LCA an Introduction

Life cycle assessment (LCA) is a decision support tool that has been developed in order to analyse the environmental burdens associated with the production, use and disposal of a product and is arguably the best way of quantifying this information (Hill 2011). The term 'product' in this context includes both goods and services. Interest in LCA grew rapidly during the 1990's and it generated high expectations, but also increasingly became the focus of criticism (de Haes 1993, Ayres 1995, Ehrenfeld 1998, Krozer and Viz 1998, Finnveden 2000). However, since that time there has been considerable progress made, with the development of international standards (ISO 14040, ISO 14044). There are also several international initiatives taking place with the aim of building consensus and developing robust methodologies. These include the Life Cycle Initiative of the United Nations Environment Program (UNEP), the Society of Environmental Toxicology and Chemistry (SETAC), the European Platform for LCA of the European Commission (EPLCA) and the International Reference Life Cycle Data System (ILCD). Although a useful tool, LCA does have its limitations. There has been criticism of the practicability of the use of LCA for the construction sector, due to the lack of availability of input data and the complexity of the LCA process, resulting in a large amount of time being taken to analyse even a small percentage of the tens of thousands of construction products available (Taratini et al. 2011). The production of an LCA does involve value choices and it has to be accepted that LCA is an imperfect tool to inform decision-making processes and that other considerations may also apply when deciding on policy instruments. Werner et al. (2007) note that LCA is to some extent subjective in nature and they refer to the mental models employed by the decision maker when conducting the analysis. This process can involve:

- Making a distinction between products, co- or by-products and waste when allocating environmental burdens
- The choice of an appropriate allocation factor
- The selection of appropriate substitutions or additional processes if system expansion is employed to avoid allocation
- How to handle lack of knowledge about processes when this occurs

LCA is inevitably a simplification of an extremely complex subject and it is important to realise that it does not capture all of the environmental aspects associated with the product system or process under study. According to Owens (1997) 'LCA may capture the global aspects of the environmental impacts by reporting impact categories such as global warming potential, ozone layer depletion, but it does not inform the analyst about more local or transient impacts and is of limited value when considering biological impacts, such as biodiversity, habitat alteration or toxicity'. It is important to be aware what LCA can and cannot do.

In order to conduct an LCA it is first necessary to determine the goal and scope (i.e., what is the purpose behind conducting the LCA and what is being included in the study). The scope must define what the system boundaries are in the study and the functional unit must be declared. For many purposes, the system boundary can be defined as ‘cradle to gate’, that is the manufacture of a specific product in a factory to the point at which it leaves the facility (modules A1-A3 in EN 15804). This usually gives the most accurate data, because this stage of a product life cycle involves the fewest assumptions and the data gathering process is *relatively* straightforward. However, a cradle to factory gate LCA does not represent the whole life cycle and may result in erroneous conclusions. For example, a low impact product, as determined through a cradle to gate analysis, may prove to require a lot of maintenance during the in-service phase of the life cycle, or there may be serious environmental impacts associated with disposal. A full appreciation and understanding of the environmental impacts associated with a product choice therefore requires the whole life cycle to be considered. This invariably introduces a higher level of uncertainty into the process, because there may be aspects of the life cycle that are not well understood and this requires assumptions to be made. These assumptions may have a very significant impact upon the LCA and there may be bias introduced if comparisons are made between competing products.

Life cycle assessment is not static and there are ongoing programmes dealing with improving various aspects of this methodology (Finnveden et al. 2009). It is important that the correct decisions are made regarding the choice of materials for the built environment and LCA can be used as a means for informing those choices. This requires that LCA is used correctly and that the decision support tools allow for comparability between products (Forsberg and Malmborg 2004, Haapio and Viitaniemi 2008a, b , Ding 2008, Audenaert et al. 2012). There are several LCA-based building assessment tools available (Bribián et al. 2009), e.g.:

- ECO-QUANTUM www.ecoquantum.nl
- LEGEP www.legep.de
- EQUER www.izuba.fr
- ATHENA www.athenaSMI.ca
- OGIP www.ogip.ch/
- ECO-SOFT www.ibo.at/de/ecosoft.htm
- ENVEST 2.0 envestv2.bre.co.uk
- BECOST www.vtt.fi/rte/esitteet/ymparisto/lcahouse.html
- BEES www.bfrl.nist.gov/oae/software/bees.html
- GREENCALC www.greencalc.com
- ECOEFFECT www.ecoeffect.se
- LEGEP www.legep.de
- EQUER www.izuba.fr

Life cycle assessment is broken down into several stages, which are described below.

