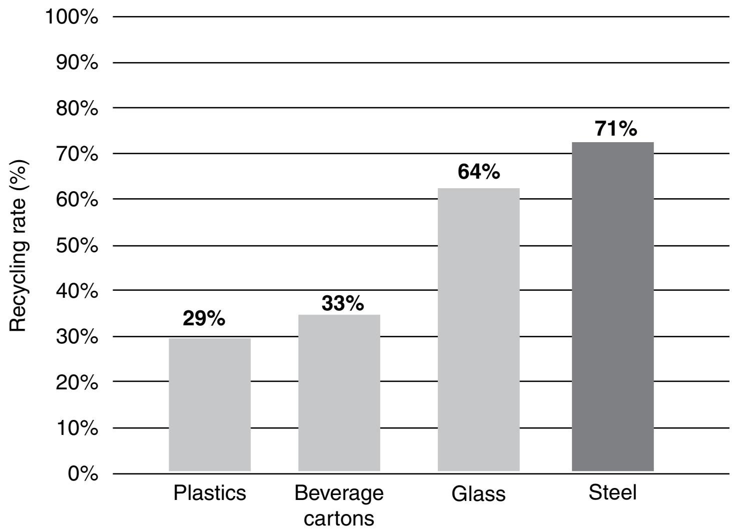

Fig. 1: EU 27 recycling rates in 2008 for different packaging materials (Source: industry experts - APEAL, FEVE, ACE, PlasticsEurope) [3]

2.2 Calculation of reduction of CO_2 emissions due to recycling

Equivalence between the recycling rate and the CO_2 emission can be calculated given the following [7]:

- Minimum recycling 0%
- Maximum recycling 100%
- CO_2 primary route 100%
- CO_2 secondary route 29%

CO_2 primary route refers to the CO_2 emissions through primary steel production route; CO_2 secondary route refers to the CO_2 emissions through secondary steel production route (European average for EAF and BF recycling routes).The link between recycling rate and CO_2 emission is given inthe following formula (Equation 1):

$$\text{Indice } CO_2 = CO_2\text{-}1 * (1\text{-}RR) + CO_2\text{-}2 * RR \qquad (1)$$

With:Indice CO_2 = Indice of emission of CO_2; CO_2-1 = CO_2 primary = 100%; RR = recycling rate; CO_2-2 = CO_2 secondary = 29%
This equation enables to calculate the indices of CO_2 emission according to the recycling rates in Table 1.

Tab. 1: EU 27 recycling rates

Year	Recycling rate [%]	CO_2 emissions [%]
1991	25.0	82.1
1992	26.0	81.4
1993	29.0	79.3
1994	34.0	75.7
1995	41.0	70.7
1996	45.0	67.8
1997	52.0	62.8
1998	51.0	63.5
1999	47.0	66.4
2000	49.0	65.0
2001	55.0	60.7
2002	60.0	57.1
2003	61.0	56.4
2004	63.0	55.0
2005	63.0	55.0
2006	66.0	52.8
2007	69.0	50.7
2008	71.0	49.2

The Table 1 data show a reverse tendency between recycling rates and indices of CO_2 emission, as is seen in Figure 2. Indeed, with an increased efficiency of recycling, the indicator of CO_2 emissions decreases.

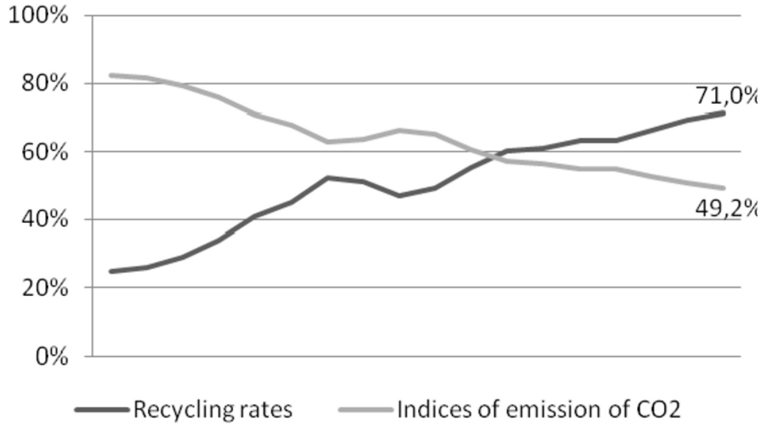

Fig. 2: Evolution of EU 27 recycling rates and equivalent reduction in indice of CO_2 emissions (from 1991 to 2008)

3 Downgauging and reduction of CO_2 emissions

The optimisation of the packaging steel production processes and the enhancement of recycling are an important part of the ecological improvement of steel packaging. On the product side the downgauging of packaging helps saving resources, energy and emissions, too. Figure 3 shows the weight reduction of some standard cans since 1990.

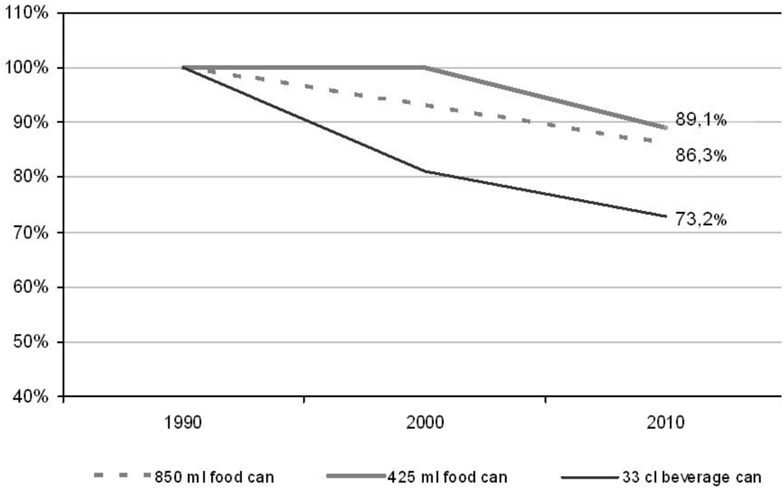

Fig. 3: Development of weights of some standard steel packing in Europe [7]

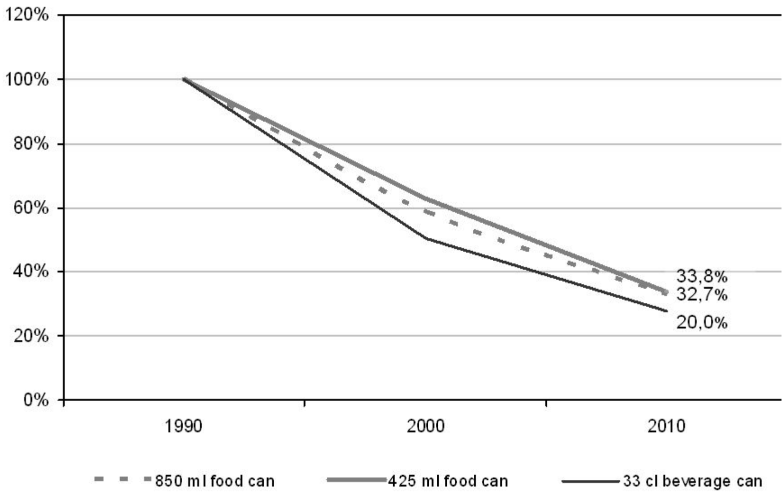

Fig. 4: Development of CO_2 emissions of some standard steel packing in Europe [7]

The downgauging process directly results in a reduction of CO_2 emissions and energy use during the steel and packaging production and - related to transports - also during the total life cycle [3,5]. In combination with the above shown effects of recycling, as shown in Figure 4, the decrease of the CO_2 emissions per packaging is enormous.

4 Steel industry and sustainability

Goods packed in steel have a high benefit for the consumer over a long period of time. Indeed, the shelf-life is increased as the packaged goods are protected by the impermeability of the can and the protection it gives to light (shelf-life refers to "the period between the manufacture and the retail purchase of a food product, during which time the product is in a state of satisfactory quality in terms of nutritional value, taste, texture, and appearance") [8, 9]. In the previous sections it has been shown that the environmental impact has decreased dramatically in the last years. This has of course been the case in the whole long history of steel packaging. Environmental improvements show only one part of the whole evolution of steel packaging. Every product, also every packaging, has to be measured by its economic and social impacts, in other words - it has to be sustainable.

The social aspect of sustainability includes safety and health in the production of steel, which always has been very important for the steel industry [10]. The social aspect is important to make food available for the consumer in a cost efficient way. When steel packaging is referred to, it is interesting to note that the distribution system requires energy only for transporting the food. No additional cost in energy is required, as to modify for instance the ambient atmosphere. Indeed, the packaging is gas tight and unbreakable. It is also non-transparent, ensuring that the filling goods are protected perfectly.

The steel for packaging industry has intensively increased the steel grades available in order to offer the material needed to make lighter steel cans possible in the market.

5 Future work - development of an LCA for steel for packaging

Researches in the world steel industry underline the importance of the use of a life cycle analysis for environmental assessment of their production [6]. Several

studies are available comparing through the means of life cycle analysis (LCA) different construction materials amongst which steel [4,11,12]
In the past APEAL has taken part in different LCA/LCI projects [5,13]. At the moment a European LCI for packaging steel in ongoing. The LCI data will be reviewed by independent experts and will be published by APEAL. This LCI data set will be updated periodically.

References

[1] Robertson G (2006) Food Packaging Principles and Practice. Taylor and Francis, New York
[2] Schüler K (2009) Aufkommen und Verwertung von Verpackungsabfällen in Deutschland im Jahr 2008.Umweltbundesamt, Dessau
[3] <http://www.apeal.org> (accessed 16.03.2011)
[4] Brimacombe L, Coleman N,Honess C (2005) Recycling, reuse and the sustainability of steel.MilleniumSteel International
[5] <http://www.worldsteel.org> (accessed 16.03.2011)
[6] <http://epp.eurostat.ec.europa.eu/portal/page/portal/waste/introduction> (accessed 17.03.2011)
[7] BUWAL (1998)Ökoinventare für Verpackungen.Band 1, Schriftenreihe Umwelt Nr. 250/1, Abfälle, Bundesamt für Umwelt, Wald und Landschaft Bern
[8] Anonymous (1993)Shelf-life of foods: Guidelines for its determination and prediction.Institute of Food Science and Technology, London
[9] Oldring PKT,Nehring U (2007) Packaging Materials 7- Metal packaging for foodstuffs. International Life Sciences Institute (ILSI)
[10] ButerbaughCP, McGrath MC, Novotny JK, Voss ZT, Brunner JM,PeasleeKD (2007)Life cycle greenhouse gas emission comparison of steel products with other materials.Association for Iron & Steel Technology AIST,AISTech Conference, Indianapolis
[11] Buchart-KorolD (2011)Significance of environmental life cycle assessment (LCA) method in the iron and steel industry.Metalurgija50(3):205-208.
[12] Zhang X, Su X, Huang Z (2007)Comparison of LCA on steel and concrete-construction office buildings: a case study<http://www.inive.org/members_area/medias/pdf/Inive%5CIAQVEC2007%5CZhang_4.pdf> (accessed 17.03.2011)
[13] Sustainability Report of the world steel industry (2008)<http://www.worldsteel.org/pictures/publicationfiles/Sustainability%20Report%202008_English.pdf> (accessed 08.04.2011)

Damage Assessment Model for Freshwater Consumption and a Case Study on PET Bottle Production Applied New Technology for Water Footprint Reduction

Masaharu Motoshita, Norihiro Itsubo, Kiyotaka Tahara and Atsushi Inaba

Abstract The effects of freshwater consumption will differ from country to country depending on the availability of freshwater resource and adaptability to water scarcity. In this study, damage assessment model focused on human health and social asset damage caused by freshwater consumption was developed based on statistical data analysis. Calculated damage factors showed that undernourishment damage due to agricultural water scarcity was a dominant effect in most countries due to the ripple effects of international food trade. Case study of PET bottle production was performed to verify the availability of calculated damage factors and quantitatively assess the effectiveness of freshwater savings by applying new developed filling technology of PET bottle. No small reduction of environmental impact by advanced filling technology will be expected through the savings of freshwater consumption.

1 Introduction

It is generally well known that large amount of water exists on the earth (almost 1.4 trillion km^3) but easily available freshwater accounts for very limited part (about 0.0075%) [1]. Population growth in the world will threaten sufficient freshwater supply and result in the expansion of suffering people from freshwater scarcity (almost 30-35% in 2025) [2]. In this context, the amount of water consumption through life cycle ("water footprint") of products and services has been highly concerned.

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Besides, water resources distribute unevenly in the world. Vulnerability to freshwater scarcity depends on not only the amount of freshwater resource but also social environment (like capacity of infrastructures for freshwater supply, economic power for avoiding freshwater scarcity, medical treatment opportunity etc.). Thus, the same amount of freshwater consumption may cause different effects. In order to discuss water footprint of products and services, it is necessary to consider specific damage caused by freshwater consumption.
In this paper, the overview of the assessment methodology for domestic and agricultural water scarcity due to freshwater consumption will be introduced. The applicability and availability of the methodology will be verified by applying to the case study of water footprint analysis of PET bottle production.

2 Damage assessment modelling

Freshwater consumption will result in freshwater scarcity for human activity and ecosystem. In this paper, the effects of freshwater scarcity for human activity were focused and modelled as a first step of damage assessment modelling. Freshwater use for human activity can be classified into agricultural, domestic and industrial use. Basic freshwater demands for human activity are for drinking and food production (agricultural crops). Thus, domestic and agricultural water scarcity caused by freshwater consumption was chosen as targets of this damage assessment modelling.

2.1 Human health damage due to domestic water scarcity

Freshwater consumption will decrease the amount of available freshwater for domestic use. The loss of available freshwater for domestic use will decrease the accessible population rate to safe drinking water. Thus, the relationship between the amount of available freshwater for domestic use and accessibility loss to safe water was modelled by applying multiple regression analysis based on country scale data with the consideration of adaptability to the shortage of freshwater resource by preparing infrastructure for water supply (GDP per capita and capital formation expenditure per capita). Loss of accessibility to safe water will lead to the increase of human health damage due to infectious diseases. On the other hand, human health damage due to infectious diseases may be related to not only accessibility to safe water but also other factors (generation of infectious sources, contact to the source, sufficient nutrient condition, medical treatment opportunity).

Multiple regression analysis was performed to model the relationship between health damage due to infectious diseases and accessibility loss to safe water. Schematic diagram of assessment flow is shown in Figure 1 and details on the modelling process and results can be referred in the reference [3]. Furthermore, the adaptability to freshwater scarcity will depend on the amount of freshwater resource. Water stress index (WSI) [4] was applied to consider the adaptability to freshwater scarcity in each country. Finally, the increase of infectious disease damage due to the loss of available freshwater for domestic use calculated based on the regression model was multiplied with WSI and the rate of domestic use to total freshwater demand in order to obtain damage factor of unit volume freshwater consumption.

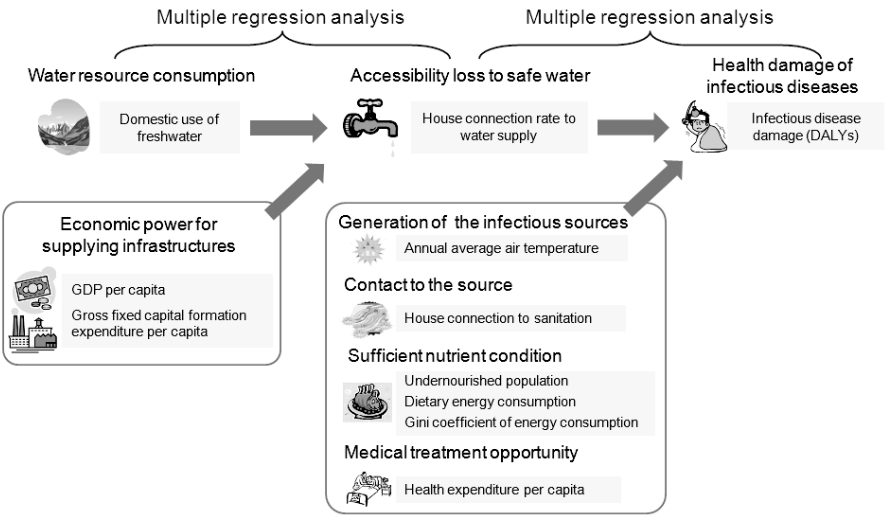

Fig. 1: Schematic diagram of assessment flow on human health damage due to domestic water scarcity

2.2 Human health damage due to agricultural water scarcity

The decrease of available freshwater for agricultural use, due to freshwater consumption, will result in undernourishment through insufficient food supply. In previous studies, human health damage of malnutrition caused by freshwater consumption was modelled by relating malnutrition damage with agricultural/fishery productivity [5] or human development index (HDI) [4]. In this study, cause-effect chain was decomposed into several processes in order to describe the damage flow of agricultural water scarcity more specifically by considering factors directly related to undernourishment.

The overview of the damage assessment flow on agricultural water scarcity is shown in Figure 2. The shortage of agricultural water due to freshwater consumption will lead to the loss of crop production. Productivity loss of crops was calculated on country scale by dividing total dietary energy of crop production with total amount of supplied freshwater for crop production. The loss of food consumption in each country as a result of crop production loss was modelled by considering international food trade relationship. The shortage of dietary energy consumption due to the loss of food consumption will result in the prevalence of undernourishment. However, other factors in addition to average nutrient condition (food consumption), like nutrient condition gap and medical treatment opportunity, may affect on the prevalence of undernourishment. Thus, prediction model of undernourishment prevalence rate was obtained by applying multiple regression analysis with three explanatory variables (average dietary energy consumption per capita, Gini coefficient of dietary energy consumption and health expenditure per capita). Health damage of undernourishment was calculated by multiplying the estimated increase of undernourishment prevalence rate due to agricultural water scarcity with health damage of undernourishment per case. Details and results of above analyses are shown in the literature [6].

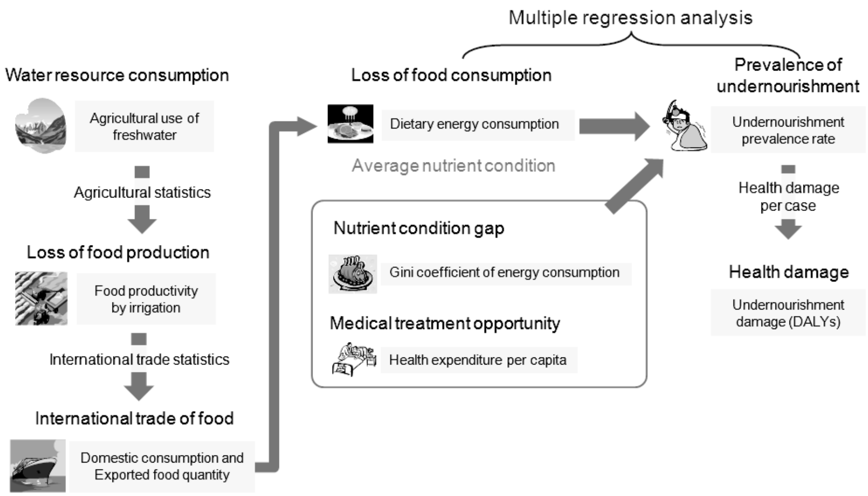

Fig. 2: Schematic diagram of assessment flow on human health damage due to agricultural water scarcity

2.3 Social asset loss due to agricultural water scarcity

Crops as agricultural commodities are one of the social assets as same as other biotic/abiotic resources. Thus, crop production loss due to agricultural water scarcity causes not only undernourishment but also the loss of social asset. The amount of lost social asset due to agricultural water scarcity could be estimated by multiplying the loss of crop production with agricultural commodity prices. Details and results of this analysis are shown in the literature [6].

2.4 Integration of damage factors on each endpoint

Damage on each endpoint is expressed by different units (DALY and US dollar). Weightings on each endpoint in LIME2 methodology (one of the LCIA methodology) were applied in order to integrate damage factors on each endpoint. Weightings on each endpoint were evaluated as economic values (willingness to pay for avoiding unit damage on each endpoint) by applying conjoint analysis based on questionnaire survey to Japanese citizens [7]. After converted into international US dollar (Purchase power price: PPP), obtained weightings are multiplied with damage factors on each endpoint and summed up for each country.

3 Calculated damage factors of freshwater consumption

The distribution map of integrated damage factors of each country is shown in Figure 3. In Figure 3, dark coloured countries show high sensitivity to freshwater consumption. Freshwater consumption in African, Asian and South American countries seems to result in relatively large damage. Basically, developing and emerging countries show high sensitivity. On the other hand, freshwater consumption in some developed countries in Europe also causes no small damage (ex. Netherland, Austria, Italy, Spain and France). Water scarcity caused by freshwater consumption in these developed countries will affect on food consumption in economically disadvantaged countries through international food trade [6].

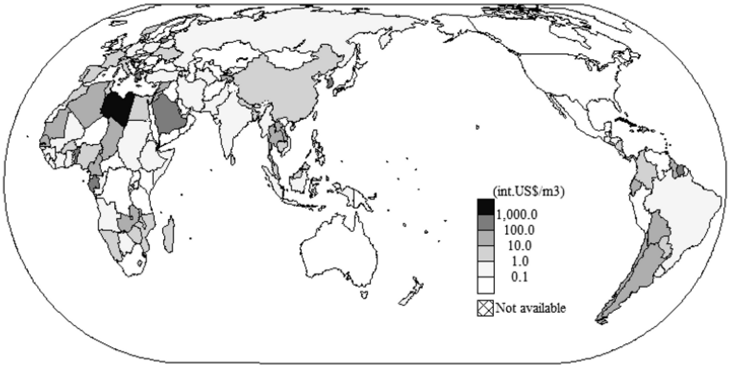

Fig. 3: Distribution map of integrated damage factors of each country

The dominant effect of freshwater consumption differs from country to country. Figure 4 shows the rate of damage on each endpoint to total damage for countries with the high-ranking rate of human health damage due to domestic water scarcity. Top three countries (Ireland, Latvia, Luxembourg) are reported to use no irrigated land for agriculture. Thus, these countries are exceptions with no damage due to agricultural water scarcity (on both human health and social asset). Most of other high-ranking countries locate in African region. Limited accessibility to safe water and existence of infectious sources seem to result in relatively high sensitivity to domestic water scarcity in these countries.

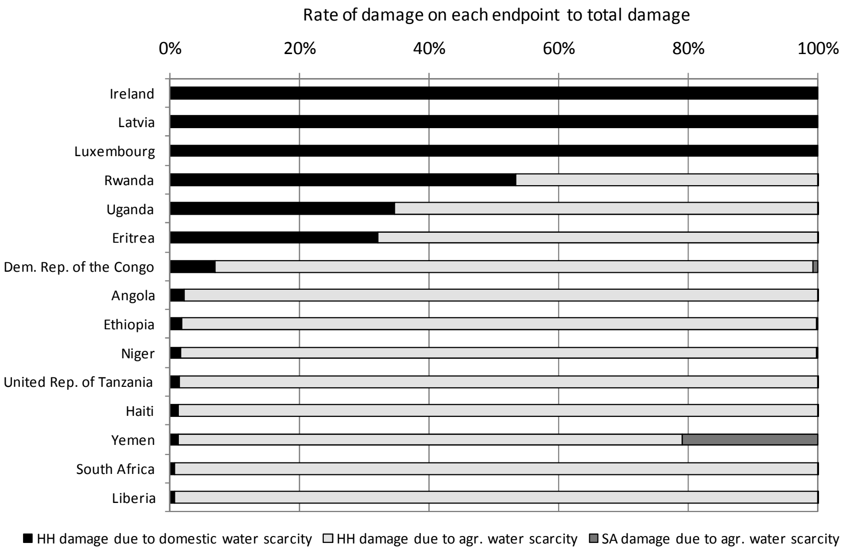

Fig. 4: The rate of damage on each endpoint for countries with the high-ranking rate of human health damage due to domestic water scarcity

Countries with high-ranking rate of social asset damage due to agricultural water scarcity are listed in Figure 5. Listed countries are countries with high productivity of crops. In other words, scarcity of unit volume water will cause large amount loss of crop production in these countries. Furthermore, listed countries can avoid the effects of crop production loss by decreasing the food amount of export or increasing that of import. Consequently, the shortage of food supply will spread to more economically disadvantaged countries (most of them are vulnerable to food supply shortage) and cause severe human health damage. Thus, dominant damage of freshwater consumption in countries, listed in Figure 5, is arisen from agricultural water scarcity compared to domestic water scarcity.

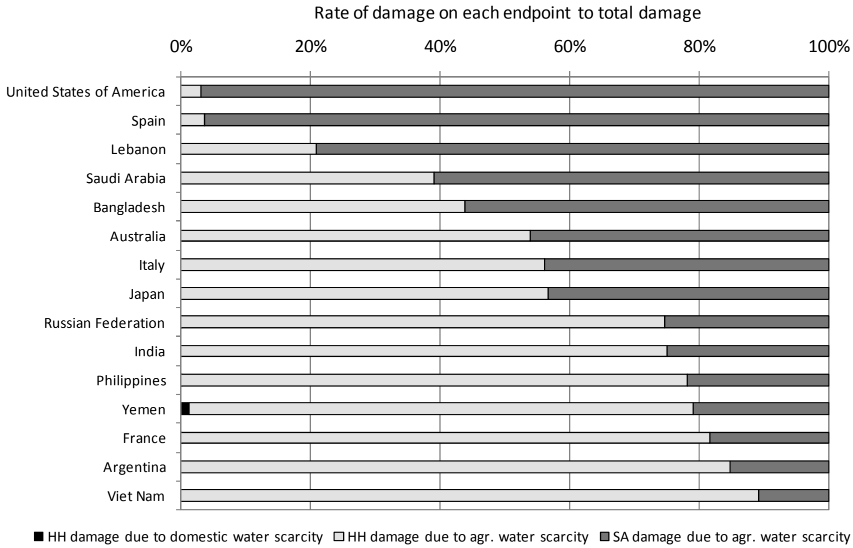

Fig. 5: The rate of damage on each endpoint for countries with the high-ranking rate of social asset damage due to agricultural water scarcity

4 Case study on water footprint of PET bottle production

In this section, case study is performed in order to verify the applicability of damage factors obtained in the previous section. The target of this case study is PET bottle production. PET bottle is mainly made from naphtha but large amount of freshwater is consumed especially for sterilization and rinsing in the filling process. Advanced filling technology has been developed and will contribute to the reduction of freshwater consumption by directly connecting blow-form machine with sterilized filling machine [8]. In this case study, the availability of developed filling technology for the reduction of environmental impacts (mainly freshwater consumption effect) is quantitatively verified by applying damage factors for freshwater consumption compared to that of conventional filling technology.

4.1 Analysis method

Functional unit of the assessment is defined as a 500mL size PET bottle. The scope of this assessment is from pre-form (including raw material production) to waste disposal as shown in Figure 6 and the production of content liquid is excluded in the analysis. Some of consumers may wash their used PET bottle as preparation for recycling. However, it will largely depend on consumers and consumed freshwater in use stage is excluded in the analysis. Input amount of materials and energy in each stage for both systems with advanced and conventional filling technologies was investigated and inventory analysis was conducted by using JEMAI-LCA pro [8].

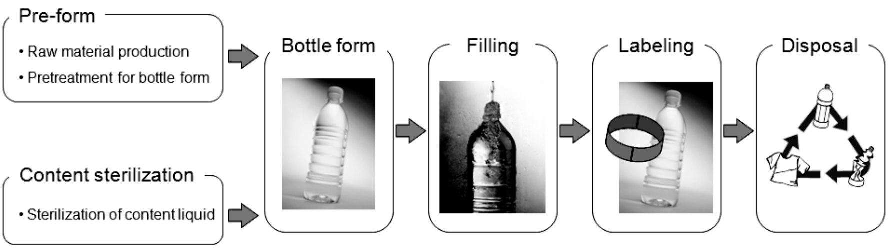

Fig. 6: System boundary of the assessment

4.2 The amount of freshwater use and consumption in each stage

As a result of inventory analysis, the input amount of freshwater in each stage of the system with conventional filling technology is shown in Figure 7. Cooling water use in the pre-form stage dominates large amount of freshwater input. However, cooling water and water for hydro power are not consumptive use of freshwater because they are normally circulated or used in-stream. Thus, their effects on environment will be smaller than that caused by consumptive water.

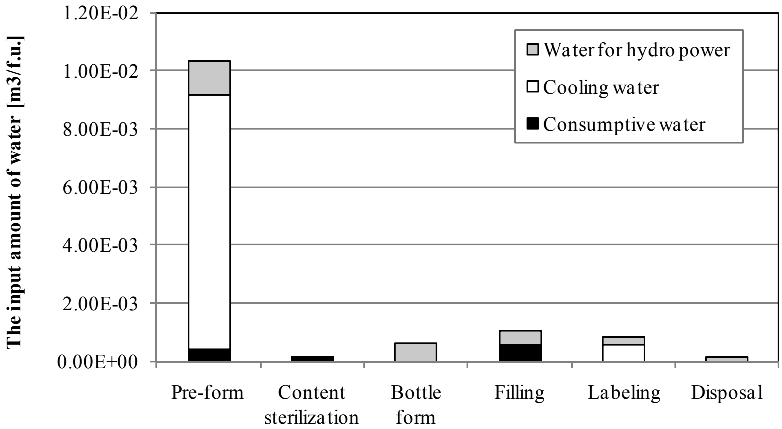

Fig. 7: The input amount of freshwater in each life stage

Figure 8 shows the amount of freshwater only for consumptive use in each stage of both systems with advanced and conventional filling technology. As shown in Figure 7, the amount of consumptive water is smaller than that of cooling water and water for hydro power. However, the amount of consumptive water (1.11E-03 (m^3)) is larger than the amount of content liquid (500 (mL) = 0.5E-03 (m^3)) in Figure 8. The introduction of advanced filling technology will reduce nearly 38% of consumptive water, which seems to be significant reduction of consumptive water.

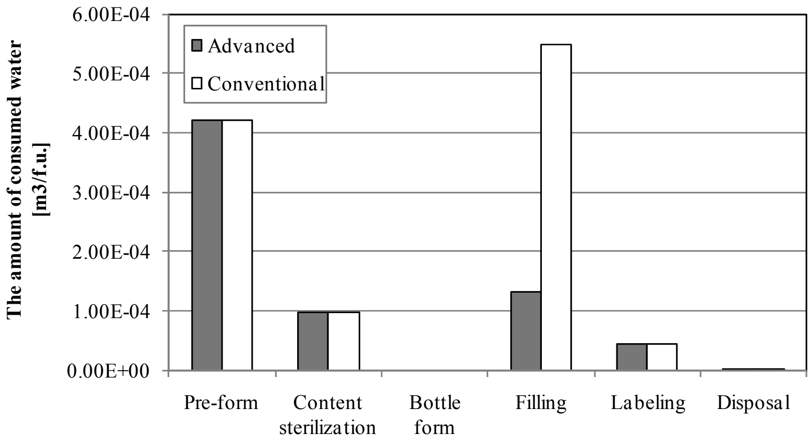

Fig. 8: The amount of consumptive water in each stage of both systems with advanced and conventional filling technologies

4.3 Significance of freshwater consumption compared to other categories

The environmental impacts of other categories are evaluated based on LCI data and LIME2 methodology in order to verify the significance of freshwater consumption compared to other categories. The rates of effects on each category to total environmental impact for the system with advanced filling technology in representative countries are shown in Figure 9. While the significance of freshwater consumption in Japan and Germany is lower compared to other categories (abiotic resource consumption, global warming, urban air pollution and ecotoxicity), the effect of water consumption accounts for the largest part of environmental impacts in China, Thailand and Algeria. The savings of freshwater consumption by applying the developed filling technology can contribute to the significant reduction of total environmental impact especially in highly sensitive countries to water scarcity.

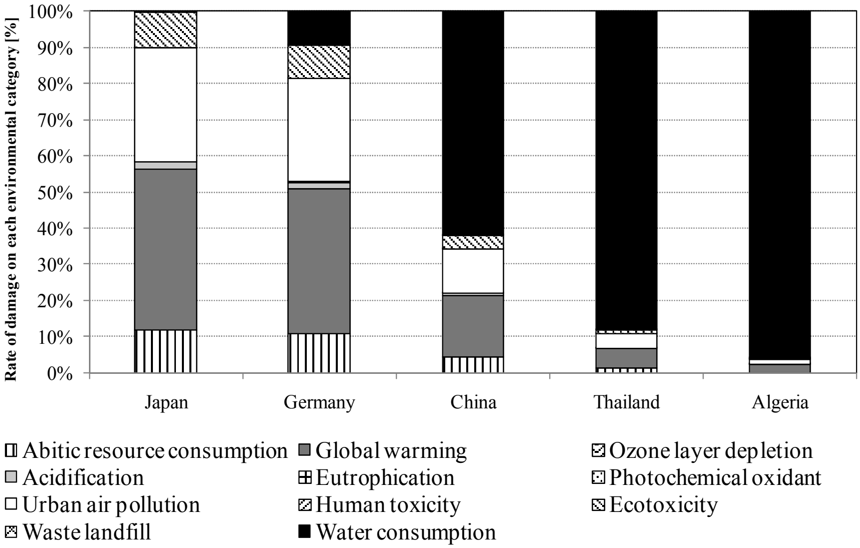

Fig. 9: The contribution rates of each category to total environmental impact related to PET bottle production system with advanced filling technology in representative countries

The contribution rates of freshwater savings to the total environmental impact in each country are shown in Figure 10. Dark collared countries have high potential to reduce environmental impact by applying advanced filling technology. The range of contribution rate of environmental impact reduction was 0~37% (median:

10%). Especially, high availability can be found in countries at North African, Asian and South American regions.

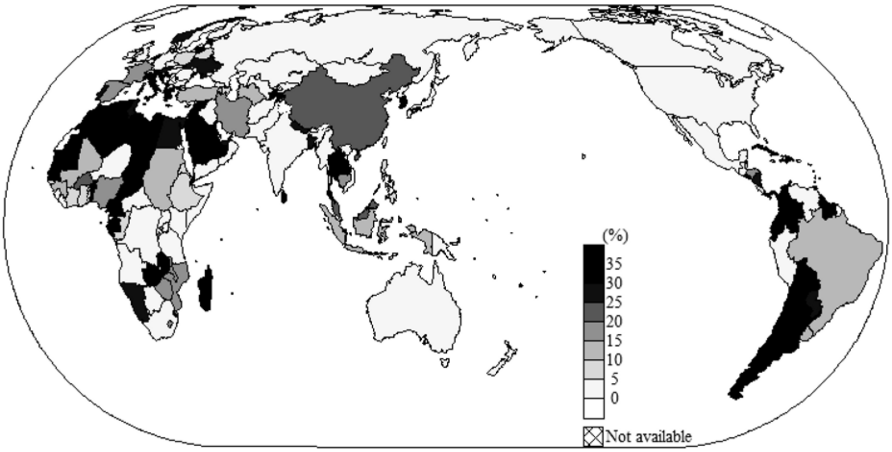

Fig. 10: Distribution map of the contribution rate of freshwater savings by introducing advanced technology to the total environmental impact in each country

5 Conclusions

Damage assessment model focusing on human health and social asset damage caused by freshwater consumption was developed on country scale. Results of damage factor calculation for each country indicated that undernourishment damage due to agricultural water scarcity dominate large part of total damage of freshwater consumption in most countries because of the ripple effects of the food supply shortage through international trade. On the other hand, infectious diseases damage due to domestic water scarcity dominated some extent of total damage in countries with low accessibility to safe water and existence of infectious source. Social asset damage due to the loss of agricultural commodity showed no negligible contribution to the total damage in some developed countries with high crop productivity.

The case study on PET bottle production revealed that advanced filling technology of PET bottle would successfully reduce the amount of consumptive water. Especially, large advantage of freshwater consumption savings by applying advanced filling technology of PET bottle can be expected many developing and emerging countries at the North African, Asian and South American regions.

Acknowledgements The authors gratefully acknowledge Asako Fujimori (Dai Nippon Printing Co., Ltd.), Ayumi Shibata (Dai Nippon Printing Co., Ltd.) for providing useful suggestions.

References

[1] Shiklomanov IA (1996) Assessment of water resources and water availability in the world. World Meteorological Organization (WMO)

[2] UNESCO (2003) World water resources at the beginning of the 21st century.

[3] Motoshita M, Itsubo N, Inaba A (2010) Development of impact factors on damage to health by infectious diseases caused by domestic water scarcity. Int J Life Cycle Assess 16(1):65-73.

[4] Pfister S, Koehler A, Hellweg S (2009) Assessing the environmental impacts of freshwater consumption in LCA. Environ Sci Technol 43(11):4098-4104.

[5] Bouley AM, Bulle C, Bayart JB, Deschenes L, Margni M (2011) Regional characterization of freshwater use in LCA: modelling direct impacts on human health and indirect through compensation. Submitted.

[6] Motoshita M, Itsubo N, Inaba A (2010) Damage assessment of water scarcity for agricultural use. Proceedings of the 9th International conference on Ecobalance, Tokyo

[7] Itsubo N, Sakagami M, Washida T, Kokubu K, Inaba A (2004) Weighting across safeguard subjects for LCIA through the application of conjoint analysis. Int J Life Cycle Assess 9(3):196-205.

[8] Dai Nippon Printing Co Ltd (2011) Report on the method of filling a PET bottle using lower water consumption. JLCA news letter, Life-Cycle Assessment Society of Japan (10-11):171-186.

PART VII:
LCM in the Energy Sector

Sustainability Assessment of Biomass Utilisation in East Asian Countries

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Abstract In order to provide decision-making methodology to evaluate sustainability of biomass utilisation in East Asian context, an expert working group has been formed since 2007 and has been conducting researches to assess the sustainability of biomass utilisation with the concept of triple bottom line focusing on environmental, economic and social pillars of sustainability. Based upon the methodology developed in 2008, the WG had conducted four pilot studies in India, Indonesia, Thailand and the Philippines in 2010 to field-test the methodology developed and investigate the sustainability of various feedstocks utilisation for biomass energy. This paper aims at introducing the sustainability assessment methodology the WG developed and addressing experiences and lessons learned through the pilot studies.

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1 Introduction

It is generally acknowledged that biomass energy can make a significant contribution to environmental improvement, energy supply diversity from fossil fuel and socio-economic development goals both in the developed and developing world owing to the following reasons. Firstly, biomass energy developments offer the opportunity for enhanced energy security and access by reducing the dependence upon fossil fuels. Secondary, biomass energy has the potential to contribute to environmental matters including the GHG emissions reduction. Thirdly, biomass energy development can create employment that will positively affect agricultural and rural incomes, poverty growth and economic growth.

On the other hand, there is a rising concern for life cycle GHG reduction effect of biomass energy, food versus fuel problem and environment disruption caused by the expansion of biomass resources production and use as energy. In view of these, there is also widespread recognition that biomass energy must be produced and used in a sustainable way, considering all the positive and negative effect from environmental, economic and social pillars of sustainability.

There are several initiatives working to develop sustainability criteria and indicators for biomass energy and their feedstocks for use in certification schemes. Several national governments had defined their own sustainability criteria especially for liquid biofuels (e.g. the UK's Renewable Transport Fuels Obligation [1], Germany's Biomass Sustainability Ordinance – BioNachV [2] or the USA's Renewable Fuel Standard [3]). The European Union is in the process of agreeing a common set of sustainability criteria through Renewable Energy Directive [4] to achieve significant greenhouse gas savings and to prevent negative effects upon biodiversity by the use of biomass energy. There are also international frameworks to discuss the sustainability of biomass energy. The Roundtable on Sustainable Biofuels (RSB), a multi-stakeholder initiative hosted by the Energy Centre of École Polytechnique Fédérale de Lausanne (EPFL), has developed a global sustainability standard and certification system for biofuel production since 2007 [5]. The Global Bioenergy Partnership (GBEP) [6], a forum where national governments, international organisations and other partners seek to facilitate effective policy frameworks and suggest rules and tools to promote sustainable biomass energy development through voluntary cooperation, has been working to develop a set of relevant, practical, science-based voluntary sustainability criteria and indicators under the Task Force on Sustainability since 2008. The criteria and indicators are intended to guide any analysis undertaken of biomass energy at the domestic level with a view to informing decision making and facilitating the sustainable development of biomass energy in a manner consistent with multilateral trade obligations [7]. International Organization for Standardization (ISO) is also under development of "Sustainability criteria for bioenergy" to bring

together international expertise and state-of-the-art best practice to discuss the social, economic and environmental use of bioenergy, and identify criteria that could prevent it from being environmentally destructive or socially aggressive [8].
Although there is high biomass energy potential in East Asia, most of the counties in this region are heavily dependent upon fossil fuel imports to meet their energy needs. Governments in this region are looking for various energy alternatives and in this regard biomass energy has emerged on the forefront, which may assure social benefits due to employment generation through its development as well as GHG reduction and energy security. Taking into these backgrounds into consideration, an expert WG (working group) has been formed under the support of ERIA (Economic Research Institute of ASEAN and East Asia) since 2007 and has been conducting researches to assess the sustainability of biomass utilisation.
In our 2007 discussions upon 'Sustainable Biomass Utilisation Vision in East Asia' [9], we suggested policy recommendations and framed "Asia Biomass Energy Principles", which were endorsed by the Energy Ministers Meeting of East Asian Summit at Bangkok in August 2008 [10]. In response to the request from Energy Ministers of the region to develop a methodology to assess the sustainability of biomass utilisation for energy production by considering specific regional circumstances, our WG started investigations to 'Guidelines for Sustainability Assessment of Biomass Utilisation in East Asia' [11] in 2008. In 2009, our WG field-tested the guidelines developed in four pilot studies conducted at India, Indonesia, Thailand and the Philippines and investigated the sustainability of various feedstocks utilisation for biomass energy [12].
This paper aims at introducing the sustainability assessment methodology of biomass utilisation our WG developed and addressing the experiences and lessons learned through the pilot studies.

2 WG methodology to assess sustainability of biomass utilisation

In this chapter, the WG methodology to assess sustainability of biomass utilisation is addressed. Please refer to WG reports [11-12] for the details.

2.1 WG aim and concept

The WG adopted the definition of "sustainable development" from "Our Common Future" of the UN world Commission on Environment and Development report published in 1987 [13], i.e., "development that meets the needs of the present without compromising the ability of future generations to meet their own needs".

The triple bottom line approach, focusing upon "people, planet, profit" is based upon social, environmental and economic criteria. To ascertain the sustainability of biomass energy development, these aspects are necessary and must be considered to overcome and minimise the problems that may occur with the expansion of biomass energy utilisation. In view of these, the WG has developed a methodology to assess sustainability of biomass utilisation in East Asian context from environmental, economic and social pillars.

2.2 Environmental indicator

Life cycle assessment (LCA) is increasingly being promoted as a technique for analysing and assessing the environmental performance of a product system and is suited for environmental management and long-term sustainability development. Although LCA can be used to quantitatively assess the extent of impact of a product system towards environmental issues of concern such as acidification, eutrophication, photooxidation, toxicity and biodiversity loss, these impact categories are currently not in the limelight as compared to climate change, a phenomenon that is associated with the increasing frequency of extreme weather conditions and disasters. Effects of climate change have been attributed directly to the increased atmospheric concentration of GHG released by anthropogenic activities. Taking other standards or frameworks for biomass energy sustainability into consideration, the WG had adopted life cycle GHG emissions that can be quantified through life cycle inventory analysis (LCI) using the collected foreground and background data as the indicator to evaluate sustainability of biomass energy utilisation.

The system boundary for LCI is consisted of three stages: feedstock cultivation, feedstock collection and biomass energy production. There is a wide recognition that the effect of land use change (LUC) towards the LCGHG emissions is significant. Although their effect can be calculated using equations and default values proposed by the International Panel on Climate Change [14], the WG recognises that there still lies uncertainties for the calculations and standardised methodologies for GHG emissions from LUC are not yet to be established. Hence the emissions from LUC are excluded from the system boundary of the WG's methodology.

The LCI for biomass energy should cover CO_2 and non-CO_2 GHGs, namely CH_4 and N_2O that are released directly and indirectly from agricultural activities. The GHG inventory is calculated as CO_2-equivalent (CO_2e) and the summation of contribution from non-CO_2 GHGs are based upon the IPCC Fourth Assessment Report (AR4) Global Warming Potential (GWP) value for a 100 year horizon.

