Market-based allocation of recycling benefits

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1. Copy of abstract

Market-based allocation of recycling benefits has already been described since the years 2000. However, this approach tends not to be largely adopted in international standards.

In practice, there are mainly three types of benefit allocation with system expansion modeling: (i) allocation of benefits to the producer bringing material on the market (that can become secondary material at end-of-life); (ii) allocation to the product incorporating secondary material and (iii) 50/50 allocation (half of the benefits of recycling at end-of-life and half of the benefits of incorporating secondary material are accounted for).

Market-based allocation is dictated by the answers to the following questions: “Which additional amount of secondary material will be exchanged on the market if supply increases thanks to the apparition of a new source of secondary material?” or “Which additional amount of secondary material will exchanged on the market if demand increases thanks to the apparition of a new producer of a good based on the secondary material?” Analysis of price elasticity of demand and supply provides answers.

This paper aims at depicting several typical supply-demand curves and at associating these to actual market situations (for example, what if there is a back obligation or subsidized waste collection, etc.). The appropriate market-based allocation can hence be dictated in each case.

Namely, allocation to the supplier (i) is to be promoted in cases where the supply is fully inelastic and where the demand is fully elastic. Examples of such markets are, among others, PET and packaging glass.

In markets where answers to supply and demand variations are more equilibrated, a 50/50 allocation should be used, as it is the case for certain types of paper and boards.

Market situations for main materials are analyzed so as to provide sound justification for standard developments.

2. Content of the extended abstract

The paper will recall / refer to the paper of T. Ekvall (2000).

Typical supply – demand curves will be schemed and associated to specific materials and allocation methods.
Recommendations for main types of material are formulated.

3. References

Using a long-term energy model for the consequential and prospective life cycle assessment of the use of biomass based synthetic diesel (BTL) in France

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1. Introduction

Consequential Life Cycle Assessment (C-LCA) has been developed recently as a modeling approach that captures environmental impacts of a product beyond direct physical relationships accounted for in Attributional LCA [1]. This approach seems particularly interesting for the evaluation of biofuels since their indirect impacts on other sectors (agriculture, forestry, electricity production) may be important. It is also considered as a prospective approach because it is used to analyze the consequences of a previous decision (e.g. investing in a new technology) so applying C-LCA for the evaluation of 2nd generation biofuels (which are not yet produced in commercial scale) is justified. Previous C-LCA works on biofuels may be divided in two groups:

- Studies that apply a step-wise approach developed by Weidema et al. in 1999 [2] to identify the technologies that are affected by a decision. The authors using this approach rely on historical data (culture yields, crop prices, etc. in the case of agricultural products), to determine market trends.
- Studies using partial or general equilibrium economic models to evaluate the direct and indirect consequences of a decision (e.g. [3,4])

In this study, we use a prospective optimization model representing the French energy production and transportation sectors (agriculture is partially included due to the presence of biofuels) to evaluate the impacts of the production of BTL in France in the time horizon 2007-2030. The model used was developed with the economic model generator TIMES (The Integrated MARKAL EFOM System). We estimate that a step-wise approach would not be applicable in this case, notably due to the lack of market data on lignocellulosic biomass.

2. Materials and methods

The partial equilibrium models developed with TIMES are generally used for prospective studies but it has already been used for environmental evaluations in the "New Energy Externalities Developments for Sustainability" European project [5]. As in this project, the first step of our work was to adapt the model for the C-LCA means. This adaptation work basically consists in integrating energy consumptions and emissions factors to all the technologies described in the model. For example, life cycle heat, electricity and diesel consumptions were associated to the production of agricultural products (rapeseed, sunflower, sugar beets, etc.) as well as the field N2O emissions. Emission factors for greenhouse gases (GHG), CO2, CH4 and N2O were associated to imported products (coal, uranium, crude oil, natural gas, etc.) to represent the emissions of the extraction and transportation of these products to France. The data used in the model adaptation is mainly extracted from Ecoinvent, most complete LCA database available at the European Union (EU) level, which allows keeping a certain homogeneity of sources. The model inputs are the following:

- Demands for energy and energy services: household and industrial electricity and heat consumptions, mobility demands, etc.
- Availability and price of primary energy resources: crude oil, natural gas, coal, uranium, biomass and other renewables (wind, solar, etc.)
- Energy policies: mandatory consumptions of renewable energies, GHG emissions cap, etc.
- Technology descriptions: energy conversion yields, emissions, investment, operation and maintenance costs, etc.

Using these inputs, the model finds the optimal solution in order to satisfy the energy demand at a minimum total cost. It simultaneously makes decisions on equipment investment and operation, primary energy supply and energy commodities trade. As a first step, the single impact indicator analyzed is Global Warming Potential (GWP). That is the reason why only GHG emission factors were integrated to the model but further work should consist in including other pollutants, water use, etc. The C-LCA model developed computes
emissions between 2007 and 2030 and time-dependent characterization factors for each GHG are used to calculate the GWP consistently considering the temporal profile of the emissions.

3. Results and discussion
The main objective of this paper is to develop C-LCA methodology and the BTL case study is suitable for that because of its interactions with other sectors. Scenarios were built in order to expose the C-LCA methodological issues we aim to discuss. Analysis consisted in observing how the model behaves when applying variations to:

- The technology used for BTL production: autothermic (biomass provides process energy needs), natural gas or electricity allothermic (external energy provides process energy needs).
- The levels of development of BTL in France: many possible scenarios for the number of BTL production units and their starting up dates (2020, 2025 or 2030). Methodologically interesting for the observation of marginal and non-marginal effects.

These C-LCA results are measures of how the development of BTL in France affects the different scenarios GWP. For instance, the usage of BTL representing less than 0.3% of the total French energy consumption could cause variations of the energy sector GWP in the order of 0.1%. To explain the GWP variations, the consequences of the development of BTL are analyzed. Preliminary results show changes mainly in two different sectors described in the model:

- The electricity production. It is possible to identify, for example, which is the source of the electricity that is going to compensate the supplementary demand for electricity due to the production of BTL using an allothermic process.
- The types of fuels consumed and the consequences on the automotive market. This analysis can help answering what kind of blends (B7, B30, jet fuel) is BTL going to be incorporated into and what is it substituting (fossil diesel, 1st generation biodiesel, etc.).

4. Conclusions
The long-term energy model can be applied for measuring the indirect effects of the development of BTL in France. This type of model may be used for C-LCA presenting some advantages in relation to other models previously used for this type of study:

- It allows a fine description of transformation steps of primary resources in energy carriers.
- It allows the observation of marginal and non-marginal perturbations on the energy sector.
- Emissions are endogenous to the model. Most of the other C-LCA studies use economic equilibrium models to quantify indirect effects (consumption of certain products) that are converted into emissions afterwards and integrated manually in another calculation tool.

Nevertheless, sectoral modeling also has limits that should be treated in further studies. In this version of the model, the agricultural sector is described partially. Only quantities, GHG emissions and prices of crops used in the energy sector are included. We understand that the evaluation of land use changes (direct and indirect), an important issue in the case of biofuels, requires the use of a global (whole world) description of the agriculture and forestry sector. One way of improving this C-LCA would be to integrate this model with a general equilibrium model.

5. References
Modelling land use changes in consequential LCI: limitations of equilibrium models

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1. Introduction

Conventional LCIs are static models not considering any mechanism of revenue maximization and price equilibrium under external constraints. An additional demand of a given commodity, irrelevant the amount, can always be supplied by the average supplier (fully elastic market). This constitutes a recognized limitation for the application of LCA to the evaluation of complex systems, like agro-systems and options for “deepening” LCA have been proposed [1], including the consideration of additional mechanisms (rebound, behavioural and price effects). In the so called consequential LCI, the relationships between the activities and processes of a life-cycle are no more seen as essentially technical connections, based on average data, instead the determining socio-economic mechanisms are considered via market information and economic models (partial or computational general equilibrium). Following this line of research, our team is currently developing a methodology for the inclusion of Indirect Land Use Change (ILUC) effects into consequential LCA of biomethane production in Luxembourg. The core methodological development is a Partial Equilibrium (PE) model representing the market constraints and reaction to the demanded production of maize at a given time horizon. The PE model provides two main results: 1) the change of crop production patterns, i.e. the primary consequences on Luxembourg's agriculture system of the planned production of maize. Forage crops are included, making the link to the consequences on meat and milk production. 2) The changes of land use type and related primary consequences in terms of modified pollutant emissions (e.g. CO₂) and land transformation impacts. The modelled changes are then propagated inside the national economy, using a computable general equilibrium model (Luxgem), and outside the national boundaries to account for additional environmental impacts. The latter propagation is performed using a conventional consequential approach [2]. This presentation aims at presenting the current state of progress, and discussing the identified limitations of equilibrium models.

2. Materials and methods

The Grand Duchy of Luxembourg has an area of 258600 ha of which nearly 130762 ha were used for agriculture and livestock activities in 2009. Legal restrictions prevent the meadows and pastures (67368 ha) to be converted to farmlands thus restricting the farming activity (cereals, forage crops, vineyards) to 63394 ha. Assuming that the pecuniary incentives (subsidy, price) for producing bio-methane actually yield the necessary quantity of gas through adequate production of maize, the PE Model analyses the land use changes incurred due to additional demand for maize to produce bio-methane. The agrarian system is price sensitive and the prices of past years play a major role on the cropping decisions by farmers. The choice to plant crops is also constrained by the need to preserve soil quality which often leads to predetermined crop rotation schedules. The crop rotation imposes a limit to the extent of substitutability between crops despite favourable price incentives. In the PE model we account for wheat, rye, triticale, barley, oats, potatoes, forage plants including maize, vineyards and meadows and pastures. The substitution between crops depends on the opportunity cost of planting a different crop from the existing one. If we replace one ha of existing crop (O) with a new crop (N) we incur gains (market price and subsidies on new crops, fixed and variable costs of old crops) and losses (vice versa). Opportunity Value \([OV(O,N)]\) in € per ha equals the net gain (gains minus losses) of replacing a crop O by new crop N. Since each of the crops mentioned above has a different yield, market price, subsidy, variable cost and perhaps fixed cost, the \(OV(O,N)\) will change for each pair of crop replacement. The PE model maximises the sum of all opportunity value pairs subject to restrictions of the maximum possible replacement permissible for each crop. Since the model is also at an annual time frame and not on a monthly frame, the crop rotation is incorporated in the constraints by controlling the maximum possible replacement for each crop. Other decision variables like intensification are also included and intensification is a function of use of additional fertilizers. The cost of fertilizers is included in the variable costs. The model is solved using GAMS [3]. The agriculture sector is also treated as a single entity and not by farm size. The data is disaggregated by size of farm into 9 types ranging from < 2 ha to
Each farm has a certain distribution of crops and the replacement constraints by each farm type would yield different land-use patterns than the aggregate land use pattern. The incorporation of livestock activities in the model is ongoing and the animals concerned are bovine, pigs, sheep, goat and chicken. Each of these also produce dung which is a potential replacement for fertilizers and thus animals on a farm would imply dung that would reduce the cost if fertilizers. The net benefit per animal will determine the number of animals that a particular animal activity (number of animals used for milk, meat) would be operated. This will be subjected a minimum level of animals alive in any given year to prevent all the animals from being slaughtered due to a possibility of high meat prices.

3. Results and discussion

We estimate a requirement of an additional 80000 tons of maize for the production of bio-methane leading to an increase of 5850 hectares. The crop outputs and land use changes are shown in figure 1 and 2. The model first tries to obtain the required output through land use changes but when that falls short it moves to intensification through additional use of fertilizers. So far no need for intensification of the existing and new areas to meet demand for maize has been observed. To study the impacts of increased agriculture prices, we increased the import prices of the “agriculture” commodity by 10% in Luxgem calibrated to the base year 2005. We find that the share of expenditure by households on agriculture products fell from 2.5% to 2%. Agriculture has a very small share in the value added in the economy and increased demand for maize which may displace existing crops will not have serious economic impacts. These findings were corroborated using the global model GTAP, which was used to evaluate the increased demand for displaced agriculture crops on account of additional production of maize. However, since the PE model is based on revenue maximization, it is difficult to properly consider non-economic constraints such as behaviours related to habits, cultural heritages or additional regulatory constraints, like for instance the fear of generating production surplus that the market would not be able to absorb.

4. Conclusions

The consideration of several farming types to specify parameters affecting the evolution of crop production patterns, and also their mutual relationships, require important efforts for inclusion in the PE model, because of the difficulty to translate these variables into costs. The modelling of the influences of crop production patterns on the food sectors is not trivial because of the difficulty of assigning a clear and robust aggregated market relation between forage crops and meat and milk. Beyond technical limitations, equilibrium models fail short in considering behaviour, which has to do with autonomous changes in time and exogenous changes. The use of agent-based modelling could be an alternative approach for proper consequential LCI.

5. References


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Integrating accident-related methods and impacts into the life cycle toolbox

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1. Introduction

Life Cycle Assessment, while a key tool in context of resource-efficiency and sustainable production and consumption, is more and more understood to be "only" one component of a wider environmental and even sustainability assessment toolbox. Enhancing Life Cycle Inventory (LCI) modelling by integrating accidents in production, use and end-of-life of products is one important step towards more complete evaluation of product systems' life cycles.

LCA according to ISO 14040ff deals implicitly with non-accidental releases ("regular") only, while some accidental releases may be included depending on how LCI data is collected (e.g. average of a site's declared annual emissions including leakages and other smaller accidents). There are however important organisational and decision-making related reasons for managing these two sets of inventories separately: Accident prevention in industry is managed separately from dealing with regular releases to the environment.

Questions such as allocation vs. system expansion known from classical LCA seem to require a specific interpretation in context of accidents. Accidents often involve the interaction of different product systems to result in an accident, other than LCA where the interventions can typically directly be related to one process, while partitioning may be needed across co-products.

A fully operational and more systematic toolbox-type approach for joint analysis of accidents and of releases under regular operation in life cycle perspective is lacking, but would have many advantages: Most relevant accident effects can be quantified in absolute numbers, summed up along the life cycle. The direct comparability between effects under regular operation and from accidents could help to focus improvement efforts on most relevant weak-points. Quantitative comparisons of accident-intensity among technologies, sites and companies would be possible, drawing on the functional unit principle know from LCA.

2. Materials and methods

Over the years, relatively few studies have been carried out on LCI-analogous accident assessments (e.g. [1], [2], [3], [4], [5], [6], [7], [8], [9]). Often these studies were application-specific and/or limited to certain accident types (e.g. fire in use phase), or they only considered certain effects or sectors or the severity of accidents (e.g. limited to accidents with at least five fatalities).

Some general method concepts have been presented (e.g. [1], [8], [10], [11]), but details "on the ground" as to systematically integrating such models of inherently different nature and managing inventories are mostly missing. Such a detailed, directly applicable methodology should consistently address both average situations in the background system and allow for a detailed weakpoint-analysis in the foreground system. Moreover it has to provide an overall view of the relative relevance of regular and accidental interventions and impacts in the life cycle perspective.

3. Results and discussion

A proposal is presented of how best to place the accident-related impacts within the life cycle toolbox and methodological implications will be explained. Data sources and management of data are another component to discuss to ensure a proper approach in support of life cycle based decision making in industry.

Joining LCI methodology and tools from Risk Assessment (Event Tree Analysis and Fault Tree Analysis) yields a powerful approach for integrated analysis of the environmental and health effects from regular operation and due to accidents.

Accidents are currently integrated with LCA data coming from at least different approaches: social LCA / Working Environment, i.e. the social pillar of sustainability (Area-of-Protection perspective) OR the risk assessment based insight (accident modelling methodological perspective). Both have advantages, while a clearer and systematic distinction of how accidents are modelled and a separation of distinct sets of
inventories regarding the kind of damages they cause (e.g. hurting or even killing persons vs. emissions to the environment due to a fire accident) is needed (see Table 1).

A systematic approach is argued to be necessary for an efficient, overlap-free integration of accident-related inventories with life cycle inventories due to regular-operation. Methodologically, an explicit modelling of cause-effect chains drawing on fault-tree analysis methods appears beneficial in this context. A systematic approach for reflecting on attributional and consequential modelling principles when modelling accident inventories has been developed.

<table>
<thead>
<tr>
<th>Type of activity / event affecting the intervention</th>
<th>Impact pathway, principle</th>
<th>Cause of impacts</th>
<th>Human management influence on impact</th>
</tr>
</thead>
<tbody>
<tr>
<td>LCA-type interventions</td>
<td>Impacts via interventions with the natural environment</td>
<td>Single process, co-functions require partitioning / system expansion; some relevance of human interference</td>
<td>Decisions on operation of processes and their implementation</td>
</tr>
<tr>
<td>Regular operation: continuous or intermittent</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Accident-type interventions</td>
<td>Impacts via interventions with the natural environment</td>
<td>Process / product system(s) interaction; high relevance of human interference</td>
<td>Decisions on safety measures and their implementation</td>
</tr>
<tr>
<td>Accidents; singular to intermittent but also continuous</td>
<td>Direct impact on humans Damage to goods / interruption of services</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 1: Some key characteristics of impacts due to regular operation and due to accidents

4. Conclusions

The last decade has seen quite some progress on life cycle accident assessments, but a consolidation to one or few well-defined, common approaches is still needed. One methodological challenge encountered is to come to a clear distinction between regular operation and accidents. A pragmatic solution needs to be found that however have to reflect on the decision context of such analysis as well as data availability; solutions for dealing with overlapping inventories are required.

5. References

Better characterising the environmental performance of intermittent power generation with help of LCA: integrating wind power into the German electricity grid in 2006

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1. Introduction

Renewable energies are high on the political agenda for resource depletion, security of supply and climate change concerns. Wind power, in particular, is continuously gaining shares in the power plant parks of many countries. Given its intermittent nature, however, the true environmental impacts of increased wind power integration remain an open question. This is because the seemingly equivalent functionality expressed in a kWh of electricity produced is in fact not directly comparable to conventional power generation techniques. For that reason, a more systemic approach is advocated.

To evaluate the environmental performance of power generation techniques, Life Cycle Assessment (LCA) is one of the most prominent ecological assessment tools. Different ways exist to carry out LCAs [1]: (i) an attributional LCA, the traditional and most widely used LCA approach, which investigates the environmentally relevant flows over a life cycle. (ii) a consequential LCA, promoted as a new alternative, which explicitly investigates the changes resulting from a specific decision.

The aim of the current study is to analyse to what extent considering substitutional effects and back-up effects quantified with help of an energy system analysis model can help better characterise power generation from intermittent and non-intermittent resources. To this end, a study on the CO₂ benefits of wind power integration into the German grid in 2006 following a consequential approach was taken as a point of departure [2]. The intention is however to perform a more comprehensive study, which presents a greater variety of environmental impact indicators.

2. Materials and methods

The study starts with describing the German electricity grid in 2006 in terms of electricity generation with and without wind power generation. This way, capacity effects, e.g. the commissioning of new plants, are disregarded. The software tool Umberto version 5 [3] was used to model the systems. Based on power generation data [4], the German electricity mix is modeled by taking inventory data of typical power plants from the database of the Swiss Centre for Life Cycle Inventories “ecoinvent” version 2.2. When disregarding wind power generation, the other power plant types were scaled up proportionately in a first scenario. By using the results from the agent-based energy system model PowerACE [2], substitutional information was obtained, describing in a second scenario which types of power plants were assumed to be replaced by wind power according to the merit order. Efficiency losses were taken into account by assuming a 7% increase in the emission factors per kWh according to Klobasa and co-workers [2][5]. These were attributed to wind power integration. Impacts were quantified according to the ReCiPe hierarchist midpoint impact assessment framework [6].

3. Results and discussion

When analysing the environmental performance of (onshore) wind power integration into the German electricity grid in 2006, the systemic LCAs led to different results (Table 1). Taking modeled consequences into account, lower environmental impacts of the German electricity mix in 2006 resulted in all the investigated impact categories (including global warming) except for mineral resource depletion and natural land transformation. In particular, this difference is due to the inclusion of two effects:

1) The substitutive effect from the integration of wind power into the grid. Electricity produced by wind power displaces other generation sources, notably power plants fuelled by fossil resources, such as coal or gas. As a consequence, the environmental impacts per kWh in the mix are remarkably reduced for several categories, such as climate change or other emission-related effects.

2) The altered operation of the conventional power plant mix. In order to compensate for the fluctuating wind power generation, conventional power plants are used to balance the demand and supply of electricity. The
altered operation of the affected plants leads to efficiency losses, thereby lowering the environmental benefits caused by the substitutive effects.

<table>
<thead>
<tr>
<th></th>
<th>Climate change</th>
<th>Fossil fuel depletion</th>
<th>Mineral resource depletion</th>
<th>Natural land transformation</th>
<th>Particulate Matter formation</th>
<th>Photochemical ozone formation</th>
<th>Terrestrial acidification</th>
</tr>
</thead>
<tbody>
<tr>
<td>Linear scaling</td>
<td>-4.3%</td>
<td>-4.7%</td>
<td>19.9%</td>
<td>-11.6%</td>
<td>-5.1%</td>
<td>-5.2%</td>
<td>-6.0%</td>
</tr>
<tr>
<td>Substitution &amp; efficiency losses</td>
<td>-6.6%</td>
<td>-6.9%</td>
<td>53.0%</td>
<td>-7.5%</td>
<td>-6.5%</td>
<td>-7.0%</td>
<td>-7.0%</td>
</tr>
</tbody>
</table>

Table 1: Changes resulting from the integration of wind power into the German electricity grid in 2006 for selected ReCiPe 2008 (H) impact categories, respectively following an attributional LCA and a consequential LCA approach

Interdependent and large-scale systemic consequences as the ones described before are best captured by a consequential approach that accounts for effects like marginal technologies, i.e. the substituted techniques.

The most important limitations of this study are: (i) Not all of the consequences from the integration of wind power into the electricity grid are taken into account such as possible grid expansions and grid reinforcement as well as capacity adjustments attributable to wind power. (ii) Uncertainties about the consequences are equally to be determined for a better assessment. One source of uncertainty is the extent of the impact of wind power generation on the environmental performance of the conventional power plants. The 7% cutback applied in this study for Germany in 2006 is not a precise assessment of this impact, though validated by other studies cited in [2]. Likewise, it is consistent with the range of numbers given in a recent Intergovernmental Panel on Climate Change (IPCC) report on renewable energies [7]. Further on, the results of this study are very sensitive to the outcome of the Power Ace model, which bears uncertainties in itself, as it relies on various assumptions and an important amount of input data. A validation of the Power Ace scenario calculations regarding the substitution of conventional techniques through wind power could be interesting in this sense. (iii) Sensitivity analyses should also be considered, not only for the consequences of wind power integration, but also for other aspects such as using other LCIA frameworks.

4. Conclusions

Although this work shows how an environmental assessment of wind power can be performed in a system-oriented LCA approach, this consequential approach could not entirely resolve the issue related to the lacking consideration of intermittency in the functional unit kWh. Issues related to the availability of different techniques persist. Two solutions are discussed that both consider bundles of techniques rather than single techniques in line with the recommendations to policy makers of the EU-funded NEEDS project [8]:

1) Transforming intermittent techniques into systems meeting base-load demand through backup and comparing those systems to other systems or single techniques meeting base-load demand

2) Analysing entire electricity systems (e.g. at the national or European scale).

Note that the first approach fails to account for the interactions in the electricity system, finally leading to the equilibrium of supply and demand. The main message concerning a comparison of different power generation techniques is that without additional technical solutions, neither base-load plants can be compared to peak-load plants, nor can intermittent plants be compared to non-intermittent plants.

5. References

1. Introduction

In May 2011, the association of issuing bodies AIB announced that 1 billion EECS certificates (which equals 1 billion MWh of electricity) have been issued in Europe since its start nine years ago. There is a large demand in renewable energy certificates, which helps electric utilities and companies from the manufacturing as well as service sector to reduce the environmental impacts of the electricity they purchase.

The international standards on life cycle assessment (ISO 14040 [1], ISO 14044 [2]) do not specify how certificates should be taken into account when performing a product or company life cycle assessment (LCA) study. The recently completed carbon footprint standard [3] are clear with regard to carbon offsetting measures (emission certificates): these are considered as an improvement measure and shall not included in the product LCA but kept separate.

Up to now, the role of renewable energy certificates has not been discussed widely. This presentation shows the mechanism and volume of RECS certificates as well as its consequences and proposes some guiding principles how RECS certificates may or may not be used within product LCAs and the eco-balances of companies.

2. Materials and methods

The international standard on life cycle assessment advises to use the “actual electricity production mix in order to reflect the various sources of resources consumed” (ISO 14044 [2]). The current draft standard on the carbon footprint of products requires that “specific electricity products, including a guarantee that the product sale and associated emission are not double counted” are used if the electricity supplier delivers such an electricity product (ISO 14067 [4]). The draft standard also notes that “some "green certificates" are sold without coupling to the electricity, which might lead to double counting”.

The trade with electricity certificates in Europe developed dynamically during the recent past. shows the trade and cancellation (use) of RECS certificates during the year 2008 [5]. It shows that Norwegian electricity suppliers are most active in issuing RECS certificates, while other countries like Poland or United Kingdom have no activities. Norway exported about 50.5 TWh RECS certificates while Belgium (25.0 TWh), the Netherlands (18.9 TWh) and Germany (14.6 TWh) are the most important countries with respect to RECS imports. RECS certificates need not to be cancelled in the same year. Belgium, the Netherlands and Germany cancelled 13 TWh, 21 TWh, and 8 TWh RECS certificates in 2008.