Goal and scope definition

The goal and scope stage comprises the writing of a series of statements at the beginning of the analysis which tell the reader the reason why the LCA was performed, who is performing the study, who the client is and what is covered in the LCA. It is at this stage that the system boundary is defined. For example, the purpose may be to undertake an LCA of the manufacturing process only (cradle to factory gate), or the whole service life may be included. Additional parts of the lifecycle, such as recycling and disposal may also be included. The purpose of the LCA may be simply to report the environmental burdens associated with a product or process (referred to as an attributional LCA), or it may examine the consequences of changing various parameters or assuming different scenarios (called consequential LCA) (Frischknecht and Stucki 2010, Gala et al. 2015). It is also necessary to specify what the subject of the LCA is. This is referred to as the declared unit, if cradle to factory gate is being analysed, or the functional unit, if other parts of the lifecycle are also being studied. Another important consideration when studying the environmental impacts associated with a product or process is the timescale involved and it is important that this is also defined at this stage. It is also a requirement to specify what allocation procedures were used during the analysis.

Life cycle inventory

This phase of the analysis requires the assembly of all of the information about the process. In order to do this, an imaginary system boundary is drawn around the process and all of the material and energy inputs and outputs are quantified. This process is usually divided into the different life cycle stages, manufacture, service life, end of life, disposal. Once the life cycle inventory (LCI) phase of the analysis is complete then data gaps are identified. In some cases, it is possible to collect the missing data, but where this is not possible, 'reasonable' assumptions have to be made. During this phase, mass balance calculations are also performed. This is a very useful tool for identifying data gaps and is based upon the principle that the mass of all matter going into the system under study should equal that of all the matter exiting the system. At some stage, the data gathering process has to be terminated and the point at which this occurs is determined by cut-off criteria. Data falls into two principal categories: primary (foreground) and secondary (background) data. Primary data is that which has been gathered by the LCA practitioner and may include utility bills, delivery notes and other information that is directly linked to the process. Secondary data is that which has not been directly obtained, but is more generic in nature; for example, if wooden pallets are used to ship the product, then it is highly unlikely that a full inventory of the pallet would be made.

Ultimately, what should result from such an analysis is a table (called an input-output table) that represents flows of materials and energy to and from nature (the ecosphere). All of the foregoing is complicated enough, but if the factory in question also produces other products (co-products) then the question of allocation of the environmental burdens to the different components in the inventory to the declared unit must be considered. For example, a utility bill for a factory will give the total electricity consumption for a year, but if the factory makes ten products then a means of correctly allocating the electrical energy (and associated environmental burdens) to the analysed product must be derived. The collection and analysis of data invariably leads to issues regarding commercial confidentiality, which can cause problems, especially when the LCA has to meet adequate levels of transparency in order to be credible.