2.3 Economic indicator

Economic sustainability of biomass utilisation relates to the exploitation of biomass resources in a manner by which the benefits derived by the present generation are ascertained without depriving such opportunity to the future generation. In the assessment of sustainability, it is equally important to determine the actual level and degree of the economic benefits brought about by the biomass industry. Specific economic indices would have taken into consideration to measure the scope of the benefits. Existing methodologies in quantifying such indicators would have to be adopted and evaluated as well. Economic indicators ultimately provide for an accurate measurement of the economic performance of a particular industry such as biomass. Based upon the various literature reviewed, the most common economic contributions of biomass utilisation are value addition, job creation, tax revenue generation and foreign trade impacts. The same indicators were taken into consideration to evaluate economic sustainability of biomass energy utilisation in WG's methodology: 1) total net profit accumulated from product conversion or processing; 2) personnel remuneration created by employment at the biomass industry; 3) tax revenues generated from the different entities within the industries; 4) foreign trade impacts in terms of foreign exchange earnings and savings; and 5) total value added, which is the sum of all the previous indicators. Each indicator can be calculated by the following equations:

$$\underline{\text{Total net profit (TNP)}} = \text{Total returns} - \text{Total costs} \tag{1}$$

where

$$\underline{\text{Total returns}} = \text{Sales from primary output} + \text{Sales from by-products} \tag{2}$$

$$\underline{\text{Total costs}} = \text{Amount of material inputs used} + \text{Labour costs} + \text{Overhead costs} \tag{3}$$

$$\underline{\text{Overhead costs}} = \text{Taxes and duties} + \text{Interest} + \text{Depreciation} \tag{4}$$

$$\underline{\text{Personnel remuneration}} = \text{Total man-days (Employment)} \times \text{Average wage per man-day} \tag{5}$$

where

$$\underline{\text{Wages}} = \text{Wage rate} \times \text{Labour requirement} \tag{6}$$

$$\underline{\text{Tax revenue}} = \text{Total taxable income} \times \text{Tax rate} \tag{7}$$

where

$$\underline{\textit{Total taxable income}} \\ = \textit{Income from main product} + \textit{Income from by-product} \tag{8}$$

$$\underline{\textit{Income of main product}} \\ = \textit{Profit per unit of main product } A \times \textit{Volume of } A \tag{9}$$

$$\underline{\textit{Income of byproduct}} \\ = \textit{Profit per unit of byproduct } B \times \textit{Volume of } B \tag{10}$$

$$\underline{\textit{Total value added (TVA)}} \\ = \textit{Total net profit} + \textit{Personnel remuneration} + \textit{Tax revenue} \tag{11}$$

2.4 Social indicators

Social issues in the growing markets for biomass energy are expected to become prominent as the producers and consumers of biomass energy may belong to different countries. Major social benefits of biomass energy include greater energy security, employment opportunities and improved health from reduced air pollution. On the other hand, possible negative social impacts of biomass energy, such as food insecurity, need to be considered seriously. While there could be some relief on energy front, the food insecurity and food prices, particularly in developing economies, may aggravate the negative social impact on people.

Measurement of social development significantly differs from economic development. Also, compared to social indicators, a plenty of economic indicators are more frequently available for all countries. However in many cases, particularly in case of some developing economies, they reflect a rosy picture that is far away from the reality. To capture the holistic picture of development across countries, the United Nations Development Programme (UNDP) has used the human development index (HDI). This essentially takes into account the measures for living a long healthy life (by life expectancy), being educated (by adult education and enrolment at primary, secondary and tertiary levels) and having a decent standard of living (by purchasing power parity). The WG had adopted HDI as the indicator to evaluate social sustainability of biomass energy utilisation. The calculation of HDI can be described as equation (12) and Table 1.

$$\textit{HDI} = 1/3 \times (\textit{Life expectancy index} + \textit{Education index} + \textit{GDP index}) \tag{12}$$

Tab. 1: Calculation of HDI

Index	Measure	Minimum value	Maximum value
Life expectancy	Life expectancy at birth (LE) *LE index* = $(LE-LE_{min})/(LE_{max}-LE_{min})$	25 years	85 years
Education	Education index = ALI×2/3+GEI×1/3 *Adult literacy index (ALI)* $= (ALR-ALR_{min})/(ALR_{max}-ALR_{min})$ where ALR: Adult literacy rate [%] *Gross enrolment index (GEI)* $= (GER-GER_{min})/(GER_{max}-GER_{min})$ where GER: Gross enrolment ratio [%]	0%	100%
GDP	*GDP index* $= \{\ln(GDP)-\ln(GDP_{min})\}$ $/\{\ln(GDP_{max})-\ln(GDP_{min})\}$ where GDP: GDP per capita [USD]	100 USD	40,000 USD

3 Results of testing WG methodology

Four pilot studies have been implemented by designated organisations under the ERIA's framework to apply and field-test the assessment methodology developed by the WG. One case study was implemented in each selected East Asian country, namely, India (Andhra Pradesh), Indonesia (Lampung), the Philippines (Quezon) and Thailand (Khon Kaen), as shown in Figure 1.

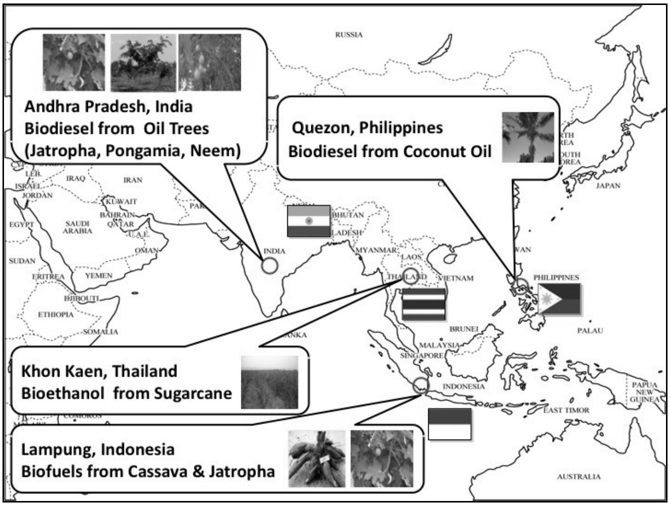

Fig. 1: Location of four pilot studies with different feedstocks for biomass energy

In each pilot study, more than hundred sets of data were obtained through interviews, calculations based upon primary data collected from pilot study sites, and secondary data from elsewhere to calculate the environmental, economic and social pillars of sustainability of biomass energy utilisation according to the WG methodology. The brief summaries of each pilot study are addressed in this section. Please refer to the WG report [12] for the details. For the economic pillar of sustainability, the original data in monetary unit were collected in each currency. Also please note that economic effect by biomass energy utilisation had been converted in terms of US dollar (USD) using the following currency conversion rates in this article: 1 USD = 48 Indian Rupee (IDR) = 9,200 Indonesian Rupiah (IDR) = 45 Philippine Peso (PHP) = 32 Thai Baht (THB).

3.1 Pilot study in Andhra Pradesh, India

In case of India, economic assessment indicates that cost incurred during the Jatropha cultivation stage is much higher than the revenue generated, which is not economically viable. In biodiesel production stage, both TVA and TNP are quite attractive, provided the raw material is available at a reasonable price. During the lifecycle of biodiesel production process, a TVA of 10,886,859 USD and a net profit of 5,840,068 USD per year were estimated. On environmental fronts, companies expect some carbon saving and an additional revenue from carbon credits. GHG saving potential estimated during the process shows a net carbon saving of 2,771,681 t CO_2e per year. On social fronts, several positive results are visible during various stages of biodiesel production, the main being employment generation for local people increasing their income, which may result in an overall improvement in their living standard.

3.2 Pilot study in Lampung, Indonesia

Biomass energy program in Indonesia was carefully designed but was not running as smoothly as planned originally. It was observed that the cassava utilisation for ethanol in Lampung Province is facing a competition for raw material from tapioca factories. Environmental assessment shows that during bioethanol production GHG emissions depend upon whether the biogas from wastewater treatment is flared or not. Economic assessment indicates that processing cassava for bioethanol increased the value added of cassava by about 0.103-0.120 USD per

L of bioethanol or about 0.0159-0.0186 USD per kg of cassava. On social assessment, the HDI values for cassava farmers in the study region were estimated lower than the HDI values for North Lampung, in general. In case of Jatropha biodiesel, although farmers in the target village receive a very low benefit from cultivation stage, utilisation of Jatropha waste increased their earnings significantly. Environmental assessment indicates that GHG emissions from Jatropha plantation and crude Jatropha oil processing were 59% and 82%, respectively. HDI estimates for Jatropha farmers in North Lampung indicate that life quality, education, and income for the people in the village were quite low.

3.3 Pilot study in Quezon, the Philippines

Economic analysis of the Philippines study shows that considering the production costs and revenues for each product, the net profit per unit of product is highest for copra production (at 0.150 USD per kg) and lowest for coconut methyl ester (CME, biodiesel from coconuts) production (at 0.122 PHP per L). The cumulative total profit for all product forms is about 844 USD per ha and the TVA from the biodiesel industry in the province of Quezon would be 305 million USD. The use of coconuts methyl ester to replace petro diesel will result in net savings or GHG emission reduction of 2,823.97kg CO_2e per ha per year. In terms of social indices, the computed HDI is 0.784 while the change in HDI is 0.004 indicating a higher level of social development. In terms of living standard, the majority (66%) of coconut farmers perceived that there has been an improvement in their living conditions due to coconut farming. In general, the results show that majority of the employees benefited from their respective employment in the biodiesel production chain.

3.4 Pilot study in Khon Kaen, Thailand

In Thailand study, environmental assessment for the lifecycle of ethanol production indicates that the overall GHG emissions associated with the ethanol production and consumption stages are slightly lower but not significantly different from that of gasoline. Increasing the utilisation of the materials produced during various unit processes in the biorefinery complex results in reducing the GHG emissions. Economic assessment of the overall process of bioethanol production indicates that the TVA for the whole biorefinery complex amounts to 116,108,080 USD and it is economically viable. For social assessment, the HDI of the sugarcane plantation, biorefinery complex, and Khon Kaen were observed as 0.736, 0.797 and 0.763, respectively. Thus, although sugarcane farmers have a lower social development than an average person in Khon Kaen or employee at the biorefinery complex, they still benefit from a steady income as a result of the contract farming, which links them to the sugar mill and guarantees an annual income. Employees at the biorefinery have a higher social development (shown by a positive change of 0.034 in HDI) as compared to the Khon Kaen.

4 Summary of the results and lessons learned from the pilot studies

It has been found from the field-testing that the methodology could successfully assess environmental, economic and social aspects of biomass utilisation projects. Highlights of the results and salient features of the pilot studies are summarised as follows:

- Indicators that have been adopted in WG methodology can quantify environmental, economic and social sustainability, respectively, of biomass energy utilisation.
- Environment indicator chosen in our methodology covers only GHG savings that is very relevant to current concerns on biomass energy. Evaluation of GHG emissions using LCA appropriately measures potential global warming intensity but other emissions and impacts can also be considered. Other than global warming, impact categories such as land use change, acid rain, eutrophication, ecotoxicity, human toxicity and resource depletion affect the locality where the emissions or depletion occur. Hence, ranking these impact categories according to local needs as a full LCA study up to the life cycle impact assessment stage may be appropriate, although collecting the information and data will be an uphill task for the developing countries.
- Economic indicators, namely, total net profit and total value added are internationally accepted. It should be emphasised that there should be a business component throughout the value chain and net profit is positive.
- Social indicators such as literacy rate, education enrolment, life expectancy, gender empowerment, etc., are relevant to the state of development of East Asian countries. Although HDI is widely applied to evaluate social impact at state, regional or national level, there is a need to develop an index or some indices that can better represent social impact at the community level. Some of the social indicators, that are

reported in the social LCA, such as child labour, minimum wage rates, forced hours, labour unions, etc., are excellent for developed countries but would not be applicable to developing economies that have to grapple with issues of poverty, employment and an expanding population that has to be provided with basic amenities through enhancing rural economy.

- Utilisation of all byproducts in the production of biomass energy is very much recommended to increase the sustainability of soil, reduce environmental impact, and optimise social and economic benefits.
- Sustainability can be viewed at different levels using appropriate indicators at community, regional and national levels.
- 'Guidelines for Sustainability Assessment of Biomass Utilisation' may be applied to each country in the East Asian region with minor locale-specific modifications. Training is recommended in order to apply the guidelines in East Asian countries properly.
- Dissemination of 'Guidelines for Sustainability Assessment of Biomass Utilisation' and experiences and lessons learned from the pilot studies may be helpful to other East Asian Countries and frameworks of sustainability of biomass energy such as the GBEP and the ISO.
- It must be noted the assessment methodology developed is tailored only for the biomass renewable resource and may not be applicable for comparison with other renewable energy resources such as solar energy, wind energy or wave energy. Although sustainability encompasses the three pillars of economic, environment and social, the specific indicators and mode of calculations including the boundaries and scope of comparison will differ. Such differences have not been considered by the WG whose focus is primarily on looking at options and issues pertaining to biomass utilisation.

Based upon the lessons learned from the pilot studies, the WG is now under discussion to upgrade the methodology. Major discussion points can be stated as follows:

- To simplify the methodology. For the analysts who are the users of the methodology, an easier way is required to collect data necessary for calculating the indicators for sustainability. Also for the decision makers who are the users of the results obtained through the methodology, social and regional characteristics of East Asian countries should be reflected in the results and the methodology should be more practical for decision makers to use.
- To assess sustainability of biomass utilisation at macro (national / state / province) and micro (community / project) level. The pilot studies were

carried out at specific sites and obtained results represent a micro level of events and characteristics of each study site. In the same manner, studies at macro level might be feasible by use of the methodology but would require pooling more comprehensive data such as those from various associations i.e. farmers, manufacturers, traders, etc. The WG is currently discussing indicators that are able to capture both levels of effects from biomass utilisation for energy.

Acknowledgements This article contains the outcome of the research activity of ERIA Working Group on 'Sustainability Assessment of Biomass Utilisation in East Asia' in 2009-2010, sponsored by the Economic Research Institute for ASEAN and East Asia (ERIA).

References

[1] <http://www.renewablefuelsagency.gov.uk/> (accessed 29.03.2011)
[2] <http://ec.europa.eu/enterprise/tris/pisa/cfcontent.cfm?vFile=120070679EN.DOC> (accessed 30.03.2011)
[3] <http://www.epa.gov/otaq/fuels/renewablefuels/index.htm> (accessed 29.03.2011)
[4] <http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2009:140:0016:0062:EN:PDF> (accessed 11.04.2011)
[5] <http://rsb.epfl.ch/> (accessed 11.04.2011)
[6] <http://www.globalbioenergy.org/> (accessed 29.03.2011)
[7] <http://www.globalbioenergy.org/programmeofwork/sustainability/en/> (accessed 29.03.2011)
[8] <http://www.iso.org/iso/pressrelease.htm?refid=Ref1282> (accessed 11.04.2011)
[9] <http://www.eria.org/research/no6-3.html> (accessed 11.04.2011)
[10] <http://www.aseansec.org/21853.htm> (accessed 11.04.2011)
[11] <http://www.eria.org/research/y2008-no8-2.html> (accessed 11.04.2011)
[12] <http://www.eria.org/research/y2009-no12.html> (accessed 11.04.2011)
[13] World Commission on Environment and Development (WCED) (1987) Our Common Future, Oxford University Press
[14] International Panel on Climate Change (IPCC) (1997) The Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories: Reference Manual

Life Cycle Inventory of Physic Nut Biodiesel: Comparison Between the Manual and Mechanised Agricultural Production Systems Practiced in Brazil

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Abstract The physic nut (Jatropha curcas L.) is an oleaginous species recently introduced into Brazil for energy purposes. The technological framework for the development of the physic nut biodiesel productive chain in Brazil is still being set up. Two production systems are in practice at the agricultural level, the small scale manual system and the medium scale mechanised system. The objective of the present research was to assess the environmental performance of these two production systems by elaborating life-cycle inventories (LCIs) using a cradle-to-gate approach. The main environmental aspects of these LCIs are the synthetic fertilisers, pesticides, land-use changes and its emissions and the occupation of the land. Making use of the residues from the agroindustrial physic nut chain and the use of biological pest control methods could improve the environmental performance of these systems.

1 Introduction

Physic nut is an oleaginous species that originated in Central America and started being introduced into Brazil for motives of energy production in 2005. Interest in this species is due to its elevated oil productivity (1.5 t/ha) and its ability to adapt to marginal, degraded areas. Currently physic nut plantations account for about 60 thousand ha in Brazil in the Central-West, North and Southeast regions, but there are estimates that this could reach 750 thousand ha by 2020.

The physic nut biodiesel productive chain is currently being established in Brazil, and the technologies of grain, oil and biodiesel production are being adjusted to the conditions found in Brazil. At the agricultural level, the production systems

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currently practiced correspond to a small-scale system employing minimal cultivation and manual labour; and a second, medium-scale system, making use of conventional soil preparation and mechanisation techniques.
Since it is an exotic species recently introduced into the country, a potential alternative source of energy to other fuels of vegetable origin whose productive chains have already reached high levels of development, such as sugarcane ethanol and soybean biodiesel, the potential environmental impact of producing physic nut biodiesel in Brazil deserves attention.
The objective of the present study was to evaluate the agricultural performance of the agricultural phase of physic oil biodiesel production using both manual and mechanised systems, by way of the elaboration of their life cycle inventories.

2 Methods

2.1 Definition of the objective and scope

The methodological structure of this study was based on the ISO 14044 norm. The objective was to evaluate the environmental performance of two physic nut grain production systems practiced in Brazil: the manual and mechanised systems. The study was justified by the recent implantation of the physic nut crop in Brazil for energy purposes, with considerable perspective for expansion. The evaluation aims at offering subsidies to orientate adjustments to the physic nut grain production systems, with a view to improving their environmental performance. The target public were researchers, extension workers and other components of the physic nut biodiesel production chain.
The manual and mechanised physic nut production systems were defined as the product systems. The manual system is practiced on a small scale (up to 10ha) employing minimal cultivation techniques and manual labour; the mechanised system is practiced on a medium scale (up to 100ha) and employs conventional soil preparation techniques and mechanised labour.
The function of both systems is to produce physic nut grains destined for the synthesis of biodiesel. The functional unit of the systems is the production of physic nut grains in an area of 1ha for 20 years. The reference flow was defined as the production of 79,500kg dry physic nut grains.
The elementary processes included at the boundary of the product systems are the production and distribution of electrical energy, diesel oil, agricultural inputs [except the manufacturing of seeds, aluminium phosphate, copper sulphate, sulphur, fipronil and nonylphenol ethoxylate and the manufacturing and transport

of raffia sacks, due to the unavailability of the data] and the production of physic nut seedlings and grains, including the post-harvest treatment. The manufacturing of agricultural machinery was not considered in this inventory. With respect to the criteria used to exclude the entrance of items, all those that attend the defined technological standard were considered in the LCIs.
With respect to the type and source of the data, the agricultural inputs correspond to the secondary data collected in adequate, up to date bibliographical sources [mainly 1] and information provided by specialists [2]; data referring to natural resources, to the manufacturing of agricultural inputs and to the production and distribution of electrical energy came from the data base Ecoinvent 2.2; the data referring to the manufacturing of diesel oil were obtained from Florin et al. (2008) [3]. The data on emissions were estimated based on models found in the scientific literature [4,5,6,7,8,9,10].
With respect to the data quality criteria, the temporal coverage includes the period from 2006 to 2010 and the geographical coverage the current Brazilian producing regions. With respect to technological coverage, the manual production system employs minimal cultivation and manual labour at all steps of the crop, harvest and post-harvest handling, whereas the mechanised system employs conventional soil preparation and mechanisation for the operations of ploughing, liming, harrowing, furrowing, chemical fertilising, leaf fertilisation combined with phytosanitary treatment, hoeing and threshing. The technical recommendations for the Brazilian savannah were adopted as the reference for both systems and for small and medium sized production scales [1,2]. All the flows involved in the physic nut grain production systems were measured or estimated in this study, and the group of data were considered consistent.
The main presuppositions assumed in this study were: a) the density of the physic nut crop was 1250 plants/ha; b) its productive longevity was 20 years; c) the productivity was 4,500 kg dry grains/ha/year on reaching its maximum productive potential; d) the crop was not irrigated; e) the husks removed during dehusking on the farm constituted the solid residue; f) the distance between the storehouses and the farms was 70 km and 100 km for the manual to the mechanised systems, respectively (considering that the manual system was practiced in small production units, mostly in the Southeast region, more densely populated, whereas the mechanised system was practiced in larger units in the Center-West region and in the State of Tocantins, less populated areas); g) the carbon dioxide (CO_2) sequestered from the atmosphere during the growth of the physic nut plants was computed as an input from nature. A recognised limitation of this study was related to the fact that the physic nut productive chain is still in its implantation phase. This represents a difficulty in obtaining reliable, representative data for the

inventory. Moreover, data obtained refer to the current technological stage of the productive chain, which could alter significantly as it develops.

2.2 Elaboration of the physic nut grain production inventory

To elaborate the physic nut grain production inventory, an annual production per ha of 200kg was considered for the first year; 800kg for the second year; 2000kg for the third year; and 4500kg for the 4th to the 20th years [2].
With respect to the natural resources, the prior use of the land for extensive pasture was assumed, transformed into a permanent crop.
The amount of CO_2 sequestered during the growth of the physic nut plants was calculated by adding up [the mass of the aerial part of the plant (excluding the leaves and fruits, 3.9 kg/plant * 1250 plants/ha) multiplied by the percent C (51.2%)] and the [mass of the plant roots (1.6 kg/plant * 1250 plants/ha] multiplied by the percent C (52%)], multiplied by the conversion factor for C into CO_2 (44/12) [11].
The consumption of agricultural inputs considered the recommendation of Dias et al. (2007) [1], adjusted by Laviola (2009) [2]. The adjustments corresponded to transforming the values originally calculated for a density of 1111 plants/ha to a density of 1250 plants/ha.
The exclusive use of limestone as an agricultural corrective throughout the entire production was considered in this study, using the amount indicated for the first years. More limestone was used in the mechanised system because it was applied as a corrective to the entire area during soil preparation, whereas in the manual system it was only applied to the holes.
The organic fertiliser (poultry manure) was only used in the manual production system, with a mean density of 0.3 g/cm^3 and N content of 3%. The formulated NPK fertiliser (urea as N; single superphosphate as P_2O_5; and potassium chloride as K_2O), corresponding to: 0 to 1 year, 20-00-15; after 1 year, 20-10-15. It was considered that urea contains 46.67% of N; potassium chloride (KCl), 63.65% of K_2O; and single superphosphate (SSP), 18.4% of P_2O_5. Only the mechanised system used leaf fertiliser. In addition to KCl, this contained boric acid, zinc monosulphate (with 20% zinc), copper sulphate (with 26.36% Cu) and sulphur. The copper sulphate and sulpher also act as pesticides (the former as a fungicide and the second as an acaricide and fungicide). The amounts of leaf fertiliser indicated in the chapter on "Custos e Rentabilidade" [1] were adopted, adjusted for a density of 1250 plants/ha, considering 5 chemical elements and equal amounts of each element.

As yet no pesticides have been approved in Brazil for use with the physic nut crop and thus the study was carried out with the hypothesis of using: glyphosate as the herbicide; fipronil as the formicide; equal amounts of thiametoxam and lambda-cyhalothrin (87% in the commercial product; piretroid chemical group) as the insecticide; abamectin (1.8%; biopesticide) as an insecticide/acaricide; methyl thiophanat (70%; benzimidazole) as the fungicide [2]. Generic inventories were used for the "production of insecticides" and "production of fungicides".

Only the mechanised production system consumed diesel oil in its agricultural operations. To calculate the diesel oil consumption, three applications of formulated fertiliser/year were considered, the first being combined with liming; two hoeings/year (as from the 2nd year) and the transport of the harvest (as from the 2nd year) [1,2]. The hours spent in the agricultural operations were calculated according to Dias et al. (2007) and Laviola (2009) [1,2]. The diesel oil consumption per agricultural operation was calculated according to Nemecek & Kägi (2007) [9].

A load factor of 50% was considered in the transport steps. In the calculation of the transport of the diesel from the refinery to the gas stations, it was assumed that: a) the diesel oil came from the refinery closest to the production area (at a distance of 246km); b) the diesel oil production LCI constructed for the refinery REPLAN was representative of all the Brazilian refineries [3]; c) the transport of the diesel oil from refinery to gas stations was done directly by road in tankers with a mean capacity of $45m^3$. The diesel oil was transported from the gas stations to the farms, distant 100 km, by road in trucks.

To estimate the change in the stock of C in the soil due to the land-use change (ΔC_{LUC}), it was considered that: a) the area transformed into physic nut crop was formerly pasture; b) grassland has a stock of C in the biomass of 5 t/ha [8]; c) the biomass of a physic nut plant contains 2.83kg of C [11]. The value for ΔC_{LUC} is calculated by difference between the stock of C in the original use of the soil and the stock of C in the current use of the soil. The CO_2 emissions due to the land-use change (CO_{2LUC}) were calculated by multiplying ΔC_{LUC} by the conversion factor of C to CO_2, assuming a discount period of 20 years (IPCC standard).

The CO_2 emissions caused by the use of dolomite limestone and urea were calculated according to IPCC (2006) [8]. The methane emissions (CH_4) resulting from the reduction in the soil retention capacity caused by the use of the N were calculated considering that for each 150 kg N/ha applied in the form of ammonia, the methane reducing capacity of the soil decreases by 1 kg/ha [5].

Estimates of the N_2O emissions caused by the grain production considered: a) the input of urea as a synthetic nitrogenated fertiliser; b) the input of manure as an organic fertiliser exclusively for the manual production system; c) the emissions of N_2O caused by mineralisation of the N in mineral soils, associated with a loss of C from the soil, as a result of the changing use of the soil or its management (F_{SOM}). The calculations were carried out according to the IPCC (2006) [8]. In the calculation of F_{SOM}, a standard value of 15 was adopted for the C:N ratio, adequate in situations involving a land-use change from grassland to cropland.

The atmospheric emissions of nitrogen oxides (NO_x) generated by the production of physic nut grains corresponded to 10% of the emissions of N_2O [5]. Specifically in the case of mechanised production, the emissions of N_2O and NO_x already considered should be summed up with the emissions of these gases generated by the combustion of the diesel oil. To calculate the emissions of ammonia (NH_3) derived from the use of urea, an emission factor of 0.1 was used, to be multiplied by the amount of N in the nitrogenated fertiliser [8].

To calculate the PO_4 emissions for the water and soil, resulting from the use of SSP, it was assumed that: a) part of the P in the fertiliser system was exported to the crop; b) the fruits harvested leave the product system; c) the biomass corresponding to the leaves returns annually to the soil; d) the biomass corresponding to the stalks, branches and roots leaves the system at the end of the productive cycle; e) the mass of dried grains corresponds to 62.51% of the total mass of dried physic nut fruits; f) the dry fruits contain 0.86% P; g) a mature physic nut plant produces 3.9kg (dry weight basis) of aerial biomass (except the fruits and leaves) and 1.6kg (dry weight basis) of subterranean biomass with a P content of 0.1% (value found in the literature for other fibrous parts of plants) [11]. The excess of P in the system is calculated from the difference between the P carried in the fertiliser and that exported to the crop. Of this excess P, 0.29% is leached into the subterranean waters and the rest accumulates in the soil [10].

With respect to the calculations of heavy metal emissions coming from the fertilisers, including the copper sulphate and zinc monosulphate used as leaf fertiliser in the mechanised system, the fractions exported to the crop were not considered, since the type of crop and the soil and climate conditions affected this exportation and no specific values for physic nut and for Brazil are available, and also, physic nut is a perennial crop and hence the majority of the biomass is maintained in the agricultural system after the annual harvest of the fruits. Thus the total amount of heavy metals entering the agricultural system will be reverted as emissions to the environment. It was considered that part of the heavy metals emitted to the soil is lost as run-off to surface waters (0.01%). The emissions to the soil were calculated by difference between the amount of heavy metal entering the system and the amount emitted to the surface waters [5]. The heavy metal contents in the nitrogenated fertiliser corresponded to the mean of the value reported by Canals (2003) [5] and Schmidt 2007 [10].

For the mechanised production system, there were emissions generated by combustion of the diesel oil consumed in the agricultural operations. The

emissions of hydrocarbons (such as NMVOC), benzene, benzo(a)pyrene, aromatic polycyclic hydrocarbons, carbon monoxide (CO), CO_2, CH_4, NO_x, N_2O, NH_3, sulphur dioxide (SO_2), Cd, Cr, Cu, Ni, Pb, Se and Zn, as also the emissions of particulate material with a diameter <2.5μm, were calculated according to Nemecek & Kägi (2007) [9].
With respect to the emission of pesticides to the environment, the amount of the active principle of the pesticide applied to the crop (per ha/20 years) was used as a base for the calculations. The metabolites generated by degradation of the pesticides were not considered. The fate analysis suggested by Hauschild (2000) [4] was adopted to estimate the emissions of pesticides into the environment.
The total amount of active principle of the pesticide applied to the crop was divided into fractions derived from the production area by the wind and reaching the surrounding environment (f_{drift}) or deposited on the plants (f_{plant}) or on the soil surface ($f_{soil\ surface}$). The fractions that reach the plants or the soil can volatilise ($f_{vol\ plant}$ and $f_{vol\ soil}$). The fraction in the soil can be run-off into surface waters ($f_{run\text{-}off}$) or by leaching into subterranean waters (f_{leach}) or to surface waters, if the soil is drained. Part of the contaminants in the soil are degraded by microbial activity (f_{degrad}) and part remain in the soil to the end of the productive cycle.
Only the pesticide applied to the border of the crop (frontier of the production area up to 30 m in the direction of the centre of the production area) suffered from the effect of wind drift, f_{drift} [5]. Assuming that the production areas had a square format, the border areas corresponded to 34347.6m^2 and 116,400m^2, or 34.35% and 11.64% of the total production area for the manual and mechanised systems, respectively. For shrub crops, in which the application of pesticide is done when the plants are fully foliated (as in the case of physic nut) [2], the pesticide drift emission factor is 0.24% [4,5]. Thus to calculate f_{drift}, the amount of active principle of each pesticide applied was multiplied by the percent referring to the border of the crop and by the drift emission factor.
The amount of pesticide remaining in the system, subtracting f_{drift}, was divided between f_{plant} and $f_{soil\ surface}$. Empirical estimates have been used to estimate f_{plant}, obtained by varying the leaf density and the concentration of the pesticide solution, obtained in studies carried out in New Zealand in apple orchards (MANKTELOW, 1998, cited by CANALS, 2003) [5], and were adopted in the present study. For the physic nut crop, the pesticide is applied when the plants are fully foliated and the concentration of the solution is relatively high, 400L/ha [2], and so a percent retention by the plant of 85% was considered [5]. Thus to calculate f_{plant}, the value for f_{drift} was subtracted from the quantity of active principle applied to the crop. The rest was multiplied by 0.85. The value for $f_{soil\ surface}$ was calculated by subtracting the values for f_{drift} and f_{plant} from the total amount of active principle applied to the crop.

The values for $f_{vol\ plant}$ and $f_{vol\ soil}$ were calculated by multiplying f_{plant} or $f_{soil\ surface}$ by their respective emission factors, calculated according to Hauschild (2000) [4]. The values used for the vapor pressure and half-life$_{soil}$ of the pesticides in this calculation can be found in the specialised literature [12,13,14]. The value for the half-life$_{plant}$ of glyphosate is 35 days [5] and for abamectin 0.21 days. The values for the half-life$_{plant}$ of the other substances are not available, and thus the mean values of 34.4 days for pesticides, or 4.6 days for fungicides [5], were used. The residence times of the pesticides (in the plant and soil) were calculated by multiplying the values for the half life (in the plant and soil) by 1.443. The fractions $f_{vol\ plant}$ and $f_{vol\ soil}$ were added together to give f_{vol}. The $f_{run\text{-}off}$ was calculated by multiplying $f_{soil\ urface}$ by 0.0001 [4].
In order to estimate the value for f_{leach}, the attenuation factor (AF) was first calculated, according to Paraíba & Miranda (2003) [7]. The data referring to the soil correspond to a type representative of the Brazilian savannah, Typic Orthic Neosol Quartzarenic Brazilian, characterised by being prone to leaching (hence the worst case of a real situation), and were obtained by Paraíba et al. (2003) [6]. The soil organic carbon partition coefficient (K_{oc}), the molecular weight and the solubility in water of the pesticides can be found in the specialised literature cited above. The value for f_{leach} was calculated by: $f_{leach} = AF * (f_{soil\ suface} - f_{vol\ soil} - f_{run\text{-}off})/L * \delta$, where L is the depth soil and δ is the soil porosity.
The soil leaves the product system after the harvest, when the remaining pesticide fraction starts being considered as an emission. Nevertheless this fraction can undergo degradation in the soil. The value for f_{degrad} was calculated according to Canals (2003) [5], considering the degradation rate of the pesticide in the soil to be the time between the annual pesticide applications and the 20th harvest of the production (which corresponds to 180 days for the herbicide; 90 for the insecticide, acaricide, fungicide and sulphur; and 240 for the formicide) [2]. The degraded fractions were calculated annually and then added up.
The emissions for the environmental compartments are calculated as: emissions to the air = $f_{drift} + f_{vol}$; emissions to surface waters = $f_{run\text{-}off}$; emissions to subterranean waters = f_{leach}; emissions to soil = $f_{soil\ surface} - f_{vol\ soil} - f_{run\text{-}off} – f_{leach} – f_{degrad}$.

3 Results and discussion

Tables 1 and 2 show the inventories for the production of physic nut grains by the manual and mechanised systems. The main environmental aspects involved in the production of the grains are the synthetic fertilisers, the pesticides, the land-use change and its emissions and the land occupation. Specifically for the mechanised production system, the aspects related to the consumption of diesel oil and its emissions must also be included.

Tab. 1: Main environmental aspects of the life-cycle inventory for the production of physic nut grains - inputs

Inputs (1 ha/20 years)	Manual system	Mechanised system
Products		
Jatropha curcas grains, at farm (kg)	7.95E+04	7.95E+04
Resources		
Carbon dioxide, in air (kg)	1.30E+04	1.30E+04
Occupation, permanent crop (ha a)	1.00E+00	1.00E+00
Transformation, from pasture, extensive (ha)	1.00E+00	1.00E+00
Transformation, to permanent crop (ha)	1.00E+00	1.00E+00
Materials and fuels		
Copper sulphate (kg)	-	1.23E+01
Fungicides, at regional storehouse (kg)	4.28E+01	4.28E+01
Insecticides, at regional storehouse (kg)	6.35E+01	6.35E+01
Limestone, milled, packed, at plant (kg)	4.60E+03	6.60E+03
Potassium chloride, as K_2O, at regional storehouse (kg)	1.85E+03	1.86E+03
Poultry manure, dried, at regional storehouse (kg)	2.00E+04	-
SSP, as P_2O_5, at regional storehouse (kg)	1.18E+03	1.18E+03
Urea, as N, at regional storehouse (kg)	2.46E+03	2.46E+03
Zinc monosulphate, $ZnSO_4.H_2O$, at plant (kg)	-	1.23E+01
Diesel, from crude oil, consumption mix, at refinery, 500ppm sulphur 500 (kg)	-	2.25E+03
Transport		
Diesel transport from gas station to farm, by van, <3.5t	-	2.25E+02
Diesel transport from refinery to gas station, by lorry transport, >32t, Euro	-	5.53E+02

The synthetic fertilisers are responsible for the emission of heavy metals to the soil (the most important being Cd, Zn, Hg and Se, in this order) and to the water (Hg and Se, in that order), substances causing impacts related to human toxicity and to aquatic and terrestrial ecotoxicity. The emissions derived from the agricultural use of urea, as in the case of ammonia (CH_3) and nitrogen oxides (NO_x) - which contribute to the impacts of acidification and eutrophication, and indirectly of methane (CH_4) - which contributes to the impacts of global warming and photochemical oxidation - are also relevant. The phosphate fertiliser generates emissions of phosphate to the soil and to the water, a substance related to the impacts of human toxicity and eutrophication.

With respect to the pesticides, the most important emissions are those of lambda-cyhalothrin, abamectin and methyl thiophanat to the soil, substances which are also toxic.

The land-use change, formally pasture, transformed into a permanent crop, causes CO_2 emissions related to the impact of global warming. The biomass produced by the crop, on the other hand, sequesters carbon from the atmosphere, which could compensate the emissions from the combustion of the biodiesel. Thus the mobilisation of the area occupied by the physic nut crop causes its own impact.

The main differences between the two production systems are due to the greater consumption of inputs by the mechanised system. Although organic fertiliser is not used in this second system, the consumption of limestone is greater and other consumables are introduced, such as leaf fertiliser and diesel. The emissions of nitrous oxide (N_2O) are smaller in the mechanised system, since organic fertiliser is not used. On the other hand, the emissions of CO_2 are greater in the mechanised system due to the greater consumption of limestone and diesel oil. The consumption of diesel oil also results in an increase in the emission of methane (CH_4). Thus the impact of the production of physic nut grains on global warming is greater for the mechanised system.

The consumption of diesel oil also results in an increase in the emission of nitrogen oxides (NO_x) and heavy metals. The use of potassium chloride, copper sulphate and zinc monosulphate as leaf fertiliser also increases the emissions of these metals. The mechanised system also causes the emission to the atmosphere of heavy metals, principally of Zn, Cd, Se, Cu and Ni (in order of importance). The emissions of pesticides to the air are slightly reduced in the mechanised system, since the emissions caused by the wind by drifting are inversely correlated to the size of the area cultivated. Comparing the two production systems, the impacts with respect to human toxicity are very close, but relatively more important in the manual system.

With respect to the emission of particulate material <2.5µm and of SO_2, resulting from the combustion of diesel oil, together with the emissions of CH_3e NO_x (common to both production systems), which make up the impact denominated as particulate material, with negative effects on human health, the weight of this factor is of importance in the general impact of the grain production systems, being greater for the mechanised system.

Despite the fact that the physic nut crop is not considered to be demanding with respect to nutrients and resistance to pests and diseases, the LCI for the production of the grains shows an elevated consumption of limestone and fertilisers, particularly of the organic and nitrogenated ones. Although the consumption of pesticides is not high in absolute terms, it is nevertheless high when compared to other perennial oleaginous crops such as palm oil.

Tab. 2: Main environmental aspects of the life-cycle inventory for the production of physic nut grains - outputs

Outputs (1 ha/20 years)	**Manual system**	**Mechanised system**
Emissions to air		
Ammonia (kg)	2.46E+02	2.46E+02
Benzene (kg)	-	1.64E-02
Benzo(a)pyrene (kg)	-	6.74E-05
Cadmium (kg)	-	2.25E-05
Carbon dioxide (kg)	2.19E+03	3.15E+03
Carbon dioxide, fossil (kg)	2.92E+02	7.30E+03
Carbon dioxide, land transformation (kg)	5.36E+03	5.36E+03
Carbon monoxide, fossil (kg)	-	1.19E+01
Copper (kg)	-	3.82E-03
Methane (kg)	1.64E+01	1.64E+01
Methane, fossil (kg)	-	2.90E-01
Nickel (kg)	-	1.57E-04
Nitrogen oxides, NOx (kg)	6.66E+00	1.01E+02
Nitrous oxide, N_2O (kg)	6.66E+01	5.40E+01
Particulates, < 2.5 µm		1.13E+01
Selenium (kg)	-	2.25E-05
Sulphur dioxide (kg)	-	2.27E+00
Zinc (kg)	-	2.25E-03
Emissions to surface water		
Mercury (kg)	9.41E-07	9.41E-07
Selenium (kg)	1.67E-06	1.67E-06
Emissions to groundwater		
Phosphate (kg)	2.34E-01	2.34E-01
Emissions to soil		
Abamectin (kg)	3.44E-04	3.44E-04
Arsenic (kg)	2.86E-02	2.87E-02
Cadmium (kg)	3.00E-01	3.00E-01
Chromium (kg)	1.69E-02	1.70E-02
Cobalt (kg)	1.03E-02	1.04E-02
Copper (kg)	4.74E-02	3.29E+00
Lambda-cyhalothrin (kg)	1.42E-02	1.42E-02
Lead (kg)	9.31E-02	9.36E-02
Phosphate (kg)	8.04E+01	8.04E+01
Mercury (kg)	9.41E-03	9.41E-03
Molibdenum (kg)	2.90E-02	2.90E-02
Nickel (kg)	4.14E-01	4.14E-01
Selenium (kg)	1.67E-02	1.67E-02
Thiophanat-methyl (kg)	2.10E-03	2.10E-03
Zinc (kg)	3.19E+00	5.65E+00

One could indicate as opportunities to improve the environmental performance of the production of physic nut in Brazil, the agricultural use of vegetable and agroindustrial residues from the productive chain itself, which could reduce the use of synthetic fertilisers, and the use of alternative technologies for the chemical control of pests and diseases, which would demand technological development.

References

[1] Dias LAS, Leme LP, Laviola BG, Pallini Filho A, Pereira OL, Carvalho M, Manfio CE, Santos AS, Sousa LCA, Oliveira TS, Dias DCFS (2007) Cultivo de pinhão-manso (*Jatropha curcas* L.) para produção de óleo combustível. UFV

[2] Laviola BG (2009) Pesquisador da Embrapa Agroenergia, comunicação pessoal

[3] Florin H, Betz M, Schuller O, Makishi C (2008) Methodology report - Pilot study diesel: contribution to the Brazilian National Life Cycle Inventory (LCI) Database. PE International GmbH/University of Stuttgart

[4] Hauschildt MZ (2000) Estimating pesticide emissions for LCA of agricultural systems. In: Weidema BP, Meeusen MJG (Ed.) Agricultural data for life cycle assessments LCANet Food, 2:64-79.

[5] Canals LM (2003) Contributions to LCA methodology for agricultural systems. Universitat Autònoma de Barcelona

[6] Paraíba LC, Cerdeira AL, Silva EF, Martins JS, Coutinho HLC (2003) Evaluation of soil temperature effect on herbicide leaching potential into groundwater in the Brazilian Cerrado. Chemosphere 53:1087–1095

[7] Paraíba LC, Miranda JI (2003) O modelo RACHEL. Rev Científica Rural, 8(2):29-42.

[8] IPPC (2006) Intergovernmental Panel on Climate Change (IPCC). Guidelines for national greenhouse gas inventories. Agriculture, forestry and other land use, Vol. 4, IPCC

[9] Nemecek T, Kägi T (2007) Life cycle inventories of agricultural production systems. Data v2.0. Ecoinvent Report 15, ART

[10] Schmidt JH (2007) Life cycle assessment of rapeseed oil and palm oil. Aalborg University

[11] Reinhardt G *(2008)* Basic data for Jatropha production and use. Updated version. Ifeu/CSMCRI/University of Hohenheim

[12] Hornsby AG (1996) Pesticide properties in the environment. Springer-Verlag

[13] Tomlin CDS (2000) The pesticide manual. British Crop Protection Council

[14] <http://sitem.herts.ac.uk/aeru/footprint/en/index.htm> (accessed on 01.02.2011)

Life Cycle Assessment of Biodiesel Production from Microalgae Oil: Effect of Algae Species and Cultivation System

Javier Dufour, Jovita Moreno and Rosalía Rodríguez

Abstract Different microalgae are widely studied as alternative sources for biodiesel production. They show higher oil productivity values (per area) than oilseed crops and are not used for food industry. For the evaluation of the energy and environmental feasibility of biodiesel coming from microalgae, life cycle assessment (LCA) methodology provides a very useful tool. In this work, we have used it to evaluate the biodiesel production from the microalga *Nannochloropsis gaditana* cultivated in three different systems: tubular and flat-plate photobioreactors and raceway ponds. Results indicate that tubular reactor has a very high energy demand leading to the lowest net energy ratio (NER). Despite the better NER results of the cultivation step when using flat-plat configuration and race-way ponds, harvesting and lipid extraction necessary for biodiesel production lead to an important reduction of NER increasing also CO_2 emissions.