Norway exports about one third of its domestic hydroelectric power quality, whereas the share of RECS imports to Belgium, the Netherlands and Germany represents about 30 %, 20 %, and 2.5 %, respectively of domestic electricity production. The RECS trade volumes exceed by far the physical trade volumes.

The question is, how to deal with RECS certificates in life cycle assessment and carbon footprint studies. This question calls for a more closer look. We may distinguish two situations: Either the RECS certificates are sold together with the respective physical delivery of renewable electricity or the RECS certificates are sold separately.

3. Results and discussion

3.1. The effect of RECS trade

In Table 1 the electricity mix and technology shares of Norway and Belgium are shown according to their domestic production and including the import of non verifiable electricity to Norway (compensating for RECS exports) and the import of RECS to Belgium. We assume that

- all RECS certificates exported and imported, respectively are actually used in the respective year.

This is a conservative assumption.
- the LCI of the electricity mix is influenced not only by physical deliveries of renewable electricity but also by trading electricity quality (RECS certificates).

- the RECS exports are compensated by imports of non verifiable electricity, consisting of electricity based on fossil and nuclear power plants.

<table>
<thead>
<tr>
<th></th>
<th>Norway production mix without RECS trade</th>
<th>Norway production mix with RECS trade</th>
<th>Belgium production mix without RECS trade</th>
<th>Belgium production mix with RECS trade</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>%</td>
<td>%</td>
<td>%</td>
<td>%</td>
</tr>
<tr>
<td>fossil</td>
<td>0.5</td>
<td>21.6</td>
<td>40.6</td>
<td>31.4</td>
</tr>
<tr>
<td>nuclear</td>
<td>0.0</td>
<td>15.3</td>
<td>55.0</td>
<td>42.5</td>
</tr>
<tr>
<td>renewable</td>
<td>99.5</td>
<td>63.1</td>
<td>4.4</td>
<td>26.1</td>
</tr>
</tbody>
</table>

Table 1: Share of fossil fuels, nuclear and renewable energy used for electricity production in Norway and Belgium excluding and including RECS trade

The environmental impacts of the national electricity mix in Norway and Belgium changes substantially if the trade of quality of electricity (renewable electricity) is taken into account (see Table 2). The carbon footprint of Norwegian electricity increases by about 2'650 %, while the carbon footprint of the Belgian electricity mix is reduced by about 22 %. With 246 g CO2-eq/kWh (NO) and 254 g CO2-eq/kWh (BE) the carbon footprint of the two electricity mixes is very similar. The amount of radioactive waste per kWh Norwegian electricity is increased by 13'600 %, while it decreases by 22 % per kWh Belgian electricity.

<table>
<thead>
<tr>
<th></th>
<th>unit</th>
<th>Norway production mix</th>
<th>Norway production mix including RECS trade</th>
<th>Belgium production mix</th>
<th>Belgium production mix including RECS trade</th>
</tr>
</thead>
<tbody>
<tr>
<td>Climate change</td>
<td>g CO2-eq/kWh</td>
<td>0.90</td>
<td>246</td>
<td>328</td>
<td>254</td>
</tr>
<tr>
<td>Radioactive waste</td>
<td>mm³/kWh</td>
<td>0.013</td>
<td>1.8</td>
<td>6.5</td>
<td>5.0</td>
</tr>
</tbody>
</table>

Table 2: Greenhouse gas emissions and radioactive wastes generated by the production of 1 kWh of Norwegian and Belgian electricity, excluding and including RECS trade

4. Conclusions and recommendations

The RECS trading scheme would substantially affect the national electricity mixes, if the purchase of independently traded RECS certificates would be considered for the national electricity mixes. Currently, the life cycle inventories of national electricity mixes are usually based on international statistics, which do not consider RECS trade but only trade connected to physical deliveries of electricity. If it were allowed to adjust the electricity mix purchased by buying independently traded RECS certificates, substantial double counting of renewable electricity production would occur.

We therefore recommend to disregard independently traded RECS certificates in product and service LCA as long as the LCI of national electricity mixes is based on international statistics disregarding RECS trade. If RECS certificates are linked to the production and delivery of renewable electricity, we recommend to include the respective share of renewables in the electricity mix.

5. References


Using water markets and consequential LCA to assess indirect impacts from water use

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1. Introduction

Using water can incur direct impacts on human health from water deprivation for domestic use or agriculture\textsuperscript{1}. However, these impacts do not occur in regions where economic resources are sufficient to allow the deprived users to turn towards technology to meet their needs. On the other hand, this technology leads to burden shifting that should be captured in a comprehensive assessment of water use in LCA. This paper proposes a consequential framework and model using water markets and marginal technology in order to assess LCA indirect impacts from water consumption and degradation, by identifying marginal processes and including their life cycle inventory. This builds on the existing inventory\textsuperscript{2} and direct impact assessment methods\textsuperscript{1} which already includes water quality aspects by distinguishing 8 categories of water quality.

2. Materials and methods

Based on Boulay et al\textsuperscript{1}, the capacity of a region to adapt (AC) to water deprivation is related to the GNI. The World Bank classifies countries\textsuperscript{3} into low, middle and high income country which was used to set out that low income region will not be able to adapt (AC = 0) and will suffer human health impacts from a change in water availability, high income countries will fully adapt (AC = 1) and not suffer any human health impacts, and middle income countries will partially adapt and partially suffer from human health impacts, proportionally to their income level (0<AC<1)\textsuperscript{1}.

Adaptation occurs when a given type of water withdrawn (type i), defined by its source and quality, is constrained. The assessment whether a water type is constrained or not is performed through the scarcity parameter $\alpha_i$, with the underlying assumption that an unconstrained resource is not scarce. The market concept, applied to water, is then used for each water type to consider all sources of water of quality i. These include available water of type i, treated water from a lower quality source of water, desalinated water, and imported water. The released water j is also considered an input to a water market. It may be the same market as the withdrawn water if the quality has not been degraded (or improved) or another market specific for this water category. This is illustrated in Fig. 1 below.

\begin{figure}[h]
  \centering
  \includegraphics[width=\textwidth]{water-markets-system.png}
  \caption{Water markets system}
\end{figure}
For each source of water into water market $i$, its *state of constrain* is assessed. The lowest cost unconstrained water source is identified as the marginal one, which is the one used by a marginal water use. Conversely, the release flow will displace the marginal source of water for this market. For each local marginal water source, the processes required to supply the market with the appropriate water need to be identified. If it comes from a water of lower quality, the treatment steps to bring the quality of this water to the market quality level have to be assessed. The processes required to obtain one water quality from another were found based on an 8-step methodology considering the input and output quality parameters\(^4\).

3. Results and discussion

Results consist of quality-specific marginal water source available for 808 hydro-economic cells worldwide, as defined in Boulay et al\(^1\), resulting in 6464 (808 cells x 8 water types) marginal processes which can then serve as input to the inventory and be assessed by any impact assessment methodology. However, regions where the adaptation capacity is zero (247 of the 808 cells) present no marginal water source, as all constrained water use is set to come from deprived human users. For middle-income countries, the marginal processes are weighted based on the adaptation capacity, since only partial adaptation occurs in these regions. The entire marginal process (i.e. a weight of 1) is used in a country where the adaptation capacity is maximum (164 of the 808 cells).

Preliminary results are plotted on a world map giving additional energy demand, for ease of communication, incurred by the marginal processes for a 1000 m\(^3\) of good quality surface water use for each 808 hydro-economic cells.

4. Conclusions

This paper presents a novel approach assessing indirect impacts from water use due to compensation scenarios. It is especially relevant for developed countries facing water scarcity and/or poor water quality by overcoming current methodological limitations that solely consider direct impacts on human health from water deprivation. This model is based on a consequential approach, identifying the unconstrained marginal water source and treatment processes required to compensate the deprivation from water use.

5. References


\(^{3}\) World Bank. World Bank list of economies. 2008

Modelling of biogenic CO₂ fluxes in LCA and their integration with the global C cycle

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1. Introduction

In Life Cycle Assessment (LCA), the same characterization factors are conventionally applied irrespective of when the emissions occur (the same importance is given to emissions in the past, present and future). When the assessment is constrained by fixed timeframes, the appropriateness of this paradigm is questioned and the temporal distribution of emissions becomes of relevance. In traditional bioenergy LCA, CO₂ emissions are usually neglected in the light of the assumption that the same quantity of CO₂ has been sequestered for growing biomass. This convention neglects the time discrepancy between emissions, occurring at a single point in time, and sequestration, generally distributed over several years, depending on the type of feedstock used. In the recent years, several studies pointed out the shortcomings and inaccuracies of this accounting paradigm (e.g., Johnson et al., 2009). Following the methodology presented in recent papers (Cherubini et al., 2011), in this work we investigate the possibility of explicitly modeling biogenic CO₂ fluxes (e.g. emissions and removals) with appropriate probability distribution in biomass systems, so to introduce treatment of time of stressors in LCA. Concerning emissions, two types of distributions are here considered to model the oxidation rate of two products: a delta function, which is used to simulate emissions occurring from biomass combustion, and a chi-square distribution, used to model the oxidation rate of wood stored as non-structural panels (assuming a mean lifetime of 30 years). The CO₂ sequestration from growing trees is modeled using a gaussian distribution centered on the year corresponding to half rotation period. These fluxes are then integrated with the global carbon cycle, represented by the Impulse Response Function (IRF) from the Bern CC Model (Joos et al., 2001), in order to get the change they cause in atmospheric CO₂ concentration (used as basis for computing the climate impact). In the following results, we assume that wood comes from a forest plantation under management with a rotation period of 100 years (but the approach is flexible and can be applied to several biomass systems). Results are elaborated according to different climate metrics, both normalized (GWPbio and GTP) and absolute (changes in CO₂ atmospheric concentration, instantaneous and integrated radiative forcing and surface temperature change) metrics.

2. Materials and methods

All types of CO₂ emissions and removals cause a perturbation to the CO₂ atmospheric concentration, thereby causing a climate impact. The change in atmospheric CO₂ concentration can be modeled by means of the Impulse Response Function (IRF), which describes the perturbation of a dynamic system caused by some external change. Among the existing IRFs, the IRF from the Bern CC model is here used to predict the atmospheric decay of anthropogenic CO₂. This function represents the fraction of CO₂ remaining in the atmosphere after a single pulse emission depending on the interactions between the atmosphere, the oceans and the terrestrial biosphere. The time-distributed emissions and removals of CO₂ follow this decay, and the analytical combination of these fluxes with the IRF for anthropogenic CO₂ provides the corresponding change in CO₂ atmospheric concentration. In mathematical terms, this is implemented with a convolution between the emission and removal functions with the CO₂ decay from the air:

\[ f(t) = \int_0^t e(t') y(t-t') dt' - \int_0^t g(t') y(t-t') dt' \]

Where t’ is the integration variable from the time since harvest, t is the time dimension, e(t’) is the emission function, g(t’) is the CO₂ removal rate from the atmosphere (due to biomass re-growth), and y(t) is the IRF from the carbon cycle climate model. This equation integrates the dynamics of the biomass system within the global carbon cycle to get the resulting CO₂ atmospheric profile. The first part of the integral represents the atmospheric CO₂ concentration response to distributed CO₂ emissions, while the second integral considers distributed CO₂ removals due to biomass re-growth, here modeled as a negative emission. Solutions are computed via numerical approximations. This function can be seen as a new IRF expressed as a function of the different biomass species (represented by different rotation periods) and is used to compute the different climate metrics listed in the previous section.
3. Results and discussion

3.1. Results

Different results can be obtained by applying different climate metrics. We here show the values for the normalized metrics Global Warming Potential (GWP) and Global Temperature change Potential (GTP). Figure 1 shows the cumulative change in surface temperature as a function of time for the two cases investigated (the climate impact from combustion of fossil fuels is also shown for comparison).

<table>
<thead>
<tr>
<th></th>
<th>GWP&lt;sub&gt;bio&lt;/sub&gt;</th>
<th>GTP</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>TH = 20</td>
<td>TH = 100</td>
</tr>
<tr>
<td>Wood as fuel</td>
<td>0.96</td>
<td>0.43</td>
</tr>
<tr>
<td></td>
<td>0.97</td>
<td>0.49</td>
</tr>
<tr>
<td>Wood as non-structural panels</td>
<td>-0.04</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td>-0.03</td>
<td>0.19</td>
</tr>
</tbody>
</table>

*Table 1: GWP and GTP for the three most common time horizon of the two investigated wood applications*

Results show the importance of explicitly consider how biogenic CO₂ fluxes are distributed over time to accurately get the contribution to global warming of biomass systems (or they mitigation potential). In particular, the selected metrics point out the large long term benefits of bioenergy against fossil energy; for storing biomass into products, the mitigation target is achieved from the beginning.

3.2. Discussion

The explicit modeling of CO₂ fluxes with probability distributions and their integration within the global carbon cycle represents a flexible and consistent methodology to consistently assess the climate impact of the various biomass and bioenergy systems (including direct combustion, wood storage, afforestation, and others). This type of dynamic analysis, adaptable to different metrics, meets the target of policy makers for studies where impacts from emissions should be assessed within specific time boundaries. When emissions are distributed over time, especially if they are near or beyond the time horizon, normalized metrics have serious shortcomings, despite their large use in the LCA community. In these cases, they should be replaced (or come with) by absolute metrics, showing how the climate impact changes over time.

4. Conclusions

A debate should be opened in the LCA community on the treatment of time, with an explicit modelling of how CO₂ fluxes are distributed over time. In addition, the choice of the metric with its TH is crucial for the climate assessment of the system, and caution should be made when interpreting results in cases where large perturbations occur close to the end of the time window. For biomass systems characterized by time distributed emissions, absolute metrics showing the variation of the climate impact over time are preferable over more traditional normalized metrics, like GWP.

5. References

Evaluating the greenhouse gas emissions of retrofitting the existing Irish housing stock – A combination of process analysis and input-output analysis

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1. Introduction

The few studies that have taken place in Ireland have failed to show clearly the true contributions of the greenhouse gas emissions attributable to imported building materials. For example, the Residential Road Map to 2050 for Ireland as published by Sustainable Energy Authority Ireland (SEAI) indicates that homes accounts for more than a quarter of Irish energy-related carbon emissions, and therefore the need to reduce emissions by as much as 50% by 2050 but failed to directly point out the proportion of it that is attributable to imported goods into Ireland. Using a combination of process analysis and input-output analysis and with a focus on imports, the total greenhouse gas emissions caused by retrofitting the existing Irish housing stock across life cycle phases have been accounted for. The impact of imports was found to be significant due to production and consumption of resources in the Irish residential sector.

The study also provides greater information on the overall depiction of the emissions associated with retrofitting the existing stock and thus asserting significant role of the residential sector as an important research area and for the generation of the information needed for evidence-based energy and emissions policy development. Indicators from models based on domestic greenhouse gas emissions are not likely to provide a broad depiction of the emissions.

The aim of this paper is:

- Assess across life cycle phases the total greenhouse gas emissions attributable to retrofitting the existing Irish housing stock based on three scenarios – “Baseline”, “Low-emissions dwellings” and “Passive house standard”

In implementing this study the following questions will be addressed:

- What is the magnitude of emissions induced by imports to overall total emissions of retrofitting the existing Irish housing stock?
- What is the relative magnitude of emissions of the different stages of life cycle analysis of building retrofits?

2. Methodology and databases

2.1. Process-based-LCA

As input-output analysis is incomplete in its ability to sufficiently evaluate the greenhouse gas emissions attributable to exports, it was decided to use process-based hybrid LCA. A process-based hybrid technique is a combination of two base approaches to LCA – input-output analysis and process analysis. Process analysis relies on process flows to steadily assemble data and calculate known environmental inputs and outputs. Input-output-LCA exploits economic operation link of different sectors of the economy to establish energy intensities as an output of monetary value. A hybrid method can adequately combine the advantages of the two techniques. While typical commercial LCA software tool was used to assess environmental impacts due to imported goods, domestic generated emissions including those due to imported products induced by addition of energy inputs were evaluated using input-out analysis.

2.2. Databases

As a first step to analysis, necessary data were compiled. In this present paper, two databases have been useful to carry out the study – the Energy Performance Survey of Irish Housing (EPSIH, 2005) and Irish National Survey of Housing Quality (INSHQ, 2001-2002). While the EPSIH was used in the current study, the INSHQ was used to check the representativeness of the EPSIH. The main database for the process-LCA
model was the EPSIH. It contains data on the physical flows of all processes that are related to the production, consumption and disassembly phases of the house in question.

From a previous Irish study, Acquaye A. (2010) on sub-sectoral embodied carbon dioxide equivalent (CO2-eq) analysis of Irish construction, input-out data for the input-output analysis was obtained. These include sub-sectoral direct embodied CO2-eq intensity and construction sector indirect embodied CO2-eq intensities due to domestic and imported goods and services.

### 2.3. Greenhouse gas emissions of retrofitting

The process-LCA model analysed emissions due to imports. The existing Irish housing stock was characterised into 13 archetypes to statistically reflect the mix of key energy related characteristics. Detailed life cycle inventories were prepared for each these archetypes and then a suite of energy efficient retrofit technologies were applied to identify the most suitable retrofit options and investigate the balance between their impact on pre-use, operational and disassembly phase energy consumption and greenhouse gas emissions.

The input-output-LCA model analysed emission due to domestic production and consumption and those of imports induced by the addition of energy inputs into imported goods. Using the input-out data in section 2.2 and the total costs (Euro) of domestically produced goods and services, and of imports, the national arising (direct and indirect) total embodied CO2-eq emissions of the Irish construction sector and the total international arising indirect embodied CO2-eq emissions induced by the addition of energy inputs into imports were determined.

Within this study the two main methods (process-analysis and input-output) were combined in order to calculate the overall emissions of retrofitting the existing Irish housing stock. Process-LCA model analysed emissions attributable to households’ use of energy based on Irish electricity grid mix. It was also assumed that household’s gas thermal energy use was based on similar production technology abroad. The emissions associated with households use of energy was then first deducted from process-LCA emissions and then added to domestic generated emissions. The total of the economic sector (input-output) is the sum of domestic and imported emissions induced by the addition of energy inputs into imports. The total of domestically generated emissions is the sum of the emissions from input-output analysis plus emissions due to households’ energy use as deducted from process-LCA. The total emissions in retrofitting the Irish housing stock is therefore, the sum of emissions generated domestically and the total emissions due to imports as obtained from process analysis.

### 3. Results and discussion

The total greenhouse gas emissions attributable to imports were found to be around 50% of the total emissions attributable to retrofitting the existing Irish housing stock. Although operational phase consumption and emissions was much greater than any other phase, there was a wide variation in the impacts on this balance across the retrofit options.

### 4. Conclusions

This paper has shown that a significant proportion of emissions attributable to retrofitting the existing Irish housing stock actual come from other countries. By using process-based hybrid LCA greater information can be provided to aid decision making.

### 5. Further research

Significant potential exists for domestically generated goods and services. Further research is required to look into production of renewable resources that are capable of substituting energy-intensive imported materials. For example, production of insulation materials from polymers was found to be responsible for a large proportion of the emissions due to the production of retrofit materials.

### 6. References


Acknowledgement

The authors gratefully acknowledge the support of the Sustainable Energy Authority Ireland for providing data for this work. The PhD programme was funded by the Dublin Institute of technology’s ABBEST.
A framework for prospective hybrid life-cycle assessment and its application to energy technologies

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1. Introduction
Life-cycle assessment and input-output analysis literature abounds with case studies that have been developed at a particular place and a particular time in the past, being as many photographic representations of specific systems. With the remarkable exception of the NEEDS project \cite{1}, future life-cycle inventory or input-output databases are not available. There is nonetheless an ever-increasing need for a life-cycle capable scenario modelling framework. We propose here a procedure to adapt life-cycle assessment (LCA) databases and multi-regional economic input-output (MRIO) tables to future years (up to 2050) according to various external parameters that have been identified as central in determining the environmental impacts of the overall economy.

The EXIOPOL project has recently produced a highly detailed input-output model; we utilize this model to build a life-cycle based framework for scenario modelling. A combination of existing prospective studies, including the International Energy Agency’s \cite{2-4} are used to produce a consistent hybrid model of the world economy which can serve as the background for prospective life-cycle assessment studies. We present the model development and provide some basic comparisons of the scenarios implemented. Finally, in order to exemplify the use of such a prospective hybrid LCA model, we apply it to the integrated environmental assessment of selected low-carbon electricity production technologies.

2. Approach
We identify the criteria that would characterise a model that would allow practitioners to obtain a proper picture of environmental impacts of energy systems. The requirements for a hybrid model include the following criteria:

- Include the global economy as a background, so that truncation errors would not be a problem. Studies show that process-based analyses neglect a large share of environmental impacts due to the system boundary definition in life-cycle assessment \cite{5}.

- A reasonable regional resolution should be offered, in order to give the chance to a LCA practitioner to model different technologies in various geographical contexts.

- Uncertainty should be taken into account. Many of tomorrow’s electricity generating technologies are only demonstration plants today, so that rough assumptions are needed to estimate their future market penetration. Also, uncertainty on prices and process data are important, since they are highly variable parameters for new energy technologies.

- Impact categories considered in the framework should be significant for the energy technologies assessed.

- The timeframe must range over a period long enough to capture all the effects of a large-scale deployment of technologies that are not mature as of 2011.

3. Model development
A proper basis for elaborating such a framework is hybrid life-cycle assessment, with a multi-regional input-output background and scenario modelling. As a starting point, we use Ecoinvent as a process-
level life-cycle inventory database, and EXIOBASE, a multi-regional input-output database, to model the background economy. Since the finality of this hybrid model is the assessment of energy technologies, we first need to disaggregate the electricity production sector in the regional economies represented in EXIOPOL to have a consistent model for a starting year. A scenario modelling phase is then needed, we choose to develop tables for the 2010-2050 period.

To accomplish the scenario modelling task accurately it is necessary to determine how the following changes affect the background input-output and process data for our model:

- increasing energy efficiency of the global economy,
- changing emissions intensity of global economy,
- increasing share of renewables in global electricity,
- increasing material efficiency of the global economy.

Data communicated by the IEA provide the parameters necessary to make this adaptation on the input-output tables. Time series are derived for the 2010-2050 timeframe. As for the process-level life-cycle inventory database, the Ecoinvent processes contributing the most to energy systems are modified. The NEEDS project has yielded a life-cycle inventory database for future years, in relation with energy technologies, which are used to accomplish the modelling task at a process-level.

Additionally, a data collection form is presented, so that foreground systems can easily be implemented. The example of a photovoltaic system is given.

4. Results

- Here is presented the results for the assessment of the large-scale deployment of polycrystalline photovoltaic energy systems.
- Results presented: global greenhouse gas abatement potential according to different scenarios, repercussions on the global economy sectors, regional detail, and structural path analysis.

5. Conclusions

The development of a hybrid LCA framework based on a multi-regional input-output background with scenario modelling seems to meet many of the requirements for an objective and relevant impact assessment of energy technologies.

Main findings are that:

- Spatial and temporal aspects can be covered. Spatial aspects encompass geographical variability of life-cycle systems, either at a downstream (type of energy conversion technology, or efficiency) or upstream (transportation schemes, electricity mixes) level; and regional environmental characteristics,
- Different data sources can complete each other efficiently,
- With slight adaptation, the framework can be utilised for other systems than energy technologies,
- Further work is needed to conciliate stressors classification across databases and to identify relevant impacts that have been disregarded so far.

6. References

1. Introduction

Every day, unsustainable patterns of consumption and production methods as well as population growth challenge the resilience of the planet to support human activities. At the same time, inequalities between and within societies remain high – leaving billions with unmet basic human needs and a disproportionate vulnerability to global environmental change. To counteract this trend, UNEP and SETAC (Society of Environmental Toxicology and Chemistry) have worked together through the UNEP/SETAC Life Cycle Initiative to develop the current work Towards a Life Cycle Sustainability Assessment (LCSA). A key objective of the UNEP/SETAC Life Cycle Initiative is to help extend life cycle assessment (LCA) methods and practices.

This approach towards a LCSA bases on ISO 14040/44, ISO 26000 and acknowledges the developments of number of international initiatives and experiences about sustainability assessment. The approach presented has been developed by 12 co-authors1 and internationally peer reviewed by more than 20 organizations and individuals. This publication is a natural step in UNEP’s work, which has in the past decade focused on developing the 10-Year Framework of Programmes for Sustainable Development and which is now also focusing on economic sustainability through the UNEP Green Economy Initiative. This publication will increase the awareness of stakeholders and decision-makers in governments, agencies for international cooperation, business and consumers’ associations who are called on to take integrated and holistic decisions on products. This publication describes methods and techniques that can measure sustainability and allow LCA to support decision-making toward more sustainable product and process systems. In this way, life cycle techniques can be used to carry out life cycle sustainability assessments. This guidance document provides a starting point for learning about the methodologies and techniques suitable for life-cycle-based ways of measuring sustainability.

2. Techniques

Different life cycle assessment techniques allow individuals and enterprises to assess the impact of their purchasing decisions and production methods along different aspects of this value chain. An (Environmental) life cycle assessment (LCA) looks at potential impacts to the environment as a result of the extraction of resources, transportation, production, recycling and use and discarding of products; life cycle costing (LCC) is used to assess the cost implications of this life cycle; and social life cycle assessment (S-LCA) examines their social consequences. However, in order to get the ‘whole picture’, it is vital to extend current life cycle thinking to encompass all three pillars of sustainability: (i) environmental, (ii) economic and (iii) social. This means carrying out an assessment based on environmental, economic and social issues – by conducting an overarching life cycle sustainability assessment (LCSA). This publication shows how all three techniques – which all share similar methodological frameworks and aims – can be combined to make the move towards an overarching LCSA possible.