Life cycle impact assessment

Once the LCI phase has been completed, it is then necessary to quantify the environmental burdens, during the life cycle impact assessment (LCIA) phase. At this stage there are several further complications that have to be considered. There is still discussion as to how to do this in order to properly report the environmental burdens, but a consensus has been developing over the past decade. The principle is to aggregate the environmental implications associated with the flows to and from nature into a small (but nonetheless meaningful) set of indicators. This methodology has essentially distilled down into two main approaches, referred to as midpoint and endpoint indicators (Bare et al. 2000, Jolliet et al. 2003, 2004, Ortiz et al. 2009, Hauschild et al. 2013). In the midpoint approach, the environmental burdens are grouped into similar environmental impact categories (e.g., global warming potential, ozone layer depletion, freshwater eutrophication, etc.). The endpoint approach seeks to model the chain of cause and effect to the point of the evaluation of damage, which makes for simpler reporting with fewer indicators, but has a much higher level of uncertainty. Midpoint is preferred because of the higher level of accuracy, but can be more difficult to interpret (Dong et al. 2014). Endpoint impact categories are reported in terms of impact on human health (e.g., DALY, disability adjusted life years), or on ecosystems (e.g., species loss). Some systems have even gone so far as to aggregate all of the impacts into one category (e.g., ecopoints), but the data reported using this approach has very high uncertainties associated with them. The environmental impacts are calculated using a variety of models (over 150) which attempt to determine the impacts of processes upon the environment. Examples of such models include:

- Midpoint: TRACI, CML, EDIP, ecopoints
- Endpoint: Eco-indicator, LIME2
- combined midpoint and endpoint: ReCiPe (Bare et al. 2000), IMPACT 2002+ (Jolliet et al. 2003).

In IMPACT 2002+ the ‘value’ of the environmental impact is reported as an ecoindicator and measured in environmental points. The accumulated ecoindicator is composed of damage categories (human health, ecosystem quality, climate change, resources) and impact categories (carcinogens, non-carcinogens, respiratory inorganics, respiratory organics, ionising radiation, ozone layer depletion, aquatic ecotoxicity, terrestrial ecotoxicity, terrestrial acidification, land occupation, aquatic acidification, aquatic eutrophication, global warming potential, non-renewable energy, mineral extraction). This requires a weighting process to be applied, which is reliant upon value judgement.

The impact categories selected should provide useful information about the product or process, taking the goal and scope of the study into consideration. When selecting the impact categories, it is also necessary to select the characterisation factors, which are the units used to report the environmental burden. To consider the example of the climate change impact category, the characterisation factor for this is global warming potential with a 100-year timeframe (GWP100) and the characterisation factor for this is kg CO_2 equivalents. The method used to calculate impacts can affect the results of the LCA study and this should always be remembered when making comparisons between products or materials in different studies (Monteiro and Freire 2012, Herrmann and Moltesen 2014, Lasvaux et al. 2015, Bueno et al. 2016). For some impact categories mixing, LCA databases is not appropriate, but for the main indicators used by the building sector (GWP and embodied energy) the information is reasonably comparable.

Another important factor is the correct allocation of environmental burdens to different co-products, if the system under analysis produces more than one product. Examples of this include the allocations between cereal and straw, or meat and wool in agricultural production systems (Brankatschk and Finkbeiner 2014). Ideally, allocation should be avoided when possible; but in many cases, this cannot be done and a choice has to be made regarding the allocation procedure used. Various approaches can be used for allocating environmental burdens, including mass, energy, or economic allocation. Guidance regarding allocation is given in ISO 14040 and ISO 14044, recommending a hierarchy of choice for allocation methods. In many cases, economic allocation is used, which gives a more realistic allocation of the burdens. This is because economic activity is the primary motivation for the manufacturing of products and this allocates the highest environmental burdens to the highest value products.

Jungmeier et al. (2002a) identified ten different processes in the forestry value chain where allocation issues can occur: forestry, sawmill, wood industry, pulp and paper industry, particle board industry, recycling of paper, recycling of wood-based boards, recycling of waste wood, combined heat and power production, landfill. These can be divided into multi-output processes (e.g., sawmill) or multi-input processes (e.g., landfill). A forest can produce wood for a variety of uses, including: solid wood, particle board, paper pulp, plywood, biomass for fuel. The question then arises how to allocate the environmental burdens associated with the forestry and harvesting operations to the different outputs. One way of dealing with this is to employ system expansion so that all of the different product streams are included within the same system boundary. The problem with this approach is that the functional unit that is now considered may not be very useful. For comparison purposes, the wood-based functional unit must be the same as the non-wood-based functional unit. If a timber frame building is manufactured with the result that the waste from the process is used for energy production, then it is possible to use system expansion to compare the functional unit as being the structural frame plus the production of x kWh of energy. However, if the wood waste or by-products go to the production of chipboard or paper, then the comparison becomes more difficult. It is almost inevitable that some form of allocation will have to be employed. In many cases an economic allocation may be the best way of allocating burdens, but prices can fluctuate. Furthermore, the forest can produce different product streams at different times (first thinnings, second thinnings, third thinnings, harvest) which adds to the problem of economic allocation over a time scale that can be as long as a century (Jungmeier et al. 2002b).