1 Introduction

Biodiesel production has become a very intense research area because of the growing interest on finding new resources and alternatives for conventional transport fuels [1-3]. As known, in the last few years various kinds of biomass have been identified as possible sources for biodiesel production, e.g. bio-wastes (food wastes, municipal wastes or agricultural wastes), edible and non-edible oil seeds and various aquatic plants (microalgae) [4]. Microalgae are basically a large and diverse group of simple, typically autotrophic organisms (CO_2 is their carbon source), ranging from unicellular to multi-cellular forms [1]. These have the potential to produce considerably higher amounts of biomass and lipids per hectare than any kind of terrestrial biomass. Traditionally, microalgae are cultivated in closed systems or open ponds, however another steps are involved in

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biodiesel production as harvesting, drying, lipid extraction, and lipid esterification/ transesterification to obtain the fatty acid methyl esters (FAMEs). For each step, there are many variables with high influence on the energy demand and environmental impacts such as the kind of cultivation systems, the method selected for microalgae harvesting and drying or the necessary chemicals for the extraction and esterification processes. Likewise, the use of algae wastes produced in the extraction and the re-utilisation of nutrients and other chemicals allow increasing the net energy ratio (NER) of the process and reduce its total environmental impact. Therefore, all these aspects must be evaluated in order to know if the production of biodiesel from microalgae is an effective alternative to solve problems associated with the growing energy demand and global warming. In this context, life cycle assessment (LCA) methodology provides a useful tool for the evaluation of different cultivation systems. In this work, three different cultivation systems have been evaluated: open raceway ponds, tubular photobioreactors and flat-plate photobioreactors. Since LCA evaluation of this work is focused on Spanish conditions, we have selected *Nannochloropsis gaditana* as the most significant alga to carry the assessment out. It has been previously cultivated [5] in the south of Spain and presents high oil content.

2 Methods

The assessment was carried out with Gabi 4.3 software by using the database Ecoinvent 2.1. The considered functional unit was 1 kg of biofuel ready to be used in a diesel engine; the boundaries include extraction and production of raw materials, facility construction and dismantling, biodiesel elaboration from microalgae, distribution and use in the engine. Figure 1 shows the inputs and output of steps considered for biodiesel elaboration from *Nannochloropsis gaditana.*

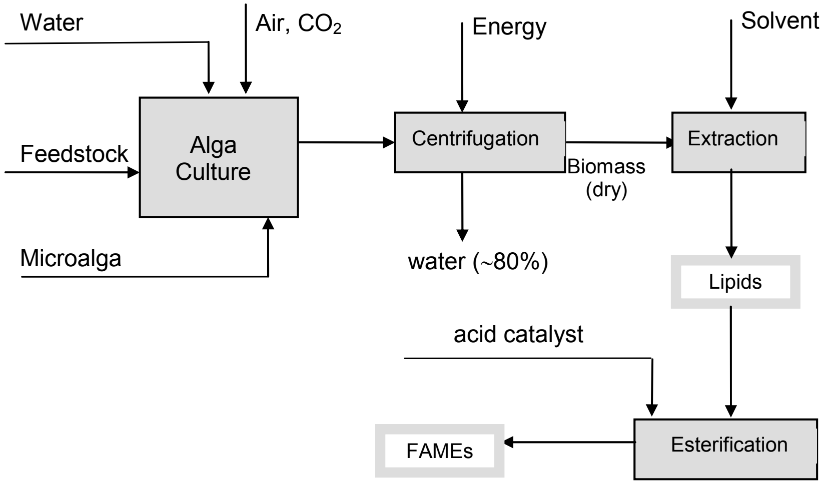

Fig. 1: Diagram of the production of biodiesel from *Nannochloropsis gaditana*

For LCA inventory, productivity, energy demand and building materials of cultivation systems were obtained from the work reported by Jorquera et al. [6] Culture medium for *Nannochloropsis gaditana* was the f/2 recipe and requirements of CO_2 were 3.5 kg CO_2/kg dry mass [7]. Microalgae harvesting was carried out by means of conventional centrifugation removing 80 wt% of water, then, a drying process under room temperature was supposed. For the oil extraction, the principal inputs were energy and hexane leading to a yield of 40 % [8, 9]. Due to the high content of free fat acids (FFAs) of the oil coming from *Nannochloropsis gaditana*, its transformation into biodiesel must be carried out by means of an acid esterification reaction. In this case, sulphuric acid was used as catalysts [10].
For evaluation of environmental impacts the following parameters were analysed: greenhouse gases emissions (GHG) expressed as kg CO_2e and calculated according to the CML-2001 method (at 100 years), energy consumption of each step and net energy ratio (NER: MJ produced by 1 kg of biodiesel/MJ used).

3 Results and discussion

In order to compare the behaviour of the different cultivation systems, Figure 2 shows GHG emissions of each step involved in the biodiesel production. As can be seen, the CO_2 emissions of the tubular photobioreactor are higher than those corresponding to flat-plate photobioreactor and raceway ponds. In fact, these two last systems present negative values of GHG emissions in the cultivation step (-2.1 kg CO_2e for flat-plate and -3.4 kg CO_2e for raceway ponds) indicating that CO_2 fixation of microalgae compensates the emissions.

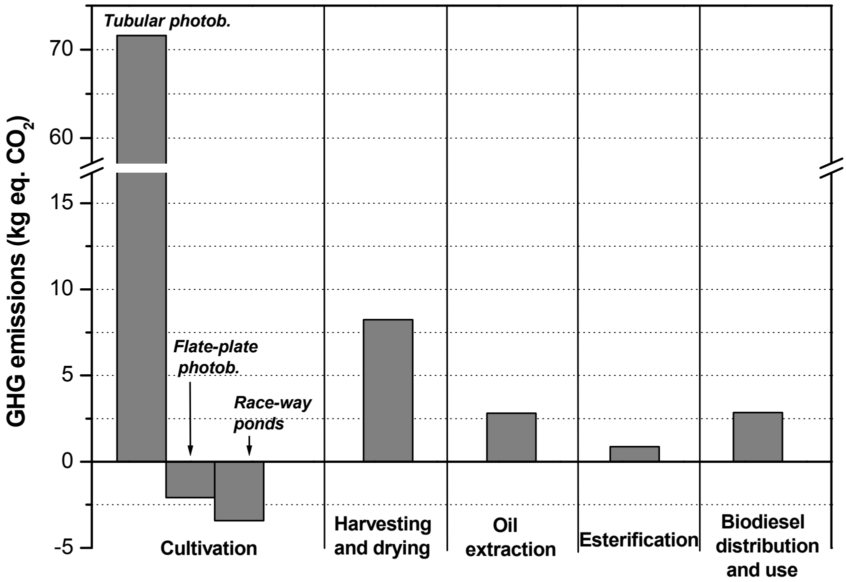

Fig. 2: GHG emissions of processes for biodiesel production from microalgae

As known, CO_2 emissions are a direct consequence of fossil fuel consumption for energy requirements. In this sense, power supply and NER values were also estimated by Gabi and represented in Figure 3. As expected, tubular reactor shows the highest energy consumption value (~1,730 MJ/kg biodiesel). These results are in agreement with others previously reported [11,12] which describe that the required power for an adequate mass transfer in tubular photobioreactors is more than 30 times higher than the necessary for flat-plate ones. The configuration of each kind of reactor leads to different mixing and aeration systems which are the principal responsible of the energy consumption (tubular photobioreactor consists of an array of plastic or glass tubes whereas flat panel photobioreactors are constructed by vertically translucent flat plates). Regarding to the cultivation open system, it presents the lowest energy requirement's because of it is the most simple configuration (raceway ponds are open systems with a closed loop recirculation channel with paddlewheel for mixing and circulation).

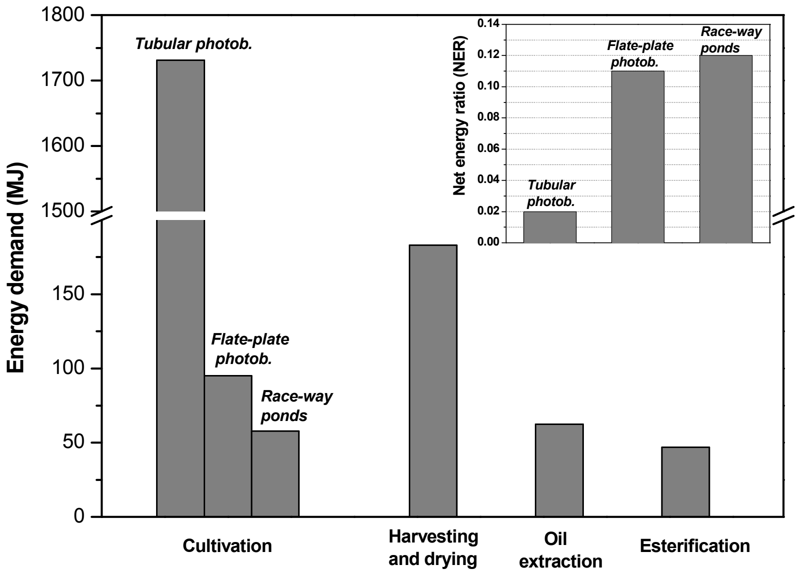

Fig. 3: Energy consumption and NER values of processes for biodiesel production from microalgae

Analysing NER values for all the systems, they are very low (~0.12 for raceway and flat-plate and 0.02 for tubular system). This is mainly due to the centrifugation process which is necessary for eliminating the larger part of water. However, these low values do not include the combustion of the dry biomass obtained after the extraction of the lipids for energy production (increasing NER). The similar values of the ratio produced/consumed energy for both raceway and flat plate systems can be explained by comparing the required energy for the construction of the raceway ponds to that required in the flat panels for mixing and aeration of the system.

In conclusion, tubular photobioreactors are not efficiently systems for biodiesel production from microalga *Nannochloropsis gaditana* while flat plate reactors are the most remarkable configuration due to the lower power supply (lower CO_2 emissions) and the lower contamination of the microalga culture in comparison to the raceway one.

Finally, alternative processes for microalgae harvesting and drying with lower energy requirements seems to be indispensable for increasing NER values and, subsequently, the feasibility of biodiesel production from algae oil.

References

[1] Mata TM, Martins AA, Caetano NS (2010) Microalgae for biodiesel production and other applications: A review. Renew Sustainable Energy Rev 14:217-232.

[2] Miao X, Wu Q (2006) Biodiesel production from heterotrophic microalgal oil. Biores Technol 97:841-846.

[3] Vicente G, Martinez M, Aracil J (2004) Integrated biodiesel production: a comparison of different homogeneous catalysts systems. Biores Technol 92:297-305.

[4] Xu H, Miao X, Wu Q (2006) High quality biodiesel production from a microalga Chlorella protothecoides by heterotrophic growth in fermenters. J Biotechnol 126:499-507.

[5] Lubián LM (1982) *Nan*nochloropsis gaditana sp. nov. una nueva Eustigmatophyceae marina. Lazaroa 4:287-293.

[6] Jorquera O, Kiperstok A, Sales EA, Embirucu M, Ghirardi MA (2010) Comparative energy life cycle analyses of microalgal biomass production in open ponds an photobioreactors. Biores Technol 2010:1406-1413.

[7] Patil V, Tran KQ, Giselrod HR (2008) Towards sustainable production of biofuels from microalgae. Int J Molecular Sci 9:1188-1195.

[8] Yusuf C (2007) Biodiesel from microalgae. Biotechnol Adv 25:294-306

[9] Razon LF, Tan RR (2011) Net energy analysis of the production of biodiesel and biogas from the microalgae: Haematococus pluviales and Nannochloropsis. Applied Energy, available on-line 15 January 2011

[10] Gerpen JV (2005) Biodiesel processing and production. Fuel Proces Technol 86:1097-1107.

[11] Molina Grima E, Acién Fernández FG, García Camacho F, Chisti Y (1999) Photobioreactors: light regime, mass transfer, and scaleup. J Biotechnol 70:231-247.

[12] Sierra E, Acién FG, Fernández JM, García JL, González C, Molina E (2008) Characterization of a flat plate photobioreactor for the production of microalgae. Chem Eng J 138:136-147.

Modelling the Inventory of Hydropower Plants

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Abstract Life cycle inventories are data intensive by definition and missing data continues to hinder more complete and accurate assessments. This article proposes a statistical approach to address data gaps in life cycle inventories applied to large scale hydroelectric power. The procedure relies on relationships between the technical characteristics of hydropower plants and the material and energy flows necessary throughout the life cycle of such systems. With highly flexible estimators known as kriging, predicting the value of material and energy flows suddenly becomes more accurate. From relatively small sample sizes, kriging allows better estimation without averaging out any of the original data. Similarly, parameter estimation and model validation can be performed through cross validation which assumes very little on the data itself. Mean absolute errors for various forms of kriging and regression show that the former performs better than the latter, more so in cases of incomplete data.

1 Introduction

Hydroelectric power generation accounts for approximately 16% of world electricity output, with China, Canada and Brazil being the leading producers [1]. Despite widespread claims of hydroelectricity being a renewable source of power, very few studies on its potential impact exist. Indeed, less than a handful of life cycle assessments (LCAs) have been conducted on hydro power plants [2]. Hydroelectric projects depend on numerous local conditions, topography, hydrology, geology as well as factors affecting human populations and the environment. This specificity of hydroelectric power accounts for much of the lack of necessary data, making inventory analysis and impact assessments notoriously difficult. Moreover, new hydropower projects became larger over time, multiplying data collection efforts and challenging the development of environmental assessment tools [3]. Generic hydropower plants hardly exist.

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Quantifying and qualifying the gains and losses to society and the environment from hydropower projects is a monumental task the World Commission on Dams managed to undertake [4]. Most studies have emphasised the site specific issues which translate into questionable comparisons with other large sources of power. Hence the goal of this article, taking a statistical approach to enable better estimation of life cycle inventory data on hydropower plants and more generally, showing how data gaps in life cycle inventories (LCIs) and associated databases can be alleviated with appropriate statistical estimators. Besides primary sources of data, this article draws upon existing inventories, namely the Ecoinvent database and Itaipu assessment [2,5]. A description of the hydropower systems follows before the methodological approach is explained. Results emphasise the importance of the construction phase of hydropower and a conclusion ends this article.

2 System characteristics

The power of hydroelectric plants usually depends on two factors, the water flow and the hydraulic head or height difference between water intake and turbine according to the relation:

$$P = \eta \rho g Q H \tag{1}$$

where P is power (W), η is a coefficient of efficiency, ρ is the density of water (~1000 kg/m^3), g is the acceleration due to gravity (~9.8 m/s^2), Q is the flow rate (m^3/s) and H is the head (m). In practice this is implemented with either low head, high flow (generally run-of-river plants), high head, low flow (generally with reservoirs) or combinations in between. Again this scale varies considerably with respect to site specifications.

Overall, hydroelectric power plants are 82 to 88% efficient [5]. The difference between run-of-river and reservoir plants lies essentially with storage which is one of the key advantages of hydroelectric dams [6]. Electricity cannot be stored such that supply must match demand almost instantaneously. Technically, hydropower plants can respond and adjust their production within minutes and are therefore well suited for both base and peak load production, particularly with a reservoir.

Despite the specificity of every power plant and reservoir, a set of characteristics was chosen based primarily on data availability. Table 1 summarises the characteristic variables of hydroelectric power plants referred to in this article.

Tab. 1: System characteristics

Characteristic variables	Ranges	Notes
Type	0/1	Run-of-river/reservoir
Total installed capacity (MW)	20 – 14000	-
Annual production (GWh/year)	90 – 90000	Average over ≥ 7 years
Hydraulic head (m)	6 – 1400	-
Surface of reservoir (km^2)	0 – 4200	-
Volume of reservoir (km^3)	0 – 140	-

3 Methodology

Life cycle inventory, as defined by international standards, requires the quantification of material and energy flows as well as emissions crossing a system's boundaries [7]. The quality of a LCI directly influences the overall quality of an LCA [8]. A number of reasons lead us to hypothesise that statistical estimation can overcome both the specificity of hydropower with respect to construction sites and operations and the limitations of current practices dealing with missing data in LCI. Analysing the inventory of electricity generation tends to yield higher uncertainties when compiled from generic data [9]. Moreover, statistical models have been relatively successful in such cases as with the assessment of chemical manufacturing [10]. Regression and other linear methods have also been applied to inventory analysis of electricity generation [11]. A more flexible estimator is presented here. Borrowed from spatial statistics, kriging distinguishes itself on several accounts. Clearly no Cartesian points exist in LCIs such that the characteristic variables described above become coordinates and the material and energy flows or emissions to be estimated, dependent variables. Kriging is a weighted linear combination of the observations. We assume a model with both a random function and a deterministic drift or low order polynomial. Provided a set of unbiasedness constraints are met, the properties in Table 2 hold:

Tab. 2: Properties of the kriging estimator [12]

Properties	Description	Expression
Unbiased estimator	Expected values of estimates and observations are equal	$E[\hat{Z}(x_i)] = E[Z(x_i)]$
Exact estimator	No loss of information	$\hat{Z}(x_i) = Z(x_i)$
Screening effect	Weights vary according to the distance from estimates	-
Size and position	Covariance functions *C(h)* to model observations in space.	$h = \|\|x_i - x_j\|\|$
Smoothing effect	Kriging varies less than the estimated phenomena.	$Var(\hat{Z}(x_i)) \leq Var(Z(x_i))$

E[ ] and *Var( )* are the expectation and variance operators respectively. The x_i's are characteristic variables and the *Z*'s corresponding material and energy flows. Covariance functions *C(h)* translate in formal terms the idea that distinct observations close to one another should agree more than if they were far apart. A number of valid covariance functions exist, linear, exponential, spherical, etc. Each function has 3 parameters, the first of which is a nugget effect which captures variations on a small scale and can be understood as an interpolating smoothing parameter [13]. Note that since kriging is an exact estimator, it is discontinuous at every observation. The second parameter is a structured variance which equals the total variance when added to the nugget effect. Third, the range of a covariance function corresponds to the distance *h* at which the covariance is nil or observations are unrelated.

If deterministic and random components as well as covariance functions impart great flexibility to the kriging estimator, another advantage is multivariate kriging, or cokriging. Multivariate kriging enables the estimation of primary variables using data from secondary variables simultaneously [14]. All estimations can be performed jointly as the order between primary and secondary variables can be interchanged [15]. For example, if data on explosives is available for the construction of most power plants and dams whereas the excavated volumes are not, joint estimation might provide more accurate results for excavation. In general, primary and secondary variables need not be observed at the same points.

The relatively small sample sizes (here $N = 27$) available for power generation systems would forfeit many attempts to validate a model and its parameters. Cross-validation however, has no prior assumption such as normality and observations take part in both trial and validation sets [16]. Leave-one-out cross-validation (LOOCV) successively removes observations and derives estimates from neighboring values exactly once. Moreover, with cokriging, the values of secondary variables for a given observation can be part of the validation data all together or one after the other. These two options are referred to as Leave-one-observation-out cross-validation (LOoOCV) and Leave-one-variable-out cross-validation (LOvOCV), respectively. Comparing the results of the two validation approaches indicate some of the advantages of cokriging, should the errors calculated from LOvOCV be lower than that of LOoOCV.

4 Results

Data on the construction, operation and removal of hydropower plants is scarce. A sample from different sources and for widely different power plants was carefully assembled. Emphasis was put on the construction phase, less controversial than reservoir flooding and responsible for most of the overall impact in many cases [2,5,17]. Time is clearly an important factor to be considered with construction. In particular, Ribeiro and his colleague [2] have shown the LCI of hydroelectric power plants to be sensitive to time horizons. Optimistically the useful lives of infrastructures and equipments were adjusted to 100 and 50 years, respectively. If dams and equipment can technically last that long, chances are operations and economics dictate otherwise.

By and large the main material and energy flows by weight or volume involved in hydroelectric power construction and operation are water, cement and concrete, steel of various grades for structures and equipment, explosives, and fuels and lubricants for machinery, transportation and operation. Their provision and the construction phase itself account for an important share of hydropower's total impact, although water use is often ignored. Because of different scales, the data was normalised, dividing variables by their mean values. Mean absolute errors (MAEs) can then be derived from performing leave-one-observation-out cross-validation (LOoOCV). As shown in Figure 1, comparisons between the different estimators are not straightforward.

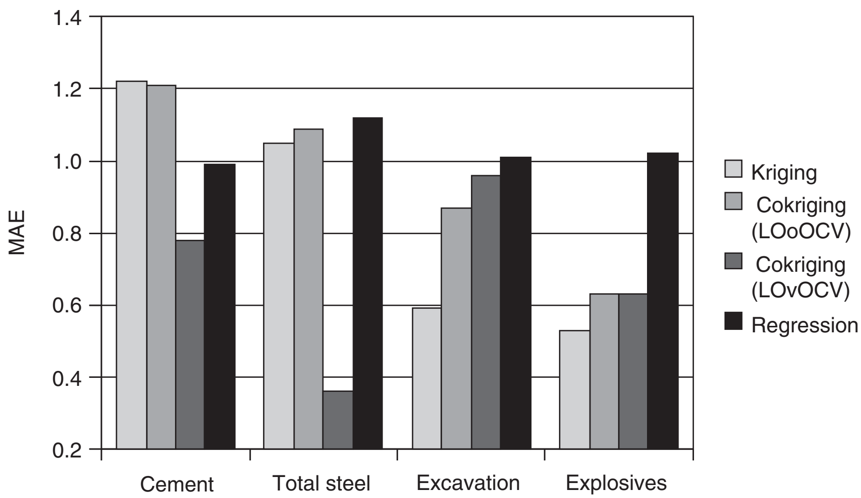

Fig. 1: Comparison of MAEs for different estimators

The most evident result is the most interesting: the kriging and cokriging errors are lower than that of regression for the two rightmost flows of the chart. This shows not only that kriging performs better in the presence of data gaps but also provides more accurate estimates. Estimating one variable at a time reduces somewhat the MAEs, as the comparison of cokriging errors indicates. Both

observations support the original hypothesis that estimators as versatile as kriging are particularly well suited in situations where data is missing and needed.
Does accounting for several variables simultaneously further reduce kriging errors? Besides providing more information from which to estimate missing values, the benefits are not obvious either. In the case of kriging, univariate or multivariate, both characteristics, installed capacity and annual production, enter in the calculation. Regression uses installed capacity only. It appears that the advantage of multivariate over univariate kriging (and regression) are particularly important when data is relatively complete, as shown on the left of Figure 1. To the contrary, univariate kriging shows lower errors when data is missing. One explanation is weaker relationships between the different material flows themselves than with the characteristic variables, which is a reason why they are characteristic. Hence the preliminary work necessary to identify appropriate variables describing the properties of a system.

5 Conclusion

The lack of data affects inventory analysis of many hydropower plants and other systems. In this article, the authors show how material and energy flows involved in the initial phases of hydropower plants can be better estimated based on their design and operating characteristics. The constraints and drawbacks with respect to data collection and data gaps can therefore be loosened, thanks to appropriate and accurate statistical estimators. The versatility of kriging allows for better estimation especially in the absence of complete data. Limitations to this procedure exist, kriging is better suited for interpolation purposes. A minimum sample size (typically $N = 30 - 50$) is necessary to establish covariance functions, such that this experiment used predefined functions and cross-validation to estimate the better model and its parameters. While no prior assumptions on the data are needed, the results of this parameter selection might differ from one data set to the next. Nevertheless, the approach presented above contributes not only to the estimate of missing data but also to much needed representative data which lacks from existing inventory databases.

References

[1] IEA (2010) Key World Energy Statistics. International Energy Agency

[2] Ribeiro FDM, da Silva GA (2010) Life-cycle inventory for hydroelectric generation: a Brazilian case study. J Cleaner Prod 18:44-54.

[3] Rosenberg DM et al. (1997) Large-scale impacts of hydroelectric development. Environ Rev 5:27-54.

[4] World Commission on Dams (2000) Dams and Development: A new framework for decision making. Sterling VA: Earthscan Publications Ltd., London

[5] Bauer C et al. (2007) Wasserkraft, Sachbilanzen von Energiesystemen: Grundlagen fur den okologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Okobilanzen fur die Schweiz. Paul Scherrer Institute Villingen, Swiss center for life cycle inventories, Dubendorf

[6] Egre D, Milewski JC (2002) The diversity of hydropower projects. Energy Policy 30:1225-1230.

[7] ISO, 14040:2006. Environmental management - Life cycle assessment - Principles and framework, International Organization for Standardization

[8] Mongelli I, Suh S, Huppes G (2005) A structure comparison of two approaches to LCA inventory data, based on the MIET and ETH databases. Int J Life Cycle Assess 10:317-324.

[9] May JR, Brennan DJ (2003) Application of data quality assessment methods to an LCA of electricity generation. Int J Life Cycle Assess 8: 215-225.

[10] Wernet G et al. (2008) Molecular-Structure-Based Models of Chemical Inventories using Neural Networks. Environ Sci Technol 42:6717-6722.

[11] Hondo H (2005) Life cycle GHG emission analysis of power generation systems: Japanese case. Energy 30:2042-2056.

[12] Isaaks EH, Srivastava MR (1989) Appl Geostatistics, Oxford University Press

[13] Marcotte D, David M (1988) Trend Surface Analysis as a Special Case of IRF-k Kriging. Math Geol 20:821-824.

[14] Wackernagel H (2003) Multivariate geostatistics: an introduction with applications. Springer, Berlin

[15] Marcotte D (1991) Cokriging with matlab. Comput Geosci 17:1265 - 1280

[16] Marcotte D (1995) Generalized Cross-Validation for Covariance Model Selection. Math Geol 27:659-672.

[17] Peisajovich A (1997) Etude du cycle de vie de l'electricite produite et transportee au Quebec. Direction principale Communication et Environnement, Hydro-Quebec

Life Cycle Carbon Dioxide Emission and Stock of Domestic Wood Resources using Material Flow Analysis and Life Cycle Assessment

Junhee Cha, Youn Yeo-Chang and Jong-Hak Lee

Abstract Lifecycle greenhouse gas emission and carbon stock of domestic wood resources and products were evaluated using material flow analysis and life cycle assessment. Carbon storage and material substitution effect was analysed for a single residential wooden house substituting typical concrete house. The embodied greenhouse gas emission of wooden type was lower than that of concrete type by 54.3t CO_2e. Furthermore, the wooden house stored 38.3t CO_2 in the wood material used for the house until the end of life of the house. To promote greenhouse gas mitigation potential of forest sector, the application of life cycle assessment and material flow analysis of wood products is required. Wood biomass such as demolition wood or wood pellet can reduce greenhouse gas emission when used for energy source by substituting fossil fuel consumption. Better option and increasing wood use for material and energy substitution can be a strategy to cope with climate change in forest sector.

1 Introduction

Wood is potentially a carbon neutral material if produced in a sustainable way. There is a growing concern for the mitigation of global warming by wood use increase such as increasing construction of wooden building. Building sector is a major energy and material consumer as well as a major consumer of wood products. Wood can substitute energy intensive materials such as concrete and steel. When wood and wood wastes are used for biomass energy substituting fossil fuels, greenhouse gas emission can be reduced because burning of wood does not increase atmospheric carbon dioxide in the atmosphere. Wood building products including framing lumber and plywood are sequestering carbon dioxide for a long time until the end of life of a building.

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This study analyses life-cycle greenhouse gas emissions of a conventional two-story single family house comparing wood frame and concrete frame. Greenhouse gas emissions (CO_2, CH_4, and N_2O) are estimated for the life-cycle stages of the house, that is, raw material extraction, transportation, building operation, maintenance, and final disposal. For this purpose life cycle assessment (LCA) methodology, a tool to evaluate potential environmental impact of a product or a service, with the mechanism of analysing input and output of material and energy of the system boundary.

Comparison between wooden- and concrete- framed houses have found that wood material based houses have low embodied energy and low greenhouse gas emission than concrete or steel based houses [3]. An LCA study done in the U.S. by CORRIM (Consortium for Research on Renewable Industrial Materials) found that life-cycle carbon dioxide emissions of wood framed houses are smaller than steel framed design by 17% and then reinforced concrete design by 31% [7]. In Japan, where the proportion of wooden house construction is more than 70% of the residential house, the life-cycle carbon emission of reinforced concrete house is about 23% greater than wooden house [4].

While around two thirds of land is covered by forest, Korea is a net importer of wood products. This is because most of the forest resources are not matured and the production cost of domestic wood is high. From the growing concern form mitigate global warming, increasing portion of renewable energy is an important environmental issue of Korea. However wood biomass is a carbon neutral energy, it requires some energy during collection, production, and transportation of the fuel. Wood resides in the forest, which amount to one third of roundwood production in South Korea are not collected and used for material or energy source for economic reason. As a result, the important renewable low carbon energy is abandoned and becomes potential greenhouse gas emitter. We study the opportunities to increase the carbon stock and to reduce greenhouse gas emission by increasing the use of uncollected wood biomass as a fuel and wood material substitution in forest sector. Wood pellet LCA study was done to know the GHG mitigation potential of wood biomass. The production and use of wood pellets has been increased greatly in the South Korea. Wood pellets are recognised as carbon neutral energy and can be made from the wood resources of sawmill residues, roundwood, and wood residues collected from forestry practices. The embodied energy use and greenhouse gas emission were compared among three types of wood biomass sources for pellet production. The wood material life-span of domestically produced wood is rather short compared to imported wood, which is mostly used for long lived wood products such as lumber in wooden house. About sixty percent of domestic wood is used for pulp production and medium density fibreboard production, which are rather short life-cycle. In conclusion, greenhouse gas reduction potential of domestic wood can be achieved by increasing use of long lived wood products storing carbon and substituting fossil intensive materials and by using more wood biomass for fuel substituting fossil fuel.

2 Materials and methods

2.1 Life cycle assessment of wooden house

Life cycle assessment (LCA), a useful tool to assess overall potential environmental impact of a product or a service, was used in this study to compare life-cycle greenhouse gas emissions between wooden house and concrete house alternative. The goal of the study is to estimate life-cycle global warming impact of a single family house focused on greenhouse gas emissions related to life cycle of a house, from raw material extraction to final disposal of a house. The three main greenhouse gases, carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) were included for the GHG estimation in mass of carbon dioxide equivalent (t CO_2e).

The studied wooden house, HANGREEN, is a two-story single family house (floor area: 190m^2) constructed in South Korea. The system boundary of the study consists of five life-cycle stages, such as, building material production including transportation, on-site construction with heavy machinery, house operation for 50 years, maintenance during house use, and final disposal of house.

2.2 Material flow analysis of wood

Wood products can displace more fossil-fuel intensive construction materials such as concrete, steel, aluminium, and plastics, which can result in significant emission reductions [8]. Material flow analysis helps to understand societal metabolism of sustainable production and consumption of staple resources [2]. There are some studies of material flow analysis of wood products including paper [5].

In 2009, Korea consumed 26,607,000m^3 of wood and wood products. Among them, 23,431,000m^3 was imported, and 3,176,000m^3 was from domestic roundwood supply. 10,967,000m^3 (41.2%) was used for pulp and chip, 5,852,000m^3 (22.0%) was used for lumber, 3,061,000m^3 (11.5%) was used for plywood and veneer, and 2,472,000m^3 (9.3%) was used for particle board and

MDF. In 2009, 819,000m^3 of wood was collected after forest tending, among which, 199,000m^3 (24%) was used for roundwood resources. Korea exported 7,000m^3 of plywood in the year 2009 [6].

Forest stock of South Korea was 659,120,000 at the end of 2008 and 696,828,000m^3 at the end of 2009. Therefore, 37,708,000m^3 of forest stock was increased during the year of 2009. From the MFA and LCA of domestic wood products, greenhouse gas reduction potential of wood material substitution effect can be measured. Carbon stock in forest sector of Korea is increasing.

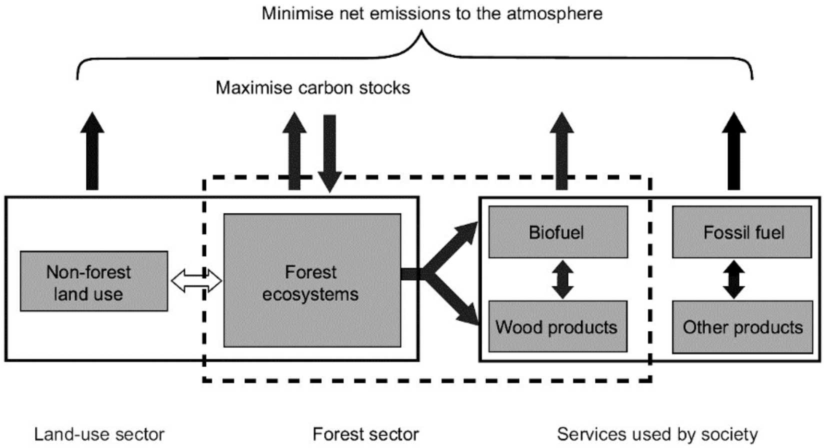

Fig. 1: Forest sector mitigation strategies need to be assessed with regard to their impacts on carbon storage in forest ecosystems on sustainable harvest rates and on net GHG emissions across all sectors (Source: IPCC Fourth Assessment Report)

3 Results

3.1 Greenhouse gas reduction effect of wooden house

3.1.1 Embodied GHG emissions

The embodied GHG emission of a house is total greenhouse gas emissions from construction material production, transportation, and construction activities. Total material input of the wooden house was 209.6 ton, while concrete house used

499.7 ton of construction material. GHG emissions of production and transportation of construction material of wooden house and concrete house were 51.27t CO_2e and 104.61t CO_2e, respectively. GHG emissions from fossil fuel consumption of heavy construction machinery are 1.05t CO_2e for the wooden house and 1.97t CO_2e for the concrete house. Embodied GHG emission of wooden house was 52.33t CO_2e and that of concrete house was 106.59t CO_2e. The embodied GHG emissions of a wooden house were lower than those of a concrete house by 54.3t CO_2e.

3.1.2 GHG emissions during house use

The service life of 50 years was assumed for lifetime of the studied houses. The service life of 50 years has been chosen for many other LCA studies on building regardless of framing material [1]. GHG emission from consumption of natural gas and electricity used during the service life of the studied houses was estimated 421.45t CO_2e regardless of house type owing to the same thermal capacity assumption. GHG emissions from maintenance of the houses were 6.95t CO_2e for wooden house and 8.17t CO_2e for concrete house.

3.1.3 GHG emissions from house disposal

Dismantling of the house and transportation of demolished material to the waste management site are included for the estimation of greenhouse gas emission of disposal stage. GHG emissions from machinery use in disposal stage were 3.11t CO_2e for wooden house and 5.25t CO_2e for the concrete house.

3.2 Greenhouse gas reduction scenario

3.2.1 Life-cycle GHG emissions

The baseline life-cycle GHG emissions, which include material, construction, operation, maintenance, and disposal were 483.84t CO_2e for wooden house and 541.46t CO_2e for concrete house. The emission difference of the two houses was 57.62t CO_2e. This implies if wooden house is constructed instead of concrete house, 57.62t CO_2e is reduced.

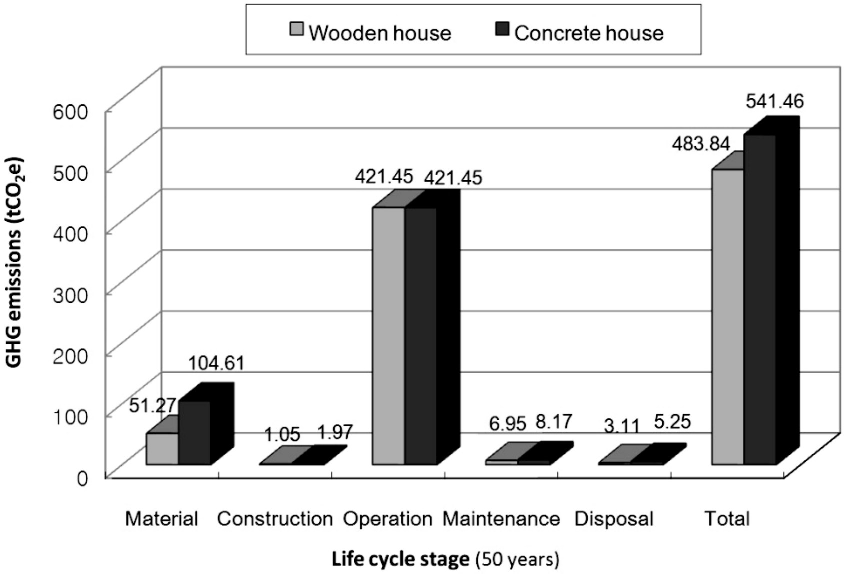

Fig. 2: Life-cycle greenhouse gas emission of wooden and concrete house

3.2.2 Energy saving in use phase

About 80% of life-cycle GHG emissions result from fossil fuel consumption in use phase of 50 years. The GHG reduction potential of energy saving by 10%, 20%, and 30% are 42.2t CO_2e, 84.3t CO_2e, and 126.5t CO_2e respectively. Photo voltaic system installed on roof of the house can substitute purchased electricity. Considering the emission factor of PV systems, the reduction potential of using PV system per 1kWh is 0.419 $kgCO_2e$. Using this factor, we assumed we get electricity from PV system by 30% and 50%. The reduction potential is 24.5t CO_2e and 40.8t CO_2e respectively.

3.2.3 Carbon storage and fossil fuel substitution

Harvested wood products (HWP) store carbon dioxide during the use. Wooden house uses a great amount of wood products for building components, such as framing wood of wall, roof and flooring. In the study, wooden house used 38.4m^3 of HWP and stored 38.3t CO_2, while concrete house used 1.8m^3 of HWP and stored 1.8t CO_2. Waste wood from demolished wooden house can substitute fossil fuel consumption when used as energy source at combined heat and power generation. If wood substitute bunker C oil and LNG in CHP, 21.8t CO_2e and 18.1t CO_2e can be reduced from using carbon-neutral biomass, wood.

3.2.4 Wood pellet use for boiler

From the life-cycle assessment of wood pellet production system in South Korea, 0.27kg CO_2e/Mcal and 0.24kg CO_2e/Mcal were estimated to be reduced by using wood pellet boiler instead of using diesel and natural gas, respectively. In the use phase of the studied house, annual energy required for heating was estimated 24,950 Mcal/year. When substituting natural gas boiler with wood pellet boiler, it comes out that about 6 ton of carbon dioxide can be reduced annually.

4 Discussion

While 6.4 million ha of forests in South Korea, which accounts for 64% of total land cover, sequester carbon dioxide, harvested wood products can help to increase carbon pool of forest sector including wood products carbon pool. The mitigation effect of biomass use is direct and permanent on condition that the biomass source was come from sustainably managed forests.

The production and use of wood pellets has been increased greatly in South Korea. Wood pellets are carbon neutral energy and can be made from the wood resources of sawmill residues, roundwood, and wood residues collected from forestry practices. The LCA study showed that the use of wood pellets as biofuels substituting fossil fuels such as natural gas and kerosene for boilers has positive effects both in reducing greenhouse gas emissions and in decreasing cost for house heating. The life-cycle environmental impacts of wood pellets were mainly from the transportation of wood resources and pellet products to the final consumer. Better utilisation of forest residues which have not been collected and used much for biomass, increasing pellet-boiler efficiency, and improving regional biomass utilisation system such as decrease in transportation distance can be measures to reduce greenhouse gas emissions in forestry sector for climate change mitigation.

The result shows that wooden house construction has a potential of mitigating climate change. The total GHG emissions of wooden house was lower than that of concrete house over the life-cycle of the studied single family house such as raw material extraction, manufacture of building materials, on-site construction, operation and maintenance, and final disposal of house.

At present, demolished construction materials are landfilled or incinerated as well as recycled as recycled concrete gravel or as wood resource for particle board production. When recovered wood materials from demolished buildings are used as biomass energy substituting for fossil fuels, greenhouse gas emissions could be more reduced. Increasing wooden house construction instead of concrete or steel house would be an effective measure to reduce fossil fuel consumption and greenhouse gas emissions.

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References

[1] Adalberth K, Almgren A, Petersen EH (2001) Life cycle assessment of four multi-family buildings. Int J Low Energy Sustainable Build 2

[2] Bringezu S (1997) Comparison of the material basis of industrial economies. In: Bringezu S, Fischer-Kowalski M, Kleijn R, Palm V, (eds) Analysis for Action, Support for Policy towards Sustainability by Material Flow Accounting. Proceedings of the ConAccount Conference, 11–12 September 1997, Wuppertal

[3] Buchanan A, Honey B (1994) Energy and carbon dioxide implications of building construction. Energy Build 20(3):205-217.

[4] Gerilla GP, Teknomo K, Hokao K (2007) An environmental assessment of wood and steel reinforced concrete housing construction. Build Environ 42:2778-2784.

[5] Hekkert MP, Joosten LAJ, Worrell E (2000) Analysis of the paper and wood flow in The Netherlands. Resour Conserv Recycl 30:29-48

[6] KFS (2010) KFS Statistical yearbook of Korean forestry, Korea Forest Service

[7] Lippke B, Wilson J, Perez-Garcia J, Bowyer J, Meil,J (2004) CORRIM: Life-cycle environmental building materials. Forest Prod J 54(6):8-19

[8] Petersen AK, Solberg B (2002) Greenhouse gas emissions, life-cycle inventory and cost-efficiency of using laminated wood instead of steel construction -Case: beams at Gardermoen airport. Environ Sci Policy 5:169-182.

Analysis on Correlation Relationship Between Life Cycle Greenhouse Gas Emission and Life Cycle Cost of Electricity Generation System for Energy Resources

Heetae Kim and Tae Kyu Ahn

Abstract In this work, we analysed correlations between life cycle greenhouse gas (GHG) emissions and life cycle cost of energy resources. Energy resources studied in this paper include coal, natural gas, nuclear power, hydropower, geothermal energy, wind power, solar thermal energy, and solar photovoltaic energy, and all of them are used to generate electricity. We calculated the mean values, ranges of maximum minus minimum values, and ranges of 90% confidence interval of life cycle GHG emissions and life cycle cost of each energy resource. Based on the values, we plotted them in two dimensional graphs to analyse a relationship and characteristics between GHG emissions and cost. Besides, to analyse the technical maturity, the GHG emissions and the range of minimum and maximum values were compared to each other. For the electric generation, energy resources are largely inverse proportional to the GHG emission and the corresponding cost.

1 Introduction

Our society has gradually entered into an electricity centred society. The total world's energy consumption was 11,730 Mtoe in 2006, and it is expected to increase by 1.6% every year until 2030 [1]. At present, energy system generates electricity mainly from fossil fuel, emits a great amount of greenhouse gas (GHG) and causes anthropogenic climate change[2]. To mitigate the environmental impact and to meet the energy need, eco-friendly policies and technologies should be adopted. These may involve the complete replacement of highly energy consuming devices by electrical equipment, e.g. electric vehicles. In fact, electricity consumption reached up to 15,665 TWh in 2006, and it is expected to constantly increase up to 20,760 TWh in 2015 and 28,140 TWh in 2030 [1]. At present, the world has been changing the existing fossil fuel centred energy system to an electricity centred renewable energy system. According to the gradual

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improvement and exchange of electrical motive force, our fossil fuel centred society can move towards a post-carbon society, i.e. an electricity centred one.

By analysing the comprehensive environmental impacts of technology, life cycle assessment (LCA) helps decision makers to make more reasonable decisions when they want to introduce new energy related technologies[3]. Because the process in launching and organising a new energy system is obscure, time consuming, and mostly irreversible, they also should consider GHG emissions at the same time. Generally renewable energy resources such as solar photovoltaic energy are usually thought to be no GHG emission source. However if they are thoroughly considered using the life cycle analysis from the first stage of material mining, production, maintenance, operation, and to final disposal, renewable energy resources also emit GHG especially during their maintenance [4]. An LCA analysis on electricity generation technologies provides practical information and helps to make more comprehensive decision within possible available information [5]. A life cycle cost (LCC) is a tool used to estimate the cost during the whole life cycle from cradle to grave of buildings [6], which can be applicable to LCC estimation of electricity generation. An LCA analysis combined with life cycle GHG emissions and life cycle cost of electricity generation can correlate their energy performance (LCC) with environmental impact (GHG emission) [4].

As for the GHG emissions and cost of the respective energy source for electricity generation, this work investigated the correlation between two values and analysed the characteristics of each energy resource. We also reviewed the energy references and the conventional analytical methods (LCA) and then consequently correlated with each other.

2 Data collection and analysis

In this work, we collected a number of reported values of GHG emissions and cost for a certain energy source and analysed them statistically to obtain the practical parameters such as a mean value, minimum, maximum, and 90% confidence interval range. Referring to the previous results by Lenzen [7], Hammons[8] and Sherwani et al. [9], we estimated GHG emissions during the generation of electricity using nuclear power, geothermal energy, and solar photovoltaic (PV) energy, respectively. To obtain other values in making more references, GHG emissions and cost for the existing fossil fuels and for renewable energies, we used the numbers from Hondo [10], L. Gagnon et al. [11], Denholm et al. [12], Uchiyama [13], Weisser [5], and Varun et al. [4]. In addition, the reported statistics by World Energy Council (WEC) [14] and Intergovernmental Panel on

Climate Change (IPCC) [15] were introduced to estimate the environmental impact and the economic cost in an energy source, respectively.