3. Implementing an LCSA

In an LCSA, the inventory and impact indicators must be related to a common product functional unit, which is the basis of all techniques applied. It is recommended that the overall LCSA system boundary contains all unit processes relevant for at least one of the techniques and that all impact categories are selected that are relevant across the life cycle of a product. In LCSA, the LCI compiles exchanges between unit processes and organizations of the product system and the external environment which lead to environmental.

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economic and social impacts. Because of the importance of achieving consistency with the three techniques, it is recommended that data is collected at the unit process and organizational level (see Fig. 1).

The availability of data is another aspect that must be considered; this may be a critical issue in developing countries and in small and medium enterprises when conducting an LCSA. One important issue to consider is the type of data that needs to be collected. S-LCA data is characterized not only by quantitative, but also by qualitative and semi-quantitative information. Therefore, it is recommended that while applying an LCSA, all three types of data are collected along the life cycle.

In the classification step, inventory data are assigned to the impact categories selected and this is feasible in LCSA. However, considering that characterization models are not available for all impact categories and impacted environments, it may neither be possible to convert all LCSA inventory data into common units nor to aggregate them within each impact category required by the characterization step. It is recommended, whenever feasible, that a combined framework for impact assessment based on the individual S-LCA, LCC and (environmental) LCA frameworks (example in Fig. 1) is used.

It is recommended that the results are read in a combined fashion based on the goal and scope definition. The evaluation results may help to clarify: if there are trade-offs between economic benefits and environmental or social burdens; which life cycle stages and impact subcategories are critical; and if the product is socially and environmentally friendly by understanding the impacts of the product's and materials on society.

4. Conclusions

LCsA has significant potential to be used by enterprises, governments, agencies for international cooperation and other organizations in society (such as consumers' associations) in their efforts to produce and consume more sustainable. Still more research and applications are needed, but its application is already feasible and encouraged to speed the learning curve of the society.

5. References

1. Introduction

Life Cycle Sustainability Assessment (LCSA) has been proposed to be a combination of three assessments: environmental, economic and social (incl. socio-economic). In this way LCSA tries to make a bridge between the traditional environmentally oriented and generic life cycle assessment (LCA) and the more site- and time-specific sustainability assessment (SA), which takes into account all three generally accepted pillars of sustainability (economic, social and environmental). In the process of trying to bridge and to draw from the strong characteristics of SA and LCA, LCSA encounters its own challenges.

These issues and challenges are being treated within the EU FP7 project Advanced Technologies for the Production of Cement and Clean Aggregates from Construction and Demolition Waste (C2CA) with application to LCSA for innovative technologies.

2. Materials and methods

A brief review of the challenges in implementing LCSA and interpreting their results is made on the basis of literature review and from a case study. The case study is on innovative technologies for the production of high-grade concrete from construction and demolition waste and address the needs of the construction industry. A brief review on methods for aggregation and interpretation of results is made. The analysis is put in the context of the transdisciplinary integration framework for LCSA proposed by Guinee et al. (2011) (Figure 1).

![Figure 1: Possibilities for Broadening and Deepening in the LCSA Framework (adapted from Guinée et al., 2011).](image-url)

3. Results and discussion

LCSA has been recognized to be the trend in the coming decade (Guinee et al. 2011). As a new and multi-disciplinary assessment, it faces certain challenges, which can be classified as methodological and interpretational.
3.1. Methodological challenges

The methodological challenges come from the nature of life cycle impacts assessment and in turn cause the challenges in interpretation because in order to make decision on a basis of the three assessments, the results of each assessment have to be based on a robust methodology. And while the methodology for the environmental LCIA has been developed for years, and there are already different approaches to LCC, it is recognized that more research is necessary before a robust methodology for SLCA is developed.

For the case study a set of indicators has been selected specifically for the construction sector from the indicators suggested by the Guidelines for Social Impact Assessment of Products. The challenges at the stage of indicator selection and definition arise both from the inability of some indicators to refer to the functional unit and from the validity of the indicator to be used as a measure for a certain impact. An advantage of the current selection process is the high relevance of the selected sector specific indicators for the project stakeholders. A drawback for the current selection process is that these stakeholders may not be representative for the whole sector. Therefore, it is necessary to assess the validity of the selected set of indicators for the construction sector according to different criteria and by a broader range and larger number of stakeholders.

The challenges in the process of data collection refer mainly to data availability and data confidentiality. These challenges are valid for both the foreground and background processes, which involve a supply chain perspective, and lead to the question of whether it is possible for a SLCA to be implemented for the same system boundary as the ELCA. At the stage of impact assessment the challenges come mainly from the need to find an appropriate method for characterization and normalization. The state-of-the-art in SLCA’s methodology is neither advanced, as in the ELCA, nor are indicators measured in a common unit, as in the ELCC. It is therefore a difficult task to come up with SLCA results, which would not be subject to discussion.

Another challenge in the case study comes from the need for aggregation of qualitative, semi-quantitative and quantitative indicators and the interpretation of the SLCA impact results and the applicable approaches to weighting. A possible way to interpret the SLCA results is the multi-criteria assessment, which can account for both different types of indicators (quantitative and qualitative) and various stakeholders (Qureshi, 1999).

3.2. Interpretational challenges

Even if the methodological challenges behind the SLCA results are solved and results from the three assessments are available, the issue of how to support decision-making on the basis of these assessments arises. According to Kloepffer (2008), the three assessments are equal, and this is in line with the concept of strong sustainability. But there is the concept of weak sustainability, as well, where the underperformance in one pillar can be compensated by a better performance in another pillar. In any case, the assessments in LCSA are measured in different units, and depending on where we are in the cause-effect chain, there may be many mid-point or a few end-point categories in each assessment. The challenge then is on the basis of what method to aggregate and interpret the results from these assessments and what is the validity of interpretation for the decision-making.

4. Conclusions

While implementing LCSA for a case study two main types of challenges have been met: methodological and interpretational. The methodological challenges come mainly from the SLCA. In our case the focus on sector-specific indicators is identified as a useful approach for dealing with the recognized site-specificity of SLCA. In regions like Europe, where regulations and policies are unified to a great extent, regional sector policies may facilitate the transition from site-specific to region-specific SLCA indicators, depending on the variation of their implementation across countries and regions.

According to the literature review, a good approach to validate the work with indicators from one or different fields, from their selection to their aggregation and interpretation, has been recommended to be the multi-criteria assessment method, which can also be combined with other methods.

5. References


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1. Introduction

The role of human labour (HL) for the society and its sustainable development is pivotal, as extensively assessed through economic and social indicators [1-3]. However, in the environmental life cycle assessment (LCA) the intrinsic dependence of production systems on HL is usually disregarded, without providing any clear argument. Apparently, HL is not considered to be related to and affected by, changes to the functional unit. During the last decade, the boundaries of LCI have progressively been enlarged to include activities which are linked to the functional unit by market driven mechanisms. If the inclusion of market driven activities is straightforward in a consequential LCI approach, it is generally not addressed in attributional LCIs. This is probably why the contribution of human labour is normally not included in LCIA. Nevertheless, from our point of view, this is an inconsistency, since human activities and specifically HL, fully support the operation of products life-cycles and are, in turn, affected by changes of the functional unit. If a direct link between the use of resources and production systems does exist, as it is normally modeled in LCIs, production systems are also connected to and sustained by human life, which in turn consumes resources and releases emissions for its well-being. An intrinsic relation among biosphere, technosphere and human life cycle exists and is tied together by the exchange of material and energy flows. In this regard, the human labour could be seen as the common numeraire among the three pillars of sustainability: environment, economy and society. This research aims at investigating the relationship between HL and LCA and at developing an operational framework to include HL in LCI and subsequently assessing its lifecycle environmental impact.

2. Materials and methods

The environmental inputs and outputs occurring during the whole life year of a worker (8760 total hours/year) were considered for the system boundary of human labour. We assumed 240 working days/year, 8 hours/day, which means that 22% of the overall annual time is dedicated to labour (1920 hours/year). Finally, 1 hour of working time was taken as the functional unit by allocating 22% of the total expenditures to labour time. In this manner, we were able to spread the environmental burdens that occur during “not-working” hours of the day (e.g. when eating or when travelling to work), but which are of essence for HL. Statistical datasets of household expenditures (HEs) allowed differentiating among human consumption behaviours (which expresses all direct and indirect inputs of resources, materials and services used by people). We defined three HL types, based on different work skills: HL-1 (qualified worker), HL-2 (technician), HL-3 (manual worker). The HEs framework of Luxembourg was used, because of data availability and was then extended to other EU-27 countries. A comparative LCIA of the HLs types was carried out using an environmentally extended input-output model of EU-27, 2005. Afterwards, ten agri-food and industrial LCAs case studies were modified for hybrid LCAs, adding HL inputs to existing LCIs [4] and using the ReCiPe (midpoint) method for LCIA. A simplified diagram of the calculation framework is reported in Figure 1.

3. Results and discussion

The LCIA comparison of HLs showed that HL-1 generates environmental impacts that are always greater than HL-2 and HL-3, e.g. one hour of HL-1, which involves workers with the highest consumption of goods and services, does generate 0.52 kg CO₂-eq, whereas HL-2 and HL-3 generate 0.46 and 0.41 kg CO₂-eq,
respectively. The impact of average HL is higher in EU countries with the highest HEs budgets, e.g. the average HL impact in Luxembourg is 28% to 79% higher than the corresponding HL impact in other EU-27 countries. Within the case studies, the HL significantly contributes to the total impact for several categories (e.g. fossil and ozone depletion up to 16% and 20%, respectively) (Figure 2).

Figure 2: Relative avg. contribution of HL to the total impact (ReCiPe midpoint); this contribution is calculated from the relative ones of the ‘Luxembourgish’ HL implemented for a) 5 industrial case studies, and b) 5 agri-food case studies (mod. from [4])

Although for many technological production chains the physical HL contribution could be negligible (at least when compared with machineries; [5-6]), in some cases HL largely contributes to the life cycle impact (e.g. in the case of handmade manufacturing processes or most of agricultural productions). As a result, neglecting (as is currently done) the input of HL to a process, despite its necessity, is potential a source of error and an eco-profile of HL shall always be added to LCI models that entail a significant direct human contribution. Our proposal could enable to establish a common framework to account for human labour under the three pillars of the life cycle sustainability assessment [7]. So far, additional cost and social/organizational data of HL might be integrated in LCIs to provide additional information for more comprehensive assessment of the real life cycle cost (e.g. through addition of salaries and wages to the Life Cycle Costing (LCC) analysis) or social quality factors of labour (e.g. through implementation of further labour impact categories in the Social Life Cycle Assessment - SLCA) in a production chain.

4. Conclusions and outlook

The case studies demonstrate that adding inputs of HL to the LCI of a product or process can improve its accuracy, since the contribution of HL to the overall LCIA may be significant. The proposed methodology can be used for a future implementation of HL at a level of unit process in LCA, as previously suggested [1]. By using a hybrid approach, possible double counting with non-human (i.e. machine driven) labour is easily avoided, since all processes required to produce and operate the machineries are already included in the process-LCA, while the EEIO model with household expenditures covers only the direct inputs from human needs. The evaluation of HL using an LCA approach allows identifying opportunities to reduce the environmental impact along with the most critical processes of the human life cycle, such as alternative strategies in food supply or in the usage of transport means. Considering HL as a part of a product system helps to reduce the distance between biosphere and technosphere, since humans are put at the same level of the technological/economic activities. The human labour is intrinsically linked to the economic and social aspects of a life cycle. Here, only physical flows of material and energy requirements were included in the evaluation of HL. However, humans are not machineries and they are driven by flows of information, knowledge, educational and cultural experience, and so forth. These are essential items for our future sustainable development but how to integrate them into human labour LCI profiles remains an open task.

5. References


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1. Introduction

Performing a sustainability assessment of new technologies is a complex task, as showed by the definition itself, which refers to two big issues: Technologies and Sustainability. Technologies can be classified in many ways, depending on the different typologies, development levels, effects and impacts on sectors, territories, markets, etc. The relation with sustainability is twofold, because technology can be considered both as the cause of many environmental problems and as the key to solve them. In fact, technologies are nowadays considered the main actor of the present industrial, economical and social evolution and the main cause of the high speed of current changes. Sustainability and sustainable development are very controversial and disputed at scientific and society level. Indeed, sustainable development cannot be considered simply a goal, but rather a social process where shared sustainability principles are taken as the starting point for assessing decisions through an interactive learning process.

Being sustainability a global concept, inevitably calls for a system-wide analysis, a perspective that is at the core of the life cycle approach. A framework for life cycle sustainability analysis has been proposed, namely LCSA [1], which requires the application of LCA, Life Cycle Costing (LCC) and Social Life Cycle Assessment (S-LCA) under specific consistency requirements. The framework has been applied to the assessment of an innovative technology, in order to test its applicability. In this paper, pros and cons of the LCSA approach are highlighted, and questions for further research are pointed out.

2. Materials and method

The LCSA framework has been applied to an innovative technology for tyres residues treatment and SiC production [2], at pilot scale. The analysis investigates the innovative system in comparison with a traditional end-of-life treatment of tyres. The reference option selected is burning in cement kilns, because it is very common and has some problems of social acceptability, and of capacity limits of the European cement industry in perspective. The technology under study is a multi-output system, that we analysed by defining two scenarios:

- Scrap tyres gasification, including production of electricity and SiC + thermal energy production from coal in cement kiln;
- Thermal energy production from scrap tyres in cement kiln + conventional production of SiC + electricity production.

The state-of-the-art of the three methodologies has been applied, represented for LCC and S-LCA by Swarr et al. [3] and Benoit and Mazijn [4], respectively.

3. Results and discussion

The analysis of the framework for this specific application highlighted two major problems:

- Applicability of the available methods of the framework. In fact, LCA, LCC and S-LCA have different degrees of maturity, and S-LCA still needs developments in particular for the impact assessment phase. Moreover, several methodological questions exist for each of the three methods in the specific application, due to the complexity of the technological system.

- Significance of the analysis performed with respect to sustainability. In fact, sustainability analysis clearly shows distinctive marks of complexity theory: non-linear relationships, feedback loops, emergent phenomena, and tangled connections among the parts. Adopting the LCSA framework, the modelling is linear and static, based on technological and environmental relations only. Thus, it

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1 For example, in LCC the monetary flows are modelled but without any modelling of economic mechanism.
is necessary to evaluate whether and to what extent the analysis performed with the LCSA framework is able to describe the most relevant aspects and thus, it can be considered a good proxy of a more comprehensive – but still not feasible – sustainability evaluation. Moreover, someone could argue that costs, as analyzed by LCC, are not the only economic element of sustainability [5].

As far as the applicability of the framework is concerned, the main aspect which emerged from the application was how to cope with the consistency requirements among them, i.e. ideally identical system boundaries. Moreover, theoretically, the most consistent solution would be to use one identical LCI for all three components [1]. However, we experimented the difficulties in guaranteeing such requirements, in particular in relation to the following aspects:

- **Functional unit.** The study at hand shows that the FU becomes tight in S-LCA. In fact, the analysis of social and socioeconomic aspects of technologies and their potential positive and negative impacts requires taking into account the following aspects: the way a technology is perceived and used in a social context; the way in which it affects or transform this context; the way it interacts with technological systems already in place and its physical context; the quantity of use [6]. Thus, the need exists to investigate whether a broader, non FU-based, perspective in the S-LCA would be more appropriate [7].

- **Data availability and their significance.** Data availability is a critical aspect since the technology is at pilot scale, while the effects we would like to measure (environmental, economic and social) are at full-implementation level. Thus, the analysis of the scaled-up system becomes necessary but critical, even more for the economic (in LCC detailed cost data can be estimated for the innovative technology while rough data are the only alternative for the compared system) and social aspects.

- **Scenarios vs product analysis.** For comparative reasons, two scenarios are analysed, an aspect which makes the analysis more complex because of the high number of parameters involved, on which our control is loose.

4. Conclusions

The application of the LCSA framework to an innovative technology is a challenging task, mainly in relation to the following aspects: data availability and their significance; functional unit definition, especially in the case of S-LCA; and scenarios vs product analysis. On the other side, the framework showed also its strenghts in pointing out the most critical aspects of the assessment, on which further analysis will be necessary. Thus, the LCSA turned out to be an important knowledge instrument: it forces practitioners to think about the different options, and leads thus to detect important aspects that at first sight could be considered negligible. However, we suggest to support the application of the present LCSA with other methods and tools, able to take into account also aspects like the social acceptance, in which different ethical values (due to the different stakeholders affected and their own perspective) and risk elements are relevant. Thus, LCSA can learn from the field of Technology Assessment the way in which the problem is dealt with: the technology is at the core of the analysis, but the infrastructure and the organisation around it are equally important ingredients.

5. References

Towards comparative life cycle sustainability assessment of road marking systems

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1. Introduction

Public purchase decisions must cope with shrinking budgets at one hand and increasing demand for environmentally friendly products on the other. This study looks at both economic and environmental impacts of road markings considering the whole life cycle from manufacturing to disposal. For the first time all four major binder-based raw material options are evaluated that can be considered to equip a road section with road markings that provide high visibility even at wet nights for a period of ten years. For the accuracy of the study an external expert panel has reviewed the assessment.

Social aspects are considered in a second step. It is not trivial to quantify these.

2. Materials and methods

By using the LCC- (based on DIN EN 60300) and LCA -Methodology (based on ISO 14040 and following) an objective comparison for the following sprayed line markings is performed: solvent borne (SB) paint, water borne (WB) paint, thermo spray plastic (TSP) and cold spray plastic (CSP). The study also includes the agglomerate systems thermo and cold plastic.

Typical material formulations in characteristic application scenarios have been modelled using the data of corresponding official approval test certificates held by a major local manufacturer of all evaluated technologies. The approval certificates issued by the German Bundesanstalt für Straßenwesen (BASt – Federal Highway Research Institute) exactly define, for instance, both the marking material and the broadcasted glass bead aggregate mixture along with the proper specific consumption per square metre that must be applied in practice on the road to comply with German performance standards.

Empirical data suggests a typical service life of the various road marking systems at a typical average daily traffic of 10,000-15,000 vehicles per day. These crucial assumptions regarding service life of various road markings are in line with recent publications[1].

Empirical data for application costs were supported by applicators.

3. Results and discussion

3.1. LCC of Road marking systems

With respect to initial application cost, the paint systems are apparently cheaper. However, once costs for tendering, disposal and remarking are taken into account, the cold plastic solution is the alternative of choice.
– not only due to its higher quality of performance, but also as it’s the most economic choice on a long-term base, saving 41% of costs in comparison to the paint solution (see Figure 1).

3.2. LCA of Road marking systems

3.2.1 All impacts per 10 years and km road

Looking at the ecological impacts of the road marking systems relative to the corresponding results evaluated for CSP (see Figure 2), once again the durability or service life of a marking system largely determines the environmental impacts. This highlights the necessity of looking beyond the paint formulator’s gate considering all effects from gate to grave particularly the dominant impacts from product usage on the road as well.

4. Conclusions

When considering road marking systems, the price of the initial installation is really only a small fraction of the real price of selecting one road marking product versus another. Looking at the overall lifetime of a road surface, the necessity to refurbish the markings on a more frequent basis can very quickly counter the initial price advantage of a cheaper product – and even make this product the much more costly choice looking at the decision from an overall road lifetime perspective. For environmental aspects that is the same way. Very good looking results at supplier’s gate can change during use phase.

For social aspects officially more durable systems significantly reduce the frequency of reapplying road markings and thereby contribute of the reduction of work zone accidents and traffic down time. We know about the positive social aspects of more durable systems, but to quantify these is not trivial.

5. References


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Evaluating multiple dimensions of sustainability in the case of bioenergy production based on multi-criteria analysis and life cycle assessment

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Abstract

Environmental impacts of products and services have been traditionally the main focus in life cycle assessment (LCA). Recently, also other dimensions of sustainability including economic, social and even cultural aspects have been received more and more attention in addition to ecological impacts. In conscious decision making, decision makers should take into account all dimensions of sustainability simultaneously when evaluating the overall characteristics of products and services.

Multi-criteria analysis (MCA) is a toolbox of methods and approaches based on utility/value theory (e.g. Keeney and Raiffa 1993; see also Leskinen 2008, Mattila et al. 2011, Myllyviita et al. 2011, Seppälä et al. 2002) that evaluate the properties of decision alternatives with respect to multiple, usually conflicting decision criteria. The aim in MCA, or multi-criteria decision analysis (MCDA), is to support decision maker(s) to make as optimal decisions as possible in a given set of decision alternatives and decision criteria.

First, this presentation discusses how MCA can be utilized in LCA when evaluating the sustainability of products and services. The most important benefits of MCA are connected to integration of impact assessments not measured directly in commensurable units and incorporation of subjective preferences into the assessment framework. Some recent methodological development also facilitates versatile possibilities for uncertainty analysis. Figure 1 gives a conceptual framework to evaluate sustainability of decision alternatives in the context of LCA and MCA. In general, the performance of decision alternatives depends on the weights given to various decision elements of the decision hierarchy and the performance of the decision alternatives with respect to the lowest level elements. Different life cycle stages capture the life cycle of the production alternatives from raw material and utilization to end use. Weighting is a voluntary step in LCA, but is enables to measure overall impacts with respect to multiple dimensions of sustainability and to take into account the overall impacts of multiple indicators measured originally at non-commensurable units. The utilization of different problem structuring techniques developed in the MCA community and related disciplines are also useful especially if the sustainability indicators are defined as case-specific. By utilizing MCA evaluations of economic, environmental, social and cultural life cycle impacts, it is possible to analyze tradeoffs between different dimensions of sustainability explicitly. However, one potential problem in considering the different dimensions simultaneously is the determination of functional unit (FU). With some social sustainability indicators, for example, a natural way the evaluate impacts can be to consider the entire production systems instead of social impacts of FU. However, this may case problems when combining such impact assessments with impact assessments that are measured with respect to FU.

Second, this presentation demonstrate the empirical results of forest biomass based bioenergy production in Eastern Finland that was the topic of recent research project funded by Finnish Funding Agency of Technology and Innovation. The project was a joint project of Finnish Environment Institute, University of Eastern Finland and Finnish Forest Research Institute. The project carried out the comparison of the alternative production chains in the methodological framework described above. The analyzed production chains were as follows: (a) Local heat entrepreneurship based on forest chips (Eno energy cooperative). The most local approach of using wood based energy is to harvest biomass from local forests and burn it in local small heat generating plants compensating the use of heating oil or gas. Eno energy cooperative is local wood energy business, which is owned by local forest owners. (b) Wood pellets produced in Finland and distributed to domestic and global markets (Ilomantsi pellet plant). Wood pellet is a processed biofuel with relatively high energy content per volume. Wood pellet is manufactured in order to decrease transportation costs of wood fuels and in order to make the use of the fuel easy and fluent. (c) Direct peat combustion in large combined heat and power (CHP) plants (Fortum CHP in Joensuu). Fortum CHP plant in Joensuu is taking main care of producing heat and electricity for central business district of Joensuu city with 50,000 inhabitants. Plant is using peat-woodchip mixture as a fuel. (d) Biodiesel produced...
from both forest biomass and peat (Varkaus experimental plant). Biorefining and large scale processing of biomass to biodiesel and other liquid fuel fractions is in the focus of large number of development projects in EU. Varkaus experimental plant project is the most progressed biorefinery project in Finland.

The empirical results of the multi-dimensional comparison of the production chains including economic, environmental, social and cultural sustainability as well as the overall performance when all dimensions are taken into account simultaneously are presented. In addition, the methodological gaps and future development needed are discussed.

![Conceptual framework to evaluate sustainability of decision alternatives, when the analysis framework is based in the combination if LCA and MCA.](image)

**References**


Evaluating of sustainability as Environmental performance of the regional energy systems

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1. Introduction

Having reached the time of sustainable strategy renewal, the demand for sustainability evaluation and environmental performance is increasing. In practice, several methods could be used to assess the objective environmental performance, such as ISO 14030, ISO 14040, DPSR framework and indicator-based assessment developed by OECD, but energy and material intensity and efficiency, measurements of greenhouse gas emission, carbon and water footprint, etc.

The basic concept of our analysis is built on the LCA standards of ISO 14040 and ISO 14044 but supplemented by economic and social elements; according to our concept all (three) pillars of sustainability should be integrated into the assessment. In the development of the concept we used several models as a starting point (EIOA, MFA, REEIO, CALCAS, GEM-E3, EVR), selecting their advantages to build a measurable, relatively simplified, transparent, and coherent model specified to our region (at NUTS 2 level), representing an underdeveloped, area that was formerly a heavy-industrial centre.

The energy sector has a significant share in environmental degradation - partly because of more intensive resource usage but mainly because of the share in GHG emissions – rightly occurred the question of the energy sector’s LCA based environmental performance evaluation. Of course the measurement should be expanded with the analysis of possible environmental savings by renewable energy substitution. In our analysis we undertook to estimate the LCA-based environmental performance of the national energy sector with a special focus on the North Hungarian region’s energy system.