At the end of the LCA process, there are additional analyses that can be performed, these are normalisation, grouping (aggregation), or weighting. These are usually used to make the environmental information more understandable (Chau et al. 2015).

Normalisation is the calculation of an environmental impact relative to some reference data, in order to give some context to the information. An example of this would be comparing the carbon dioxide emissions of a process with that of an average European citizen for one year. Although this can give information regarding the importance of a particular environmental impact, uncertainties in the characterisation factors can lead to uncertainties in the results.

Grouping (aggregation) involves combining different impact categories into a few or even one. An example would be the combining of the global warming impacts due to the emissions of different greenhouse gases and reporting this in one impact category. In this case, the commonly used unit is carbon dioxide equivalents over a 100 year' time frame (GWP100 in kg CO_2-equivalents). In this example, the science is extremely well understood, due to the huge scientific effort that has gone into researching climate change, but for other impact categories there is much greater uncertainty and debate. The challenge with LCA is to use enough impact categories to make the analysis meaningful, but not so many that it makes the LCA only of interest to a very small number of academics. It is also possible (although extremely unreliable) to group everything into one impact category, as is done with systems, such as the BRE Green Book, BREEAM, LEED, Cradle to Cradle, etc., which is a huge oversimplification of what is a complex subject. For example, if a reduction of global warming potential occurs at the expense of a huge increase in ozone depletion, then this is of dubious benefit. However, if a large reduction in global warming results in a modest increase in ozone layer depletion, then this may be a sensible choice to make. The question is how to balance a decrease in one impact category against an increase in environmental burdens elsewhere. Grouping therefore requires the assigning of different weighting factors in order of a real (or perceived) impact, or importance. The environmental impacts may be quite different globally and locally (Khasreen et al. 2009). For example, global warming and ozone layer depletion are global impacts with long time scales, whereas eutrophication tends to be much more localised and with shorter time scales. The relative importance of these impacts is therefore very different, depending on the perspective of the analysis (Yang 2016).

Weighting is a process which has to be performed before the indicator results of different impact categories are combined into a single score. This is reliable when based upon strong scientific evidence (e.g. GWP, or ozone depletion), but more often it involves value judgements to be made regarding the relative importance of different impact categories. This becomes increasingly unreliable as more impact categories are included and extremely unreliable when impact categories from different disciplines (environmental, economic, social) are combined into one overall impact category to give a measure of the 'sustainability' of a process or product. This single indicator approach makes the route by which the score was obtained non-transparent and can be subject to manipulation or lobbying. There is no consensus regarding the methodology of the weighting process, meaning that different schemes are incompatible. Ultimately, it would be desirable if the outputs from LCA could be converted into a monetary value, but this is a long way from being realised (Pizzol et al. 2015).

As noted previously, the science underpinning the relationship between the release of a substance into the environment and its impact is better understood for some impact categories than it is for others. In 2011 the European Commission Joint Research Centre (EC-JRC) Institute for Environment and Sustainability published the International Reference Life Cycle Data System (ILCD) Handbook. This examined 14 impact categories at midpoint level and three at the endpoint level. Only the IPCC method for climate change and the World Meteorological Organization method for ozone depletion (midpoint) are characterised as Level I (recommended and satisfactory) (Finkbeiner 2014). However, the EC-JRC is currently in the process of updating ILCD recommendations for 4 impact categories (water depletion, resource depletion, land use and respiratory organics). JCH Industrial Ecology Ltd is monitoring these developments closely and this information sheet will be modified accordingly when the relevant PEF product category rules have been agreed.

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