We introduced the original LCA data for each energy source in case that the authors had addressed solely the mean values. When the original data didn't have been cited but the mean value was estimated, the average values were adopted. For example, for geothermal energy, the previous results revealed its average values of GHG emissions and cost [16]. Even worse, for its cost range, two reported results are belied with large discrepancy [17], which was averaged statistically. As cost for solar thermal energy, only statistically reliable regions are addressed [17], so we set the mean value in the region and accepted the reported region as reliable area. The values of geothermal energy and solar thermal energy used in this work were relatively smaller values from actual cases. In addition, the range of GHG emissions and cost of energy resources were plotted by using error bars based the standard deviation of the values (Figure 1 and 2). The confidence interval was calculated by getting standard errors based on 90% of confidence level.

3 Results and discussion

Figure 1 shows the calculated costs for various energy resources (nuclear, coal, natural gas, geothermal, hydro, wind, solar thermal and solar PV in order) with maxima/minima (line) and 90% confidence intervals (filled box). Details are listed in Table 1 and all reference data are shown in supporting materials. The mean cost of nuclear power is merely 2.9 cents/kWh resulting in the cheapest source. Next to nuclear, coal has the second cheapest source. The most expensive source is solar PV with the largest range of confidential interval. As for wind power, its maximum value is extremely high. It is because off-shore wind plants have much higher costs than on-shore plants have.

Fig. 2 shows the reported GHG emission per kWh for energy technology, which was treated statistically. Nuclear power records GHG emissions 19.7g/kWh, which is quite small amount. The nuclear technology seems already mature so that it shows the narrowest range of GHG emissions (as well as cost), i.e. 37g/kWh. However coal is the most GHG emissive source among the energy technology. Largely the amounts of GHG emission reveals decay in order of right side; coal, natural gas, geothermal, hydro, wind, solar thermal and solar PV. Wind power exhibits merely 14.4g/kWh GHG emissions which is the lowest emissions among the energy resources. In case of solar photovoltaic generation, the GHG emissions are quite low (67.8g/kWh) but has rather large intervals. Details of numbers of

GHG emissions and the costs for various energy sources are listed in Table 1.

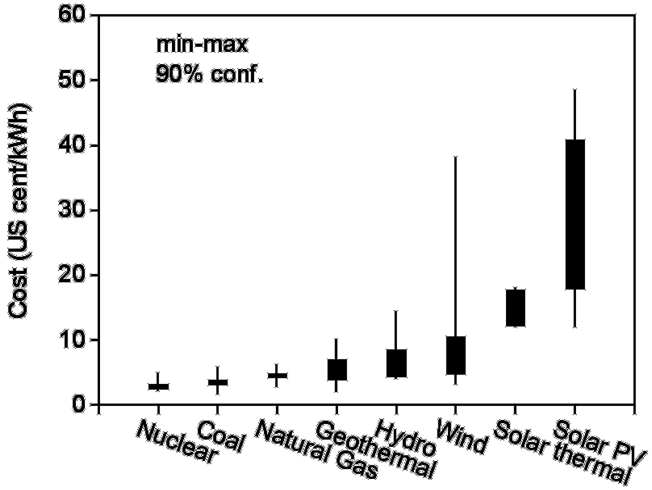

Fig. 1: Cost of electricity generation for energy resources per 1 kWh

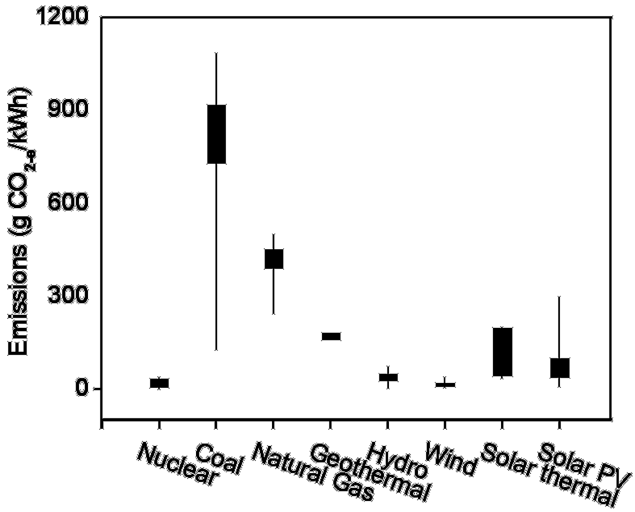

Fig. 2: GHG emission of electricity generation for energy resources per 1 kWh

Figure 2 shows the reported GHG emission per kWh for energy technology, which was treated statistically like costs. Nuclear power records GHG emissions 19.7g/kWh, which is quite small amount. The nuclear technology seems already mature so that it shows the narrowest range of GHG emissions (as well as cost), i.e. 37g/kWh. However coal is the most GHG emissive source among the energy technology. Largely the amounts of GHG emission reveals decay in order of right side; coal, natural gas, geothermal, hydro, wind, solar thermal and solar PV. Wind power exhibits merely 14.4g/kWh GHG emissions which is the lowest emissions among the energy resources. In case of solar photovoltaic generation, the GHG emissions are quite low (67.8g/kWh) but has rather large intervals. Details of numbers of GHG emissions and the costs for various energy sources are listed in Table 1 and 2.

Tab. 1: Summary of life cycle GHG emissions (g CO_2e/kWh) for electricity generation for energy resources

	Mean	Max.	Min.	Conf.	Reference
Coal	823	1085	130	729-917	[18-22]
Natural gas	421	499	245	391-450	[18-20, 24, 25]
Nuclear	20	40	3	8-31	[2, 18-20, 26, 27]
Hydro	38	75	4	28-48	[28-33]
Geothermal	170	N/A	N/A	N/A	[8]
Wind	14	39	7	11-18	[18-20, 26, 35-40]
Solar thermal	119	202	36	44-156	[41-43]
Solar PV	68	300	9	39-96	[10, 18, 20, 24,44-50]

Tab. 2: Summary of life cycle cost (US cent/kWh) for electricity generation for energy resources

	Mean	Max.	Min.	Conf.	Reference
Coal	3.5	5.7	1.6	3.2-3.8	[23]
Natural gas	4.6	6.0	2.7	4.4-4.9	[23]
Nuclear	2.9	4.8	2.1	2.5-3.2	[23]
Hydro	6.5	14.3	4.0	4.5-8.5	[23]
Geothermal	6.0	10.0	2.0	4.0-7.0	[17, 34]
Wind	7.7	38.1	3.1	4.8-10.6	[23]
Solar thermal	15.0	18.0	12.0	12.3-17.7	[17]
Solar PV	29.4	48.5	12.1	18.0-40.8	[23]

In fact, the order of energy sources listed in Figure 1 and 2 is arbitrary and random. Also hardly can the figures tell the correlation between GHG emission and their costs. There we plotted the graph of the cost versus GHG emission for various sources so that we can position the corresponding points for each energy technology in a two-dimensional graph. As shown in Figure 3, during the electricity generation, GHG emissions and costs are largely in inverse proportional. Again Figure 3 shows error bars based on the confidence error and the mean value of GHG emissions and cost of each energy resources.

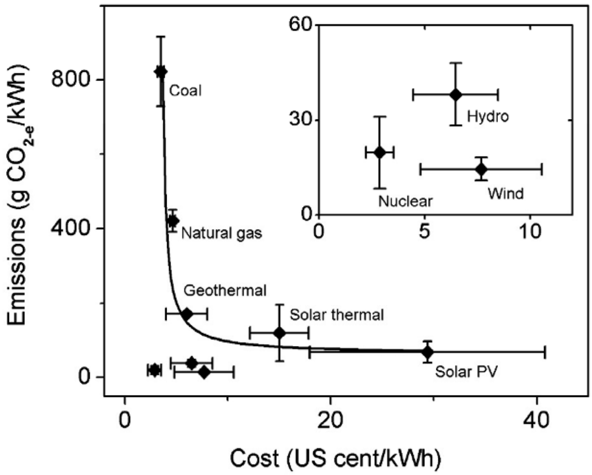

Fig. 3: GHG emissions, cost, error bar of electricity generation for energy resources and correlation between GHG emissions and cost

$$\text{GHG emissions} = 72.53 + \frac{228.95}{\text{Cost} - 3.18} \tag{1}$$

A line shows the correlation between GHG emissions and cost of electricity generation for each energy resource. Equation 1 represents their correlation. Remarkably nuclear power has superior performance in GHG emissions and cost, resulting in apart from the reversed relations between GHG emissions and costs. However, the results about GHG emission characteristics were not based on other environmental factors, such as radioactivity damage, but obtained by considering only GHG emissions.

The ranges of GHG emission values and cost values show the reliability of environmental and economic performance, respectively. Coal shows the widest range of GHG emission values. Even though coal has been used to electricity generation for a long time, the environmental performance of coal is not reliable yet. Conventional energy resources including coal and natural gas and renewable energy resources including geothermal, hydro, wind show the counter-performances. When an energy resource has high environmental reliability, it shows low cost reliability. For example, coal has higher cost reliability than wind but it has lower environmental reliability than wind. Solar PV is considered as a promising renewable energy resource. However it has the widest range of cost values so that the cost of electricity generation for solar PV is still flexible, which means that solar PV needs to optimise the technology to provide the cost reliability.

4 Conclusions

In this work, we correlated life cycle GHG emissions with life cycle cost for various energy resources; coal, natural gas, nuclear power, hydropower, geothermal energy, wind power, solar thermal energy, and solar photovoltaic energy. For the electric generation, energy resources are largely inverse proportional to the GHG emission and the corresponding cost except nuclear power. Remarkably nuclear power showed the maximum performance due to low GHG emission and low cost. In addition, more efforts are necessary to decrease of the cost of solar PV in order to apply them to electricity generation in near future.

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References

[1] International Energy Agency, IEA (2008) World Energy Outlook 2008.
[2] IPCC (2007) Climate Change 2007: The Physical Science Basis Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press
[3] ISO 14044 (2006) Environmental management – life cycle assessment – Requirements and guidelines, International Organization for Standardization
[4] Varun RP, Bhat IK (2009) Energy, economics and environmental impacts of renewable energy systems. Renewable Sustainable Energy Rev 13(9):2716-2721.
[5] Weisser D (2007) A guide to life-cycle greenhouse gas (GHG) emissions from electric supply technologies. Energy 32(9):1543-1559.
[6] NIST (1995) National Institute of Standards and Technology (NIST) Handbook 135.
[7] Lenzen M (2008) Life cycle energy and greenhouse gas emissions of nuclear energy: A review. Energy Convers Manage 49(8):2178-2199.

[8] Hammons TJ (2004) Geothermal power generation worldwide: Global perspective, technology, field experience, and research and development. Electr Power Compon Syst 32(5):529-553.
[9] Sherwani AF, Usmani JA, Varun (2010) Life cycle assessment of solar PV based electricity generation systems: A review. Renewable Sustainable Energy Rev 14(1):540-544.
[10] Hondo H (2005) Life cycle GHG emission analysis of power generation systems: Japanese case. Energy 30(11-12):2042-2056
[11] Gagnon L, Belanger C, Uchiyama Y (2002) Life-cycle assessment of electricity generation options: The status of research in year 2001. Energy Policy 30(14)1267-1278.
[12] Denholm P, Kulcinski GL (2004) Life cycle energy requirements and greenhouse gas emissions from large scale energy storage systems. Energy Convers Manage 45(13-14)2153-2172.
[13] Uchiyarna Y (2007) Life Cycle Assessment of renewable energy generation technologies. IEEJ Trans Electr Electron Eng 2(1):44-48.
[14] World Energy Council, WEC (2004) Comparison of Energy System Using Life Cycle Assessment.
[15] IPCC (2007) Climate Change 2007: Mitigation of Climate Change. Contribution of Working Group 3 to the Fourth Assessment Report of the Intergovernmental Panel on Climate Changes, Cambridge University Press
[16] Evans A, Strezov V, Evans TJ (2009) Assessment of sustainability indicators for renewable energy technologies. Renewable Sustainable Energy Rev 13(5):1082-1088.
[17] REN21 Renewable Energy Policy Network (2005) Renewables 2005 Global Status Report, Worldwatch Institute
[18] Australian Coal Association Research Program, ACARP (2001*)* Coal in a Sustainable Society.
[19] Berry JE, Holland MR, Watkiss PR, Boyd R, Stephenson W (1998) Power Generation and the Environment - a UK Perspective. Vol 1*,* AEA Technology
[20] Institut für Energiewirtschaft und Rationelle Energieanwendung, IER (1999) ExternE Externalities of Energy. Vol XX: National Implementation, Xth edn.
[21] Spath PL, Mann MK (2004) Biomass Power and Conventional Fossil Systems with and without CO2 Sequestration - Comparing the Energy Balance, Greenhouse Gas Emissions and Economics. National Renewable Energy Laboratory
[22] Spath PL, Mann MK, Kerr DR (1999) Life Cycle Assessment of Coal-Fired Power Production*,* Report NREL/TP-570-25119, National Renewable Energy Laboratory
[23] Nuclear Energy Agency/International Energy Agency, IEA/NEA (2005) Projected cost of generating electricity - 2005 update.
[24] Meier PJ (2002) Life-Cycle Assessment of Electricity Generation Systems and Applications for Climate Change Policy Analysis. Fusion Technology Institute

[25] Spath PL, Mann MK (2000) Life Cycle Assessment of a Natural Gas Combined Cycle Power Generation System. Report NREL/TP-570-27715, National Renewable Energy Laboratory

[26] VATTENFALL (2004) Vattenfall AB generation Nordic countries Certified Environmental Product Declaration of Electricity from Ringhals NPP.

[27] VATTENFALL (2004) Certified Environmental Product Declaration of Electricity from Forsmarks Kraftgrupp (FKA).

[28] Tremblay A, Varfalvy L, Roehm C, Garneau M (2004) The issue of greenhouse gases from hydroelectric reservoirs: from boreal to tropical regions. The United Nations Symposium on Hydropower and Sustainable Development, Beijing, China

[29] International Energy Agency, IEA (1998) Benign energy? The environmental implications of renewables.

[30] Varun, Bhat IK, Prakash R (2008) Life cycle analysis of run-of river small hydro power plants in India. Open Renewable Energy J 1:11-16.

[31] Varun, Prakash R, Bhat IK (2010) Life Cycle Energy and GHG Analysis of Hydroelectric Power Development in India, Int J Green Energy 7(4):361-375.

[32] VATTENFALL (2002), Certified environmental product Declaration of Electricity from Hydropower Station on the River Lule Älv.

[33] VATTENFALL (2002) Certified environmental product Declaration of Electricity from Hydropower Station on the River Ume Älv.

[34] Johansson TB, Turkenburg W (2004) Policies for renewable energy in the European Union and its member states: an overview, Energy Sustainable Dev 8(1): 5-24.

[35] Celik AN, Muneer T, Clarke P (2007) An investigation into micro wind energy systems for their utilization in urban areas and their life cycle assessment. J Power Energy 221(A8):1107-1117.

[36] Chataignere A, Le Boulch D (2003) Wind turbine systems. In: ECLIPSE-Environmental and Ecological Life Cycle Inventories for Present and Future Power Systems in Europe.

[37] Gürzenich D, Mathur J, Bansal N, Wagner HJ (1999) Cumulative energy demand for selected renewable energy technologies. Int J Life Cycle Assess 4(3):143-149.

[38] Nomura N, Inaba A, Tonooka Y, Akai M (2001) Life-cycle emission of oxidic gases from power-generation systems. Applied Energy 68(2):215-227.

[39] Schleisner L (2000) Life cycle assessment of a wind farm and related externalities. Renewable Energy 20(3):279-288.

[40] Turkulainen T (1998) Diploma Thesis, Lappeenranta University of Technology

[41] Kreith F, Norton P, Brown D (1990) A comparison of CO2 emissions from fossil and solar power plants in the United States. Energy 15(12):1181-1198.

[42] Lechón Y, d. l. Rúa C, Sáez R (2008) Life Cycle Environmental Impacts of Electricity Production by Solarthermal Power Plants in Spain. J Sol Energy Eng 130(2):0210121-0210127.

[43] Lenzen M (1999) Greenhouse gas analysis of solar-thermal electricity generation. Solar Energy 65(6):353-368.

[44] Alsema EA (2000) Energy pay-back time and CO2 emissions of PV systems. Prog Photovoltaics 8(1):17-25.

[45] Frankl P, Corrado A, Lombardelli S (2003) Photovoltaic system, Interna Photovoltaic system In: ECLIPSE-Environmental and Ecological Life Cycle Inventories for Present and Future Power Systems in Europe. <http://62.94.11.35/eclipse_eu/pubres_guide.html >

[46] Ito M, Kato K, Komoto K, Kichimi T, Kurokawa K (2008) A comparative study on cost and life-cycle analysis for 100 MW very large-scale PV (VLS-PV) systems in deserts using m-Si, a-Si, CdTe, and CIS modules. Prog Photovoltaics 16(1):17-30.

[47] Kannan R, Leong, KC, Osman R, Ho HK, Tso CP (2006) Life cycle assessment study of solar PV systems: An example of a 2.7 kW(p) distributed solar PV system in Singapore. Solar Energy 80(5):555-563.

[48] Mathur J, Bansal NK, Wagner HJ (2002) Energy and environmental correlation for renewable energy systems in India. Energy Sources 24(1):19-26

[49] Muneer T, Younes S, Lambert N, Kubie J (2006) Life cycle assessment of a medium-sized photovoltaic facility at a high latitude location. Proceedings of the Institution of Mechanical Engineers Part a. J Power Energy 220(A6):517-524

[50] Prakash R, Bansal NK (1995) Energy analysis of solar photovoltaic module production in India. Energy Source 17(6):605-613.

Development and Application of a LCA Model for Coal Conversion Products (Coal to Y)

Christian Nissing, Loïc Coënt and Nathalie Girault

Abstract TOTAL Gas & Power launched a series of development studies in order to investigate the potential for coal conversion projects (coal to Y) for the production of fuels and primary products for the petrochemical industry. A crucial role is played by the aspects of carbon capture and storage (CCS). Based on these studies, a methodology and a model for life cycle analysis (LCA) were developed in order to understand the environmental impacts associated with Coal to Y conversion routes, especially regarding GHG emissions, water consumption, and energy efficiency. The model was designed around the need for adaptability to a) the geographic location of the coal mine and the coal to Y conversion plant, and b) the final products (e.g. methanol, DME, SNG and FT diesel) and their respective markets. By applying the model to a potential coal to methanol application by utilising original data and in-house expert advice, first results were generated, giving valuable insights especially into the critical elements of the CO_2 management system. The developed LCA model is a powerful tool that can assist in analysing clean coal studies.

1 Introduction

1.1 Background and aim

Against the background of a growing global energy demand, TOTAL is thriving to diversify its product portfolio as well as its production means in order to meet this demand in a sustainable manner. Within this context, detailed engineering studies have been carried out by the chemical processes department of TOTAL Gas & Power on 5 different coal conversion routes (coal to Y) for fuels and primary materials for the petrochemical industry, including carbon capture and sequestration (CCS) options.

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Based on these studies, it is the aim of this paper to develop a life cycle assessment (LCA) of the concerned products, focused on the aspects of energy consumption, greenhouse gas emissions (GHG) and water consumption. A special emphasis will be on the development of a LCA model that is adaptable in terms of geographic location of final coal conversion product.
The results of the LCAs will help TOTAL to understand the environmental impacts associated with these production routes.

1.2 Approach

In order to achieve the aim of this paper, a number of subsequent elements will be developed. Firstly, a literature review of the main areas of interest will be presented. Secondly, a comparative LCA will be developed, including a description of the applied methodology, the scope and goal definition, as well as the generation of results and its interpretation based on a previously established data inventory. The methanol case will be detailed and put into geographic perspective. Finally, a number of conclusions will be formulated and an outlook for future work will be given.

2 Literature review

2.1 Life cycle assessment (LCA) theory

The LCA method is defined as the assessment of the environmental impact of a given product or service throughout its entire lifespan. This method is formalised through ISO norm 14040 [1]. It includes four compulsory steps, consisting of the goal and scope definition, the life cycle inventory (LCI), the impact evaluation, and its interpretation. These steps can be complemented by two optional steps, which are the sensitivity analysis and the critical review.
A number of software tools have been developed in order to assist in the performance of an LCA. The Simapro® software package has been utilised to perform the LCA presented in this paper.

2.2 LCA and its interest for TOTAL

The interest of the LCA methodology for TOTAL is manifold:

- Firstly, it can generate inputs for internal information and analysis.
- Secondly, it can provide a scientific and objective basis for external communication purposes.
- Thirdly, it is an integral part of the TOTAL EcoSolution label, consisting in a set of stringent criteria a product or service has to meet in order to be considered as having a positive environmental performance [2].
- Fourthly, it is of special interest to TOTAL's petrochemical branch, were an emphasis is laid upon the end-of-life management of plastic products [3].
- Lastly, for TOTAL's Refining & Marketing branch, it is a compulsory approach for the calculation of GHG reduction rates of biofuels [4].

2.3 Coal and the environment

Coal mining has been performed worldwide throughout history and continues to be an important economic activity. Today, coal is the largest primary source of energy used for the generation of electricity worldwide[5].
The use of coal is coupled to major environmental impacts, which are i.a. the release of carbon dioxide, a GHG which causes climate change and global warming [6], and the impact of water use on flows of rivers and consequential impact on other land-uses[7].
In order to reduce these environmental impacts, there is a trend among coal dependent industry to operate according to the highest environmental standards, which can be formalised through ISO 14001 certification [8]. Technical options for the mitigation of GHG emissions include energy efficiency measures and CCS (see section 2.5).

2.4 Coal conversion routes (CTY)

Next to coal combustion for electricity generation, coal can be converted via gasification, resulting in a hydrogen rich synthesis gas (syngas). The obtained syngas can in turn be converted to a series of products, ranging from fuels to input

feeds for the petrochemical industry. This type of coal conversion is commonly referred to as Coal to Y or CTY conversion. [10]
A CTY conversion plant is generally made up of three distinct sections, namely the coal gasification section, the syngas conditioning section, and the product synthesis section. Acid gas removal units (AGRs) are located in the syngas conditioning section, used i.a. for capturing the CO_2 stream. TOTAL investigated the potential of CTY applications. Main results will be presented in section 3.1.1.

2.5 Carbon capture and storage (CCS)

CCS is a GHG emissions reduction option which consists in capturing CO_2 at its emission source and its transportation to and injection into a suitable geological structure [9].
It includes three distinct steps:

- CO_2 capture: There are three generic capture processes for coal power plants, namely post-combustion, pre-combustion and oxy-fuel combustion capture. Similar applications exist for production plants such as cement kilns, alloy smelters, or CTY plants.
- Compression and transportation: Once separated, the CO_2 is compressed, and transported to a suitable geological storage site, either by truck, train, barge or pipeline.
- Storage: CO_2 is injected into a suitable geological storage structure, such as saline aquifers, depleted oil & gas fields, or used for enhanced oil recovery (EOR).

A good overview on LCAs performed on CCS applications is given by the IEAGHG [10].

3 Comparative LCA on coal conversion products

3.1 Scope definition

Based on the aim of this paper as given in section 1.1, the purpose of this section is to give the scope of the LCA, as well as related information such as its limitations, the functional unit, allocation rules chosen, and the applied impact evaluation methods.

3.1.1 Underlying TOTAL internal study

TOTAL investigated the potential for 5 coal conversion projects (coal to Y) for the production of fuels and primary products for the petrochemical industry [10].

Tab. 1: CTY products, LHVs and main applications (sources: [11], [12])

CTY products		LHV [MJ/kg]	Examples of application
MET	Methanol	19.9	Petrochemical base for MTO process Motor fuel
DDME	Dimethyl ether, direct and indirect route	28.4	Motor fuel (LPG blending required) Petrochemical base for DTO process
IDME			
FTD	Fischer-Tropsch diesel	44.0	Motor fuel (Standard diesel blending required)
SNG	Synthetic Natural Gas	50.0	Stationary heating applications Power generation

As given in Tab. 1, each product has a number of different final applications, possibly requiring additional downstream conversion and conditioning steps.
For each coal conversion route, a plant design was developed, considering a grass root complex able to operate in a standalone mode with only coal, raw water and start-up power available at plant boundaries. Next to the main product train, i.a. utilities and power block are included to the battery limits. The plant was set to a generic location in North America. [10]
Each conversion route has the same input feed, namely 4 Mt/a of coal. Coal requirements for utilities and power block have to be added to this amount.
The primary aim of the development study was to perform a comparison of product output, utility requirements, and technical costs. The LCA presented in this paper is based on this study.

3.1.2 Limitations

The products on which the LCA will be performed are the products at the gate of the coal conversion plant. As can be seen in the example given for the MeOH product in Fig. 1, the considered system begins with the coal mining process, followed by its transportation to the coal conversion plant. The conversion unit is split into three parts, namely the coal gasification part, the MeOH synthesis part, and the CCS part. Power is supplied to the gasification unit by a power block (integrated gas combined cycle - IGCC), on which carbon capture is applied as well.

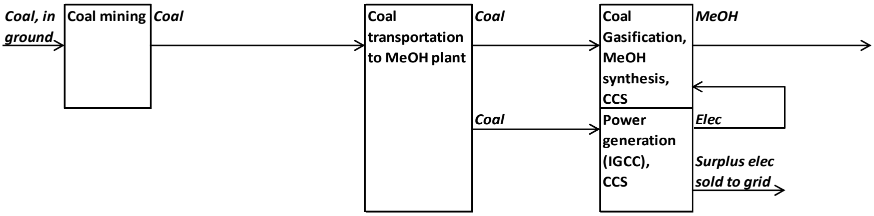

Fig. 1: System limits of Coal-to-MeOH route

It has been chosen to perform a 2nd degree LCA, excluding the infrastructure construction steps, such as mine development or conversion plant construction. The life-cycle impact of infrastructure construction has been considered as minimal, as it is depreciated over the plant lifetime (25 years).

3.1.3 Functional unit

The functional unit is defined as the quantified performance of a product system used as a reference unit in a LCA [13]. As the considered final products can all be utilised energetically (fuels), the functional unit is set to be "to produce 1 MJ of energy content in the final product".

3.1.4 Allocation rule

Inputs and outputs of a process which results in different products must be allocated between these products according to explicit rules (allocation rules). [13] An allocation problem appears for the FT route. A co-product of the FT diesel production is naphtha. As only the diesel product is considered in this paper, emissions and consumptions related to the naphtha product are subtracted according to its energy content (application of energy allocation rule).

3.1.5 Impact evaluation methods

For this paper, the first impact category considered is global warming (midpoint indicator - Simapro IPCC method) [6]. It is quantified by the corresponding amount of GHG emissions, expressed by the indicator "mass of CO_2 equivalent" (gCO_2e). Next to carbon dioxide, other GHG such as methane and nitrous oxide are considered by this indicator.
Two others impacts are evaluated: fresh water and primary energy consumption.

Fresh water consumption takes into consideration water needs along the life-cycle chain. It is expressed in "mass of fresh water used". Fresh water consumption is of specific interest, as it can lead to depletion of water resources and change in flows of rivers (see section 2.3). Primary energy consumption sums up all fossil and non-fossil energy consumptions over the life-cycle chain. The indicator used is "primary energy equivalent used" (MJ-e) [14].

3.2 Life cycle inventory (LCI)

Based on the previously defined scope, the data collected for the different life-cycle steps will be presented in the following sections.

3.2.1 Mining

In order to extract coal from the ground, a mining step is required. For this LCA, an average North-American underground mine has been considered [15]. Electricity, heat, and diesel requirements for mining operation are included. A specific coal type has been considered for the internal study [10].

3.2.2 Transportation

As the coal mine and the production plant are not located on the same site, transportation of the coal to the conversion plant is required. It is assumed that the distance between mine and process plant is 100 km. The transport system used is freight train transportation, assuming average diesel and electricity consumption [15].

3.2.3 Conversion process, utilities and power block

All GHG emissions and fresh water requirements were taken into account. All energy requirements for the conversion process and utilities are assumed to be covered by the power block [10].
Due to the relatively low number of start-ups (around once every 3 years), only emissions occurring during the regular operation phase were considered.
For the injection of electricity into the local power grid (see section 3.1.4), an average North-American electricity mix has been considered [15].

3.2.4 Carbon capture and storage (CCS)

Carbon capture is realised through AGR units downstream of the gasification trains. Overall CO_2 capture rates range from 84% for FT diesel to 99% for SNG. Energy needs for carbon capture and compression are included in the conversion process, utilities and power block. The CO_2 injection site is located approx. 150 km from the conversion plant. It has been assumed that no further energy requirements for transportation and injection were required. [10]

3.3 Impact evaluation

As can be seen in Figure 2, FT diesel generates the highest amount of GHG emissions, amounting to approx. 180g CO_2e/MJ, excl. CCS and grid injection, and a net of approx. 40 gCO_2e/MJ. Net GHG emissions for FT diesel are thus twice as high as for DDME, the second highest value. SNG has the lowest net level of GHG emissions.

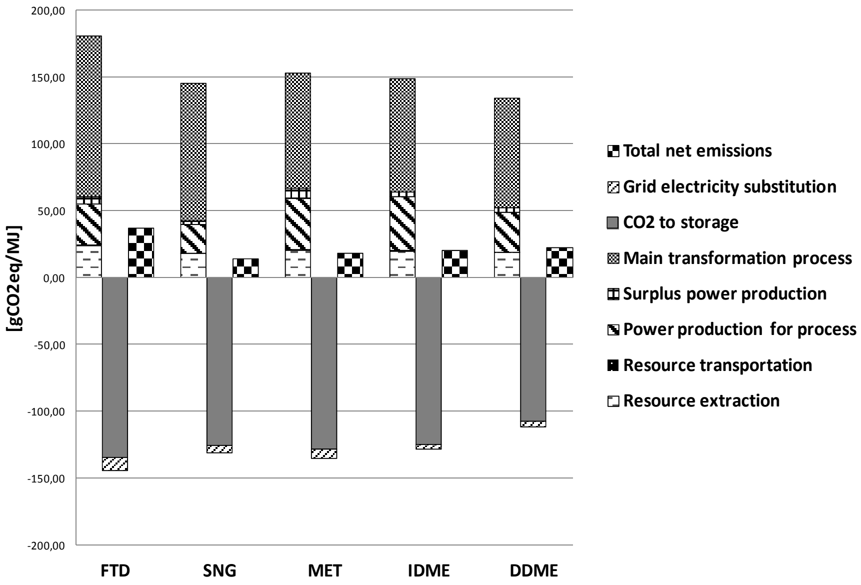

Fig. 2: GHG emissions for 5 CTY products

Surplus electricity is accounted for on the emissions side, but also on the credits side via the substitution of grid electricity.

Furthermore, emissions related to resource extraction are directly dependent on coal needs for the conversion process and IGCC. The efficiency of the conversion process and related power needs is the aspect displaying the highest variability between routes. Finally, the emissions related to transportation are negligible.
As shown in Tab. 2, FT diesel production has the highest consumption of primary energy, 1.78 MJ-e/MJ, but also the lowest consumption of fresh water, 0.34 kg/MJ. The four other routes have similar fresh water requirements. Fresh water consumption is almost entirely related to the process. The product with the lowest energy requirements is SNG, with 1.16 MJ-e/MJ.

Tab. 2: Fresh water and primary energy consumption for 5 CTY products

	FTD	SNG	MET	IDME	DDME
Water [kg/MJ]	0.34	0.50	0.52	0.50	0.54
Energy [MJ-e/MJ]	1.78	1.16	1.52	1.48	1.27

By comparing Figure 2 and Tab. 2, there seems to be a correlation between energy consumption and GHG emissions. This point needs to be substantiated statistically.

3.4 Interpretation of results

Be performing a high level analysis for the most environmentally sound product, a preliminary ranking system has been designed (see Tab. 3). For each impact category, values have been normalised by dividing it by the highest value for each category (the highest normalised value being 1). By adding up the obtained values for each product, combined impact values are obtained. At this stage, no weighting factors have been included.

Tab. 3: Preliminary ranking system

	FTD	SNG	MET	IDME	DDME
GHG emissions	1.00	0.39	0.49	0.56	0.61
Fresh water consumption	0.63	0.92	0.96	0.94	1.00
Primary energy consumption	1.00	0.65	0.85	0.83	0.71
Sum	2.63	1.96	2.30	2.33	2.32

Results show that the most environmentally friendly product is SNG, whereas the least environmentally sound product is FT diesel according to this approach.In a future step, a more detailed analysis can be developed, including weighting factors according to the attributed importance of each impact category.

3.5 Revisiting the methanol case

Methanol is an interesting case, as it is suited for a number of downstream processes such as methanol-to-olefins (MTO) conversion. Its case is thus reviewed in terms of its environmental performance (only GHG emissions) in different world regions, as well as benchmarked against other fossil routes.
As can be seen in Fig. 3, three coal-to-methanol scenarios are developed based on TOTAL internal data, respectively located in Asia, North-America and Europe. Furthermore, three scenarios are developed based on CONCAWE data [12], respectively representing a coal-to-methanol, an oil-to-gasoline and an oil-to-diesel product. All three products are produced for the European market. For the gasoline and diesel scenarios, the LCA system has been limited to the product at the refinery gate. No final combustion takes place.

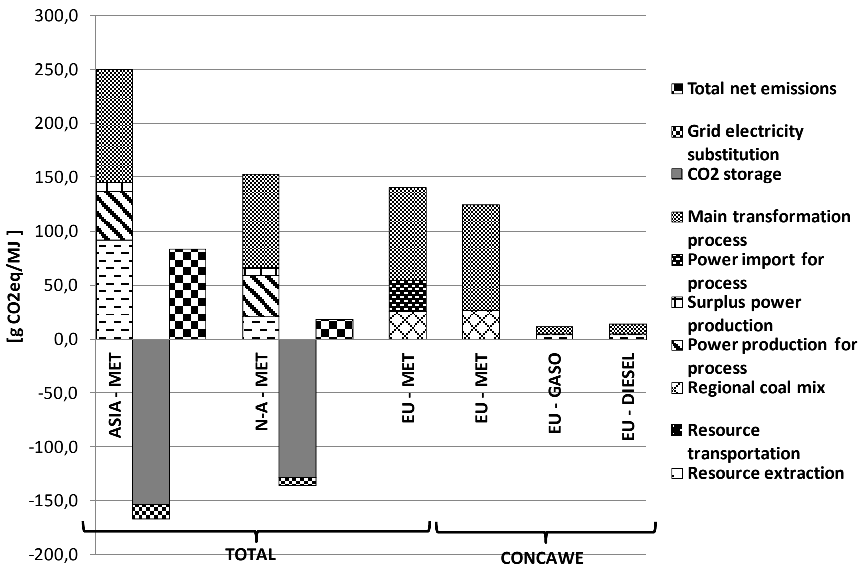

Fig. 3: Methanol from coal vs. conventional fuels

As can be seen in Fig. 3, the main difference in the first two scenarios lies in emissions related to mining activities, accounting for approx. 90gCO_2e/MJ for the

Asian case, and only about 15g CO_2e/MJ for the North-American case. This is mainly due to the difference in meeting energy demands: in Asia, needs are met via low-efficiency pulverised coal boilers, whereas in North-America, needs are met via diesel engines and imports from the electricity grid.
The second and third scenarios show that total GHG emissions are comparable for the coal-to-methanol life-cycle in North-America and Europe, although power needs for CO_2 compression have been subtracted for the European case (as no CCS). The third and fourth scenarios were developed in order calibrate the TOTAL model with the CONCAWE study.
Finally, the methanol scenarios are compared with the diesel and gasoline scenarios. It can be stated that in order to remain competitive with conventional fuels from oil on a GHG emission basis, the methanol must be produced in either North-America or Europe, and CCS is a necessary prerequisite. If such a plant was to be realised in Asia, North-American or European mining procedures and standards would have to be applied.

4 Conclusion and outlook

The aim of this paper was to understand the environmental impacts associated with coal conversion products (CTY), namely GHG emissions, as well as fresh water and energy consumption. LCA has the potential as a method to produce results in order to achieve this aim.
It was highlighted that there is a trend in coal dependent industry towards the integration of environmentally sound technology, such as energy efficiency measures and CCS.
A comparative LCA based on ISO 14040 was presented. The impact evaluation showed that FT diesel has the highest GHG emissions, while having the lowest fresh water consumption. By applying a normalisation approach incl. the three targeted impact categories, SNG showed the best environmental performance.
The methanol case was revisited by relocating its life-cycle to different world regions. Results showed that, in order to remain competitive with gasoline or diesel from oil, CCS technology is a prerequisite. Furthermore, European or North-American mining standards need to be applied.
In order to refine the obtained results, further material streams need to be looked at, i. a. sulphur and heavy metal emissions. The integration of weighting factors according to the importance of the different environmental impacts will lead to a more informed ranking system for CTY products.

References

[1] International Organisation for Standardisation (2006) Environmental management-Life cycle assessment-Principles and framework. ISO, Geneva
[2] TOTAL (2009)TOTAL Eco-solutions Guidelines Version 1
[3] TOTAL (2010) Petrochemicals, Plastics end of life management.Topics, No. 14
[4] European Parliament and Council, Directive 2009/29/EC on the promotion of the use of energy from renewable sources.Official Journal of the European Union Vol. 52 (2009)
[5] International Energy Agency Greenhouse Gas R&D Program, <http://www.ieaghg.org/> (accessed 30.03.2011)
[6] Le Treut H, Somerville R, Cubasch U, Ding Y, Mauritzen C, Mokssit A, Peterson T, Prather M (2007)Climate Change: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.Cambridge University Press, Cambridge
[7] MangenaSJ, BrentAC (2006) Application of a Life Cycle Impact Assessment framework to evaluate and compare environmental performances with economic values of supplied coal products.J Clean Prod 14:1071-1084
[8] International Organisation for Standardisation (2004)ISO 14001-Environmental management. ISO, Geneva
[9] <http://www.globalccsinstitute.com/> (accessed 30.03.2011)
[10] IEAGHG (2010)Environmental evaluation of CCS using life cycle assessment (LCA). IEA
[11] TOTAL (2009)Coal-to-Y. Final report
[12] Edwards R, LarivéJ, Mahieu V,RouveirollesP (2007)Well-to-Wheels analysis of future automotive fuels and power trains in the European context -Well-To-Tank Report Version 2c
[13] International Organisation for Standardisation (2006)ISO 14044: Environmental management-Life cycle assessment-Requirements and guidelines.ISO, Geneva
[14] FrischknechtR, Jungbluth N et al (2000) Implementation of Life Cycle Impact Assessment Methods.Final report ecoinvent
[15] Ecoinvent Centre (2007) Eco-invent data v2.0.Final report eco-invent No 1-25

PART VIII:
LCM in the Electronics and ICT Sectors

European LCA Standardisation of ICT: Equipment, Networks, and Services

Anders S.G. Andrae

Abstract Evidentially the lack of harmonisation and transparency for life cycle assessment (LCA) is a cause of concern in the information and communication technologies (ICT) sector. Building mainly on ISO 14040/44, several important stakeholders (network operators together with manufacturers of network equipment, end–user equipment, and parts) have jointly agreed on such core elements as system boundaries, recommended/optional life cycle phases, unit processes, functional units, life time, allocation methods, data quality evaluation, and cut–off rules. To show the added value of the performed work in relation to ISO 14040/44, the new standardisation approach will also be applied to common ICT equipment and networks. LCAs done according to the new approach have consistent result presentations. This is shown for some common ICT equipment, ICT networks and ICT aervices. Consistent result presentations lead to convenient quality comparisons of different ICT LCAs.

1 Introduction

Global communication based on ICT is rapidly increasing. The EU project "Energy Aware Radio and Networking technologies" (EARTH) has predicted that the data traffic between 2010 to 2020 will rise by something like 1,700% and the number of ICT equipment in use by around 100%. Both globally and in a European context major standardisation bodies like International Telecommunication Union (ITU–T) and European Telecommunications Standards Institute (ETSI) are pioneers. Digital Enhanced Cordless Telecommunications (DECT) for cordless phones and Global System for Mobile Communications (GSM) and 3rd Generation Partnership Project (3GPP) Long Term Evolution (LTE) networks are but a few examples from ETSI. In recent years carbon footprint and life cycle assessment (LCA) calculations have emerged as an important topic within the Information Communication Technology (ICT) industry [1]. However, evidently the lack of harmonisation and transparency in

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LCA methods used is a cause of concern in the ICT sector [2]. Occasionally it is difficult to know the quality of studies and judge the degree of "greenwashability". It is easy to agree with the Product Carbon Footprint (PCF) World Summit that it is high time to build credibility into carbon footprint information [3]. It would be valuable to understand, e.g., the precision with which the carbon emission values of a *Google search* have been calculated [4]. Creating a level playing field for LCA is logic due to the increased usage and importance of ICT and the demands of environmental impact estimations are increasing. The general LCA standards ISO 14040/44 led to much improvement but there is still room for industrial sector guidelines and standardisation, e.g., recently discussed by the European food and drink product industry [5]. Even if a third–party review ensures compliance with ISO 14040/44, the proposed ETSI approach will harmonise the whole ICT industry further, especially for system boundaries. A further advantage with industry specific standardisation is that it will be quickly evident why two studies disagree with one another. Japanese authors have presented papers about an initiative on "standardisation" of eco–efficiency evaluation of ICT [6–7]. Another effort is the Japanese and Korean product category rules (PCR) for electronics [8] and in Sweden the environmental product declaration (EPD) system has published several so–called *product specific requirements* for ICT systems [9]. In the US the International Electronics Manufacturing Initiative (iNEMI) is developing a streamlined LCA tool for ICT equipment [10]. However, the present ETSI standardisation proposal will have a higher granularity and a more systematic consideration for the number of minimum recommended unit processes that should be included.

2 Materials and methods – unique features of the ETSI standard

In order to handle the difficulties arising from different LCA methods applied to ICT, several efforts were started by ETSI and ITU with finish in 2011 or 2012. This paper will describe the core elements of ETSIs LCA standard for ICT.
Building entirely on ISO 14040/44, several important stakeholders (ICT Network Operators together with manufacturers of ICT network equipment, end–user ICT equipment, and parts such as batteries) have jointly agreed on such core elements as basic:

- System boundaries, including parts such as integrated circuits (ICs)
- Recommended/optional life cycle phases/unit processes
    a) Recommended raw materials and parts

b) Recommended resources and emissions

- Functional units
- Life time definition
- Allocation methods
- Data quality evaluation
- Cut–off rules.

The ETSI standard outlines four life cycle phases: raw material acquisition, production, use, and end of life treatment.

Raw material acquisition starts with the extraction of natural resources and ends with transports to part production, production starts with part production and ends with transport of ICT equipment to use, use starts with the installation of the ICT equipment and ends with the de–installation, end of life treatment starts with transport of the de–installed ICT equipment and ends with the waste treatment.

ETSIs general approach will cover "all" kinds of ICT equipment, “all” relevant environmental impact categories as well as ICT services. Consequential LCA is however out of scope at the moment. To show the added value of the performed work in relation to ISO 14040/44, the ETSI way is applied to common ICT equipment, ICT networks, ICT services.

3 Case studies

Here follows a few published LCA results of common and well–known ICT e, ICT network, and ICT services. The aim is to show the random and divergent presentations resulting from the lack of industry specific standardisation. Important points are whether the bill of materials (BOMs), system boundaries, functional units, allocations, lifetimes, uncertainty analyses, and data quality evaluations are presented clearly.

3.1 ICT equipment - laptops

This ICT equipment is almost everyone's property. Table 1 shows the CO_2/CO_2e results for three rather well–documented analyses of the laptop life cycle. IVF [11] used "lifetime" 5.6 years and Deng et al. [12] 2.9 years, and O'Donnell and Stutz [13] 4 years.

Tab. 1: Example of LCA results for two laptops

	Greenhouse gases (kg) in GWP100 [11]	CO_2 (kg) [12]	kg CO_2e [13] in China	kg CO_2e [13] in the US	kg CO_2e [13] in the EU
TOTAL life cycle	**251**	**428**	**370**	**355**	**320**
Material	71				
Material production		85			
Manufacturing	9				
Manufacturing			160	160	160
Semiconductor		61			
Circuit board		3			
Silicon wafer		5			
LCD manufacturing/ assembly		6			
Computer assembly		47			
Remaining value		62			
Distribution	10				
Transport to customer				55	40
Use	162	159	240	170	150
Disposal	7				
Recycling	-8		-30	-30	-30

Table 1 reveals considerable differences in notations and values, especially for the raw material acquisition and production phases. Some differences would of course be "allowed" by standardisation, but others would be out of line.