2. Materials and methods

We carried out a research to map the used technologies among energetic companies both at country and regional level, and collected data for primary energy production applying an input-output data questionnaire. Environmental, economic and social indicators were also included in this survey. As the share of renewable resources in the region is higher than the national average the structure is different. The life-cycle studies have been extended the use of renewable technologies. As secondary study we analyzed the earlier professional studies concerning the energy sector. The function unit was 1 MJ, the investigated system contents the all power plants. (Figure 1)

![Figure 1 The investigated system and used matrix formula](image)

We applied the our developed model compiling it with the LCA database (Ecoinvent) and the collected industrial input-output data. In this model LCA is built on a matrix structure, and the regional environmental impacts have been aggregated in an environmental performance index (REI), where $I_n$ is a n-dimension unit row vector and $T_k^T$ is a k-dimension unit column vector. The regional environmental performance is the sum of the element of matrix S, which represents the single sector environmental burden by pollutants. So matrix S can be defined as in figure 1, where the rows represent the single sectors ($i = 1, ..., n$) in the region’s economy and its columns represent the measured pollutants ($j = 1, ..., k$). Thus, the $S_{ij}$ element of S matrix is the sector i’s environmental burden caused by pollutant j. More specifically the given $S_{ij}$ element is the aggregated environmental burden of the dominant companies caused by the given pollutant j in the environmental burdens of sector i. The applied LCA method was the Eco indicator 99.
3. Results and discussion

Energy consumption is similar to the Hungarian average, but the structure of energy production is different from country level. In the North Hungarian Region the primary energy production is based on lignite and locally produced renewable, the rate of Hydrocarbon less. The RES potential is high, especially of biomass, but wind, hydro, solar and geothermal energy also increasing, what has a positive effect on the region’s employment and economy. The significant renewable resources are: biomass, hydro, wind and solar PV and thermal.

<table>
<thead>
<tr>
<th>Primary energy production by fuel, total</th>
<th>NHR [TJ]</th>
<th>Hungary [TJ]</th>
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<tbody>
<tr>
<td></td>
<td>1 610</td>
<td>10951</td>
</tr>
<tr>
<td>*of which: coal and lignite</td>
<td>1 315</td>
<td>10555</td>
</tr>
<tr>
<td>*of which: crude oil</td>
<td>4</td>
<td>1219</td>
</tr>
<tr>
<td>*of which: natural gas</td>
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<tr>
<td>*of which: nuclear energy</td>
<td>-</td>
<td>4016</td>
</tr>
<tr>
<td>*of which: renewable energy</td>
<td>291</td>
<td>1 873</td>
</tr>
</tbody>
</table>

| Table 1: The structure of primary energy production, Source: Energy Centre, Hungary |

![Figure 2: Sharing of aggregate impact in NHR](image1)

![Figure 3: Sharing of aggregate impact in Hungary](image2)

On base of above mentioned energy structure we calculated the environmental impacts in both case and the result has been significantly different. While the country level the fossil fuels has biggest impact, on regional level the respirable inorganics. The environmental load per function unit is much more higher country level than regional becouse the energy structure. 3.3. Economic and social impact

Regional data have been derived from macro level I-O tables and regional specific characteristics can be integrated by company data collection (input-output monetary flow survey). Simplification, communicative and easily understandable sustainability assessment is the only way to make them consider the predicted limits of economic and human activity.

4. Conclusion

The evaluating has some uncertainty factors and dilemmas, can be grouped into the following 4 categories: approach, indicators, statistic data and criteria of evaluation. The practical advantages are also promising: the consequent environmental, social and economic evaluation of regional performance can support regional decision-making policy in order to help the region to catch up by a more sustainable route.

5. References


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Integrating life cycle analysis, human health and financial risk assessment for the evaluation of contaminated site remediation

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1. Introduction
When evaluating remediation technologies for contaminated soil and groundwater, the beneficial effect of the remediation, namely cleaner soil and groundwater, are mostly emphasized without consideration of the environmental impact of the remediation activities themselves. Nevertheless, practitioners and decision makers can rely on a broad range of decision tools that can help them to achieve a better balance between economic, social and environmental health aspects of contaminated land remediation. A holistic approach for the management of contaminated land should ideally include an assessment of the environmental risk of the contamination, an assessment of the environmental, social and health impact of the remediation process and a cost-benefit analysis of the remediation project [1].

2. Materials and methods
In the present study, several methods that can be used to estimate the environmental, financial and health impact of a soil remediation process were compared. The environmental impact assessment was performed using two LCA-based evaluation tools, namely the REC (Risk reduction, Environmental merit and Cost) method [2] and the ReciPe method, which is incorporated in the SimaPro software program [3]. The human health risk as inferred from the REC-analysis was compared to the human health risk assessment based on a RISC HUMAN [4] analysis. For the financial risk, a risk matrix according to the PRINCE2™ method [5] was constructed. The comparison and evaluation of the different tools used to estimate the environmental and human health impact of brownfield remediation was based on 3 case studies in which a choice had to be made between soil excavation and off-site cleaning or an in situ technique (steam extraction, thermal soil remediation, soil vapour extraction) as a soil remediation technique.

3. Results and discussion

3.1. Environmental impact
A life cycle framework, including a life cycle management (LCM) approach structuring environmental activities and life cycle analysis (LCA) for a quantitative examination, can be helpful for the selection of site remediation options with minimum impact on the ecosystem and human health. Besides addressing the environmental impact of the remediation activities for, attention should also be paid to the consequence of reintroducing a remediate site into the economy [6].

For the environmental impact assessment, both the REC and ReciPe methods result in a single score for the environmental impact of the soil remediation process. The ReciPe method takes into account more impact categories, but is also more complex to work with and needs more input data. However, for the evaluation of soil remediation alternatives, estimation of the environmental impact with the REC method will in most cases be sufficient. Within the routinely evaluation of soil remediation alternatives, a detailed LCA evaluation will often be too time consuming and costly. Moreover, both evaluation methods allowed to drawn the same conclusion concerning the environmental impact of the soil remediation options: excavation and off-site cleaning has a more pronounced environmental impact than the in situ soil remediation technologies analyzed in the 3 case studies.

3.2. Human health risk assessment
The human health risk assessment gives similar results with both the REC and RISC HUMAN method, since both methods are based on the same model. However, the risk assessment part of the REC method is
easier to use compared to the RISC HUMAN method, therefore the former method is preferred in routine soil remediation evaluations. It is important to notice that human health risk assessment is also (partly) included in some models for LCA.

3.3. Financial risk of soil remediation projects

Certain soil remediation technologies, especially the more ‘gentle’ remediation technologies are characterized with a lot of uncertainty with regard to the time frame in which the final remediation goals will be achieved. Unexpected situations can result in an increase of the costs of the remediation project. Therefore, there is a need for practical tools that help practitioners in choosing the correct technology that will not only be effective but also will minimize the financial risk associated with the cleanup.

For the case study where steam extraction was compared with soil excavation and off-site cleaning, steam extraction was the preferred technique based on environmental and human health impact of the remediation. However, steam extraction represents the most important financial risk, despite its lower total cost compared to excavation. Because the quite innovative character of this method, the experiences with this method are limited and it is difficult to rely on previous experiences when (technical) problems arise during the remediation project. A successful soil remediation project that makes use of steam extraction requires a more detailed characterization of the site contamination in comparison with soil excavation. The consequences of missing or incorrect data concerning the contaminated site (characterization of soil and subsoil) carry a higher probability of financial risks, while excavation is generally characterized by a more important severity of the risks, but with a lower probability of occurrence.

4. Conclusions

Life cycle analysis offers a powerful tool for the evaluation of the environmental impact of soil remediation technologies, but is too complex for a routinely incorporation in the evaluation of the environmental impact of soil remediation operations. Nevertheless, a simplified LCA-based evaluation, in combination with a more detailedd assessment of the impact on human health and an estimation of the financial risk associated with the soil remediation can contribute to more sustainable management and/or clean-up of contaminated sites.

5. References

Developments in Social Life Cycle Assessment (S-LCA) for Life Cycle Sustainability Assessment (LCSA) – application to the construction and demolition sector in France

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1. Introduction

This paper presents the results of a research project carried out in collaboration with the French agency of environment and energy control (ADEME), University of Technology of Troyes, the Industrial Ecology Club of Aube and the French public works firm Eiffage.

The project was based on a specific case-study: the construction of a part of the ring road of Troyes city (Aube, France) where alternative construction materials and techniques, such as secondary raw materials and local natural resources, have been used. After a preliminary environmental assessment of the project highlighted its environmental benefits, the issue of its sustainability still had to be addressed. Accordingly our research aimed at developing a specific approach for sustainability analysis dedicated to such infrastructures and buildings construction projects.

2. Materials and methods

A life cycle sustainability assessment has been conducted, built on environmental and social LCA, to address the environmental and social efficiency of such construction projects. It has been applied to the case study in order to compare the global performance of the road construction with the performance of a similar case which would have been conducted under more “business as usual” conditions.

On the basis of UNEP-SETAC Guidelines [1], a social life-cycle assessment has been developed to assess the social impacts supported by the system’s stakeholders. This second step has required to adapt UNEP-SETAC methodological recommendations in order to be able to assess social impacts of construction activities in a Western European context.

3. Results and discussion

3.1. Social impacts assessment

A screening of impacts was conducted to identify potential social impacts supported by the system’s stakeholders. Specific impact categories, indicators and data have therefore been defined and searched for, considering (1) social LCA UNEP-SETAC Guidelines, (2) constraints encountered in this case study regarding data availability, and (3) specificities of the French construction and demolition sector’s activities (Table 1).

<table>
<thead>
<tr>
<th>Stakeholders</th>
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<tbody>
<tr>
<td>Workers</td>
<td>Illegal working</td>
<td>Local communities</td>
<td>Access to material resources</td>
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<td></td>
<td>Salary</td>
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<td>Delocalization and migration</td>
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<td>Working hours</td>
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<td>Safe and healthy living conditions</td>
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<td>Equal opportunities / discrimination</td>
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<td></td>
<td>Health and safety</td>
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<td>Local employment</td>
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Table 1: Social-LCA: impact categories for workers and local communities stakeholders

The analysis relies on a sectorial approach and includes identification of all unit processes related to the case studied, i.e. to the construction of the ring road in the “real” case and in a standard or “business as usual” fictitious case. This impact screening has mainly focused on two stakeholders’ categories: workers and local communities.

Results show some positive potential social impacts, especially related to employment aspects. This is due to the fact that construction activities and raw material transportation by truck are labour-intensive activities, generally offering higher salaries than other activities employing workers from similar socio-professional
categories. However, some negative potential social impacts exist, especially regarding risks for workers health and safety and for neighbours well-being. Compared to a business as usual case, the case studied is characterized by a mitigation of its social and societal potential impacts, thanks to lower raw material volumes extracted or dumped and transported: potential social impacts turn out to be highly dependent on the unit processes’ material flow size.

3.2. Developments for life cycle sustainability assessment

As an environmental impact assessment was also conducted for this case study, following international methodological references for LCA [2], this research also focused on the definition of an approach of the sustainability of human and industrial activities. This approach gives a conceptual basis for environmental and social LCA combination (Figure 1), on which life cycle assessments can rely to tend to a life cycle sustainability assessment of systems.

![Figure 1: Approach of the sustainability of human and industrial activities: conceptual basis for environmental and social LCA combination.]

4. Conclusions

Final results highlight the fact that social LCA needs today strong methodological developments:

- To produce methodological inputs for system definition, data inventory methods and reference databases adapted to different socio-economic contexts, relevant indicators and impacts characterization methods, etc.
- To address for example intra Western Europe comparative case-studies, and more generally impacts of products or systems for which unit processes are located in comparable areas in terms of socio-economical development level.

More case studies results are also needed to refine and illustrate social LCA feasibility and relevance.


Acknowledgement - The authors thank Gregory Lannou, Sabrina Brullot and Marion Stoos, from University of Technology of Troyes, Patrick Thomassin, from Eiffage, and Nicolas Juillet, from the Industrial Ecology Club of Aube, for their valuable contribution. They also thank scientific officer Laurent Chateau and the French agency of environment and energy control (ADEME) for having supported this research.
1. Introduction

Currently, electricity generation contributes to 40% of global carbon dioxide (CO₂) emissions. Carbon dioxide capture and storage (CCS) technology is widely recognized as an appropriate option to achieve ambitious CO₂ reduction targets. There are three main technology routes (post-combustion, oxyfuel, pre-combustion) that offer removal of CO₂ from combustion and gasification processes. The environmental performance of CCS beyond the reduction of CO₂ emissions has increasingly become a subject widely discussed in several studies in the last 10 years. This meta-analysis comprises fifteen LCAs of the three CCS technologies with a focus on greenhouse gas reduction for different regions (Europe, United States, Japan, global), different fuels (hard coal, lignite, natural gas), and different time horizons (between the present and 2050). The goal of this meta-analysis is to provide a structured overview of assumptions and methodological choices made and, where possible, their effects on the outcomes.

2. Results and discussion

The CCS technologies are compared considering different capture techniques and types of fuel. As most studies include coal-fired electricity production, the results shown here refer only to hard coal exemplarily. Analyses for lignite and natural gas, as well as oxyfuel and pre-combustion can be found in a comprehensive LCA study comparison, commissioned by the IEA GHG R&D programme [1]. As CCS is a future technology, no common understanding of future efficiency of commercial power production exists, let alone of energy penalties due to capture. It is often not clear which detailed technical assumptions, technological representation or emission reduction efficiencies are used for the analyses. For hard coal post-combustion, efficiency values lie between 30% and 50% (Figure 1), representing different stages of technology, but also different technology concepts. Additionally, the perception of efficiency losses due to the capture system diversifies between 6 %-points and 18 %-points across all capture technologies.

Upstream and downstream process chains (coal, solvent and energy supply) are often not represented with the same quality as the main processes. However, the results clearly show the significant influence of those processes on overall emissions and their impacts. For power plants with CCS, this influence is generally higher than for power plants without CCS due to losses in efficiency and the associated additional amount of fuel. The estimated share of CO₂ transportation and storage on environmental impacts varies by one order of magnitude, depending on the capture performance and fuel used. Also, transport distance, number of recompression steps along the pipeline, type of storage (gas field, saline aquifer), and depth of injection affect the results, but to a smaller degree.

For comparison of environmental effects only those categories are chosen for which a sufficient number of studies use the same impact indicator: global warming potential (GWP), acidification potential (AP), eutrophication potential (EP), photochemical oxidation potential (POCP), and cumulative energy demand (CED). The absolute GWP without capture varies from 765 g CO₂-eq./kWh to 1,100 g CO₂-eq./kWh,
depending on the estimated efficiency and type of coal used (Figure 2). Acidification potential values are more scattered. EP, POCP, and CED do not vary considerably between the studies.

![Figure 2: Environmental impacts of hard coal combustion technology without capture, along with normalized values related to total global emissions in 2000 calculated for lowest (best case) and highest impact values (worst case)](image)

All LCAs show the expected reduction in global warming potential but an increase in almost all other impact categories, regardless of capture technology and time horizon considered (Figure 3). The normalization shows that power generation without CCS has a considerable share on the total global GWP, with 13.2% and still 10% assuming worldwide power production with only low performance plants and best technology, respectively. The share on global AP using worst-case technologies is 3.5%, while best-case technology reduces AP to about one percent. Effects on the EP and POCP are even smaller (Figure 2).

![Figure 3: Relative impacts for power plants with post-combustion/MEA capture, along with normalized values related to total global emissions in 2000 calculated for lowest (best case) and highest impact values (worst case)](image)

Three parameter sets have a significant impact on the results: 1) power plant efficiency and energy penalty, 2) CO₂ capture efficiency and purity, and 3) fuel origin and composition. However, the normalization indicates only a small impact from CCS power plants on total global environmental impacts.

### 3. Conclusions

This meta-analysis proves that LCA is a helpful tool to investigate the variety of environmental consequences associated with CCS. However, there are differences in the underlying assumptions of the LCAs as well as methodological shortcomings that yield heterogeneity of results. A sophisticated and common understanding of the most important technological parameters is necessary to draw a clearer picture of both single CCS techniques and comparisons across techniques. Therefore, it is essential that LCAs include well documented parameters and describe uncertainties and assumptions precisely. There also remains a wide field of subjects and CO₂ capture technologies of 2nd generation (like membranes) that have not been covered yet.

### 4. References


**Acknowledgement** - The authors are grateful to Mike Haines for his contribution to the earlier, full version of this research [1]. The authors would like to thank the IEA Greenhouse Gas R&D Programme for financial support.
Simplified life cycle approach: GHG variability assessment for onshore wind electricity based on Monte-Carlo simulations

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1. Introduction

The environmental impacts of electricity production systems have been widely assessed over the past years with many published LCAs in the literature. In the special case of greenhouses gases (GHG) from wind power electricity, the LCA results variability observed is very high, for example ranging from 2 to 81 g CO\textsubscript{2eq}/kWh in a literature review performed by the IPCC [1]. Such result might lead policy makers to consider LCA as an inconclusive method [2].

The main objective of this paper is to build a representative model of onshore wind turbines environmental performances with a simplified life cycle approach. Variability of GHG performances of onshore wind turbines, generated for a representative sample, is assessed through the running of Monte-Carlo simulations to identify the key parameters having the biggest influence on the results. Finally, this methodology will enable to define generic curves of onshore wind power electricity GHG as a simplified function of these key parameters.

2. Method

The methodology developed to establish a generic GHG performance curve is described according to the 6 steps below (more details on the methodology is to be found in [3]):

1. An extended literature survey on LCAs of onshore wind power electricity is performed to obtain a clear overview of the most representative assumptions taken in the literature and their related variability.
2. GHG impacts of onshore wind turbines (WT) are analysed by disaggregating the existing onshore wind turbine inventory in ecoinvent v2.2 model [4].
3. These two steps enable to define a reference wind turbine model, representative of the current WT technologies.
4. This WT model is applied to build the sample inventories representative of the recent wind power market (the sample is made of 17 WTs).
5. By performing standard sensitivity analyses on LCAs results from this sample, the identification of two key parameters (the turbine life time and the wind speed) are identified. GHG performances generic curves are then elaborated on these two key parameters.
6. Finally a validation process is performed in order to assess the range of validity of the simplified model made out from these generic curves by comparing them with reference LCA studies.

Two limitations on this methodology are now discussed. These first results were based on a restricted wind turbine sample and local sensitivity analyses were not handling the variability issue in a comprehensive way [5]. We therefore propose here an enhanced version of the initial methodology reported in [3] by applying a Monte-Carlo simulation step to a more representative sample. With this step, the sample restriction as well as the limitation of the local sensitivity analysis approach will be overcome.

For the Monte-Carlo simulations, we used the following equation to calculate the GHG performances defined as the ratio between the environmental impacts of the onshore turbine over the electricity produced over its life time:

\[
\frac{\text{Onshore wind power electricity GHG performance}}{8760, A, L, LT, P} = \frac{\text{WT impacts}}{P \times \text{LT}}
\]

Each parameter and its related distribution need to be characterized for the Monte-Carlo simulations:

- P is the nominal WT power in kW and characterized as a discrete function of the installed WT statistics in France between 2003 and 2008, ranging from 800kW to 3500kW with a high probability for 2000 kW.
- LT is the system lifetime. According to expert discussions and literature review, this parameter range from 10 to 30 years with a normal distribution centred on 20 years, with a standard deviation (SD) of 3 years.
- A is the availability factor considering the real time when the WT is working compared to the theory. It ranges from 0.9 to 0.99 with a normal distribution centred on 0.945 with a SD of 0.015.
The WT impacts have been modelled using empirical equations to size the wind turbine as a function of their nominal power. These equations have been defined with an extended literature survey of the different existing WT. To consider uncertainty of this approach, for each flow, an uncertainty of ±15% has been associated.

- L is the load factor parameter. It represents an equivalent annual percentage when the WT is producing at nominal power. It is the product between the WT power curves and the wind speed Weibull distribution (details in [3]).

3. Results

The Monte-Carlo simulations have been performed on 25 000 runs varying each input according to the defined distributions.

Based on these Monte-Carlo simulations, we plotted GHG performances distributions for two key identified parameters: the WT life time and annual wind speed. Results are ranging from 2.7 to 119.7 g CO$_2$eq/kWh, a range which is comparable to the literature review observed in [1].

4. Conclusions

The initial methodology to generate simplified models for WT environmental performances has been improved with a better identification of the variability assessment. A systematic representative sample has been defined on which Monte-Carlo simulations have been applied. Running Monte-Carlo simulations on a representative model of onshore wind turbines (mostly 2 MW WT) has confirmed that GHG performances variability is mainly explained by the WT lifetime and the wind speed. A set of generic GHG performances curves has been defined as a function of these key parameters. The obtained results range between 2.7 to 119.7 g CO$_2$eq/kWh and can be adjusted as a function of either one or both key parameters. This methodology will be applied later for all types of electricity generation systems.

5. References


What can meta-regression analysis tell us about variations in Life Cycle Assessment (LCA) results for Greenhouse Gas (GHG) emissions estimated for advanced biofuels?

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1. Introduction

This paper presents a systematic literature review of studies using a LCA approach for the environmental evaluation of advanced biofuels. The main factors influencing the results of these studies are characterized and estimated using a meta-regression analysis. Such assessments allow to analyse and compare LCA results from various sources in order to clarify conclusions concerning environmental performances of advanced biofuels to make it easily understood for policy makers. The term "advanced biofuels" refers, here, to biomass-based transportation fuels produced from lignocellulosic materials (ethanol and biomass-based synthetic diesel – BTL) and microalgae (fatty acid methyl ester – FAME and hydrogenated vegetable oil – HVO), also known as second and third generation biofuels respectively.

The transport sector was responsible for 23% of the world's GHG emissions in 2009 and the transport fuel demand is projected to rise steadily in the coming decades [1]. Biofuels are deemed to be an alternative to petroleum based fuels and advanced biofuels are currently being developed, as they seem to be more efficient in terms of land use, GHG emissions and other environmental aspects than conventional biofuels (i.e. first generation biofuels produced in commercial scale nowadays). The interest of policy makers for advanced biofuels is reinforced by climate change issues. For instance, the United States of America and the European Union have recently developed regulation inciting biofuel production: the Renewable Fuel Standard (RFS) and the European Directive 2009/28/EC (a.k.a the Renewable Energy Directive – RED), respectively¹. Both texts include a minimum threshold for GHG emissions savings compared to fossil fuel reference as the most important environmental criteria that should be met to be eligible as sustainable biofuel.

2. Materials and methods

LCA is seen in the literature as the appropriate methodology for the environmental evaluation of biofuels and it has been widely applied by the scientific community to calculate the Global Warming Potential (GWP is an impact indicator for GHG emissions) associated with biofuel life cycle. Nevertheless, it has been shown that LCA results can vary significantly depending on various factors. In the case of biofuels, some of these leading factors are: the assumptions made to describe the biomass production step (model used to estimate N₂O emissions and inclusion of direct and indirect land use change), the data used to describe the biomass conversion into biofuel and the general LCA methodological choices (system boundaries, the method used to account for coproducts impacts, etc.).

Here, we propose an alternative approach to previous narrative surveys of biofuel LCA studies [2][3]. Only applied in few studies for LCA result analysis [4][5], the meta-regression analysis methodology is used to describe and synthesize existing estimates of the GWP of advanced biofuels. This quantitative research method has been developed to compare and/or combine outcomes of different individual empirical studies with and without similar characteristics that can be controlled for [6][7][8]. By nature, each result from an individual empirical study may be quoted to illustrate the sampling uncertainty of estimates. Estimates of previous empirical studies are grouped together in a database, according to one or more differentiating characteristics. Then, estimates are assumed to be a function of these variables and their effects are assessed by the mean of specific econometrics methods. This multivariate setup allows the identification and the effect quantification of the characteristics relating to the sample or to the author's methodological choices that lead to the greatest impact on the results. Besides, this framework gives the possibility to produce an estimate of the “mean” estimate after controlling for its moderators by establishing the extent to which the variation is systematic.

¹ While the RFS mandates the production of specific quantities of biofuels until 2022, the RED mandates the use of 10% of renewable energies in transports by 2020.
Therefore, a database has been built. It contains a vector of previous studies estimates of GWP (GHG emission indicator expressed in mass of equivalent CO₂ per megajoule of biofuel) that is the dependent variable of the model, and a vector of explanatory variables. We choose GWP expressed per MJ as the dependent variable because GHG emissions reduction is the most important environmental criteria in the American and European regulatory texts. The explanatory variables are the factors that can potentially influence LCA results (see above) as well as some study characteristics (authors, country, year of publication, etc.), which allows accounting for potential publication biases. Parameters of the meta-regression model are estimated by mean of both fixed-effects and random-effects methods. Advanced biofuel LCA results from peer reviewed articles, research reports (grey literature) and regulatory texts (RFS and RED) are included in the database through systematic research of relevant databases. All the retained studies follow the ISO 14044 guidelines to conduct a LCA. Finally, 43 LCA studies are included, providing 585 estimates.

3. Results and conclusions

Preliminary results show, ceteris paribus, that the mean life cycle GWP associated with 1 MJ of ethanol, BTL and algae fuel are 27, 21 and 83 gCO₂eq respectively. In comparison, the RED reference value for fossil diesel and gasoline is 83.8 gCO₂eq/MJ. These mean values show a clear advantage of 2nd generation biofuels compared to 3rd generation in terms of GHG emission reductions.

The factors responsible for the variability of GWP estimates in the different studies are also examined in this work. The meta-regression analysis indicates that GWP estimates are higher in studies that take explicitly into account estimate uncertainties and Land Use Changes or that include infrastructures in the system boundaries. Conversely, GWP estimates are lower in studies accounting for other impact indicators than only GHG emissions. Moreover, our results indicate that regulatory texts (RFS and RED) provide lower GHG emissions estimates than peer reviewed studies and that there is a localization effect on these estimates: estimates from European studies are statistically higher than American ones.