3.2 Fibre to the home (FTTH) networks

Below in Table 2 original FTTH LCA results are shown. The quality of these cannot easily be compared and this is a major and common problem for ICT LCAs. The FTTH studies were presented per user [14] and for 10,000 homes [15].

Tab. 2: Example of LCA results for two FTTH networks

	IPCC-Greenhouse effect (direct, 100 years) (g) 14]	Basic GWP100 result per year (ton) [15]
TOTAL life cycle	**581,248.93**	**1200**
Production of components in network devices		78
Cable and passive equipment production	42,965.95	
Active equipment production	50,036.78	
Transport	921.27	
Deployment (installation) of network devices		400
Passive fibre network deployment	48,0624.94	
End of Life	6,700.1	
Use of network		690
Network power consumption *for 1 year*	35,166.16	
1 year savings associated to services	87,042.35	
Total savings *for 1 year*	51,876.19	
Others		59

These approaches are very different which makes it relevant to show the benefit of standardisation.

3.3 ICT Services – business meeting with video conference

The last odd decade it has become popular to compare "usual" services with ICT services. Video conference, e–newspaper, e–book, e–mail, and the list goes on, but here the focus is on the CO_2 emissions caused by the ICT s itself and not by the savings made possible by using it. Below the example of having a business meeting using video conference solution is analysed as several LCA studies were found. The studies here are from Östermark and Eriksson [16], Nippon Telegraph and Telephone Corporation (NTT) [17], Ericsson [18], and Global e–Sustainability Initiative (GeSI) [19], and the common denominator is video conference solutions. Östermark and Eriksson [15] analysed a 3.5 hour meeting and two different use scenarios (Table 3). NTT [17] also presented two scenarios; once a week for one hour (48 times/year) and 240 days per year and 8 hours/day for meetings between Tokyo and Yokohama. Ericsson [18] proposed an average video conference room anywhere in the world per hour with the hypothesis 240 days/year and 4 hours /day. GeSI [19] presented the CO_2e of Cisco's TelePresence solution for video conferencing. Here 25 hours usage/week and 10 years lifetime were assumed.

Tab. 3: Example of LCA results for three video conference LCAs

	g CO_2e/ meeting [16], little usage/ much standby	g CO_2e/ meeting [16], much usage/ less standby	Kg CO_2/ year [17], 48 days/ year	Kg CO_2/ year [17], 240 days/ year	kg CO_2e/ hour [18]	kg CO_2e/ hour, [19]
TOTAL life cycle	**7,000**	**1,000**	**120**	**500**	**2.0**	**7.5**
Raw material extraction, production, disposal						3.2
Production	4,500	750	105	100		
Use	2,500	250	15	400		4.3
One video conference room					0.7	
Smartphone (voice)					0.3	
Laptop with high speed packet access connection					0.7	
Laptop in office local area networks					0.2	

It is rather challenging to compare the above LCAs.

4 Results and discussions

Here follows a discussion on the extent to which the case studies in Chapter 3 would benefit from proposed ETSI standardisation which has set up rules for basic system boundary and functional unit.

4.1 ICT equipment -laptops

There is insufficient information about the core elements (Section 2.1) in the studies [11–13] for them to be compliant with the proposed standardisation approach. It is unknown which sub–unit processes for IC production that were included in the IVF and O'Connell and Stutz studies. Nevertheless, the result presentations in [11–13] could easily be changed (Figure 1) to represent the ETSI basic functional unit *per one year of typical ICT equipment use.*

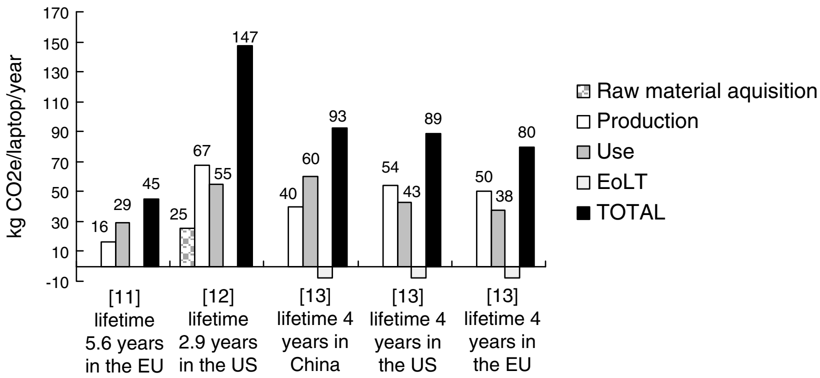

Fig. 1: Standardised result for ICT Equipment: laptops

The proposed standard would reveal that the quality of these studies is not comparable. Especially the IVF study [11] has led to much debate, which is good for the progress of electronics LCA. For the laptops [11–13] the lifetime and the production assumptions are crucial for the conclusions.

Deng et al [12] make a big point of using so called economic input output (EIO) models/methods for LCA (EIOLCA) as a means to avoid cut–offs and increase comprehensiveness. All in all, as it stands today, this method is questionable as many "corrections" have to be done between the US EIO database and studies performed outside the US. Macro level hot spot findings in the US have the best chance of benefiting from the EIOLCA approach. EIOLCA could possibly be used in the US for ETSI standard Part Category "black box modules".

The process–sum LCA (PLCA), which the ETSI standard is targeting, does not "forbid" the use of EIOLCA, especially for US conditions, but as the data quality requirements will be rather strict for specific studies, the roughness of the EIOLCA method would not be motivated in many cases. In the next years many improvements in LCI data availability is expected which narrow the gap (comprehensiveness) between EIOLCA and PLCA. The ETSI standard will require a specific PLCA approach including end–of–life treatment. Therefore O'Donnell and Stutz [13] would likely be close to ETSI compliance. Furthermore, once step up in the LCA standardisation hierarchy, the Japanese PCR for notebook computers [8] are stricter and narrower than the general ICT Equipment LCA requirements from ETSI, as the former targets a certain Japanese environmental label (EcoLeaf™) for a specific ICT Equipment.

4.2 ICT networks – FTTH networks

The ETSI standard for ICT Networks will require that the result is presented per End–user, Home, Access, Data transport, and Data centres, should all these ICT equipment be within the scope of the specific LCA study.
There is insufficient information about the core elements (Section 2.1) [14–15] for them to be compliant with the proposed standardisation approach. Below in Figure 2 the original result presentations have been changed to represent the basic functional unit *per one year of typical ICT Network use*

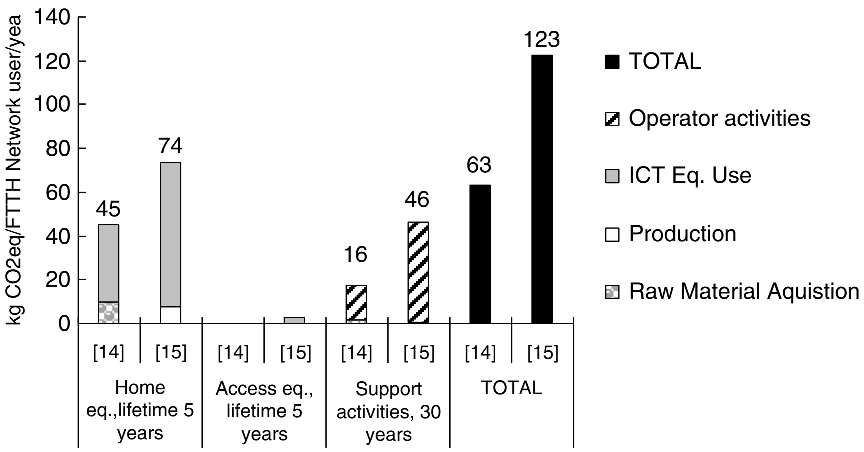

Fig. 2: Standardised result for ICT networks: FTTH

The strength of FTTH Council Europe [14] is the details for the deployment of the ICT network. However, in [14] ICT equipment production was only assumed to consist of raw materials acquisition and no division was made between access and end–user equipment. In [15] no division between part production and raw material acquisition was done. No cut–off discussion could be found in either paper.

4.3 ICT services – business meetings

There is insufficient information about the core elements (Section 2.1) in [16–19] for compliance with the proposed standardisation approach. The result presentations could not easily be changed to represent *per one year of typical ICT service use,* even though some use assumptions were given. Instead *"Per meeting per hour"* is used below in Figure 3.

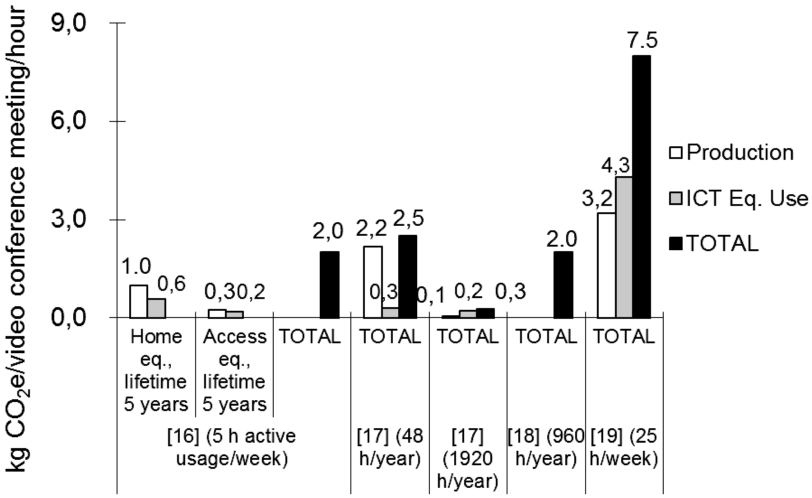

Fig. 3: Standardised result for ICT Services: Video conferences

Östermark and Eriksson [16] is peer–reviewed and rather well–documented regarding high–level assumptions. However, for parts production there are too few details. Naturally different studies would give different values, but without enough account of the system boundaries and other core elements for equipment, the reasonableness assessment is challenging. The number of users per video conference meeting is arbitrary and the studies do not give sufficient accounts for the allocation procedures, e.g., allocation of ICT networks to ICT service. Obviously there could be many valid reasons (e.g., industry secrets) for leaving out details when publishing LCA results.

Anyway, the proposed standardisation will clearly make the LCA process more streamlined and occasionally faster as the practitioner gets an "exact" guide what is recommended to include if applicable, and what is optional. LCAs done according to the proposed ETSI approach will have consistent result presentations. However, comparisons of absolute/relative values between LCA studies performed and presented by different individuals using different software/databases are beyond the scope of the new ETSI approach, as such comparisons would require that the assumptions and context of each study are exactly equivalent. ETSIs approach could especially benefit small and medium sized ICT companies who might not have the knowledge of multinationals in conducting LCA.

5 Conclusions

The new ETSI LCA standard for ICT means consistent result presentations which lead to convenient quality comparisons and re–usability of different ICT LCAs. Compared with ISO 14040/44, the ETSI LCA standard will *oblige* the practitioner to be more accurate when performing/reporting the analysis for ICT (equipment, networks, services).

6 Recommendations and perspectives

The focus of the proposed ETSI standard is more on ICT Equipment, ICT Networks and ICT Services than on Parts. The Part manufacturers (Batteries, Cables, Electro–mechanics, ICs, etc.) are encouraged to agree on "their" system boundaries. A first suggestion of sub–unit processes for Parts will be listed in the ETSI standard. Raw Material manufacturers and other stakeholders are also encouraged to agree on the core elements (Section 2.1) within their industry fora.

References

[1] Prasad R, Ohmori S, Aeimuni D. (2010) Towards green ICT. River Publishers

[2] Andrae ASG, Andersen O (2010) Life cycle assessments of consumer electronics — are they consistent? Int J Life Cycle Assess 15(8):827-836.

[3] <http://www.pcf-world-forum.org/wp-content/uploads/2011/03/5th-PCF-World-Summit-Programme-14032011.pdf> (accessed 13.04.2011)

[4] <http://www.timesonline.co.uk/tol/news/environment/article5488934.ece> (accessed 13.04.2011)

[5] Peacock N, De Camillis C, Pennington D et al (2011) Towards a harmonised framework methodology for the environmental assessment of food and drink products. Int J Life Cycle Assess 16(3):189-197.

[6] Matsuno Y (2006) The development of information communication technology (ICT) eco-efficiency evaluation guidelines. EcoBalance, Nov. 14–16, Tsukuba, Japan
[7] Matsuno Y, Takahashi KI, Tsuda M et al (2007) Eco-efficiency for information and communications technology (ICT): the state of knowledge in Japan. IEEE International Symposium on Electronics & the Environment, May 7–10, Orlando, FL
[8] http://www.cfp-japan.jp/english/gpl/product_category/electronics/ (accessed 13.04.2011)
[9] <http:www.environdec.com> (accessed 13.04.2011)
[10] Okrasinski T, Malian J. (2010) A framework work for estimating life cycle eco-impact for information and communication technology products. CARE Innovation, Vienna, Austria
[11] IVF (2007) European Commission DG TREN Preparatory studies for eco-design requirements of EuPs. IVF Report 07004 -Laptop used at home. Table 92,101
[12] Deng L, Babbitt CW, Williams ED (2011) Economic-balance hybrid LCA extended with uncertainty analysis: case study of laptop computer. J Cleaner Prod. Article in press. doi:10.1016/j.jclepro.2011.03.004
[13] O'Donnell S, Stutz M. (2010) Product carbon footprint (PCF) assessment of Dell laptop – results and recommendations. IEEE International Symposium on Sustainable Systems and Technology (ISSST), May 17–19, Arlington, VA
[14] FTTH Council Europe (2008) Developing a generic approach for FTTH solutions using LCA methodology. Methodological guide, Final version.
[15] Griffa G, Radice L, Bianco C et al (2010) Carbon footprint of next generation fixed networks. 32th International Telecommunications Energy Conference, INTELEC, June 6–10, Orlando, FL
[16] <http://www.cpm.chalmers.se/document/reports/99/1999_7a%20Del%201.pdf> (accessed 13.04.2011)
[17] <http://aimp.apec.org/Documents/2010/TEL/TEL42-DSG-WKSP2/10_tel42_dsg_wksp2_012.pdf> (accessed 13.04.2011)
[18] Guldbrandsen F, Malmodin J (2010) Life cycle assessment of virtual meeting solutions. 9th International Conference on EcoBalance, Nov. 9–12, Tokyo, Japan
[19] GeSi (2010) Evaluating the carbon–reducing impacts of ICT - an assessment methodology. Case Study 6: Telepresence system.

Product Carbon Footprint (PCF) Assessment of a Dell OptiPlex 780 Desktop – Results and Recommendations

Markus Stutz

Abstract Dell has adopted a strategy that takes into account the greenhouse gas (GHG) impacts of our products and our suppliers. By assessing the carbon footprint of our products, we are able to identify areas for improvement to reduce overall GHG emissions and also help customers do the same. We determined the carbon footprint of the OptiPlex 780 Mini Tower, a typical business desktop. The total carbon footprint has been studied for three regions (US, Europe, Australia). The differences in the three scenarios have been further assessed. The distribution between manufacturing and use has been studied to determine where the focus on environmental improvement needs to be, manufacturing or use. Further the key components in terms of impact in the manufacturing phase have been assessed.

1 Introduction

Dell recognizes that climate change is real and must be mitigated, and we support efforts to reduce global greenhouse gas (GHG) emissions to levels guided by evolving science. We are also committed to reducing GHG emissions beyond our own operations. To do this, we have adopted a strategy that takes into account the GHG impacts of our products and our suppliers. For products, we look at each stage of the product life cycle — from developing, designing and sourcing through manufacturing and operations, order fulfilment, customer use and product recovery. By assessing the carbon footprint of a desktop, we are able to identify areas for improvement to reduce overall GHG emissions and also help customers do the same.

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2 Calculating the carbon footprint of a desktop

In research conducted in 2010, Dell determined the carbon footprint of the OptiPlex 780 Mini Tower, a typical high-volume, mainstream business desktop that is representative of a range of similar desktop products. It is Energy Star® 5.0 compliant and EPEAT (electronic product environmental assessment tool) Gold registered [1].

Fig. 1: Dell OptiPlex 780 desktops; Mini Tower is at far left.

The carbon footprint of the desktop was assessed for three regions: the US, Europe and Australia. This was done to compare the impacts caused by different assembly and transport patterns and energy mixes. The GHG emissions were calculated according to ISO 14040 and ISO 14044, the two international standards governing the investigation and evaluation of the environmental impacts of a given product over its life cycle. The carbon footprint includes GHG emissions' contribution to global warming in kg of CO_2 equivalents (kg CO_2e).

The following life-cycle phases were taken into account:

Manufacturing — Includes the extraction, production and transport of raw materials, the manufacturing of components and subassemblies (including product packaging), and the final assembly of the desktop. The transport of the subassemblies (chassis, hard disc drive (HDD), optical disc drive (ODD), motherboard, cables, power supply unit and packaging) was taken into account as well. Energy consumption (electric power, fuels, thermal energy) for the different Dell final assembly sites (US, Poland and China) was also included.

Transport — Includes air, ocean and land transportation of the desktop and its packaging from the final assembly sites in the three regions to the end customer. Transport can be quite varied, depending on region, customization level and lead time.

Use — Lifetime of the desktop was estimated at 4 years. This is consistent with general business customer use models. To determine the energy consumption in use, the US Environmental Protection Agency's Energy Star® Typical Energy Consumption (TEC) method was used. This method focuses on the typical electricity consumed while in normal operation during a representative period of time and can be used to compare the energy performance of computers. The use phase was considered in each of the three regions (US, Europe and Australia), taking into account the respective grid mixes.

Recycling — It is common for desktops to be refurbished and/or reused at the end of the first customer use. For this study, however, it was assumed that the desktop was sent for recycling at the end of the first customer use. Per European recycling legislation (the Waste Electronic and Electrical Equipment Directive, or WEEE) and similar US electronics recycling requirements, we assumed 75 percent of the desktop is recycled, while the rest is incinerated to recover the energy contained. Transport to recycling as well as energy used in mechanical separation and shredding were taken into account.

3 Carbon footprint of the Dell OptiPlex 780 Mini Tower

The total carbon footprint of a Dell OptiPlex 780 Mini Tower is approximately 800kg CO_2e when used in the US, 720kg CO_2e when used in Europe and 1230kg CO_2e when used in Australia. The main reason for the differences between the three scenarios is the amount of emissions associated with the differing power generation modes in the three regions, although transports in the assembly and distribution chain also account for a significant part of the differences.

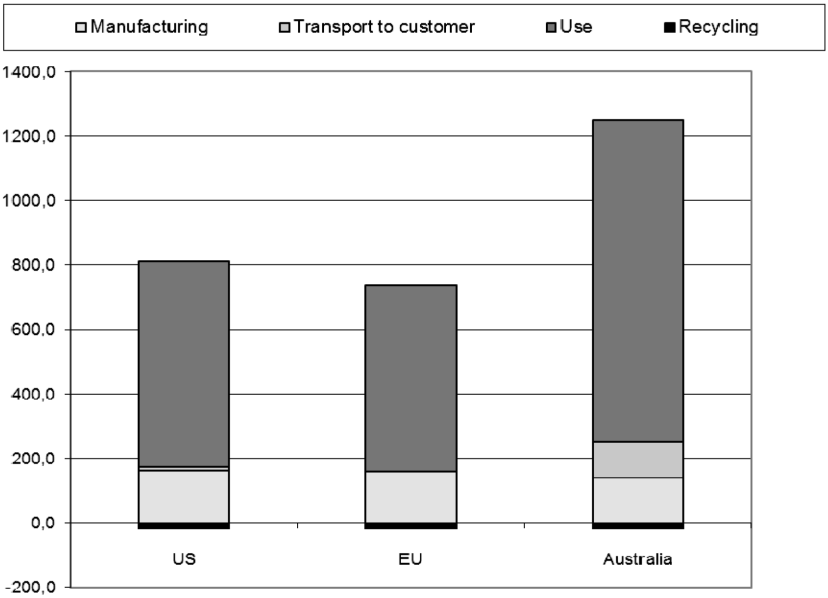

Fig. 2: Total product carbon footprint [kg CO_2e] of the OptiPlex 780 Mini Tower in the US, Europe and Australia

The GHG emissions from use (dark grey) account for approximately 80 per cent of the total life-cycle impact. This dominance of the use phase can also be observed in other electronic equipment. It demonstrates that our goal to design desktops and laptops to consume up to 25 per cent less energy by end of calendar year 2010 compared with systems offered in May 2008 is right on target. Additionally it makes the case that use of power management features, which put the desktop into sleep mode when not used, needs to be applied by users more uniformly. This is best achieved by leaving the power management in the factory-default setting.

Manufacturing (light grey), which includes component manufacturing, transport of components to assembly and assembly itself — has the next biggest impact, but it still represents only a fraction of the total GHG emissions (10 to 20 per cent, depending on the scenario). Only three subassemblies make up about 85 per cent of the total GHG emissions in manufacturing. These are, in order of importance, the motherboard, the chassis and the optical disc drive (ODD). The motherboard accounts for about 8 per cent of the desktop's total carbon footprint.

Transport to the final assembly and to the customer is not particularly relevant where it includes transport by ship or by truck. This is the case for the US and the European scenario, where subassemblies, with the exception of the hard disc drive (HDD) and the ODD, are transported by ship and truck to the regional final

assembly sites (Winston-Salem, NC, and Lodz, Poland, respectively). A different picture emerges in the Australian scenario, where the desktop is assembled in China and then transported by plane to the regional distribution centre. Air travel, given the same distance and same amount of transported goods, has a carbon footprint that is about 42 times higher than road travel and about 164 times higher than transport by ship. Regional assembly or transport by ship, if lead time allows, is therefore a very preferable option from a GHG emissions point of view.
As we assumed that 75 per cent of the desktop is recycled, a credit (or a negative impact) of approximately 20kg CO_2e resulted. This is the case where the recycled (secondary) material can be used directly to replace the primary material in new products, thereby avoiding all GHG emissions associated with primary production of the material.
The total product carbon footprint of the OptiPlex 780 MT is comparable to driving 2,700km in a Porsche Cayenne (assuming a CO_2 emission of 296g/km [2]). It is also comparable to drinking 560 litres of orange juice (assuming 360g CO_2e/250ml [3]). This is equivalent to each member of a family of four drinking 390 ml of orange juice every day for a year. These comparisons demonstrate that the GHG emissions over a four-year lifespan of the desktop are relatively modest.

4 Comparison with the carbon footprint of a laptop

Earlier in 2010 Dell published a carbon footprint study of a typical business laptop, the E6400 [4]. The laptop and the desktop are products designed with a basic set of overlapping functionalities (computing), but a larger set of differences due to the different requirements (e.g., mobility versus storage). The results of the two studies show that a laptop has an inherently lower carbon footprint due to the high efficiency in use.

5 What Dell is doing to lower the carbon footprint

By optimizing consumption of energy, we can reduce costs, shrink our carbon footprint and develop expertise that allows us to help our customers do the same.

Manufacturing — In 2008, we met our operational carbon neutrality goals for our global operations ahead of schedule. We committed in early 2009 to further reduce our worldwide facilities' GHG emissions by 40% by 2015. We require our primary suppliers to measure and publicly report their GHG emissions, and we ask them to set improvement goals of their own and set expectations for their suppliers.

Use — All Latitude laptops, Precision workstations and OptiPlex desktop can be configured for Energy Star® compliance and are among the most energy-efficient in the industry. In fiscal year 2010, we had more than 135 products registered for Electronic Product Environmental Assessment Tool (EPEAT). Dell implemented server-managed power management for customers worldwide to avoid 40,000 tons of carbon dioxide emissions between FY08 and FY12.

Recycling — Dell is committed to the environmentally responsible reuse and recycling of our products when our customers are finished with them. We are the first manufacturer to offer free computer recycling to consumers worldwide, and we have been providing responsible recycling services for more than a decade. We were also the first major computer manufacturer to ban the export of e-waste to developing nations.

References

[1] <http://www.epeat.net/ProductDisplay.aspx?return=pm&action=view&search=true&productid=2083&epeatcountryid=1> (accessed 18.02.2011)
[2] <http://www.porsche.com/germany/models/cayenne/cayenne/featuresandspecs/> (accessed 08.02.2010)
[3] <http://www.pcf-world-forum.org/2008/04/tesco-puts-carbon-reduction-label-on-20-products/> (accessed 08.02.2010)
[4] Carbon Footprint of a Typical Business Laptop From Dell <http://i.dell.com/sites/content/corporate/corp-comm/en/Documents/dell-laptop-carbon-footprint-whitepaper.pdf> (accessed 28.09.2010)

State of the Art in Life Cycle Assessment of Laptops and Remaining Challenges on the Component Level: The Case of Integrated Circuits

Ran Liu, Siddharth Prakash, Karsten Schischke, Lutz Stobbe

Abstract The current hype for product carbon footprints of IT, policy initiatives, such as the European ecodesign directive and several research activities recently lead to the publication of numerous LCA results of laptop computers. A comparison of the various studies unveils a broad variance among the results, which cannot be explained solely by technical differences: LCA for IT products still faces severe shortcomings and methodological uncertainties due to the complexity of the products, and assumptions to be made. This paper addresses the component level in the case of integrated circuits (ICs) to reveal the challenges in the data collection. The paper discusses the various approaches under consideration as reference units for IC-datasets, in order to contribute towards creating a harmonised reference unit of IC dataset.

1 Introduction

A comparison of the numerous LCA studies on laptops published in the recent past unveils a broad variance among the results, which cannot be explained solely by technical differences: LCA for IT products still faces severe shortcomings and methodological uncertainties due to the complexity of the products, and assumptions to be made.

Figure 1 illustrates the overall results of global warming potential of laptops. In this figure x-axis refers to the weight of laptops and y-axis shows the value of GWP in kg CO_2 equivalent related to the manufacturing stage. The size of balls represents the display size of laptops. Generally speaking, the GWP impact of laptops in their manufacturing stage varies from about 50 kg CO_2e to 450 kg CO_2e. The black marked balls demonstrate the GWP-value of Apple MacBook series from its environmental report [1]. The grey marked balls give picture of diverse sources of other studies. With one exemption of an Apple laptop with

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13.3" size, the increase of GWP values of Apple laptops generally shows a tendency to follow the increased display size and weight of laptops. The ecodesign preparatory studies lot 3 for desktop and laptop [2] published assessment results for a laptop with a weight of 2.8 kg and has almost the biggest display size of 15" among the laptops compared in this compilation. However, it causes almost the smallest GWP result of 80 kg CO_2e in laptops regarded. Furthermore, GWP values of laptop with display size of 12" analysed in the Ecoinvent database version 2.1 and 2.2 [3] range from about 250 kg CO_2e to 200 kg CO_2e. A recent study in 2009 on Product Carbon Footprint (PCF) assessment of a Dell Laptop [4] concluded 150 kg CO_2e in correlation with its display size of 14.1", while the EPIC-ICT project [5] reported greenhouse gas emissions of the manufacturing phase of 51 kg CO_2e, referring to another Dell laptop with a display size of 12". A presentation by PE International [6] showed that a laptop with 14" display size, with a weight comparable to an Apple laptop with a display size of 13.3", features a GWP value comparable to this Apple laptop. The GWP impact of a Fujitsu laptop awarded by Japan Ecoleaf labelling is reported with 96 kg CO_2e in the manufacturing stage [7], although its weight is similar to most laptops considered in this figure.

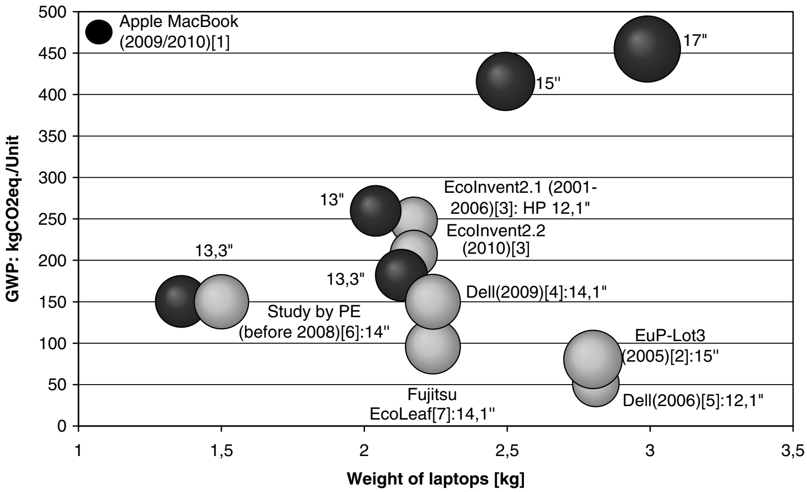

Fig. 1: Global warming potential (GWP) of laptops in the manufacturing stage

Compared to results published by other studies, GWP impact associated with the manufacturing is dramatically underestimated by the assessment under the ecodesign directive. The large range of GWP values varies between 50 and 450 kg CO2e which could not be further explained without detailed corresponding documents. Nevertheless, it is possible to conclude that the differences can trace back to the fact that configurations of products, generations of fabrication, and fabricating methods as well as the assumptions involved in the calculation are diverse. Another reason for the large spread of results is due to the databases applied and methodological determination in the balance, e.g. definition of system boundary, or cut-off criteria.

In this context, a research project, commissioned by the Federal Environment Agency of Germany (UBA), on creating a dataset on notebooks, was initiated in March 2010. This paper documents the initial challenges of this research project in terms of identifying a consistent methodological approach: This paper addresses the component level in the case of integrated circuits (ICs) to reveal the challenges in the data collection, since ICs are the basic components in all electronic applications, but also ICs are one of the main contributors to the global warming potential (GWP).

2 Integrated circuits

Integrated Circuits (IC), also called microchips, are electronic components with millions and billions of microstructure transistors. Production procedures encompass often hundreds of unit processes.

2.1 Global manufacturing processes

The production chain of ICs can be mainly divided into following three processing systems:

1) Wafer manufacturing: from extraction of quartz to production of high purity silicon wafer
2) Front-end processing: from silicon wafer to fabrication of finished wafer, i.e. integrated electrical structures on dies
3) Back-end processing: assembly, encapsulation of dies, chip packaging and testing

These three systems as well as the unit processes often occur in different countries, frequently even in different continents across the globe. In addition, it is often transported by means of air freight. The following Table 1 provides a rough overview on the distribution of production in the semiconductor sector, based on market insights and own estimates. Even within the back-end process, dies are frequently assembled at one location and are tested at other locations, as reported by [9].

Tab. 1: Silicon wafer, front-end and back-end production per region [8]

Region	Silicon wafer production [%]	Front-end processing [%]	Back-end processing [%]
Japan	66	23	10
Europe	12.5 (Germany)	8	-
USA	8.5	15	10
Korea	8.5	14	10
Singapore	4	9	15
Taiwan	-	23	15
China	-	8	15
Malaysia	-	-	15
Philippines	-	-	10

In addition, besides the energy and consumables consumption similar to other industry processes, semiconductor industry requires on the one hand high purity chemicals and gases for processing, on the other hand stringent infrastructure system, e.g. producing high ultrapure water in-house, clean working environment, controlled processing environment regarding temperature and humidity which are correlated with additional energy consumption.
Considering the technologies, the typical size of silicon wafers is nowadays 300mm (or less commonly 200mm). Regarding chips fabrication [10,11] analysed life cycle impacts of logic chips produced from 1995 through 2010 along with the technology node 350nm to 45nm, respectively. The technical development as well as the fabs generation influence directly the complexity of processes, and production efficiency along with energy efficiency.

2.2 Methodological challenges: data collection

Data collection is the fundamental step for environmental assessment and identification of hot spots of processes.
Data collection for semiconductor processes faces following specific challenges:

- manufacturing at different locations across the world
- site-specific energy correlated to the corresponding emission factor, e.g. national electricity grid regarding the different production locations
- additional global transports in between front-end, back-end, testing

2.2.1 Silicon wafers

Considering the silicon wafer manufacturing, Williams et al. investigated in 2000 and 2002 wafer fabrication and revealed inventory data of silicon wafer manufacturing [15,16]. In terms of life cycle assessment, these studies highlighted the importance of semiconductor industry and improved the understanding in public, including the LCA community. Since then, it has been considerably cited by many other studies as data base for modelling of IC manufacturing, e.g. [8,11,17]. Until now, very little update or progress of data on wafer manufacturing has been conducted, at least not publicly accessible. Hence, it is urgent to refresh data on material flows aligned with current technology generation, such as wafer size and thickness.

2.2.2 Front-end processes

Data acquisition to model front-end processes in semiconductor manufacturing has to deal with

- batch processes, frequently with a huge variance of IC designs processed in one fab (consideration of technological parameters associated with the complexity of processes and production efficiency, e.g. starting wafer area, wafer losses due to damage, good die out per wafer, die size, number of mask layers)
- role of infrastructure facilities: production effort of process versus facility
- additional up-stream production effort on high-grade purity of chemical

The production volume varies with economic cycles, chip layout changes rapidly, and so does the yield rate.
A number of life cycle assessment or product carbon footprint studies have been conducted, e.g. [9,10-11,13,18,19-23], some of them addressing above challenges. Moreover, [24] performed a detailed study of the additional purification steps needed for the key chemicals used in semiconductor manufacturing. It was concluded that a linear correlation between chemical cost and energy intensity is not accurate: The study states CO_2 factors associated with additional energy required for purification of high purity chemicals compared to standard quality chemicals. The outcome of this study allows for an estimation of additional energy demand of high purity chemicals and provides a useful basis for further study.

2.2.3 Exemplary front-end dataset

Table 2 provides a recent, generic dataset for a front-end process [8]. The reference unit is 1 cm^2 of good die out. The silicon wafer input is 1.38 cm^2 due to production line yield of 88%, wafer edge yield loss is 3% and good die yield 85%.

Tab. 2: Direct energy and material inputs and output of front-end process referring to 1 cm^2 good die out

Input	Amount	Unit	Output	Amount	Unit
Silicon Wafer	1.38	cm^2	good die out	1	cm^2
Electricity	1.27	kWh	HFC-23 (Trifluormethan)	4.21E-09	kg
Gas	0.16	kWh	Perfluorethane (C_2F_6)	9.92E-11	kg
N_2 (high purity)	6,06E-01	kg	Tetrafluormethane (CF_4)	4.81E-11	kg
O_2 (high purity)	4.13E-03	kg	Perfluorpropane (C_3F_8)	2.40E-07	kg
Ar (high purity)	2.34E-03	kg	SF_6	4.83E-05	kg
H_2 (high purity)	6.34E-05	kg	Nitrogentrifluoride (NF_3)	8.76E-06	kg
sulfuric acid (high purity)	7.33E-03	kg			
phosphoric acid (high purity)	3.32E-03	kg			
H_2O_2 (high purity)	2.04E-03	kg			
(C_3H_8O)/ isopropyl alcohol (IPA) (high purity)	2.78E-03	kg			
ammonium hydroxide (high purity)	1.09E-03	kg			
fluorhydric acid (high purity)	5.53E-04	kg			
NaOH (for wastewater treatment)	2.04E-03	kg			
Water	7.88E+00	kg			
CF_4 (high purity)	5.94E-05	kg			
C_2F_6 (high purity)	6.89E-05	kg			
CHF_3	5.66E-06	kg			
NF_3 (high purity)	3.02E-04	kg			
SF_6 (high purity)	8.96E-06	kg			

2.2.4 Back-end processes

In terms of back-end process, the energy consumption according to different sources is given in Table 3. All data is estimations according to experience since packaging types vary broadly.

Tab. 3: Overview on the energy demand of back-end process in different sources

Source	Energy demand of back-end process
MEEuP Report 2005 [25]	25% of energy demand in front-end process
ASE 2010 [26]	30% of the total energy demand
ESIA 2010 [14]	25%-37% of the total energy demand

Material input for back-end processes roughly corresponds with the material composition of the package: Several brandname OEMs disclose material declarations of typical products per package type. There are basically two different types of material composition declaration: One is to declare whether substances of concern exceed threshold levels and to report accordingly, e.g. [27], the other is a full material declaration, e.g. [28] that can help to determine material inputs for back-end processes of typical products. Regarding process consumables [14] indicated that back-end processes do not consume a significant amount of chemicals and the main input of material goes into and remains in the product. Although a typical product of back-end process might be roughly modelled for the purpose of an environmental assessment by means of above information, back-end processes still need further systematic investigation. Limitations regarding the approach to take a material declaration as a basis for calculating material related upstream impacts are as follows:

- from the silicon content in an IC package the silicon area can be estimated only with utmost caution: Silicon dies vary in thickness, depending on the application, and latest mobile products come with multi-chip packages, where several thinned dies are stacked, increasing the total silicon area significantly without increasing the absolute silicon weight in the package; thinned dies mean major upstream grinding losses of silicon
- Gold wire bonds: Only negligible process losses
- Epoxy encapsulation: minor process losses only
- Leadframe (copper): Major losses from stamping, but high recycling rate can be assumed
- Surface finishes: Losses due to wet bench chemical processes
- Interposer, if any: Material losses in production as those for complex printed circuit boards

- Test and assembly yield: Actually a certain yield loss has to be considered for packaging processes as well

Due to the difficulties and challenges in data collection as well as the extensive range of data needed, the question is what measures and parameters are both not arduous to collect and relatively easy accessible to LCA practitioners as well as convertible to other chip types in a reliable manner, at least within the same technology generation. If the relationship between environmental impacts and technology development as well as production complexity can be revealed through some parametric model, it reduces efforts on time-consuming data collection.

2.3 Methodological challenges: reference units

The reference unit is defined to provide a common basis on which inputs and outputs data is collected. It is necessary to ensure comparability of assessment results. When reviewing the literature, various parameters, such as number of mask layers, number of metal layers, processing time, one piece of finished die, one piece of good die out, technology node, whole wafer in diameter, wafer or silicon area or weight of ICs, are defined as possible reference units of ICs. This wide range of reference units applied hinders the transferability and comparability of inputs and outputs data as well as assessment results. The paper discusses the various approaches (see Table 4) under consideration as reference units for IC-datasets, covering front-end and back-end processes, in order to contribute towards creating a harmonised reference unit for IC datasets.
Choosing the right reference unit depends not only on what corresponds best with given technical specifics, but needs to consider also the intended use of any such IC data model. As shown in Table 4, some highly confidential data might make it into the parameterised model, if a manufacturer is supposed to provide PCF or LCA data as a "black box" model, but is rather not useful, if an LCA is solely based on reverse engineering where the key parameters of the model, such as potentially yield and number of mask layers, is not known to the external LCA practitioner. Moreover, parameters like yield vary depending on the complexity of products which is difficult to transfer to other products.

Tab. 4: Advantage and disadvantage of different reference units

Technology parameters as a proxy for complexity	**Advantage**	**Limitation**	**Exemplary references**
Mask layers	Often applied combined with wafer area. Could reflect the complexity of process (more mask layers, more process steps), could be used as one parameter in the parameterised model.	Highly confidential data, not known to external LCA practitioners	[18,29]
Metal layers	Reflects the complexity of process, could be used as one parameter in the parameterised model.	Other technical parameters have to be obtain, e.g. wafer yield (good die per wafer), line yield (finished wafer per wafer starts), die size.	[29]
Technology node in nm	Could reflect the complexity of process and contribute toward studying the development of technology.	Other technical parameters have to be obtain, e.g. wafer yield (good die per wafer), line yield (finished wafer per wafer starts), die size.	[10,11]
Reference units	**Advantage**	**Limitation**	**Exemplary references**
One unit or one IC package	Easy to obtain	Only feasible for back-end process. Only for specific chip, could not be transferred to other package type.	Product category rules for IC [12]
Wafer (area) out	Including information on the mask layers which are confidential data for fabs.	Yield of good die out and die size should be known.	[22]
Area of good die out	Including information on wafer yield, line yield and mask layers which are confidential data for fabs.	Confined to chip type, might only be transfer to other similar chip types. The question is that one cannot compare between chip types, if information on left side is not available.	[13,14]

Currently the European FP7 project "LCA to go" explores suitable approaches for front-end and back-end semiconductor processing, how an IC company could allocate internally environmental impacts to their broad range of IC designs, as a basis for design-specific PCF declarations. A harmonised approach among semiconductor manufacturers is intended to provide robust guidance for a coherent approach.

3 Conclusions

This paper discussed state of the art in life cycle assessment of laptops and revealed the methodological challenges on the component level, exemplarily for integrated circuits in terms of data collection and determining reference units.
Complex products and processes, multiple production locations as well as the rapid development of technology lead to challenges for data collection. The paper explored some data on the basis of literature and a recent research project commissioned by the Federal Environment Agency of Germany.
Advantages and limitations of the various methodological approaches and potential reference units have been discussed to provide a sound basis to choose the right approach for the intended use. According to current knowledge and available data, good die out in area should be appropriate reference units for front-end process. Back-end processes should be modelled on the basis of one packaged IC, but making a distinction per packaging type.
A reliable parametric model representing the correlation of environmental impacts and technological development, still needs further research and a sound definition of product category rules for semiconductors.

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References

[1] Apple Environmental Report 2010 <http://www.apple.com/de/environment/> (accessed 08.08.2010)
[2] Jönbrink AK (2007) Preparatory studies for Eco-design Requirements of EuPs: Lot 3 Personal Computers (desktops and laptops) and Computer Monitors. Final Report
[3] Ecoinvent Data v2.1 and v2.2 (2006/2010) Swiss centre for life cycle inventories, St. Gallen
[4] O'Connell S, Stutz M (2010) Product Carbon Footprint (PCF) Assessment of Dell Laptop – Results and Recommendations, Sustainable Systems and Technology (ISSST). IEEE. ISBN: 978-1-4244-7094-5
[5] EPIC-ICT (2010) Development of environmental performance indicators for ICT products on the example of personal computers, deliverable 6:

technical report samples and case studies demonstration of applicability of the EPIC-ICT method, < http://www.epic-ict.org/> (accessed 26.06.2010)

[6] Herrmann C (2008) PE International. ECOC Brussel<http://www.ecoc2008.org/documents/ECOCSymposiumPresentationCHerrmann.pdf> (accessed 28.06.2010)

[7] Jibiki H (2010) Status of type III environmental label, Eco-Leaf. Center of Ecoleaf type III environmental labelling program, JEMAI <http://www.city.sendai.jp/kankyou/kanri/icgps/pdf/special_2-3.pdf> (accessed 28.06.2010).

[8] Prakash S, Liu R, Schischke K, Stobbe L (2011) Zeitlich optimierter Ersatz eines Notebooks unter ökologischen Gesichtspunkten. Umwelt-bundesamtes, Germany

[9] Jain S (2011) A comparative assessment of the carbon footprint of AMD FusionTM products with the previous generation products. <http://www.amd.com/us/Documents/APU%20Carbon%20Footprint%20white%20paper%20FINAL%201%2021%2011.pdf> (accessed 31.01.2011)

[10] Boyd SB, Horvath A, Dornfeld AD (2010) Life-cycle assessment of computational logic produced from 1995 through 2010. Environ Res Lett 5(1). doi: 10.1088/1748-9326/5/1/014011

[11] Boyd SB, Horvath A, Dornfeld AD (2010) Supplementary data for "Life-cycle assessment of computational logic produced from 1995 through 2010". < http://iopscience.iop.org/1748-9326/5/1/014011/media> (accessed 14.09.2010)

[12] Product Category Rules (PCR) for preparing an environmental product declaration (EPD) for integrated circuits, version 1.0 (2009) <http://www.environdec.com/pcr/pcr_IC.pdf> (accessed 15.06.2010)

[13] Hermanns S (2010) Experiences from Piloting a Carbon Footprint Method for Semiconductor Manufacturing, presented at expert round-table, UBA-research project FKZ 363 01 322, Germany

[14] Questionnaire reply by the European Semiconductor Industry Association (ESIA) (2010), UBA-research project FKZ 363 01 322, Germany.

[15] Williams E. (2000) Global Production Chains and Sustainability: The case of high-purity silicon and its applications in IT and renewable energy. United Nations University, Japan

[16] Williams ED, Ayres RU, Heller M (2002) The 1.7 Kilogram Microchip: Energy and Material Use in the Production of Semiconductor Devices. Environ Sci Technol 36(24):5504 -5510.

[17] Schischke K, Kohlmeyer R (2003) ReUse Projekt, „Regionale Netze für die Wieder- und Weiterverwendung elektronischer Geräte". Arbeitspaket 6.7 Umweltbewertung des ReUse von PCs, Germany

[18] Schischke K, Stutz M, Ruelle J-P, Griese H, Reichl H (2001) Life cycle inventory analysis and identification of environmentally significant aspects in semiconductor manufacturing. IEEE Electronics and the Environment

[19] Boyd S, Horvath A, Dornfeld D (2009) Life Cycle Energy Demand and Global Warming Potenttial of Computational Logic. Environ Sci Technol 43:7303-7309.

[20] Boyd S, Horvath A, Dornfeld D (2009) Supporting Information for “Life Cycle Energy Demand and Global Warming Potential of Computational Logic”.

[21] Krishnan N, Boyd S, Somani A, Raoux S, Clark D, Dornfeld D (2008) A Hybrid Life Cycle Inventory Model for Nano-Scale Semiconductor Manufacturing, Environ Sci Technol 42:3069-3075.

[22] Krishnan N, Boyd S, Somani A, Raoux S, Clark D, Dornfeld D (2008) Supporting Information for “A Hybrid Life Cycle Inventory Model for Nano-Scale Semiconductor Manufacturing”.

[23] Liu CH, Lin SJ, Lewis C (2010) Life cycle assessment of DRAM in Taiwan's semiconductor industry. J Cleaner Prod 18:419-425.

[24] Higgs T, Cullen M, Yao M, Stewart S (2010) Review of LCA methods for ICT products and the impact of high purity and high cost materials. IEEE International Symposium on Sustainable Systems and Technology (ISSST), 17-19 May. doi: 10.1109/ISSST.2010.5507691

[25] MEEUP Methodology Report (2005) Ecodesign of Energy-using Products. VHK for European Commission

[26] Personal communication Huang V, ITRI (2010)

[27] <http://qdms.intel.com/MDDS/MDDSView.aspx> (accessed 11.01.2011)

[28] <http://www.samsung.com/global/business/semiconductor/aboutus/ESH/ESH_EcoProduct_EcoSearch.html> (accessed 11.01.2011)

[29] Yao MA, Wilson AR, McManus TJ, Shadman F (2004) Comparative analysis of the manufacturing and consumer use phases of two generations of semiconductors. IEEE International Symposium on Electronics and the Environment

The Concept of Monitoring of LCM Results Based on Refrigerators Case Study

Przemyslaw Kurczewski and Krzysztof Koper

Abstract The paper presents the concept of new approach in measurement and evaluation of results in life cycle assessment of products as a result of implementing LCM methodology in companies, based on an exemplary product of a major household equipment producer in Poland. The new approach is established on a complex analysis of economical, environmental and social consequences of an objects' life cycle. Evaluation is generated by an unification of Life Cycle Assessment (LCA), Life Cycle Costing (LCC) and Social Life Assessment (SLCA) methodology. Obtained results were registered into a matrix, enabling the identification between undertaken development operations and their results. This created a possibility to determine alterations on economical, environmental and social levels of a products' indicators. Consecutively, the range of modifications allowed a comparison between the current "state of an art" solution and the one proposed by interested parties.

1 Introduction

Life cycle management and implementation of the environmentally oriented design are the main topics of the project realised since the beginning of 2006. At the first phase, the project was financed by Polish Ministry of Science and Higher Education. It is carried out by two scientific institutions: Poznan University of Technology and Poznan University of Economics. There is also a partner from the industry, which is one of the largest manufacturers of household equipment on the Polish market. More details about the project can be found in [1].
The company involved in the project is one of the national business leaders in taking environmental image into account of its activities. Further product-oriented improvements are continually a significant part of its policy. To achieve goals of continuous improvement of the products and to improve the organisation of production processes, some initiatives have been undertaken. It clearly shows that

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this is a very good example of an organisation engaged in building a base for introduction of LCM approach into practice.

In 1994 the company implemented the Quality Management System compliant with ISO 9001 standard. Later on (in 1997), as the first Polish company, it successfully certified management systems based on ISO 14001 (Environmental Management Certificate) and the Occupational Health and Safety Certificate according to the PN-N-18001 and OHSAS 18001 standards (in 2001). Along with some other types of certificates, it allows the company to mark its products with the CE symbol.

There is a special *Eco-management and Work Safety* department in the company, where an interdisciplinary team works. It is responsible for environmental protection issues in a broad sense. In its activity, the company follows the principle of sustainable development, integrating the technical progress with the care for the environment. Management system promotes activities that:

- increase ecological awareness of the staff and highlight the environmental issues (through various publications, meetings, trainings etc.),
- facilitate a fully documented estimation of the factory's influence on the environment,
- allow a complex monitoring of activities connected with the most important effects that the factory causes for the environment,
- let reduce the negative environmental influences of the factory and allow for counteraction,
- prevent pollution of all elements of the environment,
- meet expectations of the customers concerning protection of the environment,
- increase the ecological reliability of organisation.

2 Beginnings of the LCM case study

One of the goals of the project was to establish guidelines and principles of environmental and economic improvements of refrigerators for the need of life cycle management [1-3]. To achieve this goal, a large study based on LCA and LCC of chosen refrigerator has been carried out. Because of the still discussed methodology of social life cycle assessment (SLCA), social aspects were not considered during the studies. In consequence, the main aim of the research was the evaluation and quantification of all potential environmental impacts generated during the whole life cycle of one refrigerator, chosen from a group of several

models. In a similar way, the economic aspects of the refrigerator life cycle were analysed.
The aim of the first stage of the analysis was to find answers to the following questions:

1) Can changes in material composition improve results in environmental and economic dimensions of the refrigerator's life cycle?
2) Can changes in the production processes reflect a decrease of the environmental impacts and lower the costs?
3) What activities should be implemented in the project to minimise environmental burdens and overall costs in the life cycle of refrigerator?

The data from the inventory analysis, including aggregated data concerning manufacturing operations of each element of the refrigerator, have been processed according to detailed LCA and LCC. In cases of use phase and final disposal, some simplifications have been applied at this point. It was assumed that on the base of obtained results, collection of more detailed data will be decided.
The results have been analysed separately for different life cycle stages: manufacturing, use (operation) phase and disposal. The calculation of environmental aspects was based on the end-points methodology and in consequence results were expressed as the damage indicators for eleven impact categories and – after grouping – for three damage categories (see also [4,5]).
Based on the results, the elements of the refrigerator's life cycle were ranked according to the size of generated costs and environmental impacts. Data matrix was created to show economic and environmental aspects in the composition of these components.
Likewise, LCM-oriented recommendations have been proposed for further development at this stage of the project. They are oriented particularly on:

- decrease of the energy consumption in the use phase,
- reduction and recovery of the end-of-life by-products,
- substitution of hazardous substances.

In detail, a set of rules has been established, including for example:

- easy access to replaceable components,
- easy separation of contaminated materials,
- easy disassembly to constituent parts,
- implementation of the materials identification system,
- decrease in use of "blended" materials,
- application of recycled materials where it is possible,
- upgrading of refrigerator components.

For the LCM to become a practice, it was necessary to ensure easy access to the results of the analysis and to provide easy means to present them. Analytical results of economic and environmental consequences of an objects' life cycle should be in the centre of given attention. Such data was generated as the results of life cycle assessment and life cycle costing studies during refrigerator development [2, 7].

3 The concept of LCM results presentation

The implementation of conceptual LCM started with the definition of [1, 2]:

- the object of the analysis (a refrigerator), recognised as a reference for design procedure oriented towards development in the whole life cycle,
- an idea of a refrigerator that fulfils the requirements of all the interested parties and recommendations formulated as results of LCA and LCC studies,
- a difference in technological advancement of the object that is caused by application of life cycle management.

At the task formulation stage, data including LCA and LCC recommendations and requirements, survey among the interested parties and benchmarking results were thoroughly analysed. On this basis, following aims have been formulated:

1) achievement of higher energy class (A++) and limitation of energy consumption to the level of 218 kWh/year (13.5% reduction compared to reference refrigerator),
2) reduction in the amount of harmful substances by 25%,
3) reduction of noise level in the operational use to the 38 dB (A) limit,
4) achievement of recovery rate at 80% level,
5) shortening the disassembly time to 30 minutes,
6) weight reduction by 5% compared to the reference refrigerator,
7) ensuring the availability of spare parts for 12 years from the date of manufacture,
8) provision of service for 12 years from the date of manufacture,
9) CE marking of materials,
10) development of an appropriate system for the take-back of used appliances.

This way, a set of technical solutions was determined and evaluated using multidimensional comparative analysis. The development of scenarios for the conceptual design and then detailed solutions was based on 10 variants of the changes [2]. The number and the scale of alterations was an inspiration to develop a transparent tool (matrix) for the presentation of the results of individual variants. The results for the developed scenarios should be presented not only to those responsible for product development, but also to some of the interested parties. In the discussed example, manufacturer's needs to communicate the results of life cycle management of a refrigerator caused a necessity to develop such matrix (Table 1) to present changes being the result of LCM implementation. Opportunity to present the results of social life cycle assessment of the tested object was also provided.

Tab. 1: LCM results presentation matrix

	LCA [Pts.]	LCC [PLN]	SLCA [Pts.]	Change in relation to the reference state [%]
Design	-	-	-	-
Manufacturing	-	-	-	-
Operation (use)	-	-	-	-
Disposal	-	-	-	-
Complete life cycle	-	-	-	-
Change in relation to the reference state [%]	-	-	-	-

The concept of using a matrix to present the results of life cycle analyses carried out on the object can be supported by several important arguments:

- **Ease of use** – this form of presenting the results gives a quick insight into volumes of different categories of impacts on a given phase of a life cycle or the whole life cycle of an object (horizontal analysis of the matrix answers the question of total impacts on a chosen life cycle phase, vertical analysis gives results concerning a given category of impacts in the whole life cycle),
- **Ease of preparation** – previously calculated impact volumes just need to be entered into corresponding cells of matrix,
- **Possibility to illustrate** – results can be presented graphically, e.g. in a 3-dimensional bar graph (with life cycle phases and life cycle assessment approaches in x and z-axis, and results in y-axis); however, the problem of a lack of a common denominator for the incompatible units (environmental and social points, units of currency) needs to be solved; and until then, the results can only be compared in single categories and in a relative, percentage scale,

- **Recognition of the dynamics of changes** – the matrix shows the shift of impact categories volumes of the altered object, compared to the state of reference (in means of single categories of impact at chosen life cycle phase, total impacts on a selected life cycle phase or total impacts in a given category in the whole life cycle of an object),
- **Identification of adverse effects** – e.g. transferring the environmental impacts generated at a chosen life cycle phase to another,
- **Identification of interdependence of impacts** – influence of changes in one direction (e.g. lowering the environmental impacts in the manufacturing phase) can be positive or negative for other dimensions of life cycle (increasing/decreasing economical and/or social impacts at the same time).

4 Results of LCM in the case study

Out of the previously developed 10 variants of accepted changes made in the construction of a refrigerator, three were chosen [2]:

- changing the number of chillers (refrigeration systems) (variant 2),
- using different refrigerant (variant 3),
- improving the insulation (variant 4).

Detailed research, including LCA and LCC analyses was conducted on those variants, and the results were presented in a matrix (Table 2, Table 3).
Research outcomes were then distributed into matrices (Table 4, Table 5, Table 6) showing only the comparison between specified version and object of reference life cycle analysis results. Horizontally, the changes in relation to the reference state are presented as LCA result modification/LCC result modification and given in percentages, where the result of LCA/LCC analysis for the object of reference is 100%. Vertically, the result of change is to be understood as a total change, encompassing the whole life cycle of an object, also given in percentages, and with a similar *object of reference = 100%* rule applied. Cells at the intersections of life cycle phases and life cycle analysis modes show the absolute results in units appropriate for the analysis mode.

Tab. 2: LCA results of analysed refrigerator variants

Variants / LCA results [Pts.]				
	Object of reference	Variant 2	Variant 3	Variant 4
Manufacturing	52.09	41.58 (↓20.18%)	51.81 (↓0.54%)	52.07 (↓0.04%)
Operation (use)	148.01	128.49 (↓13.19%)	134.73 (↓8.97%)	136.21 (↓7.97%)
Disposal	-14.8	-13.43 (↑9.26%)	-17.75 (↓19.93%)	-14.80 (↓0.0%)
Complete life cycle	185.30	156.64 (↓15.47%)	168.79 (↓8.91%)	173.48 (↓6.38%)

Tab. 3: LCC results of analysed refrigerator variants

Variants / LCC results [PLN]				
	Object of reference	Variant 2	Variant 3	Variant 4
Manufacturing	644.00	635.00 (↓1.40%)	675.00 (↑4.81%)	660.00 (↑2.48%)
Operation (use)	1186.35	1002.75 (↓15.48%)	1112.15 (↓6.25%)	1078.35 (↓9.10%)
Disposal	60.00	53.00 (↓11.67%)	55.00 (↓8.33%)	57.00 (↓5.0%)
Complete life cycle	1890.35	1690.75 (↓10.56%)	1842.15 (↓2.55%)	1795.35 (↓5.03%)

Tab. 4: Life cycle analysis results for variant 2

	LCA [Pts.]	**LCC[PLN]**	**Change in relation to the reference state [%]**
Manufacturing	41.58	635.00	(↓20.18) / (↓1.40)
Operation (use)	128.49	1002.75	(↓13.19) / (↓15.48)
Disposal	-13.43	53.00	(↑9.26) / (↓11.67)
Complete life cycle	156.64	1690.75	(↓15.47) / (↓10.56)
Change in relation to the reference state [%]	(↓15.47%)	(↓10.56%)	

Tab. 5: Life cycle analysis results for variant 3

	LCA [Pts.]	LCC[PLN]	Change in relation to the reference state [%]
Manufacturing	51.81	675.00	(↓0.54) / (↑4.81)
Operation (use)	134.73	1112.15	(↓8.97) / (↓6.25)
Disposal	-17.75	55.00	(↓19.93) / (↓8.33)
Complete life cycle	168.79	1842.15	(↓8.91) / (↓2.55)
Change in relation to the reference state [%]	(↓8.91%)	(↓2.55%)	

Tab. 6: Life cycle analysis results for variant 4

	LCA [Pts.]	LCC[PLN]	Change in relation to the reference state [%]
Manufacturing	52.07	660.00	(↓0.04) / (↑2.48)
Operation (use)	136,21	1078.35	(↓7.97) / (↓9.10)
Disposal	-14.80	57.00	(↓0.0) / (↓5.0)
Complete life cycle	173.48	1795.35	(↓6.38) / (↓5.03)
Change in relation to the reference state [%]	(↓6.38%)	(↓5.03%)	

5 Interpretation of results

In most of the categories, the impacts simulated for the versions of the refrigerator are lowered, and this reduction is simultaneous in both categories of impacts. In two examples (variant 3 and 4, Table 6 and Table 7 respectively) there is a negative relation - reducing the environmental burdens at the manufacturing stage raises the costs generated at that life cycle phase. Positive correlation between those two results can be found in variant 2 and that is because it assumes alterations to the original design more advanced than just modifications to the type or amount of materials used, as in the case of the two other variants. This major redesign also contributes greatly to the reduction of overall impacts in the operation phase of a refrigerator's life cycle – for example, cost reduction scale is more than twice when compared to the third variant.

Transfer of impacts is most apparent in the LCA results for variant 4 (Table 6). Insignificant reduction of environmental burdens at the manufacturing phase affects the results for other life cycle phases negatively. This can be interpreted as a certain negative attitude at environmental evaluation of any solution that includes adding weight and volume of materials to the design of an object [6].
Advantages of matrix-based presentation of LCM can also be seen using the overall conclusions as a background [1, 2] (Figure 1). The chart shows a hierarchy of proposed solutions in means of lowering total impacts in a life cycle of analysed refrigerator – the lower the points for selected variant are, the better a solution in life cycle development context is. But although this form of showing the results is good for putting in order different variants, it does not explain the reasons for such arrangement. Therefore, it is incomplete in means of communicating detailed data for the decision-making process [6]. For example, it does not address the question of reversed scale of environmental and economical impacts reduction for variant 3 and 4. At this level of generality, it is impossible to trace back deficient decisions focused on solving a single impact reduction issue that affect other areas and contribute to the transfer of impacts across the whole life cycle of an object.

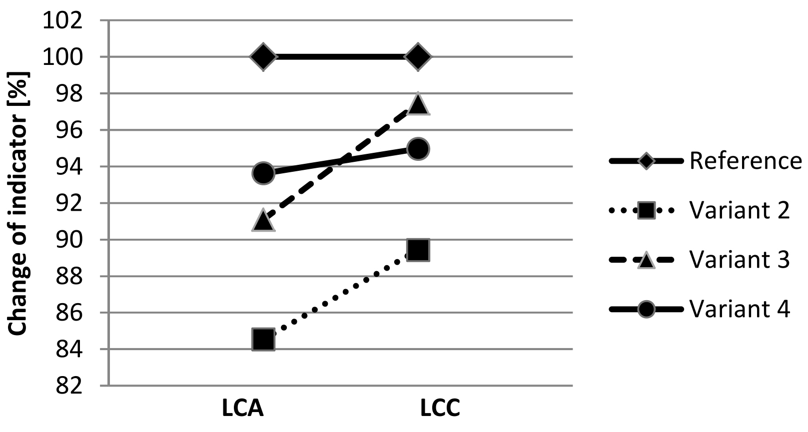

Fig. 1: Comparison of LCA/LCC results for different variants of the analysed refrigerator

6 Conclusions

Results of the conducted analysis of chosen variants show that proposed variant 2, including alterations to the original design of refrigerator should be selected to the stage of research (prototyping, testing, trial production). It is characterised by the highest degree of reduction of environmental impacts and costs that occur throughout the life cycle of analysed object. The reduction of environmental and economic indicators reaches 15.47%, and 10.56% respectively.

The proposed presentation and monitoring tool is a simple and effective way to illustrate the results of undertaken development procedures (actual or hypothetical). It can be used as an aid in dialogue between performers of life cycle analyses and decision-makers (managers responsible for product development strategies). In LCM, which is a management concept, it is fundamental that the decisions should be made based on facts. Any initiative aimed at improving the availability and legibility of acquired and interpreted data strengthens the position of LCM as default methodology for implementing the ideas of sustainable development at the object level into practice.

References

[1] Kurczewski P, Lewandowska A(2008)Ecodesign principles for managing the technical objects life cycle (in Polish).Wydawnictwo KMB Druk

[2] Lewandowska A, Kłos Z, Kurczewski P, Lewicki R (2009) Ekoprojektowanie na przykładzie sprzętu chłodniczego. Cz. I. Cele i procedura. Inżynieria i Aparatura Chemiczna 2:84-85.

[3] Lewandowska A, Kurczewski P (2010) ISO 14062 in theory and practice – ecodesign procedure. Part 1: structure and theory. Int J Life Cycle Assess 15:769-776.

[4] Rebitzer G, Schmidt W-P (2003) Design for Environment in the Automotive Sector with the Materials Selection Tool euroMat. Gate to EHS: Life Cycle Management – Design for Environment, March 17th

[5] Wimmer W, Zust R, Lee KM (2004)Ecodesign Implementation: A Systematic Guidance on Integrating Environmental Considerations into Product Development.Springer

[6] Kasprzak J, Kurczewski P, Lewicki R (2009) Zarządzanie cyklem życia sposobem na wzrost innowacyjności. Inżynieria i Aparatura Chemiczna 2:60-61.

[7] Selech J, Kurczewski P (2010) Metoda szacowania kosztu cyklu życia (LCC) i jej zastosowanie w dziedzinie budowy i eksploatacji obiektów technicznych. Inżynieria i Aparatura Chemiczna 5:105-106.

Life Cycle Management of F-Gas-Free Refrigeration Technology: The Case of F-Gases-Free Frozen Dessert Equipment

Francesca Cappellaro, Grazia Barberio and Paolo Masoni

Abstract This study aims to demonstrate the environmental advantages to adopt an innovative technology based on a natural refrigerant for an ice-cream machine, produced by Carpigiani Group. The environmental analysis has been carried out using the LCA methodology. This study has also been the starting point to introduce in the company the life cycle thinking and to create an ecodesign team integrating the environmental aspects into traditional design process. As required by several European directives the life cycle approach is recommended for all the energy using products in the residential, tertiary, and industrial sectors. The objective is to improve the overall environmental performance of these products, so to achieve the eco-innovation at the product-chain level.

1 Introduction

Commercial refrigeration covers a large variety of appliances used in several environments such as super markets, restaurant, hotels, pubs and café.

These products are estimated to consume an important portion of electricity in Europe. For these reasons they are included in energy-using products [1,2], the use of which has a relevant impact on energy consumption. The energy consumption of refrigeration systems (electrical energy used to produce the refrigeration load, lighting, etc.) has environmental impacts due to the generation of the electricity consumed. Moreover, negative environmental impacts have been also caused during their life-cycle by the material content such as refrigerants and insulating agents [3]. The aim of this study is the identification of environmental performances and improvement options, such as energy saving, for a refrigeration system: an ice-cream machine produced by Carpigiani Group. To evaluate the environmental impact associated to this refrigeration system it has been chosen the life cycle assessment methodology [4]. LCA provides a comprehensive overview of the environmental characteristics of that product or process and a more accurate

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representation of the true environmental trade-offs in product selection. LCA is useful to obtain a continuous improvement of the product and could help to increase the environmental awareness of the whole company. Moreover life cycle approach has been proposed by the European Commision as a way to evaluate the environmental impact of energy-using products and also of the refrigeration systems [5].

2 Environmental issues related to refrigerants

In 1992 the Montreal Protocol [6] banned F-gases called CFCs - chlorofluorocarbons because their responsibility of ozone layer depletion. Since then the chemical industry's most used alternative has been HFCs - hydrofluorocarbons.
Although HFCs pose no threat to the stratospheric ozone layer and were created as alternatives to ozone-depleting substances. HFCs are currently used as refrigerants and foam-blowing agents and emitted as leakage from air conditioning and refrigeration systems. As the demand for air conditioning and refrigeration increases globally, and as countries accelerate their efforts to phase out ozone-depleting substances, countries could turn increasingly to HFCs. HFCs are fluorinated compounds, when they are released into the atmosphere, contribute to climate change. They have a global warming potential similar to that of HCFCs and hundreds to thousands of times greater than carbon dioxide [7]. Table 1 shows the GWP (global warming potential) and ODP (ozone depletion potential) value for some commercial refrigerants belonging to the categories of CFC, HFC and natural refrigerants.

Tab. 1: Global warming potential and ozone depletion potential of main refrigerants [7]

Refrigerant	GWP (kg CO_2e)	ODP
R-12 (CFC)	10900	1
R- 502 (CFC)	4657	0.334
R-404a (HFC)	3920	0
R-143 a (HFC)	4470	0
R-125 (HFC)	3500	0
R-134 a (HFC)	1430	0
R-600A (isobutane)	3	0
R-290 (Propane)	3.3	0
R-744(CO_2)	1	0
R-717 (ammonia)	0	0

2.1 Alternative refrigerant technologies in commercial refrigeration

To provide significant climate protection benefits is necessary reducing projected increases in the use of HFCs in many countries. The commercial refrigeration sector accounts for approximately 32% of global HFC consumption [7]. Controlling HFCs sends an important signal to markets about the need to develop new alternatives that do not harm either ozone layer or the climate system. Technical efforts aim to reduce refrigerant emissions, to drastically lower the refrigerant charge by developing indirect systems or other concepts and to replace high-GWP refrigerants with lower GWP alternatives. [8]
Alternatives available today include ammonia (R-717), carbon dioxide (R-744) and hydrocarbons, such as isobutane (R-600a), propane (R-290), and propylene (R-1270);. Other alternatives, such as new HFCs/HFOs, are also likely to enter the market in the coming years. [8]
R134a and R-404a are two of the most commonly used refrigerants in commercial refrigeration. Although they are a good initial alternative for ozone depletion substances, R134a and R-404a are potent greenhouse gases (see Table1).
The alternatives depend on the appliance. For the ice-cream machines UNEP [9] suggest CO_2 and Hydrocarbons (HC). Since 2000, HC systems have been available and HC technology is gaining market share. In most ice cream freezers with HCs, HC-290 (propane) is used. Another alternative is CO_2. Carbon dioxide (R-744) is currently a proposed alternative for refrigeration systems. CO_2 has an ODP of zero, and a GWP of one. EPA's final rule entitled “Protection of Stratospheric Ozone” [10] listed CO_2 as an acceptable substitute for CFC-12, and noted that CO_2 is a well-known, nontoxic, non-flammable gas. Its GWP is defined as 1, and all other GWPs are indexed to it. Since it is available as a waste gas, only a pre-treatment is necessary but no additional chemical will need to be produced.

3 Ecodesign requirements of refrigeration systems

The need to improve environmental aspects into the commercial refrigeration is becoming also urgent due to the Energy Using Products (EuP) Directive, well-known as Ecodesign Directive [1,2]. The directive has required several preparatorystudies on specific product groups, including the refrigerating and freezing equipment [11]. The study on refrigerating and freezing equipment, called ENTR Lot 1, is aimed to recommend ways to improve the environmental performance of these systems and constituted the first step in considering whether and which ecodesign requirements should be set for these devices. The study includes an analysis of the relevant products (identified during the scope definition of the study) over their whole life cycle from different perspectives: market analysis, consumer behaviour, best available technologies (BAT), and improvement potential in terms of improving energy efficiency and reducing environmental impacts. The main results of the preparatory study of ENTR Lot 1 are the identification of best available technologies, energy saving and improvement options [3].

3.1 Best available technologies (BAT) for refrigerants

The preparatory study identifies the natural refrigerants as the best technology to reduce GWP of the refrigerating and freezing equipments. The kind of natural refrigerants depends on the appliance.
In Table 2 a comparison and applicability of alternative refrigerants islisted.

Tab. 2: Applicability of natural refrigerants to refrigerating and freezing equipments [3]

Refrigerant	Comments	Applications
R-744 (CO_2)	Not appropriate for small cold rooms. Used in large cascade systems, not in high ambient temperatures. Only applicable for remote blast equipment.	Blast cabinets Walk-in cold rooms Chillers
R-717 (ammonia)	Toxicity and corrosion issues.	Chillers
R-290 (propane)	May be used in small application with small refrigerant charge.	Service cabinets Blast cabinets Chillers Remote condensing unit
R-600a (isobutane)	If hydrocarbons are to be used, propane is more efficient option to choose.	Service cabinets
R-1234yf (HFO)	Used in automotive industry.	Service cabinets Blast cabinets Walk-in cold rooms Chillers Remote condensing unit

3.2 Improvement options

The preparatory study describes several improvement options available for different product groups. A significant energy saving (40%) is possible through an efficient control of the refrigeration system, the reduction of effective temperature differences on heat exchangers, by using highly efficiently compressor. The user behaviour is also very important issue for a sustainable use of the machine. Providing detailed information on best installation, use pattern and maintenance practices it can determine significant energy saving. Finally, lifetime of the machine is an important parameter that can affect the life cycle.

4 Carpigiani ice-cream machines

Carpigiani group is one of the larger manufacturers of frozen dessert equipment worldwide and takes care of research and development of new technologies and promotes new products and services. Carpigiani's ice-cream machines work in vapor-compression refrigeration cycle based actually on R404A. Driven by the "Refrigerants, Naturally!" [12], a global initiative of companies committed to combat climate change and ozone layer depletion, Carpigiani has started to investigate for a new technology that substitutes harmful fluorinated refrigerants with natural refrigerants. The company has tested two new technologies based on low-GWP alternative substances: the first using steam-compression refrigerating cycle with propane (R290) and the second one using transcritical CO_2 application. In the Table 3 are described the performances of the ice-cream machine working with different fluids.

Tab. 3: Comparison of technical performances of Carpigiani ice-cream machines

Refrigeration system	Pressure (bar)	Weight (kg)	Energy (MJ/year)	End-of-life treatment
R-404a	1,3 - 17	395	58.000	F-gas Regulation (EC) 842/2006
R-290	2.1 - 13	400	45.400	No
R-744	15 - 75	415	46.000	No

Experimental tests have proven the technical feasibility of an ice-cream machine based on CO_2 and Carpigiani has realized a prototype using CO_2 technology. The machine operates at a higher pressure than the others and requires new system design and new components. In collaboration with ENEA an environmental impact evaluation has been carried out to assess the environmental aspects of this technological solution comparing it with the machine using R-404a.

4.1 Comparative life cycle assessment

The LCA study was aimed to identify “hot spots”, to improve the overall profile and to suggest and compare design alternatives.
Functional unit is the ice-cream production for 10 years (lifetime of the machine).
The comparative LCA takes into account only phases and materials and energy flows which are different for the two machines (see Table 4). The studied components are: the compressor, the pipes, the gas cooler and the refrigerants.

Tab. 4: Ice-cream machine components considered in the comparative LCA

	HFC-machine	**CO_2-machine**
Total weight	395 kg	415 kg
Copper components	35.20 kg	26.85 kg
Compressor	80 kg	120 kg
Refrigerant fluid	2.94 kg [R-404A]	3 kg [CO_2]

The LCA considers all the life cycle phases: assembly, distribution, use and maintenance and end of life. The analysis evaluates the impacts related to the different amounts of materials, mainly iron and copper, used in tubes, in the compressor, in the intercooler and into the gas cooler. It also considers the different energy consumption during the use and the end-of-life phases. The environmental performances of the two systems were analysed by the impact assessment method CML, 2001- Dec. 07. The results of the impact assessment are described in Figure 1.

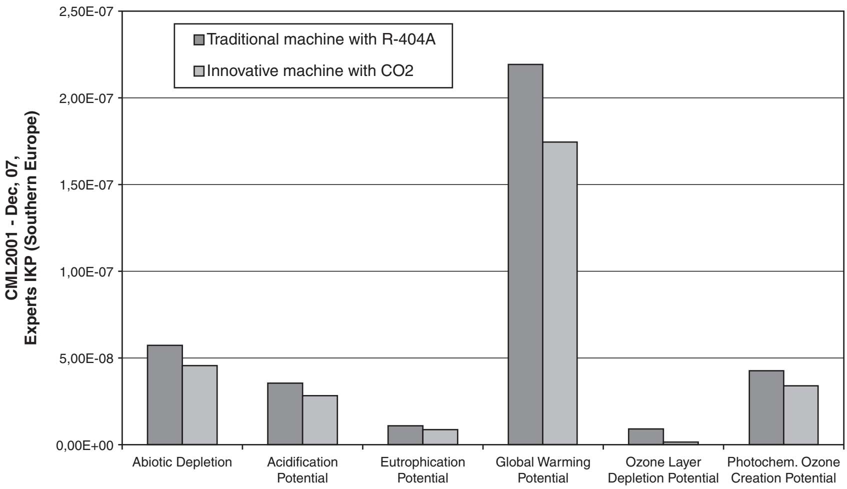

Fig. 1: Comparative environmental impact assessment between R-404a and CO_2-based ice-cream machines

4.2 Results

The results highlight that both the machines have highest impact in the use phase. The innovative CO_2-based machine shows environmental impact improvements. In particular, there are reductions:

- by 23% for energy consumption
- by 22% for global warming (GWP)
- by 80% for ozone layer depletion (ODP).

The life cycle approach has also been utilized to provide some improvement's options that can be adopt by the company during the design phase of the prototype machine. In fact, the development of the new machine was a big opportunity to introduce the ecodesign concepts in the company. The environmental assessments help to identify critical points to be improved and to compare design alternatives. The improvement's options identified for the CO_2-based ice-cream machine are listed in the following paragraph.

4.2.1 Reduce the amount of material

Materials are a key factor determining the environmental performance of a products. In order to reduce the amount of materials used it can be useful to minimize the thickness and the dimension of some components. For the ice-cream machine the compressor is the component with a major weight, the amount of steel can be reduced compacting the compressor dimension. Moreover the utilization of microchanneling can determine a reduction of amount of copper. In the Table 5 are listed the environmental improvements associated to these solutions.

Tab. 5: Environmental improvements of ecodesign solutions

Ecodesign solution	Materials' reduction	Environmental impacts
Microchanneling	< 24% copper	<30% for GWP, ODP
Compressor's compaction	< 35% steel	<5% for ODP <47%for terrestrial toxicity

4.2.2 Reduction of environmental impacts during use

As shown in figure 1 the use phase is the most environmentally important in the whole life cycle of the ice-cream machine. The optimization of this phase can help in the reduction of environmental impacts.
In the Table 6 are described the different rates of production both of the HFC machine and the CO_2 machine.

Tab. 6: Different rates of production

Rate of production	Energy consumption HFC machine	Energy consumption CO_2 machine
High	61.49 kWh	50.4 kWh
Medium	44.29 kWh	35.11 kWh
Low	55.65 kWh	45.34 kWh

The user behaviour has significant importance for reducing energy consumption [13]. Informing the user on the most efficiently use the machine (medium production) can provide significant energy saving.

4.2.3 Optimisation of the life cycle of machine

In order to minimise the environmental impact can be useful to increase the life time of machine and optimize the end of life. The prototype test has shown a better quality of the machine, so it is possible to extend the machine life up to 12 years. Moreover, it also possible to increase the material separation and the recovery (i.e. metals) to obtain a reduction of the resources consumption.

5 Conclusions

This study has described the case of a Carpigiani ice-cream machine based on a low GWP technology using the CO_2 instead of a HFC refrigerant (R-404A). This innovation was analysed by means of life cycle approach. The comparison of the traditional ice-cream machine using HFC with a new machine with natural gas technology has shown the environmental advantages of this alternative. The LCA results have also been the starting point for the Carpigianicompany to introduce the ecodesign in the traditional design process. The use of the LCA helps to identify the design alternatives and to improve the environmental performances of innovative CO_2-based ice-cream machine. The ecodesign solutions emerged by the LCA study are aligned with the requirements of the preparatory study on refrigerating and freezing equipment, required by Energy Using Products directive. The preparatory study of refrigerating and freezing equipment provides detailed specifications only for some refrigeration system. Actually, for the ice-cream machines there are not released specific requirements, but general specifications. The considerations emerging by this study could be useful to extend the specifications of the refrigerating and freezing equipment to the ice-cream-machines and to achieve the eco-innovation at the sector level. Putting in practice the environmental considerations could generate as much innovation as the creation of a new line of sustainable products.

References

[1] European Parliament and Council (2005)Directive 2005/32/EC Establishing a framework for the setting of ecodesign requirements for energy-using products and amending.

[2] European Parliament and Council (2009) Directive 2009/125/EC Establishing a framework for the setting of ecodesign requirements for energy-related products.

[3] MudgalSet al (2010)Summary Document of Refrigerating and Freezing Equipment: Service cabinets, Blast cabinets, Walk-in cold rooms, Chillers, Remote Condensing Units, Water dispensers, Minibars, Wine storage appliances, Ice makers, Dessert and Beverage machines. Available online at http://www.ecofreezercom.org(accessed 14.04.2011)

[4] ISO (International Standard Organization) (2006)ISO 14040:2006, Environmental management - Life cycle assessment – Principles and framework.

[5] European Commission (2005)Final Report Methodology for Eco-design of Energy-using Products (MEEuP).Available online at: http://ec.europa.eu /enterprise/policies/sustainable-business/ecodesign/methodology/files/ finalreport3_en.pdf(accessed 14.04.2011)

[6] UNEP United Nations Environment Programme Ozone Secretariat (2000) Montreal Protocol on Substances That Deplete the Ozone Layer. Available online athttp://www.unep.org/ozone/Montreal-Protocol/Montreal-Protocol2000.shtml (accessed 14.04.2011)

[7] EPA Environmental Protection Agency (2010)Transitioning to low-GWP alternatives in commercial refrigeration.Available online at http://www.epa.gov/ozone/downloads/EPA_HFC_ComRef.pdf(accessed 14.04.2011)

[8] Velders GJM,Fahey DW, Daniel JS, McFarland M, Andersen SO (2009)The large contribution of projected HFC emissions to future climate forcing. Proceedings of the National Academy of Sciences of the United States of America.PNAS Edition106(27):10949-10954.

[9] UNEP (2006) Report of the refrigeration, air conditioning and heat pumps technical options committee. Available online at: http://ozone.unep.org/teap/Reports/RTOC/rtoc_assessment_report06.pdf (accessed 14.04.2011).

[10] EPA (1993) Environmental Protection Agency Protection of Stratospheric Ozone. U.S. Federal Register 58(236).

[11] < http://www.ecofreezercom.org/> (accessed 14.04.2011)

[12] < http://www.refrigerantsnaturally.com/>(accessed 14.04.2011)

[13] ShailendraM et al (2010)Task 3 Working Document of ENTR Lot 1 on Refrigerating and Freezing Equipment.Available online at: http://www.ecofreezercom.org(accessed 14.04.2011)

PART IX:
LCM in the Mobility Sector

Assessment of the Environmental Impacts of Electric Vehicle Concepts

Michael Held and Michael Baumann

Abstract Under the impression of current discussions on depleting resources and environmental questions, the transportation sector is aware of its responsibility and pushes the development of alternative power train concepts. Especially electric vehicle concepts are investigated, since they decrease the dependency from oil based fuels and reduce local noise and emissions during the vehicle operation. By using renewable energies (e.g. wind power), e-mobility can contribute to a significant reduction of the climate balance of transportation. However, there is only little known about the life cycle impacts of e-mobility and respective vehicle components. Based on the method of life cycle assessment (LCA), this paper gives a first quantification on the environmental profile and relevant indicators of e-mobility.

1 Introduction

In the framework of the Fraunhofer System Research for Electromobility (FSEM), the department life cycle engineering (GaBi) investigates the environmental performance of different electric vehicle (EV) concepts. The main aim of the study is to give a first estimate on the bandwidth of the environmental potential of e-vehicle concepts and to identify the relevant indicators of e-mobility. Based on the outcomes of the study, the demand on further research is evaluated. Within this work, the production and use phase of different electric vehicle concepts, like battery electric vehicles (BEVs) and plug-in hybrid electric vehicles (PHEVs) are investigated. For a classification of the environmental performance of the EV concepts, the LCA results are compared to conventional vehicles with combustion engines (CVs). In addition, as a broader market entry of electric vehicles is expected in the next years (e.g. the national roadmap for electric mobility of the German federal government expects around one million electric vehicles on German streets until the year 2020[1]) the study analyses future scenarios to estimate the environmental profile of future concepts and developments. In a first

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step the scenarios investigate the environmental profile of future developments of the battery systems and the German electricity grid mix.

2 Boundary conditions

The data used for this study is based on internal information within the FSEM project, available literature and results from previous studies. For keeping the transparency of results, the components and vehicles are modelled in a hierarchical structure and combined in a system model according to the structure presented in Table 1. The modelling is carried out with support of the LCA software GaBi 4 and its databases [2]. The reference location of the vehicle production and utilisation is Germany. Since the power grid mix (PGM) changes during use phase of the vehicles, a dynamical adjustment of the environmental profile of the power generation according to the lead study of the German federal ministry for the environment [3] has been taken into account. In addition, an alternative scenario is investigated where a straight use of additional installed wind power is assumed. Within the study, exemplified configurations of two battery electric vehicles (BEV) and one plug-in hybrid vehicle (PHEV) are evaluated. The environmental profiles of the CVs and the vehicle platforms are based on the material mix and of comparable vehicles classes [4-6] which have been scaled according to the vehicle and platform mass. The electric motor is based on the material mix of a permanent magnet synchronous motor (PMSM) which is scaled by the weight-to-power ratio [7]. The mass share of magnetic materials has been adjusted according to the average consumption in electric vehicles from around 2 kg, whereas the share of rare earths e.g. neodymium (Nd) or dysprosium (Dy) accounts for up to ~30% in sum [7-9].
The configuration and technical specification of the battery system is based on internal data of the FSEM project. Since there is currently only little information on the LCA of the production of EV batteries and related materials available, assumptions have been made based on from previous studies, available literature and material safety data sheets (MSDS) of Li-Ion battery cells (e.g. [10-14]). The power electronic components (e.g. inverters) are based on existing models of comparable components.
At present, there is no information on vehicle or user specific utilisation profiles available. Therefore, the calculation of the energy and fuel consumption of the vehicles is based on the ADAC Eco-Test [15] and NEDC [16] driving cycles. The calculation method is based on a systematic approach developed during the FSEM project [17]. Due to the lack of available data, the impacts of the end of life phase

and possible battery recycling concepts are not considered at this stage of the project.

Tab. 1: General boundary conditions

	BEV* (mini-class)	BEV* (compact-class)	PHEV hybrid* (compact-class)
Battery technology	Li-Ion (LiNi1/3Co1/3Mn1/3O2)		
Gravimetrical energy density of cells [Wh/kg]	135		
Energy content of battery [kWh]	20	40	14
E-motor PMSM [kW]	43	70	68
Power electronics [kg]	34	35	56
Combustion engine [kW]	-	-	41
Generator [kW]	-	-	41
Car platform and other parts [kg]	736	1,115	1,115
Total mass of EV [kg]	1,037	1,670	1,505
Energy consumption, electrical** [kWh/100 km]	18.7	22.9	20.4
Fuel consumption [l/100 km]***			6,9
Share of electrical mode [%]			80
Lifetime of battery system [years]	8		
Lifetime of other components [years]	12		
Mileage (daily/annual/lifetime) [km]	39/14,300/171,600		

*Exemplified vehicle concepts; **Calculation according to ADAC Eco-Test; ***Emission profile based on HBEFA3.1 [18]

3 Results of the production and use phase

The results of the life cycle impact assessment (LCIA) are presented at the example of the global warming potential (GWP) and acidification potential (AP).
It shows that due to the additional EV specific components, the share of the production phase of EVs considerably increases compared to the CVs, which is up to around 2 times in the GWP and around 2 to 4 times in the AP. The main contribution is due to the production of the battery system. Depending on the dimensioning in the EV concepts, the battery system causes between 30 and 60% of the GWP and around 50 to 75% of the AP. The assumed battery lifetime of 8 years is lower than the vehicle lifetime of 12 years, which requires a maintenance step during the use phase. Thus, the relevance of the battery system in the vehicle life cycle further increases.
Depending on the EV concept and vehicle class, the dimensioning of the battery systems in the vehicles differs (mini-class BEV 20kWh; compact-class BEV

40kWh; compact-class PHEV 14kWh) and therefore the relative share of the battery system in the vehicle life cycle. Depending on the required energy content of the battery system, the additional mass in the vehicle varies between 150 and 450 kg. Since, especially the EV specific components require relatively high amounts of high-tech materials with accordingly high environmental impacts in the raw material extraction and material production, the impacts of these components are considerably higher than the vehicle platform which mainly consist of steel, plastics and light weight metals. The impacts of the battery systems are mainly caused in the extraction and production of the required upstream materials for the cathode production, whereas the impacts of the cobalt production dominate.
Also today's PMSM-motors are equipped with rare earths like neodymium and dysprosium, with relatively high impacts in the material production. However, due to the lower mass proportion of the e-motor, the share to the EV production phase is comparatively low.

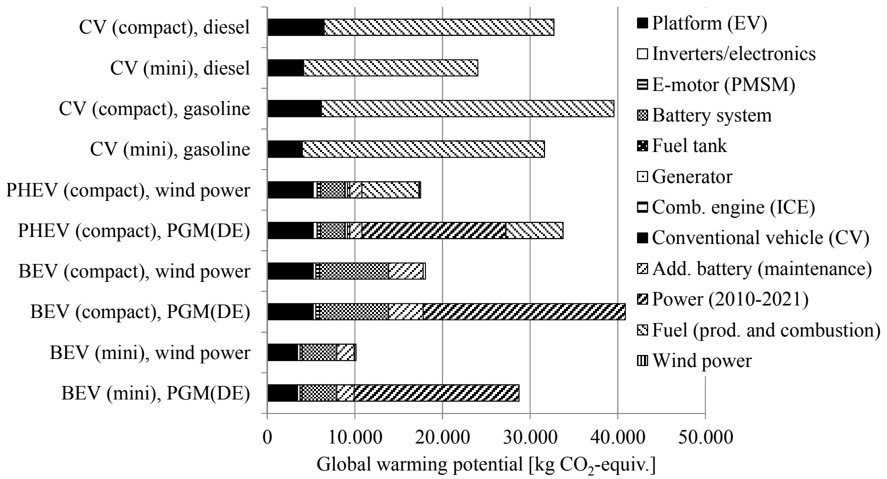

Fig. 1: Global warming potential of the production and use of EVs and CVs

Regarding the use phase, the EVs cause less environmental impacts than the conventional vehicles, which currently describe the environmental benefit. Based on the defined boundary conditions, the impacts of electric vehicle concepts to the GWP are in a comparable range to gasoline vehicles. Diesel vehicles are not reached, yet. In the case that the EVs are solely charged with wind power, significant improvements in the climate balance are achieved compared to the CV. However, this implies that the wind power is provided by additionally installed wind power plants.