4. References

Quantification of uncertainty of characterisation factors due to spatial variability

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1. Introduction
Except for a minority of foreground production sites, the geographic location of elementary flows is usually, at best, known at country resolution. More precision is needed to choose the correct spatialized characterisation factors (s-CF) from a set. Countries are usually covered by several s-CF, and their value might span over several orders of magnitude [1][2]. So far, how this source of variability is affecting the uncertainty on the CF has not been quantified. This paper presents a general method quantifying the uncertainty created by the dichotomy in spatial resolution between inventory and impact assessment, and its application to a water assessment method.

2. Materials and methods
For each country, a generic CF (g-CF) is calculated with the s-CF covering it. Two elements are necessary to build the g-CF: the probability density function (PDF) of each s-CF, and a set of weighting factors. The weighting set can be provided by several source. When available, reported rates of emission (for example, Toxic Release Inventory in the United States) should be used as a weighting set. Otherwise, another quantity deemed a good approximation to the distribution of actual emissions should be chosen, for example, surface area or population density, assuming that likelihood of emission are correlated with those. The weighting set provides a likelihood of emission on the zone covered by each s-CF, an information lacking in the generic unit processes.

Figure 1: Construction of the PDF of a g-CF based on weighted s-CF.

The average and standard deviation of the g-CF is not directly calculated from the weighted s-CF. Instead, the PDF of the g-CF is built by adding the weighted PDF of each s-CF:

$$PDF_g = \sum_i weight_i \times PDF_i$$

As shown by the generic example in figure 1 (bottom), the resulting PDF might be multi-modal. The average value of the g-CF can be calculated from the PDF and used in deterministic calculations. The standard deviation is useful if the resulting PDF can be approximated by a normal or lognormal. Otherwise, the full PDF should be reported and used directly in a Monte Carlo simulation.

Finally, the coefficient of variation (CV = average/standard deviation) of the g-CF can be compared to the CV of the s-CF. The CV of the g-CF will be higher, and this augmentation of uncertainty reflects the spatial variability that cannot be taken into account, due to the lack of spatial information in the generic unit processes. In other words, unaccounted spatial variability is transformed in uncertainty.

The impact category chosen to demonstrate this methodology is water use on human health [3], mainly dependant on adaptation capacity of countries and scarcity of water in watersheds. Therefore, the
delineation is the intersection of countries and watersheds, resulting in 808 cells, for 17 different water qualities. The likelihood of water use in each cell is based on data from the Watergap model [4].

3. Results and discussion

Figure 2 shows the result for good quality water (S2a) for China (20 watersheds). Every s-CF contains 4 parameters with quantified uncertainty. Using Monte Carlo simulation, the s-CF are found to be lognormally distributed, with CV between 0.6071 and 0.6218. The only notable exceptions are the 8 watersheds, where good quality water is not scarce, therefore, with a s-CF equal to zero and without uncertainty. The PDF of the g-CF (Figure 2, third graph) has an average of $1.233 \times 10^{-4}$ DALY/m$^3$ with a CV of 7.63. This means that if a use of good quality water occurs somewhere in China, without further information on the watershed of consumption, it will be characterized with a g-CF of a value potentially much greater or lower than with the right s-CF, and roughly 10 times more uncertain. The PDF of the g-CF is bimodal, due to the majority of water being used in 5 watersheds with s-CF values around the two modes observed. A possible simplification to facilitate the use of such distribution would be to approximate it by the sum of 2 lognormals.

![Figure 2: PDFs of s-CF and the resulting g-CF for good quality water in China.](image)

4. Conclusions

It is now possible to calculate the uncertainty due to the lack of spatial resolution in the inventory, compared to spatialized LCIA methods. This new information could help LCA practitioners justifying time spent collecting spatial data about foreground and background unit processes. It also puts pressure to include the same data in LCI database.

5. References


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Matrix-Based Sensitivity and Uncertainty Assessment for Evaluating Human Intake of Pesticide Residues in Food

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1. Introduction
Multiple pathways contribute to human exposure towards pesticides, i.e. inhalation after emissions to air, ingestion after emissions to soil, ingestion of drinking water after emission to water and, most importantly, ingestion of directly sprayed food crops. Different models exist for assessing emissions (usually spatial or nested steady-state tools), some with 1 or 2 generic plant compartments and some without considering plants at all. Currently, none of these models is valid for estimating residues from direct pesticide application to food crops, which requires dynamic assessment. We, hence, developed a dynamic multicrop model for assessing pesticide residues in food crops and subsequent human intake [1, 2]. However, full uncertainty analysis is still lacking in this approach. We already identified half-lives in plants and time between substance application and crop harvest playing a crucial role [2], but other aspects may also be important. A deterministic analysis is needed to provide more understanding and to allow incorporating our dynamic models into spatial frameworks by linearizing the complex system into a simplified regression.

2. Materials and methods
We designed a simple but accurate regression model approach with time from application to harvest, half-life in plants, residence times in the environment and key substance properties as influential input variables. Additional aspects influencing the pesticide behavior in the dynamical system are identified by assessing relative sensitivities of a wide range of input variables according to [3], thereby addressing parameter correlations and defining geometric standard deviations for all relevant input variables as a function of their base uncertainty and spatial/temporal variability as presented in Table 1.

<table>
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<th>base uncertainty</th>
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<tr>
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<td>high</td>
<td>4.0</td>
</tr>
<tr>
<td>extrapolation from uncertain data</td>
<td>very high</td>
<td>5.0</td>
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Table 1: Classification of geometric standard deviations for input variables (excerpt).

Based on relative sensitivities of input variables, parameter correlations and information about input variable uncertainties expressed by their geometric standard deviations we studied the uncertainty propagation of the model output, i.e. the relative contributions of input variables to overall model output uncertainty. Thereby, we applied a new approach of calculating the overall output uncertainty as a function of the matrix of relative sensitivities of input variables and the covariance matrix expressing their correlations.

3. Results and discussion
Model output, i.e. human intake fractions from consumption of food crops treated with pesticides, showed highest relative sensitivities across substances to half-lives in plants and on plant surfaces as well as to the time between substance application and crop harvest (Figure 1). Total output sensitivity is a function of the crop species and is highest for leafy vegetables (lettuce) and fruit trees (apple). Soil-related input variables
contribute most to output sensitivity for root crops (potato), whereas for all other crop types plant-related input variables are predominant with increasing importance of soil-related input variables for pesticides with higher persistence in the environment.

![Figure 1: Contribution of input variables to model output sensitivity for fungicide azoxystrobin.](image)

Uncertainties of input variables increased with both increasing contribution to model output sensitivity and increasing geometric standard deviation. In addition, co-variances with other input variables further increased uncertainties of particular variables.

In addition to the substance-specific uncertainty we studied the variability across pesticides, from which we derive crop-specific regression models predicting residues in food crops across pesticides within a factor of 10 of those calculated with the complex model as a function of only a handful of input variables. These simplified models are adequate to assess direct residues for multimedia models used for risk and impact assessment and, hence, enable the user to calculate direct pesticide residues by only providing a very limited set of input information.

4. Conclusions

We analyzed the relative sensitivities and uncertainties of a wide range of input variables important for dynamically assessing pesticide residues in food crops with implications for human intake. Results indicate that time between substance application and crop harvest as well as half-lives in plants and on plant surfaces are the most important aspects contributing to model output uncertainty, of which the latter has very high input variable uncertainty, if no measured data are available, which is usually the case. This highlights the need for estimation of plant half-lives for improving current modeling practice. Building on the findings from the sensitivity analysis, we provide simplified regression models for different food crops based on linearization of respectively only a handful of input variables.

5. References

Uncertainty classification and implementation in life cycle impact assessment: application to freshwater ecotoxicity of pesticide application to Maize in The Netherlands

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1. Introduction
How to deal with uncertainty has become a key challenge for integrated assessments. As yet, the application of an uncertainty analysis is not common practice in life cycle assessments. A proper analysis will be facilitated when it is clear which types of uncertainties exist and which tools are available to deal with them. The aim of this research is to set up a framework to classify levels of uncertainty in life cycle impact assessment (LCIA). This is done by reviewing the literature. Operationalization of uncertainty quantification in LCIA is demonstrated with a case study example on freshwater ecotoxicity assessment of pesticide application in The Netherlands.

2. Materials and methods
We distinguished three levels of uncertainty:
1. Statistical uncertainty arises from measurement errors, analytical imprecision, and limited sample size [1]. It is any uncertainty which can be characterized in probabilities [2].
2. Decision rule uncertainty arises whenever there is ambiguity or controversy about how to quantify or compare social objectives [3]. Both ambiguity and controversy are linked to the existence of multiple choices resulting in value judgments that need to be made by decision makers.
3. Model uncertainty is defined as uncertainty about the relations and mechanisms being studied [4]. Model uncertainty is introduced through simplifications of reality that may exclude relevant variables from the analysis.

To quantify parameter uncertainty, uncertainty distributions need to be derived for input parameters. Decision rule uncertainty can be made operational with the help of a choice analysis. To show the effect on life cycle assessment outcomes the choices included in the model need to be identified, possible options to quantify every choice identified need to be defined, and the results need to be calculated for all combinations of choices or for a limited number of choice combinations. Model uncertainty can only be quantified when the simplifications of the model are identified and the real situation and model simplification can be determined.

A case study that addresses the application of pesticides to maize in the Netherlands was performed to show the application of the various uncertainties in freshwater ecotoxicity. The Impact Score for freshwater ecotoxicity (IS_{ecotox} in yr per kg maize) was calculated as follows:

\[ IS_{ecotox} = \sum_{x=1}^{n} \left[ AR_x \cdot \left( M_{x, fw} \cdot CF_{x, fw} + M_{x, s} \cdot CF_{x, s} \right) \right] \]  \hspace{1cm} (1)

where \( AR_x \) is the application rate of pesticide x (kg per kg corn), and \( M_{x, fw} \) and \( M_{x, s} \) are the emissions of pesticide x to freshwater and agricultural soil (in fraction of total emission), respectively. \( CF_{x, fw} \) and \( CF_{x, s} \) are the characterization factors for emissions to freshwater and soil, respectively. The 10 pesticides responsible for 99% of the pesticide use on maize in The Netherlands in 2004 were included. These are dimethenamid-p, terbuthylazine, sulcotrione, glyphosate, pyridate, bentazone, dicamba, nicosulfuron, mesotrione and bromoxynil.

Statistical uncertainty was calculated by performing a Monte Carlo analysis on all chemical-specific input parameters to derive the CFs. Decision rule uncertainty was determined by identifying choices in the derivation of the CFs and subsequently determining CFs for every choice scenario. The following choices were included in our case study: (1) Availability of toxicity data; (2) Choice of effect factor model, i.e. linear or non-linear; (3) Damage factor, i.e. modeling up to midpoint or to endpoint. Time horizon and red list species are not of importance for pesticide application.
3. Results and Discussion

Figure 1: Damage per kg Maize due to pesticide use in 2004 in The Netherlands per kg Maize. The center of each box equals the median score, the edges of each box the 25th and 75th percentiles, and the whiskers the 5th and 95th percentiles. Figure 1A compares scenario’s where all chemicals are included and chemicals for which at least 10 toxicity data are available, and on midpoint and endpoint. Figure 1B compares model uncertainty on midpoint and endpoint: on the right side results when transformation products (TPs) are included.

Figure 1 shows the results for the linear effect factor method. Going from midpoint to endpoint applying an uncertain damage factor decreases the damage slightly, but hardly changes uncertainty. Excluding the chemicals for which less than 10 species toxicity tests were available slightly decreases the damage per kg maize. As only for parent compounds toxicity tests for at least 10 species are available, model uncertainty can only be compared for the scenario when all toxicity tests are included. The inclusion of effects on transformation products increases median damage, especially on midpoint, and increases uncertainty from 2 to 14 orders of magnitude. When transformation products are included, going from midpoint to endpoint increases median damage clearly.

4. Conclusion

This research showed the interaction between statistical, model, and decision rule uncertainty. Results indicated that damage can vary substantially depending on the value choices made, and, moreover, parameter uncertainty can increase to a large extent when a more accurate model is applied.

5. References


Uncertainty analysis in macro-level life cycle assessment

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1. Introduction

The need to assess environmental impacts of human activities has led to the development of complex models given the availability of more powerful computers and the increasing understanding of interactions between human activities and the environment. This is especially true with the introduction of economic modeling in consequential life cycle assessment (C-LCA), an LCA approach designed to study the environmental consequences of economic interactions due to changes occurring within life cycles. Traditionally, the application of C-LCA has focused on marginal changes. The complexity of the environmental modeling activities is further increased when macro-level changes occurring in the future are modeled. The macro LCA (M-LCA) approach, which builds on C-LCA and prospective LCA, was proposed to model the medium- or long-term environmental consequences caused by macro-level changes occurring simultaneously in multiple life cycles [1]. The basis of the M-LCA approach is to use the GTAP model [2], an economic general equilibrium model (GEM), to compute economic consequences of life cycle changes that are expected to affect the economy and then to use LCA methodology to convert economic consequences into environmental impacts. Considering the high uncertainty related to complex models such as the economic GEM [3] or LCA [4], uncertainty assessment on M-LCA results appears to be a relevant issue. For this purpose, an uncertainty analysis was conducted on results of a case study were M-LCA was applied to compare two European Union (EU) energy policies (business as usual vs. bioenergy) for the time period of 2005-2025 [5].

2. Method

After identifying M-LCA sources of uncertainty, an hybrid approach, based on developed uncertainty scenarios [6] and Monte-Carlo (MC) analysis [7], was conducted to assess and manage global M-LCA uncertainty. Combining potential values for macroeconomic and technological variables used by GTAP, 27 scenarios were created. GTAP simulations were run for each scenario drawing a tree of possible futures given assumptions made in each scenario. MC simulations were then conducted for the 27 scenarios to manage the uncertainty on the environmental impacts computed according to LCA methodology. This approach provided two kinds of results: the comparison of environmental impacts caused by EU energy policies in each scenario and a certainty index, based on MC simulations, representing the confidence in the result of EU energy policy comparison.

3. Results and discussion

It was previously found that the bioenergy scenario causes fewer impacts on human health, climate and natural resources than the baseline scenario, but more impacts on the ecosystems [5]. These results are reproduced in the N scenario (reference scenario) of Fig 1 which presents results of the uncertainty analysis for the natural resources damage category.

Uncertainty analysis reveals that, with the exception of a few scenarios characterized by low economic growth or low technological development, the difference in environmental impacts between the EU energy policies do not differ significantly from the reference scenario. Also, the identification of the policy that has the lesser impacts is more certain for climate change and natural resources (certainty index averages of 99.4% and 95.8%, respectively) than for human health and ecosystem (certainty index averages of 83.1% and 80.4 %). Assuming an equal probability of occurrence for each scenario, the results of the M-LCA comparison of the EU energy policies presented in [5] are statically valid in 77% of the cases (average on all damage categories). However, this result should be interpreted with caution because some sources of uncertainty were not taken into account in the current uncertainty analysis. Specifically, the linkage of GTAP and LCA databases, the endogenous uncertainty of the GTAP model and the uncertainty on IMPACT2002+ method were not included in this study due to lack of data regarding these uncertainty sources.
Figure 1: Comparison of EU energy policy environmental impacts and certainty index for the natural resources damage category.

Results also show that the MC analysis is not adequate to study uncertainty propagation in complex models like M-LCA. Indeed, due to the significant quantity of uncertain parameters, the time required by MC simulations makes this approach unfeasible unless extensive computational resources are available. Therefore, another approach more adapted to uncertainty management in large models, such as Gaussian quadrature [8] or Fourier transformations [9], would improve the management of uncertainty in M-LCA.

4. Conclusions

The comparison of international policies based on M-LCA can be perceived as quite robust for most of the impact categories. However, this conclusion should be treated with caution as not all sources of uncertainty have been taken into account. The review of uncertainty sources in M-LCA highlights the need for additional data, improvement of data quality, and development of uncertainty management methods adapted to computable GEM, impact assessment methods in LCA and prospective studies. Computational resources appear to be an important issue to conduct MC analysis in the context of GEM and LCA used sequentially. Therefore, another approach is needed to manage uncertainty in such complex models.

5. References


Acknowledgements - The authors would like to acknowledge the financial support of the industrial partners of the International Chair in Life Cycle Assessment and Dr Kakali Mukhopadhyay for his GTAP expertise.
1. Introduction

Life Cycle Assessment (LCA) is very dependent on the quality, relevance and reliability of the Life Cycle Inventory (LCI) data sets selected by the LCA-practitioner. When modeling a system, the representativeness of the LCI data set is complemented by the appropriateness of the data set in the context of the specific system. The appropriateness characterizes, in how far a data set in a system model represents the truly required process or product. According to the ILCD handbook [1], the use of not fully appropriate data is justifiable only if this is not relevantly changing the overall LCIA results compared to using fully representative data; otherwise the lower achieved representativeness shall be documented in the report. But when the required data is missing the practitioner has limited possibilities to evaluate any differences between the required data and the available data.

This paper aims at evaluating the appropriateness of different LCA-practitioner choices regarding the geographical, technological and time related representativeness in the modeling of a product life cycle. The examples are taken from two technologies from the same industry domain that behave differently and require different LCA methodologies.

First, different LCI data sets for spinning and weaving processes in the textile industry are inventoried. The precision, completeness, representativeness and methodological consistency of these LCI are briefly described. Then, the following data quality tools: gravity analysis and sensitivity analysis [2] are applied at the assigned LCI results. The gravity analysis reveals the main contributors to indicator scores. The sensitivity analysis measures the change in the indicator results for induced changes in LCI results. Practically, the LCI results for different geographical situations, for different technologies (time related or not) and for different scenarios (average world scenario and worst case scenario) are compared.

2. LCI data sets and their representativeness

LCI data sets for different technologies, different geographical locations and at different time periods are described. The variable parameters studied for each representativeness type are listed in table 1. Technological representativeness of manufacturing processes takes into account: the type of input, the reference flow characteristics, the machinery, the specific treatment protocol e.g. chemical auxiliaries and processing time, the outputs and the treatment of these outputs e.g. water emissions or wastes. For spinning technologies for instance different LCI data sets of spun yarns are compared.

<table>
<thead>
<tr>
<th>Type of representativeness</th>
<th>Variable parameters taken into account</th>
</tr>
</thead>
<tbody>
<tr>
<td>Technological representativeness</td>
<td>Input type e.g. cotton, wool</td>
</tr>
<tr>
<td></td>
<td>Machinery e.g. open end, carded, combed</td>
</tr>
<tr>
<td></td>
<td>Output type e.g. Thick or thin yarn</td>
</tr>
<tr>
<td>Geographical representativeness</td>
<td>country specific energy</td>
</tr>
<tr>
<td></td>
<td>country specific waste treatment</td>
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<tr>
<td></td>
<td>country specific water production</td>
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<tr>
<td></td>
<td>country specific water treatment</td>
</tr>
<tr>
<td>Time related representativeness</td>
<td>Technology progress in carded spinning</td>
</tr>
</tbody>
</table>

Table 1: Variable parameters in spinning for different types of representativeness

To ensure the geographical representativeness of the LCI data sets the following adjustments are required: energy production, water production, solid waste treatment and waste water treatment need to be country specific. For spinning: China, India, Bangladesh, Pakistan, Indonesia, Turkey, Brazil account for almost 90%...
of 2009 installed capacities, according to the International Textile Machinery shipments statistics [3]. So carded yarn spinning production is represented in each of these countries and compared with a global spinning data set.

Time related representativeness is closely related to technological representativeness as technology is developed and changes over time. To assess time related representativeness, the evolution of the process technology for the same product output is studied. Each LCI data set is made of several unit of process from different sources but in this paper only the main technology changes (machinery and process) over time are considered.

3. LCIA results

3.1. Geographical representativeness analysis

The significant environmental issues for the process vary with the geographical setting: the relevance of the use of water is thus highly variable between countries. Elementary flows act differently depending on where they are withdrawn, e.g. surface water or ground water, or on where they are emitted. Since few spatially differentiated impact assessment models are available, spatially differentiated flow indicators results are mainly analysed.

3.2. Technological representativeness analysis

Technical guidance is needed to represent the specific processes of the system and to ensure that the data has the sufficient technological accuracy. The gravity analysis identifies the type of input, the type of output and the machinery as the main contributors to spinning LCI data sets indicators scores. For weaving LCI data sets, the nature and quantity of sizes used is also a contributor to indicator scores. The respective changes in the indicator results for induced changes in technological parameters are also measured via a sensitivity analysis.

3.3. Time-related representativeness analysis

Within the same industrial sector, some technologies are developing faster than others, e.g. multiphase weaving speeds kept increasing in the past few years [4] whereas carded spinning technology changes are slower. To avoid poor time-related representativeness, the expiry date of a data set inventory should be defined according to the specific technology development speed.

4. Conclusions

According to the missing or available data, the different LCA practitioner modeling choices are presented: worst case scenario, best available scenario, most representative scenario and most appropriate scenario. The different LCIA results are compared in order to guide to the appropriate choice. The paper shows that the three types of representativeness are not equivalent and must be handled differently. Changes in geographical settings or in technological settings do not raise the same environmental issues of concern. So the geographical representativeness may be assessed on a specific set of environmental indicators different from those to assess technological representativeness. The development of new spatially differentiated impact assessment models would help assessing the geographical representativeness of an LCI data set. Time related, geographical and technological representativeness are process specific and a thorough investigation of the process industry and market is necessary to assess them.

5. References


Acknowledgement - The authors thank Bureau Veritas CODDE for providing the EIME LCA software.
1. Introduction

Uncertainties underlying life cycle assessment (LCA) results are often raised as a major obstacle to a broader use of LCA in decision making. Uncertainties arise at different stages of the life-cycle analysis and can be usually ascribed to three types: model, scenario and parameter uncertainties [1]. Thus uncertainty analysis has become a major topic of methodological development over the last years [2].

In the absence of any recommended method, several approaches are being undertaken by LCA practitioners to evaluate sensitivities and uncertainties of their results. These range from simple scenario analysis, whereby different alternatives are tested, to more advanced uncertainty propagation methods. Uncertainty propagation typically consists in first defining the uncertainties affecting model input data to secondly propagate them through the model and thus evaluate the uncertainty of the LCA results. Several methodologies can be used for defining and propagating uncertainties in LCA studies, according to the nature of available information. Two such methodologies are provided by probability and possibility theories. Probability theory is the most commonly used [3] and consists in defining uncertainties as single probability distributions and then propagating them through stochastic modelling (e.g. by Monte Carlo or Latin hypercube simulations) as shown e.g. by [4] or analytically (e.g. in [5]). For cases where available information does not justify the use of single probability distributions for representing uncertainty, possibility theory provides an alternative method which consist in defining uncertainties as nested intervals (or fuzzy sets e.g. [6]).

The objective of this paper is to illustrate the fundamental differences between these approaches using on the one hand classical stochastic modelling, fuzzy calculus, and finally a hybrid method which combines both approaches. The advantage of the latter is that the manner in which model input uncertainty is represented, can be more consistent with what is actually known regarding the input (available information).

2. Materials and methods

2.1. Presentation of the case study

The case study investigates the benefits of source-separating the organic fraction of household waste to send it to anaerobic digestion instead of incinerating it together with residual waste, in Danish conditions. The functional unit is the collection and treatment of 1 tonne of biowaste from Danish households in 2011. For the purpose of clarity, only the global warming potential impact is considered in this case study. Moreover, due to uncertainties common to the two scenarios, only the differences between the results of the two scenarios are presented.

Prior to the uncertainty propagation, a sensitivity analysis enables to reduce the number of parameters to 26. For each of these parameters, data was gathered through a literature review among different LCA databases and articles. This served to define the uncertainties underlying each parameter, using either a single probability distribution representation, or else a fuzzy-set representation.

2.2. Implementation of the stochastic modelling

Probability distributions were selected for each parameter based on the data available. Most of the distributions were chosen as log-normal as this distribution only allows positive values. Uncertainties were then propagated using a Monte Carlo analysis of 10 000 runs.

2.3. Implementation of the fuzzy calculation

For each parameter fuzzy sets were defined based on the same data. The fuzzy sets were defined using two nested intervals: an interval outside which values were considered not possible (the support of the fuzzy set)
and an interval of values considered most likely (the core of the fuzzy set). The uncertainty propagation was performed using the method of \( \alpha \)-cuts presented by [7].

### 2.4. Implementation of the hybrid method

In this third method, some parameters were represented by single probability distributions (because they could be justified by available data), while others were represented by fuzzy sets (because available data was incomplete and/or imprecise). The joint propagation of these different modes of uncertainty representation was performed using the IRS method (independent random set) proposed by [8]. This method performs a random sampling of both probability distributions (leading to point values) and fuzzy sets (leading to intervals). An optimization technique then serves to find the extrema (minimum and maximum) of the model output for these point values and intervals. Multiple iterations, as for the classical Monte Carlo method, provide a model output in the form of a family of probability distributions, the spread of which reflects the incomplete nature of input information.

### 3. Results and discussion

Comparison between the three propagation methods illustrates the very conservative nature of the purely fuzzy calculation, which encompasses a family of probability distributions, of which the result of the purely stochastic calculation is but one representative among others. Results of the hybrid calculation on the other hand are more precise than the fuzzy calculation (the spread of the probability family is less important) but of course less precise than the purely stochastic result which assumes that single probability distributions are perfectly known for all input parameters.

### 4. Conclusions

In real-world situations of LCAs, available data relative to model input parameters are typically of different natures: the data may be “rich” (e.g. many measurements) therefore justifying the use of single probability distributions, or it may be “poor” (e.g. expert judgment, literature data, etc.), which is more adequately represented by the nested intervals of fuzzy sets. In such a case, the proposed IRS method can serve to jointly propagate the different types of uncertainties in the LCA. Such an approach is deemed preferable to arbitrarily defining single probability distributions in presence of imprecise/incomplete information, thus conveying an illusion of precision that is not justified by available data.

### 5. References


The impact of processing natural resources on uncertainties in Life Cycle Assessment

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1. Introduction
Natural resources are often connected to variable compositions. Ore from mining operations seldom provide the same ore concentration of their production every day and the further processing has to deal with it. The same effect is visible in renewable material flows like wood, gras, and crops for example. By processing these materials the process has to be either adapted to the changing quality of input material or it is run on average level. An average process control level may have an inefficient effect on the process quality. Ore processing is further developed than the processing of renewable materials related to the higher economic value of most mining products. The financial profit of renewable material flows is definitely lower than the profit of mining ores. Therefore, the effort on process control on high level is mainly not as high as in mining operations.

During the last years we experience the pressure on raw materials [EU Commission, EPA] and its influence on nearly all production processes since we are dependent on energy supply (oil, gas, coal, renewable energy) and the material supply to production processes. This can result into conflicts since we need biomass for the generation of renewable energy (wood, biogas) and we need the biomass for food (corn, crops, grass).

In our research we are not able to solve the whole problem how to handle the materials between the conflict of energy generation and food processing. But we concentrate on the question: How to run a material and energy efficient production process with less environmental impact?

2. Materials and methods
Our study case is the production of pig feed in a commercial feed processing plant in Northern Germany. The annual production adds up to 240,000 tonnes of feed. The most part represents pig feed with 80 % of the production, followed by chicken feed (15 %) and 5 % bovine animal feed. For various customers the plant is producing 200 different feed recipes. All these recipes are produced batchwise on one single plant. The energy consumption of the year 2009 accounts to 5.8 Million kWh. 50 % of the whole energy consumption is related to the compacting process (pressing).

Our aim is to relate the energy consumption not only to the process steps but to the various recipes. By identifying the carbon footprint for each recipe we are going to optimize the energy input into the process without influencing the product quality. This is our highest challenge, since the quality of the end product cannot be changed due to the animals demand. For example industrial products (like automobile, aircraft, etc.) can be constructed by reducing the weight without losing quality properties. But in the food and feed processing industry reducing weight means that we put the animals or ourselves on diet (Comment: May be we should be on diet ourselves but it is not the goal of feed processors). Our research focus lies on the investigation of energy efficiency of the production process without changing the endproduct quality. This can be achieved by monitoring the process by online measuring systems and developing an expert system that is able to control the process by using fuzzy rules.

The definition of the fuzzy rules are dependent on the identified uncertainties within the process. During the handling of renewable materials uncertainties can be found in their composition due to the fact that we are handling natural resources with varying properties like their water content for example. The varying input quality effects the process control because of different conditions during the process. A higher water content in the input results in adding less fluids during the processing for example. Additional to water there are other parameters like proteine, ash and starch content, etc.

The uncertainties in the feed process can be divided into:

- uncertainties that can be influenced by the process control (adjusting the water content by adding water)
- and uncertainties that can not be influenced by the process control (chemical composition)
3. Results and discussion

LCAs are being performed for chosen pig feed recipes with a focus on the carbon footprint. Since the energy consumption during different recipes of pig feet, each recipe has different values. To cope with these values we are performing Monte Carlo Simulation to evaluate the impact of the changing values. Our interest focuses on the approach to handle the varying values as uncertain data and to identify missing data (data defects) in the process.

4. Conclusions

The development of the expert system of the production process (case study pig feet) is performed according to uncertainty management. Uncertainties are present at any time in the process and has therefore taken into account during the decision making process. The uncertainties are formed into fuzzy rules and are implemented as a new software tool that supports the process control. It will be possible to simulate the production process with varying parameters to evaluate the result of the simulation. This helps in assessing the process set up and in logistics question (what is the best cyclic order of the recipes to consume the least energy?). The project is running until July 2014.

5. References


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Introducing an Uncertainty Analysis Methodology, in an International Carbon Footprint Accounting for Decision Making, Renault Group case study

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Topic : E07 - Latest developments in uncertainty management - adding value to LCA studies
Keywords: Carbon Footprint; Uncertainty; International; Decision-making

1. Introduction
Climate change and energy dependence are two major challenges for the transport industry. Tackling these topics needs major decisions and therefore to integrate Greenhouse Gases (GHG) management in companies. The two first actions to go through are accounting over the complete life cycle and involving all the employees and key stakeholders in the progress.

An exhaustive carbon footprint inventory is undertaken with the help of a hybrid method that combines product and organisational approaches. Due to the breadth of the approach, it is of utmost importance to introduce an uncertainty evaluation in order to ensure the best decision-making.

2. Materials and methods
Since the case study is a car manufacturer, its knowledge is based on the product. The core of the hybrid accounting method is thus based on the product Global Warming Potential calculated [1] and reviewed according to ISO Life Cycle Assessment norms. Nevertheless, LCA often neglects employee mobility and behaviour in their everyday work. Therefore these items are added according to the GhG Protocol [2, 3].

Interpretation of the results is crucial and a specific uncertainty analysis method has been developed. Uncertainty is analyzed separately for Physical Input (PI) data and Emission Factors (EF). The pedigree matrix [4,5,6] takes into account reliability, completeness, and temporal, geographical and technological representativeness of data according to five different quality levels. Another topic is to increase accuracy by introducing a specific factor regarding the availability of Physical Input primary data versus secondary. Finally Emission Factor quality are specific of industrial activity. The uncertainty factors are used together to evaluate the global uncertainty.

3. Results and discussion
The share of each life cycle stage has been calculated and this allows the company to drive actions.

The main share of the carbon footprint (85%) calculation in the “good” level, while 10% are in a rather “middle” uncertainty and finally 5% are evaluated as “very good”. (Figure 1)

By using a sensitivity factor representing weight of each category and the total uncertainty calculated we evaluate the risk level for each parameter.

When taking a decision, the question will be to prioritise the action budget toward the most important topics. This uncertainty approach will enable deciders to choose in the best conditions. The risk factor can be considered in decisions and it helps us to evaluate and improve the inventory each year.

Uncertainties will also raise questions when companies make a carbon reduction commitment of less than 10%, which is the within the uncertainty.
All uncertainty factors are based on Renault expert judgment and should be challenged by external experts (e.g., Quality of background LCI database).

In this method the qualitative data quality assessment results are related to uncertainty ranges for different parameters. In the end total uncertainty is calculated based on uncertainty ranges.

4. Conclusions

Setting a Carbon Footprint accounting is a major step towards an Environmental Footprint management by introducing the life cycle perspective and involving the whole company.

Nevertheless such a wide perimeter implies a complex calculation protocol and the uncertainty calculation will have a huge importance to tackle the complexity of this management and implement the best decisions.

Finally, the authors stress that even if uncertainties exit, these shall not be a barrier for taking action.

5. References

Uncertainty and variability in the carbon footprint of U.S. coal-fired power production

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Introduction

In 2009 electricity generation was the single largest contributor to greenhouse gas emissions (GHGs) in the United States [1]. Life cycle assessments (LCA) of the emissions associated with electricity generation typically use an “average” or “typical” model to represent emissions sources from cradle to grave. However, regional and technological differences (variability) in product manufacture, use, and disposal may cause a life cycle impact (e.g. GHG emissions) to be lower or higher than that of an “average” life cycle [2, 3]. Others have shown that accounting for uncertainty in inventory data or life cycle impact assessment methods may change the conclusions of a study relative to one employing “typical” values of key parameters [e.g. 4].

The insights that researchers, practitioners and users gain when assessing the influence of uncertainty or variability on LCA results differ due to the distinct natures of these two sources of variation. Whereas uncertainty is defined as lack of knowledge and can potentially be reduced by additional research, variability is an inherent feature of the physical environment and cannot be reduced. However, by taking into account variability the LCA of GHG emissions, life cycle environmental performances of power plants may be ranked and preferable production methods or regions of operation may be identified. At present, no life cycle assessments have been conducted to characterize the carbon footprint of electricity production over a continent, separately accounting for both variability and uncertainty. The aim of this study is to quantify the uncertainty and variability in the carbon footprint of coal fired U.S. electricity production. We account for both spatial and technological variability as well as parameter and decision rule uncertainty via Monte Carlo simulation and the employment of large scale databases maintained by the U.S. EIA [5].

Methods

The functional unit for our study is 1 kWh of electricity produced in U.S. coal fired power plants. Three life cycle stages were regarded: mining (production), transport (from mine to plant) and burning of the coal in power plants (use). Provision and disposal of the necessary infrastructure, such as mines, railways and plants were also included. The inventory included emissions of CO₂ and CH₄. In this study, the IPCC AR4 Global Warming Potentials (GWP) were used to estimate the carbon footprint [6]. Data for the life cycle inventory and uncertainty ranges were taken from scientific literature and publically available information [e.g. 7, 8, 9].

Variability and Uncertainty

Two types of variability were taken into account in this study: spatial variability and technological variability. Spatial variability arises from regional differences in methane emissions among coal basins and differences in transport distances between mines and power plants. Technological variability refers to differences in emissions from underground and surface mining, differences in power plant efficiencies and differences between modes of transport of coal from mine to plant. Variability was quantified by conducting the analysis on the level of individual plants. This was possible by utilizing available information on plant fuel usage, and power generation. Variability of power plant efficiencies and coal transportation distances was estimated via the U.S. EIA 906, 920 and 923 databases.

Two types of uncertainty were also distinguished: decision rule uncertainty and parameter uncertainty. To account for decision rule uncertainty, results were calculated for the three global warming time horizons reported by the IPCC AR4: 20, 100, and 500 years. Uncertain parameters in this study include: emission factors for burning of coal and liquid fuels, and the amounts of infrastructure required for 1 kWh of electricity. To quantify the effect of parameter uncertainty on the LCA results, probabilistic modelling was applied. A separate Monte Carlo (MC) simulation was performed for every power plant including its individual “upstream” steps (mining and transport).
Results and discussion

Results of our investigation are presented in Figure 1, which includes the results of all MC simulations for all power plants. Most of the variation of emissions is due to variability, which is represented by the differences in locations of the boxes. Examples include variability in methane emissions by coal mine, and differences in the distances of transport from mine to plant. Among these factors, power plant efficiencies introduce the greatest variability. The distribution of emissions is broadened by uncertainty: in transportation distances and emissions associated with train transport. Our findings are consistent with results of previously published studies [7, 10], and reveal that variability is the primary reason for the breadth of coal GHG emissions.

Conclusions

- Both uncertainty and variability contributed to the spread of life cycle GHG emissions for coal power.
- Site to site efficiency differences of power plants are the primary cause of differences in coal life cycle emissions; efforts in increasing plant efficiency will make the greatest impact in reducing the carbon footprint of coal.

References

A protocol for approaching uncertainties in life-cycle inventories Monte Carlo analysis – a practical example using aquaculture feeds

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1. Introduction

While life-cycle inventory (LCI) values often are presented in absolute numbers, uncertainty and variability are common traits of both foreground and background data. Uncertainties result from limited temporal, spatial and technological coverage, as well as limited precision of measurements of both economic and environmental flows. In LCIs of food production systems, however, the influence of variability (ontological uncertainty) may be increasingly predominant as production is governed by natural fluctuations (e.g. yields) [1]. Outcomes of LCA results describing the same production system may therefore vary with up to an order of magnitude.

The ecoinvent 2.2 database, to a large extent, relies upon a NUSAP approach for estimating standard deviation (SD95). The pedigree adopted for the purpose originate from Weidema and Wesnaes [2] and categorises the origins of uncertainty into: reliability, completeness, temporal correlation, geographical correlation, further technical correlation and sample size. Variability is, however, not taken into account while this also can have large influences on agricultural processes.

Aquaculture is a rapid growing sector and aquaculture feeds rely on a wide range of raw materials, including wild fish, soy, rice, wheat, maize, rape seed, palm fruit bunches and microingredients. They are also a major driver for many environmental concerns related to aquaculture production and a limiting factor for further expansions within the sector [3].

2. Materials and methods

Aquaculture feeds are here used as a practical example to demonstrate a new approach to uncertainties in LCIs. The modelling includes foreground data collected from feed mills in Asia and literature sources, while background data derive from the ecoinvent v 2.2 database. Each process within the system boundary was assigned a mean, a standard deviation and a distribution to most of its economic and environmental flows using a standardised protocol (Figure 1). Each of these flows were based either on primary data, weighted averages derived from a meta-analysis or a numeral unit spread assessment pedigree (NUSAP).

The NUSAP pedigree suggested by Weidema & Wesnaes (1996) and later adopted by Frischknecht et al. [4] was adopted here also with the

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**Figure 1: Protocol for sourcing and applying inventory data**
addition of variability. The same pedigree was used for calculating the weighted means and standard deviations. The inventories were later simulated using a Monte Carlo analysis to generate confidence intervals for individual flows.

3. Results and discussion

Table 1 presents the inventory for Brazilian soybeans from ecoinvent v.2.2 in comparison with seven other reports and articles. The ecoinvent database relies on a five year average for yields but does not take inherent variability into account, excluding a standard deviation of ±10.6% for yields during these years [4]. The estimated uncertainty is, moreover, correlated to the year of publishing the process which underestimates temporal and technological correlation. It is also made evident that several available inventories have been cross-referenced between studies, resulting in outdated data in the most recent publications. Similar analyses were made for all major raw materials utilized and simulated for aquaculture feeds from Thailand, Vietnam, Bangladesh and China.

4. Conclusions

Knowing the uncertainty (including variability) of LCA data and related results is crucial to justify results and derived decision-making. Knowledge in this field is still fragmented although growing. In this presentation, we will illustrated a new approach to analyse and characterize the uncertainty quality of LCA data and results a practical brief case on aquaculture feeds. The new approach allows for critical analysis of data and provides a constructive strategy for highlighting areas of great uncertainty and identifying data gaps.

5. References


Acknowledgement - This work is part of the Sustaining Ethical Aquaculture Trade (SEAT) project, which is co-funded by the European Commission within the Seventh Framework Programme - Sustainable Development Global Change and Ecosystem (project no. 222889). www.seatglobal.eu.
1. Introduction

For decision support, results of LCA often have to be weighted to allow aggregation of impacts, and evaluation or ranking of alternatives. Impacts can be expressed in monetary terms to facilitate appropriate taxing or subsidies, and cost-benefit-analysis, etc. If impacts on human health and the environment are expressed in monetary terms they are referred to as environmental damage costs (EDC). The share which is not internalised into economic decisions is called external costs. The calculation of EDC can be done in a consistent and transparent way by following the so called impact pathway approach (IPA). The IPA enables estimating the impacts of a certain release of substances or other pressures into the environment. By valuation of these impacts based on measurements of the preferences of the affected population the welfare loss is expressed into (EDC). It is essential to take into account that the results of the assessment are time and space dependent, i.e. the damage caused by the release of one unit of substance depends on where and when the release occurs. Also the preferences of the affected people vary with location and time because they depend on socioeconomic variables such as income and risk perception.

Within the ExternE (External Costs of Energy) project-series [1] methods for estimating EDC have been developed, covering the damages of releases of air pollutants like NOx, SO2, CO, particulates, NMVOC, and other pressures such as heavy metals, greenhouse gases, POPs, radionuclides and noise.

The German Federal Environment Agency (UBA) published a convention of the method for calculation of environmental costs already in 2007 [2], [3]. Within the European integrated projects NEEDS, CASES and ExIOpol the methodology was improved in order to integrate the EDC into different modelling frameworks (macroeconomic, energy and LCA models). The online computer tool for calculation of EDC is EcoSenseWeb [4]. The methodology has been further improved in the EU Projects HEIMTSA and INTARESE. In 2011, within the project [5] for the German Federal Environment Agency cost factors have been re-calculated for different transport activities, heat & electricity supply in Germany including operation and up- and downstream processes outside Germany. The figures for EDC presented are based on the most recent existing studies and extensive own research in this field. They have been transformed into ‘ready to use’-data sets for economic and practical evaluation studies. The focus is on air pollutants, noise and greenhouse gases. The EDC are expressed as costs per unit of the respective environmental damage, and as costs per unit of output activity sector. EDC factors are calculated as averages for whole country.

2. The Approach to Calculate Environmental Damage Cost Factors

The IPA for air pollutants implemented in EcoSenseWeb starts with the emission of the pollutant into the environment at the location of the source, models its dispersion and chemical transformation in the different environmental media and identifies the exposure of the receptors within Europe and the remaining Northern Hemisphere. Based on concentration response functions for particulate matter (PM) and ozone, derived in epidemiological studies, the related impacts to human health are calculated for emissions of the precursors of nitrates, sulphates and ozone, namely SO2, NOx, NMVOC, NH3 and for primary PM. Impacts to crops, building materials and biodiversity are also calculated based on concentration or depositions of air pollutants and corresponding response functions. Based on questionnaires (e.g. contingent valuation) and other methods the willingness to pay (WTP) of individuals to prevent an impact has been elicited. By averaging the results the WTP of the society is derived. Hence, monetary values for the weighting of different morbidity and mortality endpoints are available. The methodology has been extended by estimating and valuing potential biodiversity loss (due to acidification, eutrophication and land use change).

Based on this approach marginal damage cost estimates per unit of emission are calculated for different source characteristics, such as urban and rural emission sources, and different heights of release, etc.

With regard to greenhouse gas marginal cost estimates for different decades (up to 2100) and for various assumptions regarding discounting and equity weighting have been derived with the FUND 3.0 model [6]. Based on these damage costs and by comparison with meta-analysis of avoidance costs studies a value of
77 Euro\textsubscript{2010} per Tonne of CO\textsubscript{2}equiv. has here been selected for central estimates of EDC per functional unit (FU) of activities.

The emission per FU, i.e. kWh heat, kWh electricity, person-, vehicle- and tonne-kilometres are derived from several references for emission factors, e.g., [7], [8], [9]. Finally, the emissions per FU are evaluated with EDC per unit of emission of different pollutants in order to calculate EDC per FU of activity.

3. Excerpt of Results

A large amount of results is available for differentiation between energy carriers and technologies for heat and electricity. Moreover, with regard to transport different fuels, modes and different technologies have been assessed. For example, the EDC per tonne of goods on road transport varies between 1 and 25 €-Cent depending on the size, the fuel, the location and whether it is a EURO-0 or EURO-V standard. A link to the results will be available at the [www.ExternE.info](http://www.ExternE.info) page. Exemplarily, here in this publication in Table 1 the EDC factors for emission due to road transport processes and large combustion power plants are depicted. In Figure 1 the German fleet average environmental damage costs per vehicle kilometre for road transport (urban rural and autobahn (AB)) in 2010 is shown. The results are differentiated into greenhouse gases (GHG), impacts to human health and the environment due to non-GHG air pollutants (non-GHG), impacts due to noise (Noise), impacts to human health due to particulates from abrasion and re-suspension (Abrasion) and impacts to nature and landscape due to separation and soil sealing by roads.

<table>
<thead>
<tr>
<th>Euro\textsubscript{2010} / t Emission</th>
<th>Transport</th>
<th>Large Combustion Power Plant</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Urban (large city)</td>
<td>Urban (small city)</td>
</tr>
<tr>
<td>PM2.5</td>
<td>390,900</td>
<td>360,100</td>
</tr>
<tr>
<td>NOX</td>
<td>21,300</td>
<td>21,300</td>
</tr>
<tr>
<td>SO2</td>
<td>18,500</td>
<td>18,500</td>
</tr>
<tr>
<td>NMVOC</td>
<td>3,000</td>
<td>3,000</td>
</tr>
</tbody>
</table>

Table 1: EDC for emissions due to transport processes and large combustion power plants

![Figure 1: Average Environmental Damage Costs per Vehicle Kilometre for Road Transport in 2010](image-url)
4. Conclusions

The updated and extended impact pathway approach provides results which are more reliable and conclusive because the IPA combines site specific assessment, as far as possible, but also includes average values for different source characteristics. The approach serves as a basis for environmental assessment with a practical evaluation for policy assessment. The results for electricity and heat generation and different means of transport enable a discussion on relevance of different impact categories. Moreover, cost-benefit-analysis is possible if internal costs of the technologies are known. The uncertainties of the approach are quantified and described. The user can check the robustness of his results when comparing alternatives. The uncertainties due to spatial variability inherent in many LCIA methodologies are transparent as country specific values are further differentiated into sub-regions, urban and rural environment, and different heights of releases. The results can be used to support policy decisions and to test more simplified approaches in the future.

Next steps are: evaluation of emissions and impacts taking place in other regions, i.e. coverage of the whole world, including adjustment of CRF and WTP, assessment of exposure (including indoor exposure), extension of the list of substances, Monte Carlo simulation and other tools for uncertainty assessment regarding the robustness of the results.

5. References

1. Introduction

In potable water production, the question of finding a fair trade-off between the water quality increase, from raw water to at least the potable water standard, and the total effort to be spent to reach that objective is very common despite not trivial. Production managers and decision makers have to decide whether it is better to treat a low quality water, with consequent higher effort in terms of economic resources and lifecycle environmental generated, or to treat a high quality raw water, putting lower efforts. The same reasoning does apply to the comparison of existing plants, which could treat similar raw water, still leading to different potable water quality, and with different efforts. Few comparative LCAs of potable water plants have been published, mostly disregarding the water quality increase in the functional unit and therefore obtained erroneous conclusions. Water quality gain can be evaluated by using water quality indicators, further aggregated into a single score through a statistical approach. Regarding the total effort spent to reach the gain, monetisation of LCIA results is indeed a very effective and practical approach to obtain a single numeraire, which can be further combined to the total operational costs of the water treatment plants to be fully understandable by investors and decision makers. In the framework of an international research project, we developed a novel performance index (PI), combining the two scores, which allows to properly compare alternative potable water production plants. The aim of this presentation is to illustrate the methodological framework and an application to two existing plants (site 1 and site 2), to demonstrate the added value provided by monetisation, which allows a comparison where conventional LCIA methods have failed.

2. Materials and methods

An attributional LCA has been carried out on site 1 and site 2, which are existing plants managed by Suez Environnement in France, based on historical production data (see [1] for details). ReCiPe [2] was considered as reference single score LCIA result. Monetized LCIA results were obtained using Stepwise2006 [3] and Eco-costs2007 [4]. Stepwise applies at the damage level and is based on budget constraint for human health, willingness to pay for ecosystem quality and marginal costs for resources. Eco-costs are based on costs of prevention applied to midpoint LCIA categories. The monetisation approaches are therefore quite different. The total effort generated by the treatment plant (TE) is represented by the LCIA single score, augmented by the total operational costs of the plant in case of monetized LCIA.

Water quality is characterized by turbidity, suspended matter, total organic carbon, total coliform and total pesticide. Based on measurements before and after water treatment for each plant, a water quality gain per indicator is calculated. Then, the Quality Valuation System for Water (SEQ-Eau in French [5]), certified by the French legislation and used to control water quality of rivers but also of drinking water production, is used to normalize the indicator results (having different units) and aggregated them into a single score (WQ). Normalization is performed using quality curves obtained through observations and risk assessments for each indicator, converting the raw value of a measurement into a quality single score between 0 and 100. The performance index is then defined as the ratio WQ/TE and calculated per water quality indicator. Higher is PI, the better is the performance of the treatment plant. In order to assess whether the average of the PIs calculated per water quality indicator is different from zero (i.e. whether there is a net increase of PI for the studied treatment plant), a paired-t test is performed. A confidence interval is then calculated for the average of quality gains thanks to standard error and points of the t (Student) distribution at 95% of confidence. If a net increase of PI per plant is observed, the average PIs of the two plants (and related confidence intervals)
are then compared by means of an independent t-test, to check whether a net preference between site 1 and site 2 is discernable.

3. Results and discussion

In figure 1-left, the plant comparison using the PI calculated with ReCiPe is not conclusive, since the confidence interval includes zero. This means that the variability of water quality gain, weighted by the ReCiPe single score, is too high to discern a preference between the plants. In figure 1-right, the same comparison is done using monetized single scores, augmented by operational costs which, surprisingly, are very close in magnitude. In both the cases, a clear preference can be observed for Site 1, despite the different absolute results and monetizing perspectives. This is a very significant result, which provides a clear guidance in the comparison of alternative treatment plants and technologies, and is made possible by monetizing. The comparison of operational costs with monetized environmental impacts leads to the same order of magnitude, using Stepwise2006 or Eco-costs2007.

The question about the uncertainty of the LCIA results, which could affect the comparison, maybe arisen. We are currently investigating how to combine into the performance indicator PI the confidence intervals of monetized LCIA results from Monte-Carlo simulations with the confidence intervals on WQ obtained from paired t-test. The different sampling size (WQ is based on few samples whereas MonteCarlo simulation involves thousands of samples) prevents a straightforward integration.

4. Conclusions

A performance index combining water quality gain and environmental impact score is proposed in order to be able to compare alternative water treatment plants. The case study presented clearly demonstrates the added value provided by LCIA monetization methods. First, they provide transparent guidance on the choice of the best alternative (Site 1), whereas the sole ReCiPe single score evaluation is not conclusive. Also, monetization makes the comparison between operational costs and environmental costs possible. Further research objectives are the addition of infrastructures inventories for LCIA calculations and the implementation of LCIA uncertainties into the performance index, which are both under development in the framework of an international research project.

5. References

1. Introduction

Soils are one of Earth's essential natural resources, supporting nearly all terrestrial life. In life cycle impact assessment (LCIA), midpoint potential impacts due to land use are calculated as the product of surface occupied (or transformed), occupation (or transformation) time and a parameter describing the land quality loss (ΔQ). This latter is so far solely related to terrestrial biodiversity (PDF.m².year) and it is certainly not representative of all impacts caused by human interventions, as described by the European Commission [1].