Figure 2 presents the results for the AP. Even though the EVs have lower impacts to the AP during the vehicle operation, the higher impacts of the production phase cannot be compensated. Due to the high contribution of the production and maintenance of the battery system, the impacts of the BEV concepts are up to 2 times higher than for comparable CVs. In case of the wind power scenario, the mini-class BEVs reaches a comparable range of the CVs.

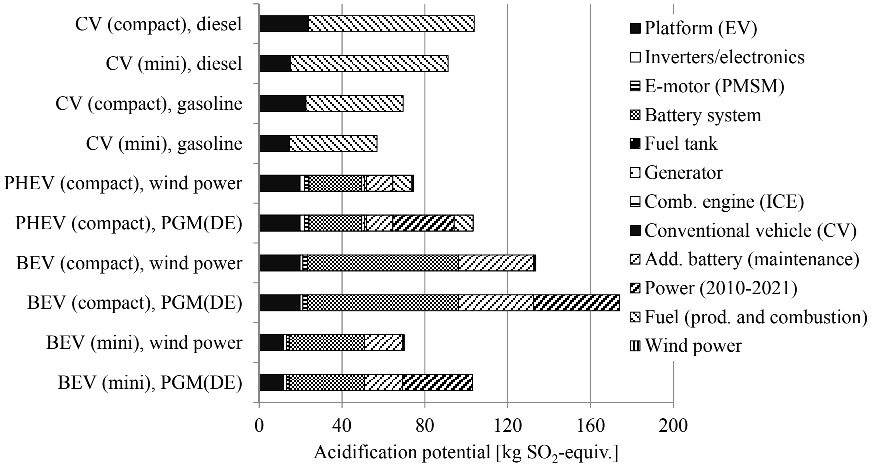

Fig. 2: Acidification potential of the production and use of EVs and CVs

Due to the smaller sized battery system, the contribution of the PHEV to the AP is lower than for the BEVs. Thus, the PHEV results in a range between the CV concepts. By the use of wind power, considerable reductions to CVs are achieved.
The results of the impact assessment show that the power generation mix, the battery system and the utilisation profile of the vehicle have a considerable impact on the environmental profile of the vehicle life cycle. Regarding the battery system, the production of the high-tech materials, the lifetime and dimensioning of the battery system are the main indicators.
Depending on the chosen driving cycles (NEDC vs. Eco-test) and EV concept, the impacts to the EVs vary between 15-20% in the GWP and 7-10% in the AP. In case of the PHEV, the chosen driving cycle also affects the share of the electrical and combustion driving mode. Depending on the evaluation method, the results vary by around 16%. Since the driving cycles do not necessarily reflect the conditions of realistic utilisation profiles of EV concepts, vehicle and user specific utilisation profiles have to be evaluated in future work to allow a fair comparison to conventional vehicles.

4 Scenario analysis

In contrary to the conventional vehicles with more than a hundred years development, today's alternative power train concepts are in a relatively early development stage. Most EVs battery systems are produced in small scales and have the potential for further improvements. One key aspect for e-mobility is to enable higher driving range with low battery weights. It is expected that the energy density of Li-Ion batteries will increase from currently around 135 Wh/kg to max. 200 Wh/kg in 2020 [19-20]. In long term view, higher energy densities are to be realised with new battery technologies like Li-S or Me-O. Besides the future increase of the energy density of battery cells, it is also expected that lifetime of batteries will be enhanced. Manufacturers target a battery lifetime up to ten years within the next years [21]. In long term view this study assumes that the battery lifetime endures a vehicle life of 12 years. Besides the battery system, improvements are also expected for the e-motor, power electronics and in the vehicle design. As presented above, the power generation mix, the battery system and the use profile for calculating the energy consumption of the vehicle have a considerable impact on the environmental profile of the vehicle life cycle. Hence, the following scenarios concentrate on these aspects by analysing the effects of the future developments of the battery systems and the German power grid mix. The scope of the scenarios is for the years 2010 to 2020. In a second scenario, the dependency of the chosen utilisation profile to the vehicle life cycle is analysed at the example of a mini-class BEV used as a city car. For keeping the comparability of results, a fixed electric driving range of EVs is considered. Therefore, improvements of the batteries result in a lower demand of battery cells and hence a lower impact in the production phase. The fuel consumption and emission profiles of future CVs are based in the scenarios of HBEFA 3.1. The development of the fuel production mix for CVs is not addressed in this study.

4.1 Results of scenario 2020

The results of scenario 2020 show that the improvements of the battery systems, mainly due to the extended battery life time, contribute to a significant reduction of the GWP of the EV life cycle (Figure 3).

In addition, due to a continuous increasing share of regenerative energies in the German power mix (around 58% until 2030), the total impacts of the EV use phase are further reduced. Based on the assumptions of the scenario 2020, all EV concepts show significant improvements against the gasoline CVs and slight improvements against the diesel CVs. However, the significant reductions are still reached by using renewable energies for the battery charging.

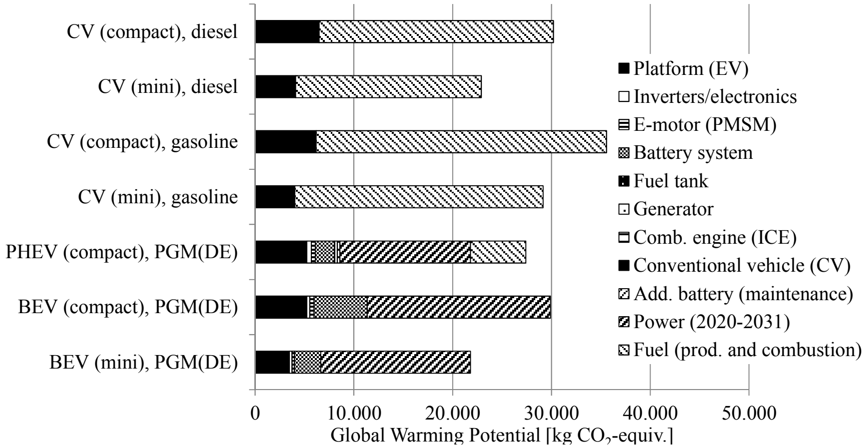

Fig. 3: GWP of the production and use of EVs and CVs (scenario 2020)

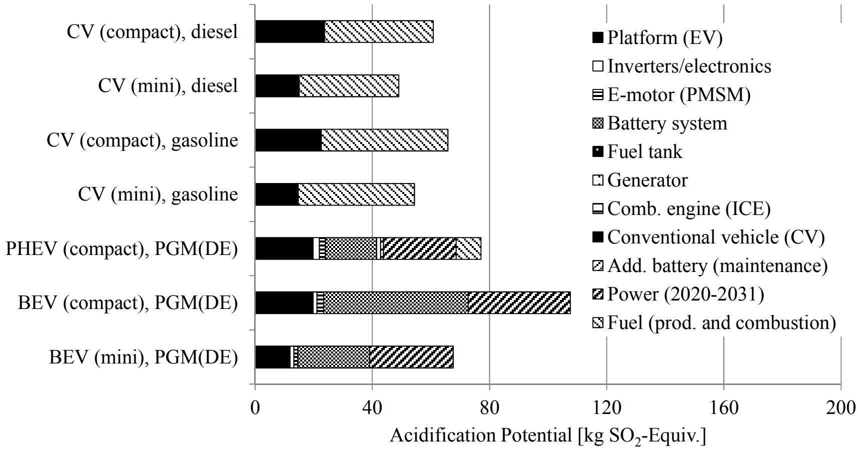

Fig. 4: AP of the production and use of EVs and CVs (scenario 2020)

Figure 4 presents the LCIA results for the AP. Also for the AP, significant improvements are reached by the extension of the battery lifetime and increased energy density, whereas the total contributions of the EV concepts are still considerably higher compared to the CVs. A still open question that has to be addressed in further studies is the evaluation of the future developments of the environmental profiles of used rare high-tech materials. We assume that due to the higher demand, also the environmental impacts of the raw materials extraction and the material production will change.

4.2 Results: User specific utilisation profiles

The calculation of the energy and fuel consumption of the EV concepts is based on the Eco-test and NEDC driving cycles. Due to the defined driving profiles, the driving cycles enable the comparison of the fuel consumption of different vehicles. However, these driving cycles do not necessarily reflect real driving conditions or utilisation profiles of electric vehicles. We rather expect that due to the limited driving range and longer charging times, the utilisation profiles of the EV concepts, especially the BEVs, will differ from the CVs. Furthermore, due to a broader implementation of e-mobility also the mobility itself might change. Thus, based on the following scenarios, the relevance of the defined utilisation profiles to the environmental profile of EV concepts is investigated at the example of a mini-class BEV used as city car. To do this, two cases with varying mileages are evaluated.

Case 1: Mini-class BEV in city use, low mileage

The first case assumes an annual mileage of 8,000 km/year (~21 km/day), e.g. used for shorter trips to work, shopping etc. According to the previous scenarios the considered vehicle lifetime is 12 years, the battery lifetime 8 years. The fuel consumption of the CVs is based on the city cycle of HBEFA 3.1. In contrary to the CVs, where the fuel consumption increases by around 10 to 20%, the energy consumption of the BEV decreases at lower speeds. The calculated energy consumption of the BEV for the city scenario is 0.15 kWh/km. Figure 5 presents the results of the scenarios for the GWP. Due to the lower total mileage of the BEV (96,000 km), the relative share of the production phase increases in the vehicle life cycle. However, the lower energy consumption leads to a GWP of the mini class BEVs which is in a range between the gasoline and diesel vehicles. Based on an increased use of renewable power significant reductions of the GWP could be reached, whereas a straight use of additional wind power would represent the best case scenario. However, we assume that in near- and mid-term, the use phase of the EVs will be in a closer range to the power grid mix scenario.

The future improvements of the battery system and a higher share of renewable energies in the power grid mix in scenario 2020 lead to significant reductions compared to the CVs. The break-even to the CVs is then between 40,000 km and 50,000 km.

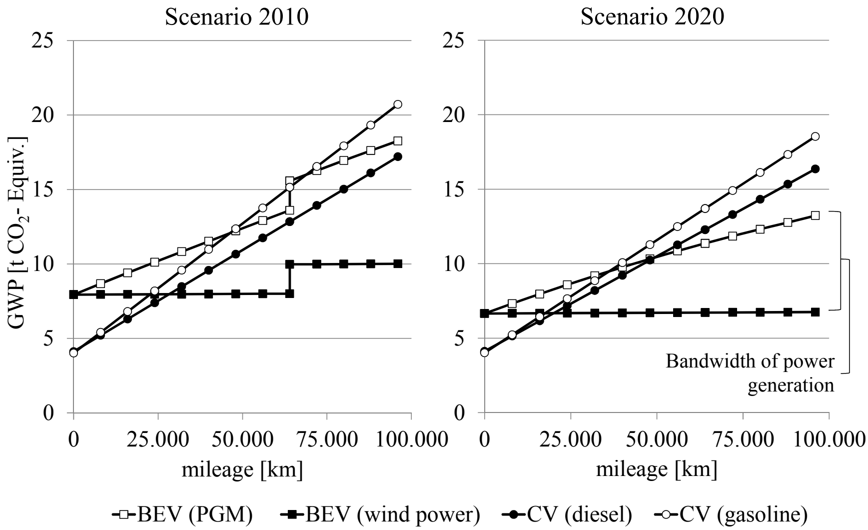

Fig. 5: GWP of a mini-class BEV in city use, low mileage

Case 2: Mini-class BEV in city use, high mileage
The second scenario investigates the case of a city car with a higher utilisation capacity, e.g. a vehicle used in a car sharing fleet. The assumed annual driving distance is 14,300 km (39 km/day). All other parameters remain unchanged. Figure 6 presents the results of the scenario.

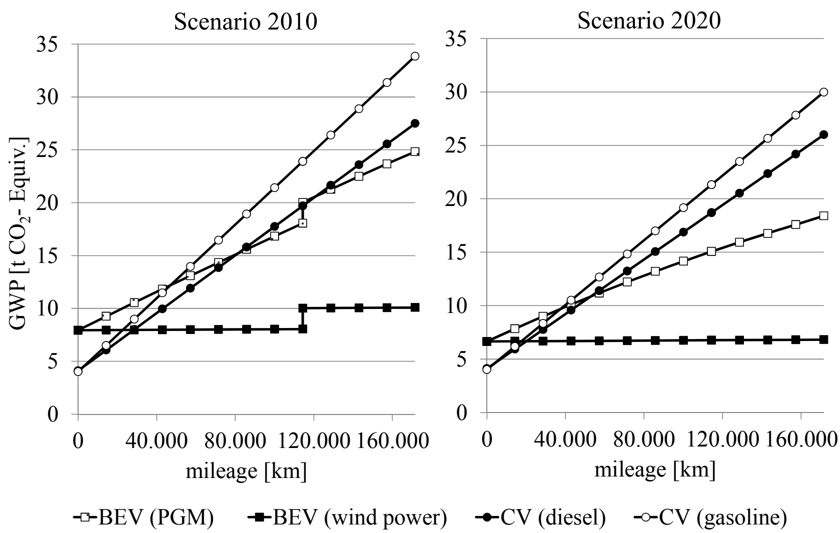

Fig. 6: GWP of a mini-class BEV in city use, high mileage

Due to the higher mileage of the BEV clear improvements are achieved. Even with the use of the power grid mix the impacts to the GWP are considerably below the comparable gasoline CVs and are in a comparable range to the diesel vehicles. The break even to gasoline vehicles is at around 50,000 km and around 125,000 km for the diesel vehicles. The results of the scenario 2020 show that depending on the field of application, the EVs have the potential for a significant reduction of the GWP, compared to the CVs.
The investigated scenarios show, that from a today's perspective, the environmental benefits of e-mobility concepts are strongly dependent on the field of use and the assumed vehicle specific utilisation profiles. For a better understanding of e-mobility, more comprehensive and representative data on vehicle specific utilisation profiles are necessary.

5 Need for further research

The results of the assessment of the EV concepts show, that there are still several open questions which have to be addressed in more detail to allow a more reliable evaluation of the environmental profile of e-mobility. Future work has to be done for gaining a better knowledge on the production and upstream processes of the relevant materials (Li, Co, Ni, Mn, Nd, Dy) as well as the future development of the environmental profiles, availability and recycling concepts. Another significant aspect is the future development of components, in terms of technologies, alternative concepts, material use and potential substitutes. Since there is only little information on the environmental profiles of EV battery technologies available, detailed LCA studies have to be carried out in cooperation with battery producers and relevant players in the supply chain and raw material production. Concerning the use phase, there is the need for the investigation of vehicle and user specific utilisation profiles to ensure the representativeness of LCA results and to enable a fair comparison to CVs. These profiles will also provide a basis for the identification of beneficial fields for the application of EV concepts. Furthermore the study showed that there are many variable parameters in the life cycle of EV concepts which can significantly influence the LCA results. Therefore, a common agreed approach for the LCA of e-mobility is required to ensure the comparability and consistency of future LCA studies.

6 Summary and conclusion

The paper presented a first quantification of the environmental profile of the production and use phase of different electric vehicle concepts. The results show that the relevant parameters of e-mobility are the used power generation mix, the dimension and lifetime of the battery system as well as the driving profiles and total mileage of the EVs. Due to the use of rare and high-tech materials, the additional components of the EVs lead to an increased relevance of the production phase, whereas the battery system has the main contribution. During the use phase, the EVs cause lower environmental impacts compared to the CVs, which currently represents the environmental benefit of e-mobility. Using the German power mix, the GWP of current EVs are in a comparable range with gasoline vehicles. However, a significant reduction of the GWP is reached when the EVs are charged with additional installed renewable energies, e.g. wind power. Regarding the AP of the EVs, the study showed that the contributions of the EVs are considerably higher than the CVs, whereas also the main impact is due to the production and dimensioning of the battery system. The results of the future scenarios presented that considerable improvements in the EV life cycle can be reached in a long term view.
However, the study also shows that there are still many open questions that have to be further addressed for a reliable assessment of e-mobility.

References

[1] Federal Ministry of Economics and Technology (2009) Nationaler Entwicklungsplan Elektromobilität der Bundesregierung. <http://www.bmwi.de/> (accessed 11.10.2010)

[2] GaBi 4 (2011) Software and database management system for Life Cycle Engineering. LBP University of Stuttgart in cooperation with PE International, Leinfelden-Echterdingen

[3] Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (2009) Langfristszenarien und Strategien für den Ausbau erneuerbarer Energien in Deutschland, Leitszenario 2009. <http://www.bmu.de> (accessed 10.8.2010)

[4] Daimler AG (2010) Mercedes-Benz Cars, Environmental Certificate A-Class 2008. <http://sustainability.daimler.com> (accessed 1.8.2010)

[5] Daimler AG (2010) Mercedes-Benz Cars, Environmental Certificate B-Class 2008. <http://sustainability.daimler.com> (accessed 1.8.2010)

[6] Volkswagen AG (2010) The Golf - Environmental commendation, background report 2008. <http://www.volkswagen.com/> (accessed 1.8.2010)

[7] Lindegger M (2009) Wirtschaftlichkeit, Anwendungen und Grenzen von effizienten Permanent-Magnet-Motoren, Zusammenfassung und Update. Eidgenössisches Departement für Umwelt, Verkehr, Energie und Kommunikation UVEK, Bundesamt für Energie BFE

[8] Rommel W (2009) Szenarioanalyse als Methode für strategische Entscheidungen am Beispiel des Ressourcenmanagements, Wirtschaftsphilologentag, Passau

[9] Pressetext Deutschland (2010) Forschergruppe optimiert Hochleistungs-magnete-Funktionalität von Elektromotoren soll trotz Rohstoffknappheit erhalten bleiben. <http://www.pressetext.de> (accessed: 08.09.2010)

[10] Gaines L, Cuenca R (2000) Cost of Lithium-Ion Batteries for Vehicles. Argonne National Laboratory, Center for Transportation Research

[11] International Battery (2011) Lithium Nickel Cobalt Manganese Oxide – Material Safety Data Sheet (MSDS). <http://www.internationalbattery.com> (accessed 01.03.2011)

[12] KOKAM (2011) Material Safety Data Sheet – Superior Lithium Polymer Battery (SLPB). <http://www.pse.nl> (accessed 1.03.2011)

[13] Dewulf J et al (2009) Recycling rechargeable lithium ion batteries: Critical analysis of natural resource savings. Resour Conserv Recy doi:10.1016/j.resconrec.2009.08.004

[14] Ishihara K et. Al (2002) Environmental burdens of Large Lithium-Ion Batteries Developed in a Japanese National Project. Central Research Institute of Electric Power Industry, Japan

[15] ADAC Fahrzeugtechnik (2010), Der ADAC-Autotest. http://www.adac. de (accessed 11.10.2010)

[16] European Union (2010) Richtlinie 70/220/EWG des Rates zur Angleichung der Rechtsvorschriften der Mitgliedstaaten über Maßnahmen gegen die Verunreinigung der Luft durch Emissionen von Kraftfahrzeugen. <http://eur-lex.europa.eu> (accessed 11.10.2010)

[17] Baumann M (2010) Development of a methodical approach for the description of the use phase of electric mobility concepts in life cycle assessment. Fraunhofer Institute for Building Physics, Leinfelden-Echterdingen

[18] TREMOD (2010) Handbook Emission Factors for Road Transport (HBEFA) version 3.1

[19] Wallentowitz H (2007) Strategien zur Elektrifizierung des Antriebstranges: Technologien, Märkte und Implikationen. Vieweg+Teubner Verlag, Wiesbaden

[20] EUCAR (2010) The Electrification of the Vehicle and the Urban Transport System 2009. <http://www.acea.be> (accessed 11.10.2010)

[21] Barenschee E (2010) Wie baut man Li-Ionen-batterien? Welche Herausforderungen sind noch zu lösen? Evonik Industries, <http://media.nmm.de> (accessed 11.10.2010)

A Consistency Analysis of LCA Based Communication and Stakeholders Needs to Improve the Dialogue on New Electric Vehicle

Stephane Morel, Tatiana Reyes and Adeline Darmon

Abstract The launch of new technologies such as electric vehicles will be a major change on several levels such as new business models and possible changes of consumer's habits. The results of the life cycle assessment (LCA) are key as they will be used for decision support for governmental policies, for vehicle design, and finally to disclose environmental data to specific stakeholders around the world. It was clear that only a multidisciplinary approach could open a successful way in this work. A new methodology is proposed, by crossing the capability of understanding and action, through the human development index and level of awareness by using the environmental performance index. Based on these indexes it is possible to assess the "eco-maturity" level of many countries. Main stakeholder needs in environmental information are then screened. The various life cycle based communication are analysed on four axes, feasibility, life cycle coverage, ability to help the decision and finally educate the consumer. An example is calculated to show concrete facts and the LCA communication strategy wheel is created to determine the right effort to provide, toward the right target, in the most efficient way.

1 Introduction

The launch of new technologies such as electric vehicles will be a major change on several levels such as new business models and possible changes of consumer's habits. The results of the life cycle assessment (LCA) are key as they will be used for decision support for governmental policies, for vehicle design, and finally to disclose environmental data to specific stakeholders around the world.

Within the first stage of LCA, the definition of an appropriate impact assessment method is a key point of the study. In our case, this choice significance is emphasised when coming to comparative assessment of new products & services.

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As a consequence, a thorough analysis is required to clarify actual needs for LCA communication between stakeholders and an appropriate way to disclose results for the comparative assessment of technologies. This paper discusses this issue and explains how a methodology was created to make recommendation to disclose the new electric vehicle LCA results.

2 Materials and methods

For each country, specific needs are identified by crossing the human development index and the environmental state. Eight stakeholders have been identified, among them; we will find policy makers, suppliers, company's decision makers, fleet customers, financial investors, etc. For each of them, their needs have been qualified and a ranking as been made based on the importance of LCA information disclosure in the dialogue for their specific countries. It will allow us to determine the right communication path for each stakeholder and country.

2.1 The human development index & the environmental performance index

The human development index (HDI) is a summary measure of human development. It measures the average achievements in a country in three basic dimensions of human development: a long and healthy life, access to knowledge and a decent standard of living (see Table 1). The HDI is the geometric mean of normalised indices measuring achievements in each dimension as describe in [1].
The second index chosen for this work is the environmental performance index (EPI). This index tracks national environmental results on a quantitative basis, measuring proximity to an established set of policy targets using the best data available [2] (see Table 2).

Tab. 1: Construction of the human development index from three dimensions to the final aggregated index

3 Dimensions =>	4 Indicator =>	Dimension index=>	Aggregated index
1/ Long and healthy life	Life expectancy at birth	Life expectancy index	Human development index (HDI)
2/ Knowledge	>Mean years of schooling >Expected years of schooling	Education index	
3/ A decent standard of living	Gross national income (GNI) per capita	GNI Index	

Tab. 2: Construction of the environmental performance index from ten policy categories to the final aggregated index (adapted from [3])

10 Policy categories* =>	2 Objectives =>	Aggregated index
Air pollution (effects on ecosystems)	Ecosystem vitality	Environmental performance index
Water (effects on ecosystems)		
Biodiversity & habitat		
Forestry		
Fisheries		
Agriculture		
Climate change		
Environmental burden of disease	Environmental health	
Water (effects on humans)		
Air Pollution (effects on humans)		

*25 performance indicators describe the ten policy categories.

For this paper, we will focus on the ecosystem vitality index to reflect the environmental condition of the countries.
By crossing these two indexes, it is possible to enlighten the situation of 153 countries around the world, covering all the continents as shown in Figure 1.

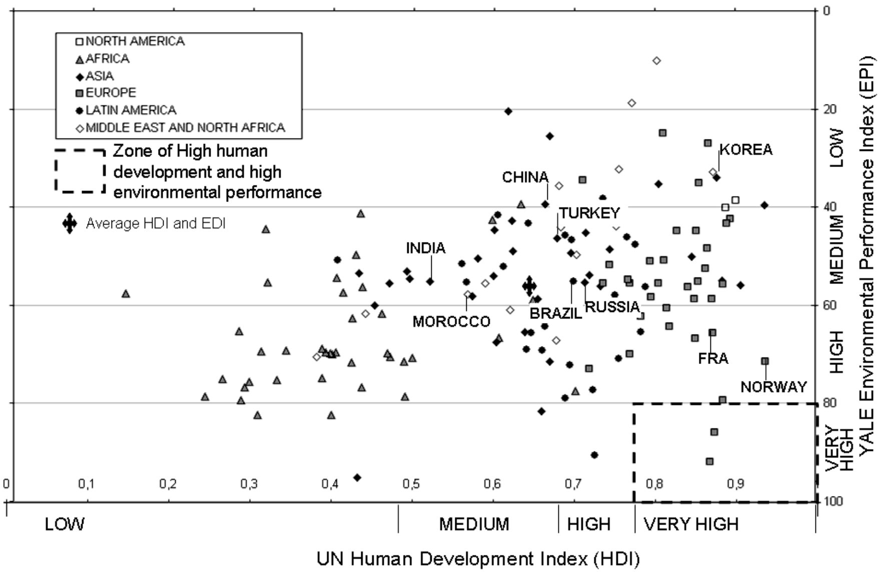

Fig. 1: Mapping of countries according to the HDI and EPI (ecosystem vitality) indexes

Eight countries will be emphasised since they cover 50% of Renault group worldwide sales and are representative of each region: France, Korea (Republic of south), Brazil, Turkey, Morocco, Russia, China, India.

Norway is a world leading country on the HDI Index. Costa Rica is one of the leaders in term of environment. Iceland, Switzerland and Sweden are the three countries reaching the "sustainability zone" where human development is achieved without compromising the environment.

The selected countries can be sorted in 5 "eco-maturity" levels presented in Table 3.

Tab. 3: Classification of countries in "eco-maturity" levels

"Eco-maturity" level	Countries	Human development index	Environmental performance index
Level 1: Mastering	Norway France	Very high	High
Level 2: Managing	Korea (Republic of)	Very high	Low
Level 3: Understanding	Russian Federation Turkey Brazil	High	Medium
Level 4: Recognising	India Morocco	Medium	Medium
Level 5: Recognising	China	Medium	Low

Note: A leading level "Excellence" could be given to Iceland and Switzerland

2.2 Stakeholders information needs

According to [4], stakeholders are in principle any party that has an interest (“stake”) in a company or its products. We will focus on customers, employees, public authorities, financial analysts, and suppliers who are considered as primary stakeholders.

Life cycle assessment is widely considered as the most transparent and complete tool to dialogue toward various stakeholders for its comprehensive approach with broad scope and system boundaries [5]. Nevertheless, expectations are different between each stakeholders and each country environmental maturity.

Based on a full life cycle study, it is possible to envisage several level of communication.

2.3 LCA based communication format

It is widely admitted that life cycle assessment is a trustful tool to evaluate the environmental potential impacts of products. Results can be published under the ISO 14025 scheme. Unfortunately, LCA results are often too complex to understand for a wide public.
It is possible to simplify the dialogue by reducing the environmental scope, such as product carbon footprint focussing only on greenhouse gases or focusing only on the use stage of the product.
A second possibility is to sum several impacts into a single score. In that case, it is of utmost importance not to disclose this single score alone to the customer. This would be misleading and insufficient for a decision.
Finally, an ecological label is a seal or a logo indicating that a product has met a certain set of environmental attributes. As an example, Renault decided to propose an innovating approach with a life cycle environmental performance oriented label versus the classical "technology" oriented label. The effective impact reduction for the planet is possible only if the product remain economically accessible for all. This is the base statement for Renault ecological and economical label: Renault eco².
The different type of environmental communication are describe by four criteria [6] linked to the sustainable production (for the industry) and consumption (for the consumers) concept.

Tab. 4: Summary of the 7 life cycle communication possibilities

	Production		**Consumption**	
+: yes **- : no**	Is it easy to calculate?	Is it complete on life cycle & multicriteria	Does it help consumers to choose	Does it educate consumers
CO_2: ranking from [A..G]	+ + +	- - -	+	+
TECH: Technology label	+ +	-	-	-
PERF: Performance label	+ +	+	-	+
PCF: Product Carbon Footprint	+	+	+	+
SCOR: EcoScore	+	+	+ +	- -
ECO: EU Ecolabel	+	+ +	+ +	+ +
LCA: such as environmental product decleration	-	+ + +	-	+ + +

3 Results

3.1 LCA case study, the electric vehicle

- The studied product is a Renault Fluence ZE
- Definition of the functional unit according to the ILCD handbook [7] and the reviewed study of Renault Laguna [8]:

what	Transportation of persons alone or with passenger
how much	150 000 km
how long	10 years
in what way	Respect of the norms for M1 type approval

- Functional unit: Transportation of persons alone or with passenger for a distance of 150 000 km, during 10 years, in the respect of the norms for M1 type approval
- Product system: Renault Fluence Z.E. in Europe, 2011

3.2 LCIA priorisation

An LCIA allow the practitioner to transform inventories in potential impact on the environment. These are various in term of importance for each specific product and present more or less uncertainties on the characterisation methodologies.
Several workshops conducted by Renault with more than 40 eco-design/LCA researcher or student. Several methods were tested such as willingness to pay, preferences, auctions or cultural choices. The result for the automotive products is a larger interest in global warming potential 100 years (GWP), abiotic resource depletion potential (ADP) and photochemical ozone creation potential (POCP). We will keep those for this paper even if Renault assessed other potential impacts categories in its studies.

3.3 Application on the 7 communication levels

Electric vehicle LCAs vary very significantly from one country to another. This is a consequence of the political choices regarding the primary resources selected to produce the electricity, necessary to fill the battery and drive. As an example, four countries are compared in Table 5.

Tab. 5: Summary of the 7 life cycle communication possibilities

	2010 average car in Europe	**Switzerland (hydro 50% and nuclear power plant)**	**France (nuclear power plant, 77%)**	**Netherland (natural gas power plant, 57%)**	**Germany (coal power, 49%)**
CO_2: ranking from [A..G]	C	A	A	A	A
TECH: Technology label	dCi or TCe	Renault ZE	Renault ZE	Renault ZE	Renault ZE
PERF: Performance label		Renault eco²	Renault eco²	Renault eco²	Renault eco²
PCF: Product Carbon Footprint and EcoScore	100%	32%	36%	72%	69%
SCOR: EcoScore UBP	100%	27%	59%	61%	69%
ECO: EU Ecolabel	NO	NO	NO	NO	NO
LCA: CML2001 GWP(100yr) ADP POCP	 100% 100% 100%	 32% 54% 23%	 36% 91% 24%	 72% 68% 25%	 69% 60% 25%

Note: Electric vehicle results are normalised in regard of the 2010 European average Renault passenger car sold in Europe.
Note: The results are calculated with the GaBi Software(R) and Renault simplified LCA model

3.4 Results for the communication and discussion

On the basis of the human development index and the environmental performance index, it is possible to assess the "eco-maturity" level of one country. Main stakeholder needs in environmental information are screened. The various life cycle based communication are analysed on four axes, feasibility, life cycle coverage, fulfil its role to help the decision and educate. An example is calculated to show concrete facts.
When crossing these analysis and ranking, it opens the possibility to define with communication is suitable for each stakeholder in each country, especially toward the supplier and the market as shown on Figure 2 with the new created LCA communication strategy wheel.

This assessment tool will allow the group to adopt the right communication level for each party today. The LCA communication is now adapted to the capability to understand and then the capacity for acting.

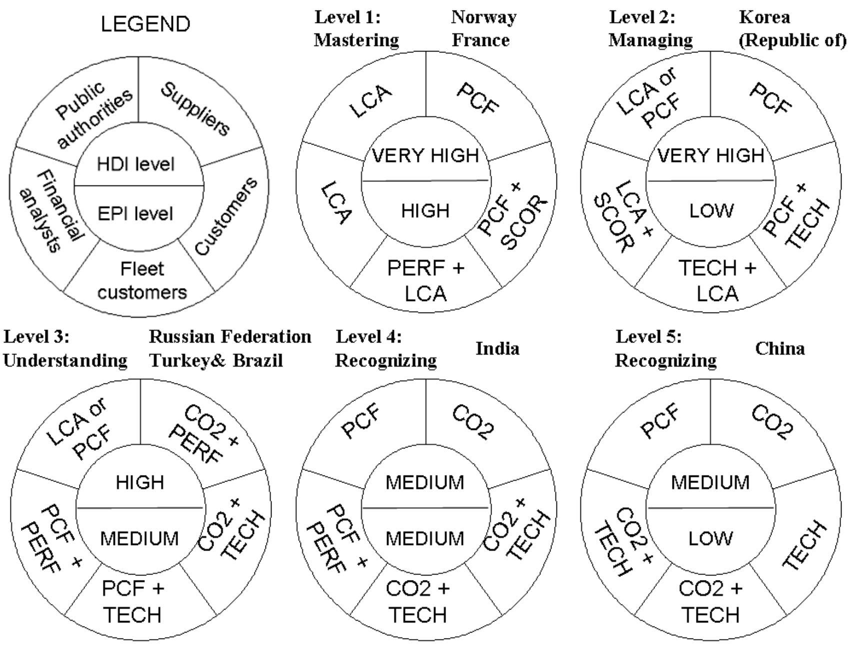

Fig. 2: LCA communication strategy wheel for each "eco-maturity" level

In most advanced countries, LCA could be widely used, it could be sometime useful to add a score index to an LCA or PCF, but Score shall never be used alone for public information as stated in the ILCD guidelines [9].
Staring from the lowest "eco-maturity" level to the highest, it is possible to read the road to follow to achieve an LCA communication in line with people expectations.

4 Conclusions

When performing an LCA study, the choice of the LCIA method is a key point to ensure the quality of the dialogue within the actors of the value chain. Since LCA development is accelerating worldwide, it brings the necessity to deepen this question.

This article describes a methodology to consider and propose relevant indicators for the dialogue on automotive technologies. The communication type are nevertheless related to the automotive sector environmental stakes and shall be adapted if this method would be considered for electronic or textile products were the water is a more important topic. Water footprint were not considered at this time due to the current development on-going but could be surely added later.

Governmental level was difficult to link with the indexes and a ground study should be performed with a network of public affair responsible. Another possibility would be to assess the potential role of LCA when applying the ISO 26000 [10] which highlights the 7 principles of social responsibility including the environment, consumer issues and community involvement topics.

Finally, this approach also open the way for a time dynamic strategy making by looking forward on the human development index and environmental performance index, it makes it possible to forecast the coming years needs in term of environmental information.

References

[1] United Nations (2010) Human Development Report 2010. United Nations, New York

[2] Emerson J, Esty DC, Levy MA, Kim CH, Mara V, de Sherbinin, Srebotnjak T (2010) 2010 Environmental Performance Index. Yale Center for Environmental Law and Policy, New Haven

[3] Emerson J, Esty DC, Levy MA, Kim CH, Mara V, de Sherbinin, Srebotnjak T (2010) 2010 Environmental Performance Index – Summary for Policy Makers. Yale Center for Environmental Law and Policy, New Haven

[4] UNEP (2007) Life Cycle Management - a business guide to sustainability. UNEP, Paris

[5] Schmidt WP, Morel S, Ruhland K, Krinke S, Kunne B (2010) Feasibility of a globally harmonised Environmentally Friendly Vehicle concept. Int J Veh Des 53:4 – 15.

[6] Morel S et al (2010) From LCA studies to the first automotive self declared life cycle environmental label. SAM4 Conference, Nancy

[7] European Commission (2010) ILCD Handbook-General guide for LCA, European Union, Brussels

[8] Morel S et al (2007) Analyse du cycle de vie de nouvelle Renault Laguna. Renault, Paris

[9] European Commission (2010) ILCD Handbook-LCIA Framework and requirements. European Union, Brussels

[10] ISO 26000 (2010) Guidance on Social Responsibility. ISO, Geneva

Design for Environment and Environmental Certificate at Mercedes-Benz Cars

Klaus Ruhland, Rüdiger Hoffmann, Halil Cetiner and Bruno Stark

Abstract Life cycle assessment (LCA) is used as a tool for design for environment (DfE) to improve the environmental performance of the Mercedes Car Group products. For new models a brochure including an environmental certificate and comprehensive data for the product are published. This environmental certificate brochure reports on processes, data and results based on the international standards for life cycle assessment (ISO 14040/44) [1,2], for environmental labels and declarations (ISO 14020-21) and for the integration of environmental aspects into product design and development (ISO TR 14062), which are accepted by all stakeholders [3]. Furthermore, the DfE process is representing the key element of the environmental management system (ISO 14001) of the R&D organisation at Mercedes-Benz Car Group. The compliance with these international standards and the correctness of the information contained in the certificate are reviewed and certified by independent experts. In 2005, the Mercedes-Benz S-Class became the world's first automobile to receive an environmental certificate. It has now also been granted to the C-Class, the A-/B-Class, the GLK, the E-Class, the new CLS and SLK, and the S 400 HYBRID [4].

1 Introduction

If the environmental compatibility of a vehicle is to be improved, it is important that its emissions and consumption of resources is reduced throughout the whole of its life cycle. The extent of the ecological burden caused by a product is already largely defined during the early development phase. Later corrections of the product design are only possible at great cost and effort. The earlier "Design for Environment" is integrated into the development process, the greater the benefits in terms of minimising environmental effects and costs. Process- and product-integrated environmental protection must be realised during the development

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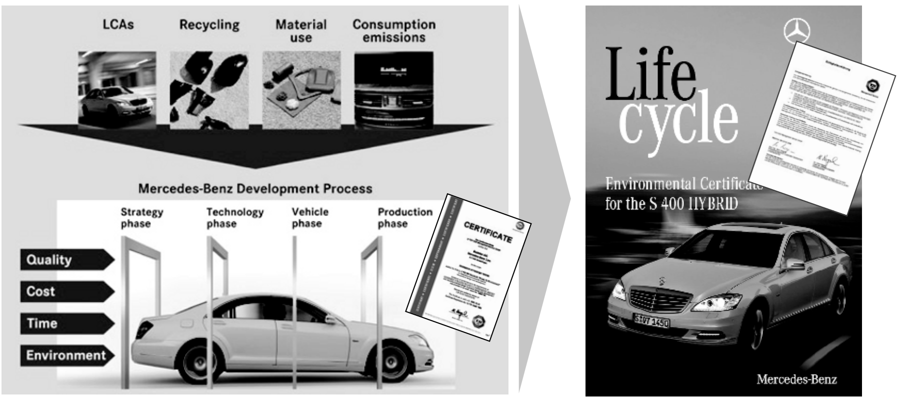
phase of a product. Later on, environmental effects can often only be reduced by downstream, “end of-the-pipe” measures.
“We develop products which are particularly environmentally compatible within their market segment” – this is the Daimler Group’s second environmental protection guideline. The aim is to improve environmental compatibility in an objectively measurable way, while meeting the demands of the increasing number of customers who are mindful of such environmental aspects as lower fuel consumption and emissions or the use of environmentally friendly materials.
The implementation of the Design for Environment Process at Mercedes-Benz will be explained in the following sections and demonstrated by the exemplary case study of the S-Class Hybrid model (S 400 HYBRID).

2 Design for environment process and organisation

The responsibility for improving environmental compatibility was an integral part of the organisation of the S 400 HYBRID development project. The so-called DfE team was made up of experts from the fields of life cycle assessment, dismantling and recycling planning, materials and process engineering, as well as design and production. This guarantees complete integration of the DfE process in the vehicle development



Design for Environment Process of MBC/ D certified according to ISO 14062 by TÜV Süd.

Brochure documents the improvement in environmental compatibility.

Fig. 1: Process design for environment and brochure environmental certificate

The integration of Design for Environment into the process organisation of the S400 HYBRID development project ensured that environmental considerations would be taken into account in the early development stages, and not just prior to

market launch. Appropriate objectives were agreed upon early on and reviewed at relevant quality gate stages during the development process. Together with the S-Class HYBRID project management, the DfE team had defined concrete environmental objectives in the areas: prohibited substances, recyclability, use of recycled plastics and renewable raw materials and reduction of all substantial environmental burdens caused by the S-Class HYBRID during its lifecycle (ecological life cycle assessment).
To provide the best possible service to as many interested parties the results were documented in a brochure called Environmental Certificate. This brochure documents the clear improvements which have been achieved through the implementation of hybrid technology, as compared with the S 350 reference vehicle. Both the process of design for environment and the Environmental Certificate have been certified by independent experts according to internationally recognised standards. The process carried out for the S 400 HYBRID meets all the criteria for the integration of environmental aspects into product development which are described in ISO standard 14062.

3 LCA results: S 400 Hybrid in comparison with S 350

Over the entire life cycle of the S 400 HYBRID, the Life Cycle Analysis calculations indicate, for example, a primary energy consumption of 1093 gigajoules (equal to the energy content of about 33.5 tonnes of premium-grade petrol) and the input into the environment of around 78 tonnes of carbon dioxide, about 32 kilograms of nonmethane hydrocarbons, about 40 kilograms of nitrogen oxides and about 68 kilograms of sulphur dioxide.
In addition to the analysis of overall results, the distribution of single environmental impacts among the different phases of the life cycle is investigated. For carbon dioxide emissions and also primary energy consumption, the use phase dominates with a share of over 85 per cent. However, it is not the use of the vehicle alone which determines its environmental compatibility. Some environmentally relevant emissions are caused principally by its manufacture, for example the sulphur dioxide emissions. For this reason the manufacturing phase must be included in the analysis of ecological compatibility.
For a great many emissions today, the dominant factor is not so much the vehicle operation itself, but the production of the fuel, for instance for hydrocarbons and nitrogen oxides and for the environmental impacts which they essentially entail: photochemical ozone creation potential (POCP: summer smog, ozone) and acidification potential (AP).

Parallel to the examination of the S 400 HYBRID an LCA was made of the comparable S 350 petrol engine model in the ECE basic variant version.

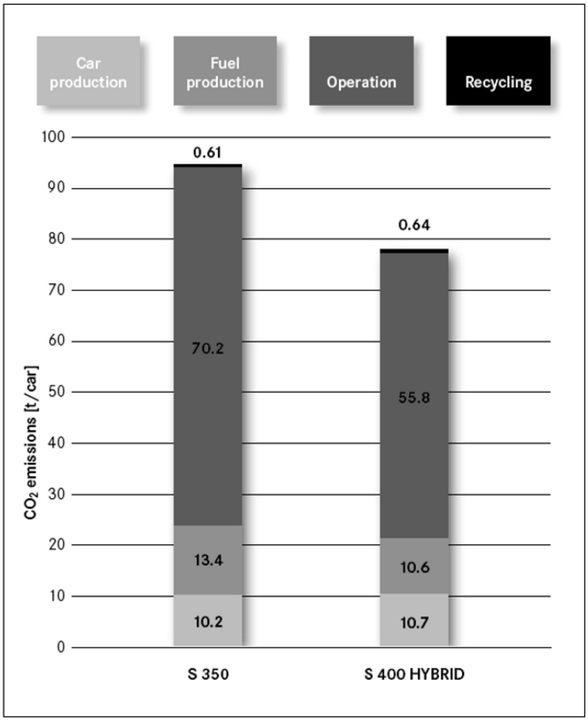


Fig. 2: Comparison of carbon dioxide emissions - S 400 HYBRID vs. S 350 [t/car]

As shown in Figure 2, the production processes for both vehicle models show similar levels of carbon dioxide emissions. But a clear advantage emerges for the S 400 HYBRID over the total life cycle. At the beginning of its life cycle, production of the S 400 HYBRID causes slightly higher carbon dioxide emissions than the S 350 reference vehicle (10.7 tonnes of carbon dioxide in total).
The reason for this is the - in part - demanding production of additional components for the drive system (above all, the battery). In the operating phase which follows, comprising fuel production and vehicle operation, the S 400 HYBRID emits approximately 66 tonnes of carbon dioxide; the total, therefore, for production, operation and recycling comes to 78 tonnes of carbon dioxide.
The production of the comparable S 350 with petrol engine uses 10.2 tonnes of carbon dioxide. In the operating phase, the S 350 emits 84 tonnes of carbon dioxide due to its higher fuel consumption. Carbon dioxide emissions come to about 94 tonnes in total. When looking at the total life cycle – comprising

production, operation over 300,000 kilometres and recycling - the S 400 HYBRID causes 18 per cent (16.6 tonnes) fewer carbon dioxide emissions than the S 350.
In Figure 3, emission into the air and the relevant impact categories are shown compared across the individual life cycle phases. In each case, the results for the S350 in the production phase are slightly more favourable; however the S 400 HYBRID displays a clear advantage over the total life cycle.