Recently, an improved land use impact assessment method has been developed by the LULCIA project of the UNEP/SETAC Life Cycle Initiative [2]. This method relates land use to six new indicators in addition to biodiversity: biotic production (BPP), erosion regulation (ERP), fresh water regulation (FWRP), mechanical and physicochemical water purification (MWPP and PCWPP) and carbon sequestration (CSP) potentials, which represent provision and regulation ecosystem services, as defined in the Millennium Assessment [3].

Ecosystem services is considered a relative new concept, which makes the link between ecological functions and their use by humans: they become services. Bridging environmental, economic sciences and decision-making policies, they have become a great area of research, particularly in ecological economics [4].

On one hand, the LCIA methodology becomes more comprehensive in regards to relevant pathways linked to land use, but on the other hand, this development can potentially reduce the capacity of LCA to be used as a decision support system, as it can increase up to seven the number of land use indicators required.

This project aims therefore to develop a method (conversion factors in $/indicator units) using economic valuation as a common thread to aggregate the new midpoint indicators to a single area of protection representing ecosystem services loss.

2. Methodology

2.1. Economic valuation: conversion from functional indicators to the area of protection

The values of BPP and CSP are respectively estimated with productivity loss (using FAO data) and carbon tax values. The other regulation services values are estimated through current compensation costs, as they are considered essential and to be replaced immediately (conservative approach).

Mechanical and physicochemical water purification potentials correspond to the natural equivalent of primary and secondary & tertiary water treatment, respectively. Current world water qualities from Boulay [5] are extrapolated without this natural filtration to identify the potential compensation technology required. The corresponding costs are calculated with the Water Treatment Estimation Routine (WaTER) from US-EPA [6].

The economic values of FWRP are estimated using urban water supply prices and the values of ERP are based on the World Overview of Conservative Approaches and Technologies database [7].

2.2. Spatial considerations and final characterization factor (CFs) calculation to endpoint

a. Since LCA assesses spatially-global product systems, particularly in land use with the indirect land use change issue, midpoint CFs were developed within the LULCIA at the world scale. Because land use is a local impact category by nature, spatial variability prevails and this is why they are also regionalized. In the same way, site-specific economic values were used for the conversion factors and the local availability of the compensation systems is taken into account when possible.

b. Each of the six midpoints was developed for different land type uses (or land covers, representing human activities). A harmonisation was made to create common land covers. Finally, all the converted midpoints indicators (in 2.1) of the same land cover were summed into a single indicator representing the potential damage costs due to the degradation of land ecosystem services caused by the considered cover.
2.3. Case study: bio-based polymers

CFs to endpoint were applied on the comparison of different production locations of bio-based polymers.

3. Results and discussion

3.1. Conversion and characterization factors

Conversion factors (Figure 1 for the BPP) were calculated according to available data, mostly at the country level. Economic valuation assumptions will be discussed. The choice of compensation systems depends indeed of the performances required (increasing with the severity of the impacts modelled in the midpoint level) and their real availability: a technology may not be available or affordable in a given country.

Multiplied with the midpoints, they generate CFs to endpoint for all the nine land covers developed in 2.2, at a global level. Biogeographic and economic boundaries were combined together to draw new frontiers for the ecosystem services. For instance, a map for an agricultural land use (or cover) (Figure 2).

![Figure 1: Map of the conversion factors for the BPP.](image1)

![Figure 2: Map of the BPP endpoint for an agricultural cover.](image2)

3.2. Aggregation of the CFs in a study case

The endpoints can also be aggregated into an integrated indicator, representing ecosystem services loss in $. Results show that potential impacts are specific to both the type of ecosystem service and the location. The midpoint level assesses the effects on ecosystem services, while the endpoint goes a step further, as it takes into account the local and actual need of the ecosystem services. Having an integrated indicator is useful in this case to rapidly compare potential locations to produce the biopolymers (Figure 3).

![Figure 3: Land use aggregated FC for different biopolymer production locations.](image3)

4. Conclusions

In conclusion, the development of the conversion factors allows to express all the midpoints proposed by the LULCIA approach to economic values, bringing a whole new level of interpretation as natural ecosystem services loss. Their economic valuation may potentially allow future LCAs to assess other impacts related to land use, such as aesthetics and recreational aspects.

Moreover, it allows the aggregation into a single endpoint indicator related to the ecosystem service loss area of protection, which also represents a good indicator for land use to work with at the decision-making level.

5. References

How to correct price for monetising non-renewable resource consumption?

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1. Introduction

Intertek RDC has developed a methodology based on life cycle thinking and monetisation to evaluate in a quantitative way the environmental, social and economic impacts of a product, service or policy. Monetising aims at reflecting the way human well-being is affected by an activity, with one euro meaning the additional welfare brought by one euro of additional income to a mean European (having a mean income).

This paper focuses on the impact category “Non-renewable resource consumption”. The cost for the society of consuming non-renewable resources is to be assessed. It corresponds to the well-being loss due to the increased resource scarcity in the future as a consequence of its consumption today. From the economic point of view, the term scarcity encompasses the stock finitude, the increasing difficulty of resource extraction and the substituability of the resource/function provided. In theory, the welfare loss can be assessed through the opportunity cost of the resource, defined as the welfare loss (the cost) of being deprived of an extra unit of resource for its most valuable use, i.e. the one providing most utility.

As for example summarised by Pickin (2008) [1], varied conceptions of scarcity can be found in the literature. The extent to which this opportunity cost is reflected in the current market price is discussed in this paper. In other words, the extent of damage internalisation is assessed.

The drivers for resource pricing in the actual markets versus perfect markets are first discussed. As a result of this analysis, a method for calculating monetisation factors for elementary flows related to resource consumption (mineral and fossil) is then proposed and discussed.

2. Analysis of actual versus perfect markets

2.1. Optimal price path

There exists an optimal price path that allows optimal resource allocation, i.e. maximising the inter-temporal welfare gained from consuming the resource stock at an optimal rate. In a perfect market (perfect information, perfect competition, all property rights distributed, etc.), the equilibrium market price equals the optimal price and the cost for the society of consuming a resource unit. Assuming a constant or null marginal extraction cost and the absence of a substitute technology, Hotelling’s rule [2] has established that the equilibrium price increases with time at a rate equal to the interest rate, following an exponential curve.

2.2. Actual versus perfect markets

In a more realistic market, the question is: “Can the actual market price be used as a proxy of the cost for the society to consume resources?” or “Are the price determinants likely to introduce bias in resource price fixing so as to drive it away from the optimal price?”

Pricing in actual market depends mainly on the anticipation by the agents of the following drivers:

- the future opportunity costs of the resource
- the substitution technology potential (or backstop technology)
- the marginal extraction cost
- the physical stock of the resource
- the discounting rate
- the taxes (depending on their natures)

For the four first drivers, we conclude that (i) mechanisms of pricing in the actual market can be supposed similar to those in a perfect market and (ii) the market agents possess together more information and are more entitled to integrate these elements into the price than a modeller. The discounting rate used in actual market pricing is higher than the social rate, as generally admitted (cf. Cruz 2006 [3]), while they would be the same in a perfect market. It results in a bias. The current market price is a lower bound of the cost for the society since the anticipated opportunity cost of the last unit consumed is discounted at a too higher rate.
(see chapter 3 for more details). Concerning the tax system, the world average tax on resources is to be counted as a driver for price fixing since demand and supply take these into account.

Factors like lack of competition, transaction costs, speculative bubbles, extraction from ore as co-product, recycling, quotas, etc. have been analysed for different metal resources. For most of them, no obvious market failure can be identified.

3. Developed monetisation method

According to the analysis described here above, we consider that market prices can be used to monetise non-renewable resource consumption if they are corrected for the excessive myopia of the market. The idea is to calculate the optimal price on the basis of the current market price and the assumed values of the market and social discounting rates and the number of years before the last unit is consumed. The correction is shown in per euro of current price. At this stage, the same correction factor is applied for all resources.

1) The current market price is obtained from a 5-year trend with appropriate accounting of the associated taxes.

2) The opportunity cost of the last unit consumed (anticipated by the market) is determined assuming that the current market price increases to this value according to the Hotelling’s rule with the market discounting rate (assumed to be 8%) and a 20 years period for correction. This value is 4.7 €/$.

3) After, the social (optimal) discounting rate (assumed to be 3%) is used to calculate the optimal price (or societal cost) by discounting the calculated opportunity cost.

4) The calculated correction factor is hence 2.6.

A common time horizon is fixed for all resources, here 20 years. It is assumed that very few resources will be exhausted by 20 years and that over 20 years, market and social discount rates are equal. Sensitivity analyses have been carried out for taking uncertainties of the various data into account.

The monetisation factor is obtained for each resource by multiplying the trend market price by the factor 2.6. If only the externality is to be accounted for, the multiplying factor becomes 1.6. The factor is applied for both mineral and fossil resources, with energetic resources being treated as oil using the HHV as equivalency factor.

4. Discussion

Limits of the use of the calculated correction factor are

- The factor derives from a partial equilibrium assumption, i.e. demand adjustments resulting from potential internalisation of the calculated externality is not considered, though it would affect namely the time before depletion.
- It is applied on the whole price, while in theory, it should only be applied on the part of the price related to the resource scarcity, i.e. excluding the extraction cost.
- It is common to all resources while resources can be classified into various types of resources with different types of reserves, demand and geopolitics

In practice, when assessing for example various end-of-life scenarios for electric and electronic waste, the total monetised impacts obtained for all impact categories are often dominated by the contribution due to resource depletion (mineral and fossil).

5. References

1. Introduction

Nestlé’s commitment to innovative eco-design for its products and activities is asserted in its corporate policy on environmental sustainability. PIQET, a packaging ecodesign tool, has been used since 2007 to assess the environmental performance of packaging systems. Its low cost and capacity to generate results fast allow for a systematic application throughout all packaging innovation & renovation projects at Nestlé, generating knowledge and promoting innovation in the design of packaging systems with reduced environmental impacts.

Acknowledging that packaging is generally only a minor contributor to environmental impacts of food products, and building on the success factors of an ecodesign tool such as PIQET, Nestlé has now developed a product ecodesign tool that takes into account the entire life cycle of the product, including the agricultural production of its ingredients, processing, packaging, distribution, consumer use and end of life.

The approach taken by Nestlé in the development of the product ecodesign tool is discussed here.

2. Potential barriers identified

The development of a product ecodesign tool for a company as large as Nestlé and with such a varied portfolio of products can encounter some barriers:

- Recipes for product formulation and manufacturing are complex, not only due to the number and variety of ingredients, but also due to the processing steps involved. Therefore, the ecodesign tool has been integrated into the systems currently used by the company for the management of recipes and trials of new products. Thus, recipes containing the ingredient specifications, quantities used and processing steps are automatically uploaded to the ecodesign tool, considerably reducing the manual input from the users (Figure 1).

![Figure 1: Integration of databases in the product ecodesign tool.](image-url)
• Food products are fast moving consumer goods and have short innovation cycles. Thousands of different products exist in the company portfolio and many product renovations occur at a given time. Therefore, life cycle assessment and ecodesign cannot be performed by central functions but rather need to be delegated to project managers. For this to happen, the ecodesign tool features a simplified interface targeted to a non-LCA specialist user base, the goal and scope have already been pre-defined, and the internal database comprises a list of relevant LCI profiles and pre-defined LCIA methodologies.

• The range of food products manufactured by Nestlé on a global scale requires the use of thousands of raw materials and ingredients, for which, to date, reliable and transparent LCI data are insufficient or non-existing. LCI profiles sourced from the Ecoinvent database for a range of key ingredients have been incorporated into the tool. For other ingredients, Nestlé is collaborating with database developers, suppliers and other food companies in the preparation of a more comprehensive and consistent public food LCA database.

3. Success factors

3.1. Integration with current work practices at Nestlé

From a user perspective, the familiarity with the operation of the recipe management system and working environment reduces the training requirements and ensures a swift adoption of the new tool.

The product ecodesign tool is embedded in the stage-gate product innovation process at Nestlé. This approach allows a systematic improvement of the environmental performance of all products and tracking the outcomes of the ecodesign assessment throughout the innovation process.

3.2. Peer-review of the product ecodesign tool

The results obtained through the use of the product ecodesign tool will support internal decision making. Nevertheless, for external communication of results and claims, ISO 14040 and 14044 standards require the preparation of peer-reviewed LCA studies.

As an integral part of the development of the tool, the methodological choices incorporated into the product ecodesign tool have already been subjected to an independent certification and peer-review process according to the ISO 14040 & 14044 standards. The limited scope of this certification (tool methodology, LCI & LCIA data) is clearly documented. Then, the certification of further ecodesign studies based on the tool will focus on the aspects that have not yet been certified (e.g., appropriate input of data by the user, assumptions & simplifications). This is how the ecodesign tool greatly simplifies the process of communicating LCA results to the public.

4. Further work

The credibility and consistency of the product ecodesign tool depends on the use of transparent, independently generated LCI data, as well as on the alignment with independently established, internationally agreed LCA and ecodesign methodologies (e.g. the EU Food Sustainable Consumption & Production Roundtable). Nestlé will incorporate developments in these areas to the tool as they become available. Moreover, once consensus is reached in how to assess key environmental impacts such as land and water use, these methodologies and indicators will be added to the range already evaluated by the tool.
Mainstreaming Life Cycle Management: using a sector based and regional approach in Northern France in the textile, seafood, packaging and mechanical sectors

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1. Introduction

Life Cycle Management, or LCM, is a business management approach that can be used by all types of business to improve their sustainability performance [1]. Most major global companies now have internal programmes to analyse and manage the sustainability performance of goods and services under across the life cycle. At least 80 private companies, such as Veolia, Arcelor Mittal, Unilever and Walmart are financing important collaborative Life Cycle research through the CIRAIG, The Sustainability Consortium and the UNEP/SETAC Life Cycle Initiative. LCM is also impacting public policy related to resources use and recycling, and more recently product-related environmental policies. For example the European Commission stressed the importance in 2003 of developing policies that look at the whole of a product’s lifecycle, including the use phase to ensure that environmental impacts throughout the life-cycle are addressed in an integrated way [2].

To make significant progress towards sustainable production and consumption patterns, LCM needs to be taken up by whole supply chains that, by definition, include many small and medium enterprises (SME). From a business perspective, this represents a competitiveness issue, as these SMEs are increasingly under pressure from clients and legislators to provide more information about the impacts of their products, and take responsibility for them both up and down the value chain. For example, the national testing of France’s LCA based environmental labelling scheme [3] is already impacting hundreds of suppliers.

Unfortunately, the majority of SMEs lack the financial capacity or human resources to implement LC tools on their own, and are wary of working with support organisations outside of their sector or local area.

In Northern France, professional support organisations, including clusters, business federations and Chambers of Commerce, have come together under the auspices of the [avniR] LCA Platform to explore ways to help their businesses adopt LCM. The strategy for achieving this is based on some key concepts:

- **LCM capacity should be built into existing support organisations**, so that businesses receive advice and tools from organisations that they already know and trust

- **Tools and actions to support LCM integration need to be adapted to different sectors** to make them as relevant as possible to SMEs

- **Training and research capacity also needs to be developed** to respond to needs of businesses

After 2 years of applying these concepts in an opportunistic fashion, the textile, seafood, packaging and mechanical sectors are developing strategic action plans to mainstream LCM into their businesses, education and research organisations.

2. Approach

The LCM action plan development process is managed by a professional organisation for each sector: MAUD competitiveness cluster for the packaging sector, CETIM for the mechanical sector, Aquimer competitiveness cluster for the seafood sector, and UP-tex competitiveness cluster for the textile sector.

All four sectors follow the same general process for developing their LCM Action Plan, as shown in Figure 1 below. The benchmark identifies life cycle based initiatives and tools relevant to each sector, focussing on, but not limited to North West Europe. The maturity of businesses, education bodies and research centres in
the region with in relation to LCM practices is undertaken via interviews with key stakeholders. Stakeholder engagement is a key aspect of the needs identification and action plan development phases, not only to ensure that the proposed LCM action plan “fits” the needs of the sector, but to create ownership for the implementation phase.

The first four LCM Action Plans are developed in parallel from November 2011 to June 2012, enabling the sectors to identify cross cutting actions. This process is managed by the regional LCA Platform [avniR], through a network of “Life Cycle Champions”. Champions have been trained in the eight sectors; and meet regularly to exchange experiences in implementing life cycle approaches and identify cross cutting projects. The four other sectors will undertake the strategic action planning process in a second “wave”, to incorporate learnings from the first process, as shown in Figure 2.

3. Conclusions

This innovative approach to mainstreaming LCM leverages sectorial and regional networks to help overcome barriers to implementation. From a business perspective, integration with existing professional organisations means that SMEs access advice and tools through organisations that they already know and trust. Working with several sectors in parallel through the Life Cycle Champion network encourages a multidisciplinary approach, essential to improving decision making across entire supply chains.

4. References


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Driving proactively the sustainability agenda for the European detergents and maintenance products industry, via the A.I.S.E. Charter for Sustainable Cleaning

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1. Introduction
A.I.S.E., the International Association for Soaps, Detergents and Maintenance Products, has a long tradition of proactive work towards sustainable production and consumption. Its main overall scheme is the A.I.S.E. Charter for Sustainable Cleaning, encouraging the adoption of sustainability management practices at all stages of the product life cycle; launched in 2004 in all EU countries plus Norway, Iceland and Switzerland, the Charter covers all products categories of the detergent, cleaning and maintenance products’ industry.

Participating companies report regularly progress on Key Performance Indicators, leading to the publication of an annual sustainability report.

As at November 2011, commitment to the Charter is a success with 160 companies committed, covering approximately 85% of the total industry’s production output;

An update of the Charter has been rolled out in summer 2010, introducing a product dimension to the existing Charter requirements. This fits very well with the overall objectives of the European Commission SCP/SIP Action Plan, and particularly the Ecodesign Directive.

2. Materials and methods
The Charter stipulates of a set of twelve Charter Sustainability Procedures (CSPs) which companies implement in their management systems. These are the threshold requirements for signing up to the Charter and are verified by an independent external verifier when entering the scheme, and then, every three years.

Once the criteria are met and verified, companies are entitled to use the appropriate Charter logo, which certifies that the company is following sustainability principles, giving priority to improvements in people’s safety, environmental friendliness, and to other key aspects of sustainability, without compromising product performance. Companies then have to report annually on 11 KPIs linked to the CSPs, which are externally aggregated and published annually by A.I.S.E.

From the outset, regular upgrades of the Charter were envisaged to ensure that it continues to offer the most advanced sustainability assurance scheme for promoting best practice within the industry, using LCA and science as a basis.

The first major upgrade, Charter Update 2010, is now launched and introduces a product dimension, enabling companies to give a sustainability assurance for individual products, with a specific logo. These can be obtained by companies committed to the Charter Update 2010 through voluntary compliance with the new Advanced Sustainability Profiles (ASPs) for product categories. Those describe the product group characteristics which the industry considers represent a good sustainability profile that is ambitious but reasonably achievable by all. The parameters are defined based on a life cycle analysis. The system works by the company self-certifying its product against tables of values covering environmental safety and certain key drivers of sustainability. Random external verification is also organized in order to secure some control of the system.

The methodology used to assess which parameters are relevant per product category relies on Life Cycle Analysis. It is on that basis that A.I.S.E. and its experts gather scientific evidence and market knowledge, and ultimately, propose the relevant ASP parameters.

Each product category will have a set of threshold values and other potential requirements focused on the three or four most important parameters affecting the sustainability profile of the category according to industry expert life cycle analysis.
3. Results and discussion

3.1. Life cycle analysis as basis for a comprehensive sustainability scheme

This section will focus on the rationale led by A.I.S.E. for working on a life cycle approach to address better sustainability profile for the sector via the Charter. It will also highlight the successes achieved (KPI deliveries, market committed), as well as political relevance of the project (especially in the EU context).

3.2. Life cycle analysis as a practical way to drive product advanced standard

Taking laundry detergent as an example, this section will focus on the work done by the A.I.S.E. experts to define the relevant criteria and thresholds, such as dosage, packaging, low temperature washing. It will also detail how the industry has addressed the ecotoxicity profile of such products (to be aligned ultimately with the REACH legislation), through the development of the Environmental Safety Check tool. This is designed to check whether all ingredients are below the predicted no-effect concentration for the aquatic environment at the concentrations they are being used in detergents. The values are based on published scientific toxicity data alongside very conservative product tonnage and other benchmarks.

4. Conclusions

Experiences of progress, best practice sharing, stakeholder feedback, developments of sustainability criteria for other product categories will be shared. Through the Charter for Sustainable cleaning, A.I.S.E aims at providing the most advanced sustainability scheme to drive progress in the detergents and maintenance sector, ahead of legislation and in close stakeholder feedback.

5. References

1. Introduction
The automotive industry is facing major challenges and the stress is strong in order to reduce production cost and usage value within the capacity of the nature to support the mobility growth.

This goal will be achievable only when a co-innovative product design will be set up, involving the whole value chain [1]. This is an ambitious target and this paper will show how life cycle assessment evaluation tool could be a major contributor to bring the various parties altogether.

2. Materials and methods
When conducting a life cycle assessment, five main steps shall be fulfilled: Goal definition, Scope definition, Inventory Analysis, Impact Assessment and finally Interpretation. [2]

A single practitioner can of course complete all these actions alone, nevertheless, it can be a tremendous opportunity to open the dialogue with several primary or secondary stakeholders as define in the UNEP/SETAC LCM Guide [3]. Several examples are identified, from the more obvious which is the critical review during which the practitioner will open his work to Research Institutes / Universities or Environmental Non-governmental Organizations.

Fig1: Evaluation of the potential for LCA collaboration

For the Electric Vehicle LCA, Renault worked on the Functional Unit definition from a customer point of view, also with Nissan and key battery producer on the inventory, with research teams (Poitier University and EcoSD network [4]) on impact assessment and finally with a panel of experts and NGOs during the critical review process.
3. Results and discussion

One result example is the definition of the functional unit for electric vehicles LCA [5]. The main difficulty [6] is to define a function equivalent between thermal and electric vehicles. The ILCD handbook [2] recommends a functional unit define by four items: What, How much, How long and in What way. A literature survey shows that no vehicle LCA study fulfills these recommendations. Nevertheless statement are homogenous regarding the three first items: transport of passenger for a certain distance (150 000 kilometers) and number of years (10). So far the function are totally equivalent and this FU could fit to the electric vehicle as well. Therefore the reason for any doubt take place in the last item, in what way. This question was studied under two approach, one “product centric” based on functional analysis and one “people centric” based on customer (emotional) needs. These needs were studied through market understanding tools and can be grouped in four emotional needs: reassurance, aesthetic & personalisation, smart & fun, environmental friendly. Then we can enlight the main differences between the thermal vehicle strengths - reassurance and aesthetic - versus electric vehicles ones – environmental friendly and fun-. Therefore, equivalent functional units shall add a reassurance dimension to the electric vehicle functional unit such as an easy access to a “long autonomy range” vehicle when needed.

Another progress was realised regarding the collaboration between an industrial company and Environmental Non-governmental Organizations during the critical review of electric vehicles. The choice was made to create a mixed panel composed by three LCA expert and two NGOs. This collaborative work proved that LCA is a major opportunity to exchange points of views and debates, bring truth or collective position on beliefs or share what are the major LCA topics to tackle. Nevertheless, managing such a mixed panel is not easy since the various actors are seeking different goals. This conduct to a recommendation to create different status and role within the critical review panel: Practitioner, Expert reviewer, NGO observer and Mediator.

4. Conclusions

For a company, it is nowadays a basic necessity to carry the life cycle assessment of its products to ensure a comprehensive environmental strategy, a transparent communication, and identify progress opportunities.

This experience on the electric vehicle shows also that LCA can be a very useful tool to bring the stakeholders (unusual ones as NGOs as well) around the table with a positive and constructive scientific approach.

Life Cycle Assessment is defenitively a powerful tool to involve the companies’ stakeholders, by opening a dialogue based on science and facts, in final creating trust among the value chain.

5. References

LCT in the floor-covering industry: the strategy of Tarkett.

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1. Introduction

Tarkett is world-wide leader in innovative flooring and sports surface solutions, providing integrated solutions to professionals and end-users. The company is concerned since a long time about environmental issues, despite not-harmonized actions. Indeed, production sites are certified according to ISO14001, and Tarkett contributes to the establishment of environmental standards for floor-covering products at the European level. Some specific projects have also been launched, aiming to develop recycling of post-used products, to reduce VOC product emissions and to develop vinyl products with alternative plasticizers. Tarkett regularly publishes five environmental “Key Performance Indicators” regarding their global production sites: energy consumption, water consumption, waste generation, VOC emissions and recycling amount. This set of indicators remained the main tool reflecting environmental performances of Tarkett activities and products, and having an influence on decision making. Therefore Tarkett came to the question “Are we really doing the right things?” regarding environmental performances of products. From this question, the need for a more comprehensive sustainable development strategy to be integrated into the daily working practice was identified. The challenge was multifold: how to evaluate the environmental performance of all products families and identify improvement opportunities; to face the growing regulation constraints; to keep a coherent and unified approach among the different international branches of the company, and to generate an added value from this huge investment? To tackle this endeavor, Tarkett has chosen the support of the Public Research Centre Henri Tudor, Luxembourg.