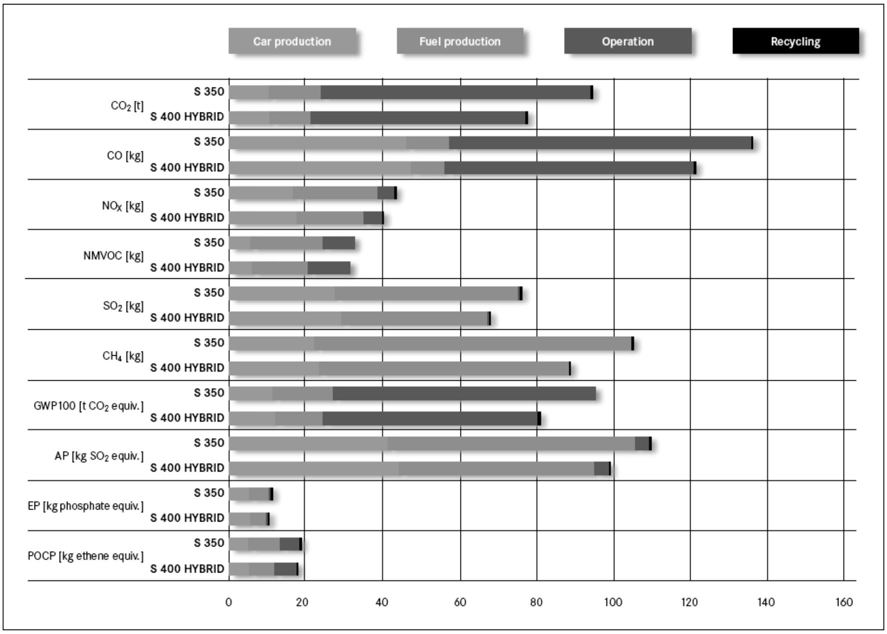


Fig. 3: Comparison of selected parameters for the S 400 HYBRID and S 350

Total consumption of resources is reduced by 18 per cent as well (ADP = abiotic depletion potential). The individual values named below show the changes in detail (cf. Figure 4): through the slight changes in material mix, the material resources demand also changes for the production phase of the S 400 HYBRID.

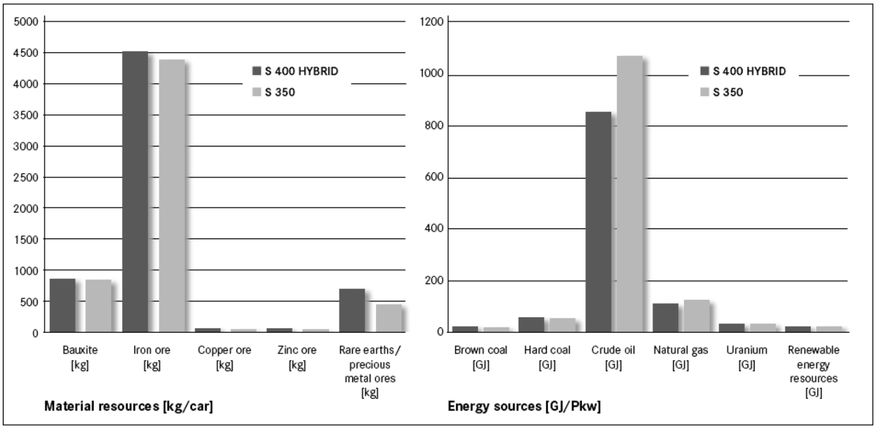


Fig. 4: Comparison of selected material and energy sources [unit/car]

For example, the bauxite demand is higher because of the use of more aluminium, as is the iron ore consumption. The lower energy resources demand (natural gas and crude oil) is above all due to the significantly reduced fuel consumption in vehicle operation. Compared to the reference vehicle, 17 per cent of primary energy is saved over the total life cycle. The reduction in primary energy demand of 231 GJ is equivalent to the energy content of around 7000 litres of petrol.

4 Recycling concept of the S 400 HYBRID

The method for calculating the recyclability of cars is laid down in the ISO standard 22628 and is divided into the following four steps:

1) Pre-treatment (removal of fluids, tyres, the battery and catalytic converters)
2) Dismantling (removal of reuse parts or components for material recycling)
3) Separation of metals in the shredder process
4) Treatment of non-metallic residual fraction (shredder light fraction, SLF)

The recycling concept for the S-Class HYBRID was designed in parallel with the vehicle development process, including analysis of the individual components and materials for each stage of the process. On the basis of the quantitative flows stipulated for each step, the recyclability and recoverability rate for the overall vehicle is determined. The deployment for the first time of a lithium ion battery in a hybrid series model has presented new challenges in the area of disposal and recycling as well. Working together with the suppliers and waste disposal partners, an innovative recycling concept has been developed which makes it possible to recover valuable content materials.

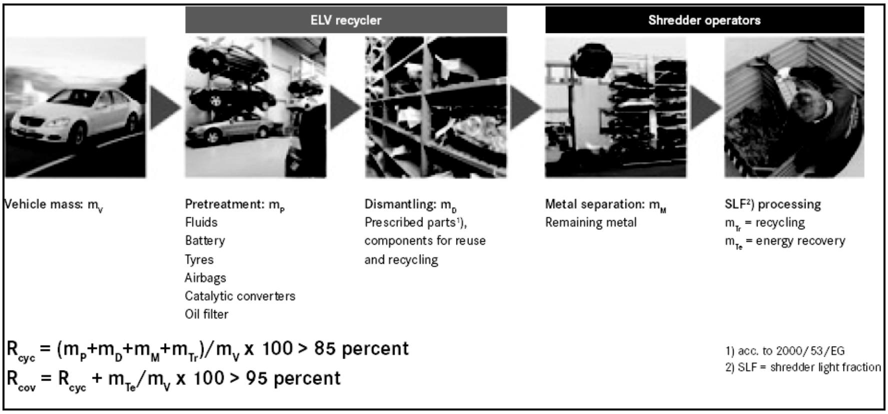


Fig. 5: Recycling concept of the S 400 HYBRID

To improve recycling, numerous components are dismantled -either for direct sale as used replacement parts or for material recycling with economically worthwhile methods. These include aluminium and copper components as well as certain large plastic parts. All plastic components are marked in accordance with the international nomenclature. During the subsequent shredder process for the remaining body shell, the metals are separated for recycling in raw materials production processes. The remaining, mainly organic fraction is separated into different categories and reprocessed into raw materials or energy in an environmentally sound manner.
All in all, with the process chain described and in accordance with the ISO 22628 calculation model, a recyclability rate of 85 per cent and a recoverability rate of 95 per cent has been established for the S 400 HYBRID within the framework of the approval for the vehicle model.

5 Avoidance of potentially hazardous materials

The avoidance of hazardous materials is the top priority during the development, production, operation and recycling of our vehicles. Since as early as 1996, for the protection of both humans and the environment, our in-house standard DBL 8585

has specified those materials and material categories that may not be incorporated into the materials or components used in Mercedes passenger cars.
This DBL standard is already available to designers and materials specialists at the pre-development stage, during the selection of materials and the planning of production processes. Heavy metals forbidden by the EU End-of-Life Vehicle Directive are also covered by this standard. Materials used for components in the passenger compartment and boot are subject to additional emissions limits which are also defined in DBL 8585 as well as in component-specific delivery instructions. The continuous reduction of interior emissions is a major aspect of component and materials development for Mercedes-Benz vehicles.

6 Use of secondary raw materials

The main focus of the research into the use of recycled materials accompanying vehicle development is on thermoplastics. In contrast to steel and ferrous materials, to which secondary materials are already added at the raw material stage, recycled plastics must be subjected to a separate testing and approval process for the component in question.

Fig. 6: Use of recycled materials in the S-Class

In the S-Class, a total of 45 components with a total weight of 21.2 kilograms can be made in part from high-quality recycled plastics. The potential application

areas for the use of recycled plastics are restricted to non-visible areas of the vehicle and have by now been by and large exhausted. Typical applications include wheel arch linings, cable ducts and undercarriage panels, which are mainly made from polypropylene (see Figure 6).
Another objective is to obtain recycled materials from vehicle-related waste streams where possible, thereby creating material cycles. For example, the front wheel arch linings of the S-Class are produced from reprocessed vehicle components: starter battery housings, bumper coverings from the Mercedes-Benz Recycling System, and production waste from cockpit units.

7 Use of renewable raw materials

The use of renewable raw materials in vehicle production focuses on interior applications. The natural fibres predominantly used in series production of the S-Class are coconut, cotton and wool fibres in combination with various polymers. In total, 27 components with a combined weight of just under 43 kilograms and made using natural materials are used in the S-Class. An overview of the types of renewable raw materials used and their fields of application are shown in Figure 7.

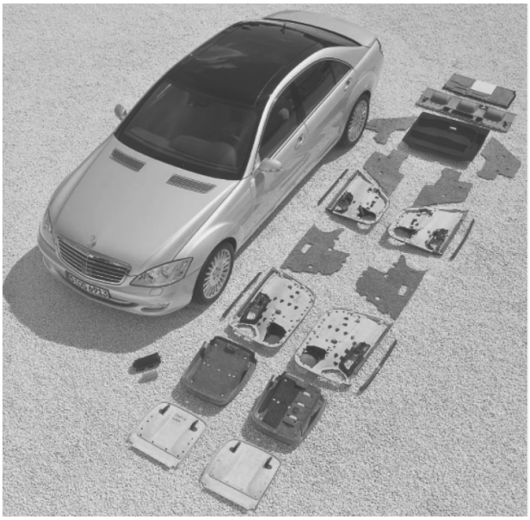


Fig. 7: Use of renewable raw materials in the S-Class

The fastening elements on the backrest trim are made directly from the production scrap in the manufacture of the trim part, thereby enabling an internal materials circulation process to be realised. The fasteners are produced using an injection moulding process, the first time renewable raw materials have been used in such a way in series production.

8 Conclusion

The new Mercedes-Benz S-Class S 400 HYBRID not only meets the highest standards in terms of safety, comfort, responsiveness and design, but also satisfies all current requirements with regard to environmental compatibility.
The environmental certificate documents the clear improvements which have been achieved through the implementation of hybrid technology, as compared with the S 350 reference vehicle. Both the process of design for environment and the product information have been certified by independent experts according to internationally recognised standards.
Mercedes-Benz was the world‘s first vehicle brand to possess this demanding certification, which was awarded in 2005 by TÜV Süd. Thus, the S 400 HYBRID sets not only new standards with regard to engineering, innovation and driving enjoyment. S 400 HYBRID owners can also enjoy a vehicle that, relative to other cars in its class, has very good fuel consumption, very low emissions, a comprehensive recycling concept, and uses a high proportion of renewable raw materials and high-quality recycled materials – in short, an excellent result in terms of life cycle assessment.

References

[1] ISO 14040 (2006) Environmental Management – Life Cycle Assessment – Principles and Framework. ISO, Geneva
[2] ISO 14044 (2006) Environmental Management – Life Cycle Assessment – Principles and Framework. ISO, Geneva
[3] Finkbeiner M, Hoffmann R, Ruhland K, Liebhart D, Stark B (2006) Application of Life Cycle Assessment for the Environmental Certificate of the Mercedes-Benz S-Class. Int J Life Cycle Assess 11:240-246.
[4] Daimler (2009) Environmental Certificate of the S 400 Hybrid. Daimler AG, Stuttgart - http://www.daimler.com

Implementing Life Cycle Engineering Efficiently into Automotive Industry Processes

Stephan Krinke

Abstract Life cycle assessment (LCA) is a powerful tool which supports life cycle engineering. It can be used as an environmental management instrument within the product development. For successful life cycle engineering the formal incorporation of life cycle thinking into the company policy is a necessary pre-requisite. Additional success factors which have to be met are the transformation of LCA results into measurable targets for engineers. Based on given environmental targets, such as a certain target value for greenhouse gas emissions, LCA can be used to calculate a specific technical target such as the weight of a component or the fuel consumption of a vehicle. The transformation of LCA results into measurable targets, show the added value which LCA can give in terms of life cycle engineering.

1 Introduction

The automotive industry is since decades one of the industrial focus areas for environmental technologies and environmental protection. But how can the environmental performance of a complex product such as an automobile be measured? The aspects which directly or indirectly influence the environment are manifold:

Starting with the production of raw materials and going along the value chain the entire production affects the environment. Especially the automotive industry is one of the industry sectors with a very complex value chain, including nearly all kind of materials such as metals, polymers, glass and ceramics. The usage phase of the vehicle also effects the environment due to the combustion of fuel and the

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herewith linked emissions such as CO_2, contributing to climate change. Other tailpipe emissions such as carbon monoxide, nitrogen oxides and hydrocarbons contribute to local air quality (e.g. summer smog). The quantity of these impacts depends on the fuel consumption and the emission standard of the vehicle. And last but not least the driving behaviour of the customer, which influences the fuel consumption, has an impact on the environment. During the end-of-life phase materials are recovered or recycled from the end-of-life vehicle and can be used as secondary (raw) materials in other applications.

Therefore the environmental assessment of a vehicle has to cover the entire life cycle. One of the most suitable instruments to measure the potential environmental impact of a product is life cycle assessment [1, 2]. Volkswagen started in 1991 with a research study for the life cycle inventory (LCI) of the Golf III which was published as first automotive LCI of a complete vehicle worldwide in 1996 [3]. In the following years LCIs of different vehicles of the Volkswagen Group were published [4, 5, 6, 7]. Today the Volkswagen Group has incorporated life cycle thinking as a main principle of the product development.

In the following chapters the environmental strategy of the Volkswagen group and the implementation of life cycle engineering into the environmental management will be explained.

2 Environmental strategy

The key element of the Volkswagen group strategy 2018 is to position the Volkswagen group as a global economic and environmental leader among automobile manufacturers. We intend to set new environmental standards in vehicles, powertrains and lightweight construction.

The main environmental challenges for the automotive industry are climate protection, health and local air quality and resource protection. Therefore these three items are incorporated into the environmental principles product of the Volkswagen Group.

The environmental strategy of the Volkswagen group has two main pillars: The environmental standard production and the environmental principles product.

Fig. 1: Environmental strategy Volkswagen Group

The Volkswagen Group is committed to developing vehicles and components in such a way that they have better environmental properties than their predecessors over the entire life cycle. Life cycle assessments are used to analyse and document the environmental performance of vehicles, technologies and processes.

3 Success factors for life cycle engineering

Whereas the first ideas of life cycle based analysis were developed 30 years ago, it still remains a challenge to implement life cycle assessment successfully into business processes. In this chapter we describe the key success factors:

- Integration into company policy and processes
- Reasonable time demand
- Reliable, meaningful and measurable targets for product development
- Communication strategy

3.1 Integration into company policy and processes

The commitment of the top management is crucial for the success of any environmental policy and strategy. For the Volkswagen group the environmental standards production and product are the basis for the environmental strategy. But for a successful environmental management system this is only the starting point.
For the Volkswagen brand we developed an environmental management scheme according to Figure 2. Responsible for the implementation of environmental aspects into the product development of the Volkswagen brand is the environmental officer product.

Fig. 2: Environmental management at Volkswagen

3.2 Reasonable time demand

LCA is a very comprehensive tool which offers a detailed insight in the environmental profile of a product. Performing an LCA of a complex product can be divided into two steps: The first is the data collection of the product. For every single component the material composition must be quantified and the herewith linked production process chain must be defined.
The second step is the transfer of this information into a LCA model. Within Volkswagen we developed an in-house solution which automatically builds up the

LCA model. The advantage of this approach is not only a huge reduction in time demand but also huge increase in consistency and quality of the LCA results. Therefore LCAs, even for complex products with a huge variety of different parts and materials, can be performed with a reasonable time demand [8].

3.3 Reliable and measurable targets for product development

Life cycle assessment is an environmental management tool that delivers scientific sound results, not less but also not more. A good environmental management system is therefore characterised by the capability to transform LCA results e.g. in terms of a greenhouse gas profile of different technologies into a technical target which can be understood, measured and monitored by an engineer. These targets can be fuel consumption, electric power consumption or the weight of a certain component. The real challenge of life cycle engineering is to bring together two different worlds: The world of the LCA expert, who models the product in terms of environmental impacts versus the world of the engineer who develops the product and takes technical measure that influence the environment.
In order to derive reliable, meaningful and measurable targets the following aspects should be considered.

Stable and reliable results

As LCA is a model outcome every LCA result should be tested by sensitivity analysis in order to prove whether the result, or much more important, a derived technical recommendation, is stable while varying certain model parameters or assumptions.

Data quality

Foreground data, such as the bill of materials, information about production processes or energy consumption in the use phase, describes the characteristics of the product. From our experience this foreground data is of major relevance and has a strong influence on the overall LCA results.
For the modelling of special materials, which are new on the market or based on new technologies, a data acquisition in cooperation with the respective supplier is strongly recommended. At that point a formal requirement for data acquisition of life cycle inventory data in the supplier contracts is very helpful [9].
For highly innovative industries, such as the automotive industry, it is of utmost importance to use high-quality inventory data in order to assess the environmental

profile of new materials and new technologies. This is a challenge because the knowledge of new technologies and new materials is of course limited. But this does not mean that LCA cannot be applied. Also in this case the area of unknown knowledge can be transformed by LCA into a measurable target value. If, for instance, the energy consumption of a new production process is still unknown, an LCA can be used to calculate the acceptable maximum energy consumption in order to achieve a certain environmental target.

Anyhow there is a parallel between the automotive industry and life cycle assessment: High-tech powertrains need high-quality fuels – high-tech LCA need high-quality inventory data. Therefore today and in the future the provision of high-quality inventory data will remain an important and necessary task, especially for science and consultancy.

3.4 Communication strategy

It is most important that products with special environmental features or technological improvements are also communicated as such. A company which invests in environmental performance and technology leadership should also use these characteristics in the communication and marketing. Therefore the challenge in that area is to translate LCA results into a communication that will be understood by the respective target groups. It is notable that economic and environmental optimisation oftentimes are in line, especially with regard to the fuel consumption. In the case of the automobile industry we have to differentiate between private customers and fleet customers as two different kinds of target groups. In most cases the customer focus is on fuel consumption and emission levels in terms of costs. But more and more customers reflect what we call total cost of ownership (TCO), which means that higher purchase prices can be amortised over life time at the customer.

For the Volkswagen brand we developed the so-called environmental commendations (www.environmental-commendation.com). Environmental commendations for new vehicle models and technologies highlight ecological progress compared with predecessor models and previous technologies. We use environmental commendations to inform our customers, our shareholders and other stakeholders how we are making our products and production processes more environmentally compatible and what we have achieved in this respect. The underlying LCA not only covers the time when the vehicle is on the road but its entire life cycle from production through to use and disposal. We engage in dialogue with our suppliers to identify environmental measures that can be taken. The information of environmental commendations is based on an LCA, which has been verified and certified by the technical inspection organisation TÜV NORD. The TÜV certificate confirms that the LCA is based on reliable data and that the methods used to compile it comply with the requirements of the ISO standards 14040 and 14044.

4 Examples of strategic and operational implementation of LCA

4.1 Intelligent lightweight design over life cycle

In this chapter we show how an LCA can be applied as environmental management tool within the product development. Lightweight design measures are chosen for this example.

Modern lightweight design is one of the key technologies for an efficient future mobility, because it contributes to reduce the fuel consumption and the herewith linked CO_2-emissions. Based on the Volkswagen's environmental standard for products this means that any lightweight design measure should also have an environmental benefit over the entire life cycle. Approximately one third of the fuel consumption in the NEDC (New European Driving Cycle) depends on the mass of the vehicle.

A comparison of different material concepts can be done by including the entire life cycle and by assessing products which fulfil the same functions. E.g. a body-in-white with the same crash performance will have different weights depending on the material-mix applied and the design of parts.

For lightweight concepts the herewith linked CO_2-emissions in the production phase are often, but not always, higher than those ones for conventional construction. These additional CO_2-emissions which occur in the production phase should be compensated as fast as possible during the use phase due to lower fuel consumption of the lightweight design. Only when we achieve the ecological break-even, we speak of an “intelligent” lightweight design as shown in Figure 3. Volkswagen has different strategies to reduce the weight of a vehicle. Besides different material concepts the integration of different functionalities contributes to reduce the number of parts. For the body-in-white construction, which accounts for up to 35% of the overall vehicle weight in the series production of passenger vehicles steel- and aluminium lightweight are established technologies.

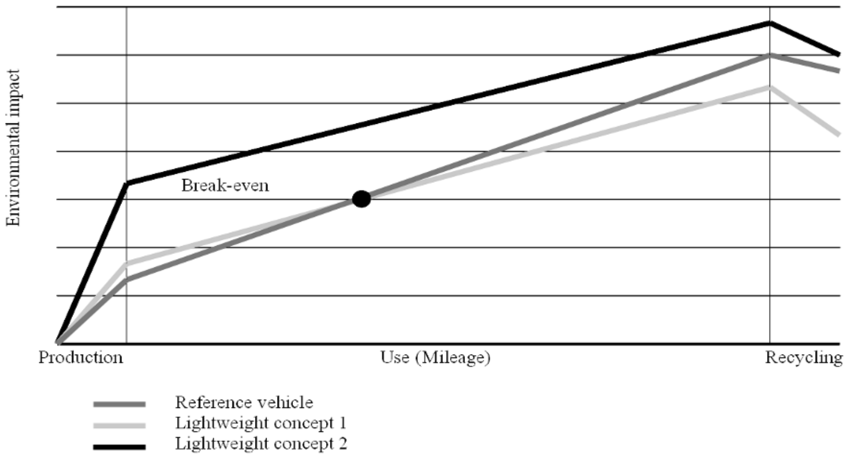

Fig. 3: Lightweight design and environmental break-even

The question whether a specific lightweight concept has a better greenhouse gas balance than a competing concept, can be answered only by assessing the entire vehicle. An important question is whether a lightweight measure will lead to a smaller engine or not. Of course any lightweight measure reduces the fuel consumption. But this reduction is low (0,15 l/100km*100kg for gasoline engines) compared with a fuel reduction of 0,35 l/100km * 100kg which can be achieved by additional measure of the engine.

What is clear from the above examples is, that it is not possible to make general claims along the lines that „material A is always better or always worse than material B“. Whether a lightweight design measure reduces life cycle greenhouse gas emissions or not will primarily depend on the following factors: the extent of the weight savings by material and material-adopted design, whether powertrain modifications can be implemented and the quality of secondary materials derived from the end-of-life vehicle. In practice, one and the same lightweight design measure might allow powertrain modifications to be made on vehicle A but might not, by itself, be sufficient to warrant such modifications on vehicle B. Therefore lightweight measures should always be assessed from the perspective of the entire vehicle and not from the part perspective.

Figure 4 shows the shows the main actors – and potential actions – at the different life cycle stages. For example the material manufacturing stage offers the opportunity to reduce specific CO_2-emissions per kg of material produced by reducing specific energy consumption and/or through the use of renewable, low-carbon energy sources. The use of secondary materials can help to reduce environmental impacts – for example cast alloys already use up to 90 % recycled content. On the process side, measures such as use of climate-friendly shielding gases as a replacement for SF_6, or the reduction of offcuts and scrap are important. In the vehicle use phase, fuel-efficient vehicle design measures by the OEM must be complemented and maximised through the optimal usage by customers e.g. by eco-driving trainings. Finally, at the recycling stage, reprocessing of lightweight materials into high-quality secondary materials will influence the range of potential applications of such materials and thus the associated positive environmental impacts.

LIFE CYCLE STAGE AND MAIN ACTORS	ASPECT	MEASURE
PRODUCTION (SUPPLIERS, OEM)	: Energy demand : Energy mix : Secondary resources : Process-specific aspects	: Reduced energy consumption : CO_2-reduced energy mix : Increased use of secondary resources : Optimized consumables
USE STAGE (OEM, CUSTOMER)	: Lightweight effect (massinduced, secondary measures, e.g. downsizing) : Customer behaviour	: Strict design in favour of fuel efficiency : Avoidance of unnecessary ballast, optimal tyre pressure, etc.
END-OF-LIFE (RECYCLING-INDUSTRY)	: Recovery of materials : Availability : Scrap quality : Secondary material quality	: Installation of collection systems : Development of separation technologies : Enhancement of yield/quality

Fig. 4: Environmentally friendly lightweight design, aspects and measures

4.2 Well-to-wheel analysis

As part of the steady evolution and variation of power train concepts, there is an increasing demand for comparing the impacts on global warming of different configurations. Due to the rise of electric mobility it becomes obvious that not only tailpipe emissions have to be accounted for. Also upstream emissions have to be quantified in order to obtain a holistic image concerning the Global Warming Potential of power train technologies. Furthermore the overall energy demand along the whole value chain is of major interest at a time of decreasing energy resources. So as to support the strategic alignment of Volkswagen's power train and fuel strategy, a well-to-wheel analysis gives a concise overview of the environmental performance of future mobility pathways.

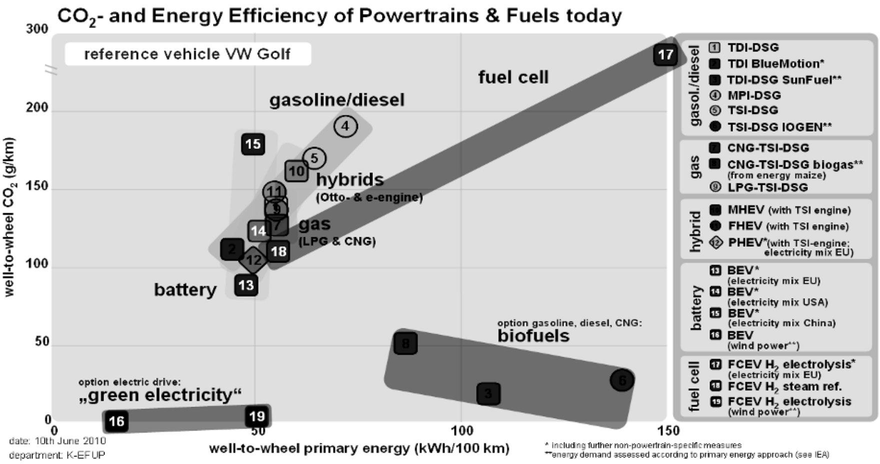

Fig. 5: Well-to-Wheel analysis of power trains and fuel options

Figure 5 illustrates well-to-wheel primary energy demand applied to well-to-wheel greenhouse gas emissions. Different power train technologies are being analysed in combination with the usage of different fuels. Taking into account the fuel production pathway as well as the tailpipe emissions, the production of the cars and their power trains is not included.

Internal combustion engine (ICE) driven cars show decreasing CO_2 emissions as well as energy demand in the course of advanced product development. Highly charged ICE technology (TSI) helps the gasoline driven cars to move from the multipoint injection (MPI) closer towards turbocharged diesel (TDI) efficiency. The fuel-efficient BlueMotion diesel Golf (TDI BlueMotion) gives an impression of the potential of ICE technology. Likewise, gas fuelled vehicles using natural gas (CNG) as well as liquefied gas (LPG) show similar efficiencies as the diesel technology.

Successive electrification – as shown by mild, full and plug-in hybrids (MHEV, FHEV, PHEV) – demonstrates a successive improvement in CO_2- and energy efficiency. The battery electric vehicle (BEV) powered by European electric energy (grid mix) shows the best overall performance. But when considering the same power train technology in another regional context, there is a wide range with respect to differing forms of power generation. Coal-biased electricity generation has a negative impact on the well-to-wheel performance. Otherwise the introduction of renewable energies shows a positive influence.

A fuel cell electric vehicle (FCEV) does not make sense if the used hydrogen derives from electrolysis powered by the grid mix. To be competitive with regard to other current technologies, hydrogen should be produced by natural gas steam reforming or by electrolysis with low CO2-emitting energy.

The most promising pathway for future mobility refers to electrically operated vehicles. Concerning the CO_2 profile, it does not make a difference whether the technology of choice is a FCEV or a BEV as long as the power comes from regenerative sources exclusively.

Also combustion engines are far away from reaching the end of their technological potential. Additionally, biofuels, as another expedient future option, can provide a meaningful contribution to sustainable mobility. Biogenic fuels offer an interesting prospect to reduce overall CO_2 emissions. Although the option to displace fossil fuels in a global scale is limited, it is possible to achieve a substitution rate of 10-20% in the market. Due to its comparatively extensive process chain of fuel supply there are disadvantages for most of the biofuel options in the field of energy efficiency.

What has emerged very obvious from this study is that in the context of evolving power train technologies upstream energy process chains are of increasing importance. In the end, future mobility has to implement alternative energy sources in combination with energy efficient technologies. Therefore, to move towards a sustainable mobility it is unavoidable to integrate life cycle aspects into technology assessment and the respective strategic decision processes.

5 Conclusions

As demonstrated above, LCA is a powerful tool which can be used as an environmental management instrument within the product development.
For successful life cycle engineering the formal incorporation of life cycle thinking into the company policy is a necessary pre-requisite. Additional success factors are the transformation of LCA results into measurable targets for engineers. Based on given environmental targets, such as a certain target value for greenhouse gas emissions, LCA can be used to calculate a specific technical target such as the weight of a component, the fuel consumption of a vehicle or the minimum amount of recycled content in a product. The transformation of LCA results into measurable target values, which can be understood by engineers, show the added value which LCA can give in terms of life cycle engineering. Even for very complex products with a huge variety of different materials and a complex value chain life cycle assessment can be performed within a reasonable time demand, with good quality and integrated efficiently into business processes.

References

[1] ISO 14040 (2006) Environmental Management – Life Cycle Assessment – Principles and Framework. ISO, Geneva

[2] ISO 14044 (2006) Environmental Management – Life Cycle Assessment – requirements and guidelines. ISO, Geneva

[3] Schweimer GW, Schuckert M (1996) Sachbilanz eines Golf. In: VDI Bericht 1307 Ganzheitliche Betrachtungen im Automobilbau

[4] Schweimer GW (1998) LCI 3-L Lupo. Volkswagen AG, Wolfsburg

[5] Schweimer GW, Bambl T (1999) LCI Seat Ibiza. Volkswagen AG, Wolfsburg

[6] Schweimer GW, Levin M (2000) LCI Golf A4. Volkswagen AG, Wolfsburg

[7] Schweimer GW, Roßberg A (2001) Sachbilanz Seat Leon. Volkswagen AG, Wolfsburg

[8] Koffler C, Krinke S, Schebek L, Buchgeister J (2007) Volkswagen slimLCI – a procedure for streamlined inventory modelling within LCA of vehicles. Int J Veh Des 46:172-188.

[9] VW Standard 91100. www.vwgroupsupply.com, Wolfsburg

Environmental Product Declaration of a Commuter Train

Kathy Reimann, Sara Paulsson, Yannos Wikström and Saemundur Weaving

Abstract The design for environment (DfE) approach of Bombardier applies a complete life cycle perspective using the methodology of life cycle assessment (LCA) and using environmental product declarations (EPDs) to provide transparent and reliable information on the environmental performance of Bombardier trains. This paper gives insight into the EPD of an electrical train and how the compiled life cycle information is used for comparison as well as improvement of Bombardier trains. The results of the performed LCA clearly show that the use phase dominates the environmental impact of the analysed commuter train in all of the input-related impacts as well as the selected output-related impact categories. In addition raw material extraction shows a significant impact in most impact categories. Through these results key environmental performance drivers are found and used for future design projects.

1 Introduction

The design for environment (DfE) approach of Bombardier applies a complete life cycle perspective using the methodology of life cycle assessment (LCA) according to ISO 14040 and ISO 14044 as an integrated part in the design process [1, 2]. Maximising energy and resource efficiency, minimising hazardous substance use and related toxic emissions as well as enhancing the overall product recyclability rate is the result of a high quality working process applied to product design and cascaded down the supply chain. Environmental product declarations (EPDs) following the international EPD® system provide a transparent and reliable way to communicate the efforts taken to improve the environmental performance of Bombardier trains. This paper gives insight into the EPD of an electrical train and how the compiled life cycle information is used for comparison as well as improvement of Bombardier trains.

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2 Development of the environmental product declaration

The EPD described in this paper provides environmental data gained through life cycle assessment. It also includes additional environmental information which is not based on LCA, e.g. noise and technical properties of the product which have an impact on the environmental performance. Apart from product related environmental information the EPD contains general technical information on the product and a short overview on the DfE process at Bombardier as well as on the environmental management systems (EMS) implemented at Bombardier. The EPD and underlying information are externally validated by independent verifiers for both the EU eco-management and audit scheme (EMAS) and the international EPD® system.

2.1 Design for environment at Bombardier

The integration of environmental sustainability into product development has a long proven history at Bombardier Transportation, where it has a core function in designing state-of-the-art rail transportation equipment.
The design for environment (DfE) approach, applying a complete life cycle perspective, is central to Bombardier's product responsibility strategy. Maximising energy and resource efficiency, minimising hazardous substances and related toxic emissions as well as enhancing the overall product recyclability rate is the result of a high quality working process applied to product design and cascaded down our supply chain. The Bombardier Transportation Design for Environment Centre of Competence, together with a wider DfE expert network, acts as a catalyst by providing the essential tools, expertise and central coordination in projects worldwide.

2.2 Life cycle assessment

An LCA according to ISO 14040 and 14044 was performed for an electrical train using the applicable Product Category Rules for Rail Vehicles (PCR 2009:05) as a basis for necessary assumptions [1, 2, 3]. The considered life cycle stages consist of the upstream module including raw material extraction and component production, the core module of final assembly and the downstream module including use and end-of-life (EoL). The data presented here is part of the verified EPD.

2.2.1 Scope and assumptions

The LCA was conducted for a functional unit of transporting one passenger over 100 km. The life cycle inventory (LCI) includes all resources from nature and emissions to nature for the upstream and downstream modules. Material and energy flows into and emissions from the core module are considered based on the cut-off rules set by the PCR. The total mass of the materials (components) not included in the LCA based on the cut-off rule must not exceed 5% of the total mass of the product, except for substances of very high concern (SVHC) as defined by the regulation (EC) No.1907/2006 – REACh for which no cut-off criterion is defined [4].
Data originates from different sources and is a mix between site/product specific data and generic data. Selected generic data representing country or regional averages is used for electricity, heat, steam, fuel and materials production processes. For the material composition specific data was acquired from the suppliers and for the core processes site-specific data is used.
The information regarding the end-of-life phase is based on specific scenarios which are technically and economically feasible and compliant with current practices and regulations.
The following impact categories were selected according to the PCR:

- Acidification potential (AP)
- Eutrophication potential (EP)
- Global warming potential (GWP)
- Ozone depletion potential (ODP)
- Photochemical ozone creation potential (POCP)

The life cycle impact assessment (LCIA) was calculated according to the CML 2001 impact assessment method [5]. LCI and LCIA were carried out both with and without regenerative braking.

2.2.2 Results of the life cycle inventory

The material composition of the studied train is shown in Fig. 1 including materials used during production and maintenance for 32 years of operation. An overview of resources used over the life cycle of the train without taking into account regenerative braking is shown in Tab. 1 and including regenerative braking in Tab. 2.

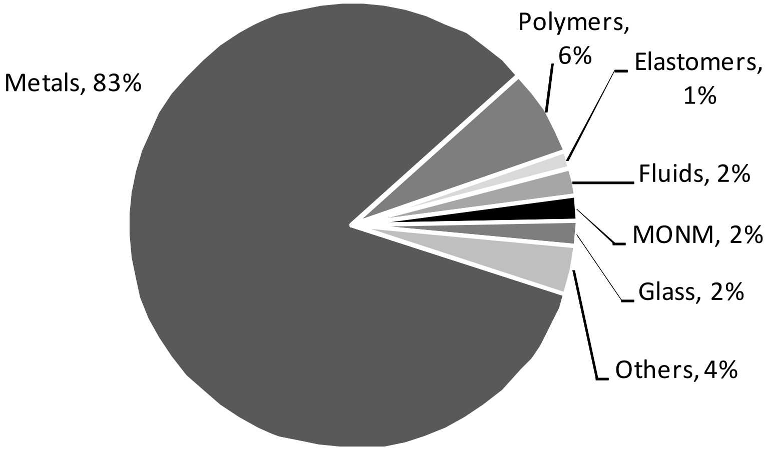

Fig. 1: Material composition of the studied train based on production and maintenance

The recyclability and recoverability rates of the studied train are calculated in accordance with ISO 22628 [6]. The potential recyclability and recoverability rates of the train are 92% and 96% respectively.

Tab. 1: Use of resources per functional unit for the entire life cycle of the studied train without regenerative braking

	Upstream module	Core module	Downstream module		Total
			Use	EoL	
Non-renewable resources					
Material [Kg/pass.100km]	0.55	0.33	32.5	0.007	33.4
Energy [MJ/pass.100km]	0.29	0.23	29.2	0.003	29.7
Renewable resources					
Material [Kg/pass.100km]	0.38	0.24	25.6	0.006	26.3
Energy [MJ/pass.100km]	0.03	0.01	4.8	0.0001	4.8

2.2.3 Results of the life cycle impact assessment

The LCIA was performed according to the CML 2001 method [5] at the characterisation level of the impact categories. No subsequent normalisation or weighting were performed. The potential environmental impact is shown for each of the life cycle stages and the considered impact categories in Tab. 3.

Tab. 2: Use of resources per functional unit for the entire life cycle of the studied train with regenerative braking

	Upstream module	Core module	Downstream module		Total
			Use	EoL	
Non-renewable resources					
Material [Kg/pass.100km]	0.55	0.33	18.6	0.007	19.5
Energy [MJ/pass.100km]	0.29	0.23	16.7	0.003	17.2
Renewable resources					
Material [Kg/pass.100km]	0.38	0.24	14.7	0.006	15.3
Energy [MJ/pass.100km]	0.03	0.01	2.7	0.0001	2.7

Tab. 3: Potential environmental impacts per functional unit

	Upstream module	Core module	Downstream module		
			Use		EoL
			Rheostatic	Regenerative	
AP (kg SO_2e)	1.2E-04	2.6E-05	2.6E-03	1.5E-03	5.7E-07
EP (kg PO_4^{3-}e)	1.0E-05	3.2E-06	2.4E-04	1.4E-04	7.0E-08
GWP (kg CO_2e)	2.3E-02	1.6E-02	1.7E+00	9.6E-01	7.3E-04
ODP (kg CFC11e)	1.2E-09	2.5E-09	3.1E-07	1.7E-07	2.9E-11
POCP (kg C_2H_4e)	1.0E-05	3.7E-06	1.9E-04	1.1E-04	3.9E-08

2.3 Additional environmental information in the EPD

The train under consideration shows several features designed to reduce the environmental impact directly and are therefore highlighted in the EPD.

2.3.1 Material selection

Strong emphasis has been put on material selection which ensures avoiding or reducing materials potentially harmful to human health and environment. The approach to material selection also prioritises materials which result in a high recyclability of the train.

2.3.2 Energy consumption and recuperation

Part of the approach to reducing energy consumption is reflected in the data shown in Tab. 1 and Tab. 2 is: recuperation of brake energy. Additionally concepts such as optimised parking or a driver support device enabling energy efficient operation are part of the train's approach for the reduction of energy consumption.

2.3.3 Flexible design

With regard to lifespan the considered train has been developed with the idea of high flexibility by using a modular design for different components enabling quick and cost efficient customisation of the trains.

2.3.4 Environmental management system

The EMS in place at Bombardier is certified according to ISO 14001:2009 as a minimum and additionally according to the European ecomanagement and audit scheme (EMAS) for several sites, thereby achieving continuous improvement not only with regard to the management system but also with regard to the actual environmental improvement [7, 8]. Besides achieving certification for the Bombardier sites, suppliers are also encouraged to implement an environmental management system.

3 Conclusions based on LCA and EPD

The results from this LCA clearly show that the use phase dominates the environmental impact of the analysed commuter train in all of the input-related impacts as well as the selected output-related impact categories. Furthermore it is shown that the most significant impact is caused by secondary emissions resulting from the energy production for the operation of the train.
The results indicate that the highest potential for improving the environmental performance of the studied train is associated with energy consumption during the use phase, i.e. future work should be focused on reducing the amount of energy used but also on reducing the emissions caused by energy production and consumption, e.g. by increasing the amount of renewable energy resources. The significance of the use phase is to some extent already addressed through the measures described in paragraph 2.3.

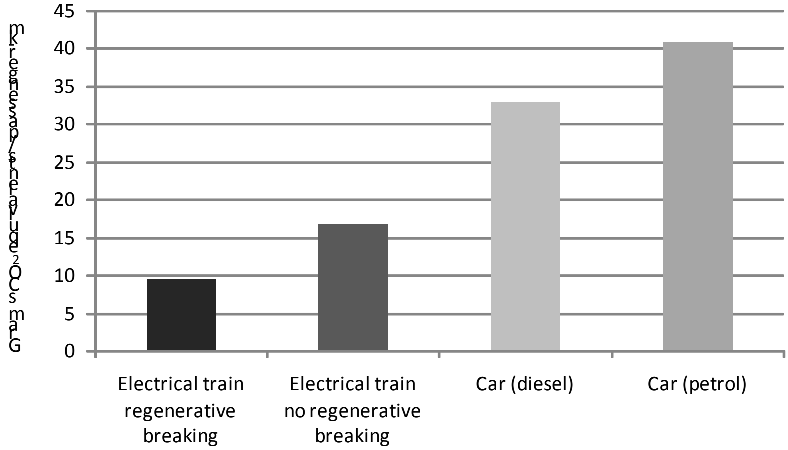

Fig. 2: Comparison of emissions of CO_2e for different modes of transport (calculations and assumptions are carried out by company Bombardier Transportation and are based on NTM's methods and data available at http://www.ntmcalc.se/index.html)

The overall results suggest that even though there is potential for improvement the results also reveal that the impact on global warming potential, which is highly influenced by energy consumption, is significantly lower when travelling on the assessed train instead of travelling by car, see Fig. 2. Therefore, with regard to global warming potential and climate change, commuter journeys made by train are the environmentally preferred mode of transport.

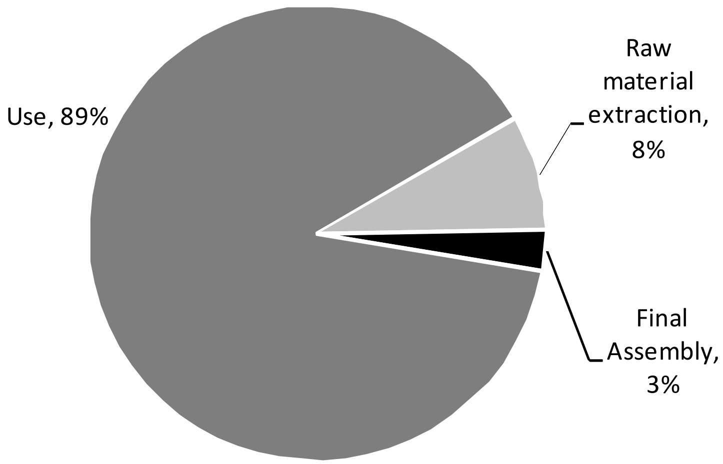

Fig. 3: Contribution of life cycle phases to POCP for the studied train

However, even though impacts are dominated by the use phase the phase of raw material extraction (including component production) also shows a significant impact for most considered impact categories, see Tab. 3 and exemplarily for POCP in Fig. 3. Therefore investigating optimisation potentials focusing on raw material extraction, including material selection itself is also worthwhile when improving the overall environmental impact.
By evaluating the results of the LCA and additional environmental information gathered in the EPD key environmental performance drivers can be determined which enables us to focus on them in future projects. Thereby every project benefits from the experience gained in the previous one and helps to continuously improve the overall environmental performance of Bombardier products.

References

[1] ISO 14040 (2006) Environmental management – Life cycle assessment – Principles and framework. ISO, Geneva
[2] ISO 14044 (2006) Environmental management – Life cycle assessment – Requirements and guidelines. ISO, Geneva
[3] PCR 2009:05 (2009) Product Category Rules (PCR) for preparing an environmental product declaration (EPD) for rail vehicles, UNCPC CODE: 495
[4] REACh (2006) Regulation (EC) No.1907/2006 concerning the Registration, Evaluation, Authorisation of Chemicals. European Union, Brussels
[5] Guinée JB (ed) (2002) Handbook on life cycle assessment: operational guide to the ISO standards. Kluwer Academic Publishers, Dordrecht
[6] ISO 22628 (2002) Road vehicles – Recyclability and recoverability – Calculation method. ISO, Geneva
[7] ISO 14001 (2009) Environmental management systems – Requirements with guidance for use. ISO, Geneva
[8] EMAS (2009) Regulation (EC) No.1221/2009 on the voluntary participation by organisations in a Community eco-management and audit scheme. European Union, Brussels

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