2. Materials and methods

Tarkett built its own team dedicated to environmental strategy development in the R&D centre in Wiltz, Luxembourg. The team identified Life-Cycle Thinking (LCT) as a central approach, since the products are at the core of the environmental strategy system. Production at Tarkett deals with a wide range of market segments, 30 worldwide production sites, manufacturing eight families of products (vinyl and linoleum products for 60% of sales, as well as wood, rubber, laminate based products or sport surfaces), which therefore induce a wide range of raw materials. In order to implement a LCT perspective, a multi-criteria approach is required to address the recyclability of materials and the relevant environmental impacts from the whole life-cycle of products.

The implementation of LCA as a fact based practice for process and product environmental improvement, in a two-year term, has therefore become a priority, using the scheme provided in Figure 1. Although LCA is used as an eco-design driver and a strategic tool to position Tarkett as an industry leader regarding
environmental performances of products, the company is aware that this tool is not sufficient in its current level of development to address specific issues. Therefore Tarkett is engaged in complementary development studies and processes. In 2010 a research study was ongoing within the LCIA program of the UNEP/SETAC Life Cycle Initiative in order to develop a methodology for the assessment of impacts from chemicals on indoor air quality during flooring products use phase. Since beginning of 2011, Tarkett is also engaged in a "Cradle to Cradle" approach. The main product families are eco-designed by assessing materials toxicology, optimising energy sources during production processes and redesigning the products for an optimized recycling at the end-of-use.

3. Results and discussion
In order to progressively integrate LCA as a decision-making tool, Tarkett first acquired the necessary knowledge on LCA methodology and LCA software, through training sessions in Europe and the US. Then, systematic LCAs of product families have been realized. Dedicated IT software is also being developed to facilitate the access to LCA models and results within commercial LCA software. The tool allows Tarkett professionals (non LCA experts from R&D and Production teams) to run eodesign simulations. The approach is kept coherent and unified for all Tarkett’s branches through an overall strategy (Figure 2).

4. Conclusions
The positive feedback from this test integration phase convinced Tarkett on the ability and reliability of LCT to drive, in a complete and accessible way, the implementation of environmental strategy at the decision level. It also allowed the identification of requirements and interests of each team-profile in order to understand the strategy and be involved as an actor in its development internally.

Further development phase is now launched in order to extend the first phase to the various product families and to other internal clients, such as purchase and marketing departments. The vision is to combine the complementary tools already used by Tarkett in the LCT process, in order to set a make robust decision making based on eco-innovation criteria. This presentation aims at presenting the Tarkett strategy, its current and future deployment in practice, illustrating the bottlenecks and the solutions foe effective implementation. The example of Tarkett could be seen as showcase of implementation of life cycle management in an industrial group worldwide.

Figure 2: Strategy deployment within Tarkett team – LCA case
Promoting use of life cycle management in Finnish companies – challenges, benefits and suggestions for future value networks

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1. Introduction

Life cycle thinking (LCT) and life cycle assessment (LCA) along with other life cycle (LC) methods are important tools in order to assess environmental impacts of products and services, and to support environmental decision making in companies. Use of those methods can also help companies to identify unsustainable strategic decisions, and therefore to find out business opportunities by taking precautionary actions. Even though the use of these methods is supported by many stakeholders, in practice, many challenges hinder the use of them to support companies’ environmental management.

The aim of this study is to present the main drivers and barriers for companies to apply LCT, LCA and other LC methods to support their strategy processes and decision making. We will also suggest a roadmap for the promotion of these methods in companies and their value networks. The focus of our study is on Finnish companies, but the results can be widely applied when use of life cycle approaches is promoted to support decision making on environmental sustainability.

2. Materials and methods

We first assessed the methodological challenges of the LCT, LCA and other LC methods from the company perspective. The applicability of those tools were assessed in several case studies that were related to new metal materials, biomaterials, construction sector, re-use of industrial by-products and paint industry. The assessments were based on the data from literature and companies. The LC methods tested included LCA, EE-IO, carbon footprint, ecological footprint and thermodynamical methods. In addition, several stakeholder workshops were organized to get the practical views on the use of life cycle methods. The current and future use of life cycle methods in companies were also studied using an internet questionnaire and a two-round thematic interview for selected company representatives.

3. Results and discussion

In Finland, as well as also globally, there is only little research on the application of LCT and LCA in companies in practice (Cooper & Fava, 2006; Teixeira & Pax 2011). The questionnaire and interviews within this study revealed that the companies can coarsely be categorised in four main company types regarding their knowledge, resources and potential to utilize life cycle methods in practice. The findings were supported by the case studies. Three company types were named as "interested outsiders", "learners" and "forerunnes". In addition, there is a large group of companies that are not interested or even aware of the use of life cycle methods and their potential benefits. The questionnaire revealed that the most popular tool to measure environmental performance of products is use of specific waste or emission indicators, but surprisingly many companies also tell that they are performing LCA or carbon footprinting (CF) studies (Fig 1a). These results, however, are most likely giving a distorted picture, because most of the respondents were advanced in LC thinking. Evidently, the demand for life cycle based product oriented environmental management is expected to grow in future (Fig 1b).

The main drivers for the environmental management of companies included legislation requirements, improvement of cost-efficiency, customer requirements, raw material price, brand issues, megatrends, business opportunities and environmental reporting. These reasons are also the main benefits for the companies to support sustainable life cycle management. Despite the fact that a group of forerunner companies are using life cycle methods on routinely basis and even participating in theoretical and practical development of methods and tools, a large number of companies are not familiar with life cycle methods and their potential benefits. Additionally, many companies, the group consisting especially of small and medium-sized enterprises (SMEs), often have no temporal, human and economic resources to learn and educate themselves on the benefits of using LCT and LCA. Another challenge is that in many cases, the companies consider their role in the value chain to be minor and therefore to have a very limited impact of the product.
4. Conclusions

LCT, LCA and other LC methods can be used to support long term environmental management of companies, which means to identification and realization of environmental key aspects and bottlenecks for companies' strategies, and to support companies operations. In order to break life cycle management of products and services and sustainable consumption and production through the whole society, life cycle thinking and methods need to be taken into practice in product design and strategic decision making. Application of LCT and LC methods to assess wide systems also creates a basis for identification of industrial ecology type of synergies between companies and actors and thus to reduce the environmental impacts and improve energy and material efficiency of the systems. Even though LCT and LC methods are commonly used in the forerunner companies, a large part of the companies are lacking knowledge and resources to apply LCT and LC methods to environmental management. LCA is often too complicated and resource demanding process for companies and therefore simplified practices need to be introduced. However, the main challenge is probably related to the communication problem between the LC researchers and companies, and therefore much more practical approach need to be used when spreading LCT in wide scale. The next step in promoting the use of LCT and LC methods in decision making in practice will be a pilot project, in which all the actors within a region, value-chain or sector will be introduced to work with each others in to a networking process, including special type of training for SMEs.

5. References


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More information: www.ymparisto.fi/syke/finlca
1. Introduction
The UNEP-SETAC Life Cycle Initiative is a joint venture formed in 2002, among several purposes, to foster the development, validation, and dissemination of methods, tools and practices for lifecycle assessment and management. Over its ten year history the Initiative has conducted dozens of projects which have contributed to the efficiency, effectiveness, and rigor of life cycle approaches on a global scale. This presentation will address those contributions from the viewpoint of meeting critical needs for maintaining and enhancing the quality of the science underpinning life cycle approaches. It will also place those developments in the context of emerging research and practice from academic, government and business organizations, providing a perspective on the significance of the Initiative's contributions to the science of LCA.

2. Methods
Technical work products of the Initiative generally fall into three broad categories - consolidation and cohesion of life cycle inventory and impact methods (particularly with a view towards how those may support social and sustainability assessments), development of practice guidance and support tools for data and databases, and creation or validation of models for various life cycle impact categories. Instances of these developments will be described along with commentary on their consequences for the LCA practitioner community.

3. Results and discussion
One of the areas of strength of the Initiative is its global reach and ability to marshal expertise from developed as well as developing regions. At the end of 2011, the Initiative’s network consisted of over 2,000 members. In conducting technical studies, the Initiative has a rigorous process of soliciting proposals, evaluating those proposals, and applying a set of internal procedures to ensure the validity and integrity of the science. The Initiative maintains its own Technical Review Committee (TRC) which peer evaluates and critiques all developments within the science arena of work products. Before a technical work product is sent to the governing body of the Initiative, it is reviewed by the TRC and if found deficient the TRC chair together with the work area lead coordinates with the study director or principal investigator to rectify the issues or the output does not get the approval of the International Life Cycle Board for designation as an Initiative output. As an additional measure, works of a seminal nature are encouraged to be submitted for publication in peer reviewed scientific journals or presented at judged symposia or scientific meetings.

Contributions of the Initiative to the scientific body of knowledge regarding life cycle approaches and methods vary widely. Early work supported the codification of inventory and impact assessment methods. Together with parallel work by SETAC in organizing Pellston-type workshops to formalize the methods, considerable input was provided to updates and extensions of the international standards for LCA. More recently, this effort extended to ancillary, but still lifecycle-based, methods for carbon and water footprint calculations.

Data and databases has been a focal point as well, with recent developments including a database registry, format converter for database interoperability and a comprehensive set of sound global guidance principles for LCA databases. The latter resulted from a global Pellston workshop involving 48 scientists and
engineers from 23 countries to ensure broad applicability and leading to increased quality data generation and availability to practitioners world-wide[1].

Impact methods are another focus area with the most significant of the developments emanating from the Initiative being the application of a multi-year scientific consensus building process to create and validate a model for human and environmental toxicity within the LCA framework. This highly successful effort resulted in recommendations of specific models and characterization factors and their validity and applicability in life cycle impact assessment. The resulting model, known as USETOX, is rapidly becoming the standard within the practitioner community for these characterization factor categories[2]. Work on water use in LCA still in progress also shows promise of resulting in a consensus model framework for this impact category.

4. Conclusions

The UNEP-SETAC Life Cycle Initiative has a 10 year legacy of serving the needs of both the professional practitioner community and users of life cycle-based methods to support decision making. Serious impediments to advancing the practice of LCA have been overcome by a combination of consensus building processes, including deployment of the SETAC Pellston workshop model, and innovative science that recognizes and respects the systems nature of LCA as reflected in both functional unit and spatial/temporal scale.

5. References

Top 10 points about LIFE CYCLE every government decision maker should know

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1. Summary

The presentation focuses on a specific contribution the Life Cycle Initiative has made to government decision makers who are working towards the goals of the next World Summit scheduled to happen in Brazil on June 2012. This contribution relates to a simplified and general list of ten bullets that government officials should remember about Life Cycle Assessment when considering sustainability decisions, as it was disseminated for the first time amongst government representatives during the High-Level CSD Intercessional Meeting on Sustainable Consumption and Production (SCP) in Panama on January 2011.

The structure of the presentation provides for a brief recount of the Initiative’s mission and objectives for its current second phase of work which is almost completed, (2007-2012), as well of the 10 Year Frame Programs (10 YFP - Marrakech Process). The top 10 points are listed and key related aspects are highlighted to stress the relevance of a system based scientific approach for decision analysis and action guidance for government officials and politicians. Attention is called to follow the underlying logic in the way the 10 bullets are listed, in summary: form the environmental to human perspective, from impact identification to decision making and finally pinpointing the relevance of the choice each individual has in shaping the path towards sustainability.

The conclusion of the presentation sheds light on role of the Initiative to further contribute to government and private sectors at a very timely moment when the new mission for the next phase of its work is being shaped.

2. About the UNEP-SETAC Life Cycle Initiative and the 10YFP

The Life Cycle Initiative is a joint effort of the United Nations Environment Program (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) which, for the past 10 years, has successfully responded to the call from governments for a life cycle economy as stated in the Malmö Declaration (2000).

"To bring science-based life cycle approaches into practice worldwide" is the Mission of the Initiative which has guided work through Phase 2, extending from 2007 and up to 2012. To achieve this mission, three main objectives were established: a. enhancing the global scope of life cycle data and methodologies through research and innovation, b. facilitating the use of life cycle approaches worldwide by business, governments and the general public and, c. expanding capabilities worldwide to apply and to improve life cycle approaches through education and training.

Sustainable consumption and production (SCP) is referenced in the Johannesburg Plan of Implementation (JPOI) adopted at the World Summit on Sustainable Development in 2002, as one of the three overarching objectives of, and essential requirements for, sustainable development. The JPOI calls for action to “encourage and promote the development of a 10-year frame work of programs (10 YFP) in support of regional and national initiatives to accelerate the shift towards SCP”.

Pursuing its mission and specifically considering objective (b), the Initiative has supported the development of the 10 YFP (Marrakech Process) to promote SCP patterns, bringing to light and providing in all regions of the world the available tools for a holistic science based decision approach both for the private and the public sectors, as illustrated by the list of “10 points to remember about LCA” presented to government officials and other interested stakeholders at the CSD Intersessional Meeting in January 2012.
3. Ten points to remember about LCA

The UNEP/SETAC life cycle Initiative targeted government officials and CSD stakeholders in January 2010, to disseminate 10 points about LCA as follows:

- Every product causes environmental impacts in its supply chain, during its use and in its disposal
- To improve and preserve the world, the different impacts of products must be identified and quantified
- Life cycle assessment is the only standardized international tool for identifying and quantifying impacts of products
- By identifying impacts, they can be understood and managed by companies and governments
- Environmental impacts affect ecosystems in many different ways
- Ultimately, all environmental impacts affect humans
- Companies and the decision of governments must be made based on which impacts are most important and which should be addressed first
- Importance of impacts depends on your perspective
- All life is connected, how those connections are managed is up to you
- When making sustainability decisions, you must consider the entire life cycle including trade-offs

This presentation centers discussion on the list and how it achieves its dual purpose to first introduce the targeted audience to the relevance of science based systems approach for sustainable decisions and second, to enhance the responsibility each individual has about his or her choices in relation to the trade offs such decisions generate. Indeed, as the reader goes through the list it is clear the first five bullets introduce key concepts such as: products, environment, LCA -as the best currently available tool for identifying and quantifying impacts- and the relation impacts have on ecosystems. On the sixth bullet, the audience is further geared to directly link environment with humans putting in context the relevance of impacts be them possitive or negative. The last four bullets compell the reader to realize the responsibility of his or her decisions: the personal reference in the final three bullet further emphasises this fact and invites the reader to think about systems (connections) and burdens (trade-offs).

It is interesting to highlight the final bullet as it specifically recommends the reader to always consider the entire life cycle and the trade-offs when making decisions for sustainable development. This recommendation is key for government officials and politicians because it puts in context, for sustainability decisions, a time frame which corresponds to the life cycle approach in contrast with other time references such as the 4 to 6 year government terms.

4. Conclusions

As world economies become global, a common metric is required to understand the impacts that products and services have on sustainability. Life Cycle Assessment (LCA) is that metric. It starts with Life-Cycle Thinking, or the understanding that the environmental impact of the entire lifecycle of products and services must be considered. Life cycle thinking and more specifically life cycle tools must be integrated into government sustainability decision making, this is particularly true when considering SCP objectives. Governments set policies which directly impact production and since they also represent one of the largest consumer groups in countries, not only do they set example for better consumption patterns but also power more sustainable businesses for the society. Government officials and politicians have in the Life-cycle Initiative an excellent resource to access information, experts, guidance and tools to responsibly work towards the sustainability of the societies they represent.

5. References

www.lcinitiative.unep.fr
1. Introduction

During the last decade, environmental and social standards have emerged in global trade to define mandatory, semi-voluntary, or voluntary criteria, by which producers and exporters in developing countries must comply to gain or maintain access with markets and positions in global value chains. As these standards develop in terms of numbers, complexity and range, the costs and technical challenges are of increasing concern not least to small- and medium scale companies, which lack the necessary financial, institutional or technical capacities. Northern customers drive the introduction of product-related environmental and social standards as criteria for regulation in international trade flows and within global value chains. The UNEP-SETAC Life Cycle Initiative (LCI) was launched in 2002 and has since initiated a range of activities among them some to encourage the practice of Life Cycle Assessment (LCA) in developing countries. As LCA is the most recent addition to the toolbox of environmental management in developing countries, a review of the international transfer of this knowledge system benefits from the earlier experiences of capacity development for environment, e.g. environmental impact assessment (EIA) and cleaner technologies.

2. Materials and methods

Based on a capacity evaluation model giving equal emphasis to the practising organisation in focus and to the enabling environment, the two case studies detail capacity constraints in the process of performing a life cycle assessment. The focus of the capacity analysis is the steps of an initial LCA for the selected products. The case studies: one on pangasius aquaculture in Vietnam and the second on palm oil production in Malaysia, draw on stakeholder interviews and documentation to further identify future options for strategizing efforts, which target transition towards sustainable production and consumption.

3. Results and discussion

First of all, capacity constraints most importantly concern the efforts to collect valid and reliable data. As the first step, prior to the assessment of environmental impacts of processes, life cycle inventories (LCIs) need to be built to replace the generic databases available in current LCA software programs. The generic databases have been developed from data collected in Europe and US. However, to achieve the most accurate impact assessment, data need to be collected from the processes as they are operated in developing countries. The application of LCA is costly in terms of time and payment of licenses for LCA software, which is a serious constraint for domestic stakeholders.

Secondly, in both case countries, LCA is at an early stage of adoption as a tool of environmental management. Most examples of LCA practised primarily seem to be motivated by concerns about the public image of a particular product in the market place. This bears resemblance to an earlier wave of ISO certification, by which companies included environmental audits according to the ISO 14000 series of standards as part of their branding, while the basis for such certification reflected a variety of practises of the environmental management system. The most significant driver currently seems to be a situation, in which a vital product in the export profile of a given developing country is contested on the export markets for its environmental impacts.

Thirdly, LCA is practised as basic research and documentation by academics, except for subsidiaries of foreign companies, in which LCA is practised as part of the corporate environmental management system. Only these companies will practise LCA as a tool for transformation of processes and products. Thus, while subsidiaries of foreign owned companies take the lead in adopting a life cycle approach in environmental management as a result of corporate policies, the large sector of medium and small scale companies have
few or no incentives in their national context at a time when export markets introduce conditionality concerning sustainable production practices.

Fourthly, while fragmented activities such as awareness and short training workshops, accessibility to databases, and university research are conducted, an integrated enabling environment for LCA practices by companies, including a framework of legal regulations and incentives, has yet to be established. Current policies on sustainable production and consumption, incentives and other private sector supporting schemes need to be reviewed as the starting point for defining and implementing regulation dedicated to the promotion of sustainable production systems based upon LCA thinking.

Finally, the early response of some developing countries saw the LCA agenda as part of green protectionism and as a push for industrial modernization, which may involve direct collisions between environmental and socio-economic concerns. Thus, the adoption of LCA practices in a given country must be understood in the context of trade and environment policy positions of the national government, the level of export dependency of the national economy, and the scope for civil society, in particular the priorities of environmental non-governmental organizations.

4. Conclusions

While inputs such as methodologies, software, training modules may be readily available, both in donor-funded and in commercially based programs, strategic considerations on how these inputs may produce outcomes contributing to objectives of transition towards sustainable production and consumption are lacking. To move beyond the awareness of life cycle thinking and the communication of ‘greener’ brands, the linear model of rolling-out and implementing LCA practices in developing countries, which basically is derived from requirements of the LCA methodology itself, needs revision. Establishing knowledge and data resources, facilitating communication and training, providing limited support for application, and creating international networks for research definitely contributes to capacity development. However, a coherent strategy – drawing on the paradigm shift initiated by UNDP in 2002 and subsequent research on the concept of capacity development - is needed to identify options for interventions for direct stakeholders, for entities in the enabling institutions, for foreign investors, and for regulatory authorities in economic and industrial policies and foreign trade relations.

The international development cooperation on LCA needs to reconfigure the relationship between ‘sender’ and ‘recipient in at least three areas:

(1) In relation to government policy, LCA/LCM methodologies need to respond to the specific context of developing countries to fully incorporate socio-economic concerns in developing countries; also, programs of action for LCA in developing countries must be strategized to integrate with the current level and scope of environmental management in a given country.

(2) In relation to company practices, research on simplified tools for small producers must be stepped up, and manuals for application must build upon examples relevant to production and services in developing countries.

(3) In relation to the actors in domestic and export markets, the application of LCA in developing countries must produce immediate and tangible benefits as a contribution to transition towards national objectives of sustainable production and consumption, and as enabling steps to maintain or access positions in global value chains.

Trends in international development cooperation offer new opportunities for mainstreaming the life cycle perspective and the application of simplified life cycle assessment tools. Donor support to private sector development in African countries increasingly targets interventions in global value chains to upgrade producers and improve the livelihood of smallholders and workers. Pro-poor and green growth strategies include efforts for environmental efficiency, although most often on a limited basis. At the macro level, the Dutch Sustainable Trade funded by the Government of Netherlands takes on the challenge to motivate the dominant companies in fifteen global commodity chains to move towards sustainable production. In both cases, LCA has an important role in scrutinizing claims about sustainability and clarify dilemmas between developmental and environmental concerns.
Water use in life cycle assessment and water footprinting: outputs and prospects of the working group WULCA

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1. Introduction

Water use has been largely neglected in LCA of the first decades. While some water use data has been collected in inventories, environmental impacts related to use and consumption of this vital resource have not been addressed until 5 years ago. At this time different research groups have started to develop methods to quantify impacts, and the UNEP-SETAC life cycle initiative has established the working group for water use in life cycle assessment (WULCA).

2. The Working Group

The WULCA working group is organized by a core of researchers actively contributing to the development of the field and practitioners from various organization representing different sectors. A larger group of interested stakeholders is involved in rounds of feedbacks and invited to meetings in order to allow input from various perspectives in a very dynamic field. In total, the group is composed of approximately 30-40 members, representing the scientific community, the industry, and consulting interested in following the development on this topic. The more active part of the group is mainly composed of ~20 persons.

3. The Working Group’s Deliverables

3.1. Defining a framework

The first goal of the group was to compile a consistent framework for addressing the environmental impacts related to the various types of water use and areas of protection affected. Another challenge was to establish a common terminology. As a result, the working group published a scientific article which builds the basis for future research in this area \cite{1}. The framework is depicted in Figure 1.

3.2. Assessing existing methods and providing guidance

In a second phase, the group focused on comparing different available methods, comparing their scientific quality, completeness and applicability on order to provide guidance to LCA practitioners and other interested people about the status of inventory data and impact assessment methods. They provided recommendations on inventory and impact assessment. Further, requirements for future research were elaborated and practical advice for practitioners was given. In this context, existing water footprint methods and approaches have also been compared.

3.3. Education and training

The group is also involved in compiling material and providing up-to-date training on behalf of the UNEP-SETAC Life-Cycle Initiative. Examples of past trainings include the training series on “Capacities for Sustainable Resource Management in Latin America and the Caribbean” in Mexico (Spring 2011) and Argentina (Fall 2011).
4. Closing gaps and harmonization

Future action points are the harmonization of existing LCIA methods and inventories. Building up on the methodological review, a quantitative comparison of different approaches targeting similar impacts are performed to derive estimates of robustness, while research of missing impact pathways is fostered. Inventory data collection has been of lower priorities so far, however, with increasing demand for both robust water use information and related impact assessment, the supply of representative water use data becomes a relevant issue, especially due to the high spatial variability of water use and related impacts. Ultimately, a complete and harmonized water impact assessment methodology arising from the comparison efforts with past and future developments is foreseen.

5. References


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1. Introduction

The nexus between security and safety and environmental performance of packaging in food and beverages is a key question driving the market for sustainable packaging today. Therefore, an articulation of the benefits of the life cycle approach to design, manufacturing, use and end of life management of packaging for food applications is important to further examination of the role of packaging.

Key items of focus for this research and analysis included:

- What is the value of a life cycle approach for beverage and food products and packaging?
- What is the value of including all life cycle stages in evaluating the packaging/food systems to reduce overall life cycle impacts?
- What is the value of including multiple impacts in evaluating the packaging/food systems to reduce overall life cycle impacts?
- What is the value of including the food and/or beverage into an evaluation of the packaging life cycle impacts?
- What characteristics of future LCA studies should be considered when evaluating the food/packaging life cycle?
- Examples of how the waste management hierarchy and LCA results interface/connect

This presentation will present the results of study to examine the Value of a Life Cycle Approach in Evaluating the Environmental Impacts of Packaging for Food and Beverage Applications.

2. Materials and methods

This work, coordinated by the UNEP/SETAC Life Cycle Initiative, brought packaging and finished goods manufacturers together to advise research that would mine existing LCA studies to enable an articulation of the benefits of the life cycle approach to design, manufacturing, use and end of life management of packaging for food applications.

The boundaries of the project included evaluation of the following types of packaging:

- Primary and secondary, including bags, boxes, cans, cartons
- Finished product packaging as well as wholesale
- Packaging for prepared foods as well as fresh
- Food and beverage packaging

The UNEP/SETAC Life Cycle Initiative brought to the project a unique combination of benefits that cannot be obtained from other sources. These benefits include:

- Neutral, objective, authoritative, and recognized forum for advancing understanding of packaging life cycle for food applications
- Global dissemination of the report
- Proven ten (10) year history of solid project deliverables

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1 Value is defined within a sustainability context to ensure that decisions regarding packaging and food product systems maximum benefits to society while minimizing risk/impacts and includes ensure that those decisions do not create unanticipated and unexpected impacts to different life cycle stages, or regions.
3. Results and discussion
This effort educates audiences (throughout the UN network of countries) including organizations in the packaging value chain, regulators, and other stakeholders about the value of a life cycle approach to better inform business decisions and policy making.

4. Conclusions
This presentation will present the results of study to examine the value of a Life Cycle Approach in Evaluating the Environmental Impacts of Packaging for Food and Beverage Applications.

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