SUSTAINABILITY

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CONTENTS

EMBODIED ENERGY

The embodied energy (or energy cost) of a material or product used is often defined as the primary energy used in the manufacture, which includes all of the energy used in the production, as well as the primary energy used in the transport of materials and goods required for the production process (Hill 2011). Embodied energy can be subdivided into initial embodied energy (the energy used for resource extraction, processing, transport and manufacture of the product) and recurring embodied energy (energy used for maintenance or replacement). Definitions of embodied energy can vary from study to study. Embodied energy analysis can be used as a proxy for determining the environmental impacts associated with a product or service (Huijbregts et al. 2006), but more usefully it can be used in the calculation of energy payback times, or energy balances. An example would be calculating the energy savings associated with using different insulation materials in a building, compared with the energy saved.

The Ukrainian socialist Sergei Podolinsky was the first to attempt a description of economics from a thermodynamics perspective, performing energy analyses of society that were a century ahead of their time (Martinez Alier and Naredo 1982, Martinez Alier 1990). Robert Costanza adopted the term 'embodied energy' to describe the total energy cost of a product or service and proposed that this was a better measure of the true price rather than the market price. He argued that a perfectly functioning free market would give prices that were proportional to the embodied energy content (Costanza, 1980). The link between embodied energy values and price is strongly influenced by the economic cost of energy.

The embodied energy is invariably reported according to the cumulative energy demand (CED) method, which states that the embodied energy is assessed as the primary energy used for the manufacture, use and disposal of an economic good (product or service), or which may be attributed to it with justification. The method distinguishes between non-renewable and renewable energy use. The cumulative energy demand (CED) represents the primary energy used (both direct and indirect) during the life cycle of a product (Huijbregts et al. 2006). This includes the energy consumed during the extraction, manufacturing and the disposal of the product and raw and auxiliary materials. Different methods for determining the primary energy demand exist. For example, the lower or higher heating values of primary energy sources may be used, the use of renewable energy resources may not be included or it may be reported separately. Huijbregts et al. (2006) found a good correlation between fossil CED and GWP and resource depletion, but the correlations with acidification, eutrophication, tropospheric ozone formation, stratospheric ozone depletion and human toxicity were much lower and for land-use they were absent.

JCH Industrial Ecology mainly conducts embodied energy analyses for materials used in the built environment. Buildings and building materials are responsible for the consumption of nearly 40% of global energy (Dixit et al. 2012). The total life cycle energy use of a building comprises the operational (or direct) and embodied (or indirect) energy. The sum of these two (direct and indirect) can be considered to be the embodied energy of the building, but it is useful to report them separately. The embodied energy of the materials in a building seldom exceeds more that 30% of the total energy demand (embodied plus operational energy), unless the building is deliberately designed to be a low energy structure. End-of-life energy requirements, demolition and disposal also make a very small proportion of the energy use of conventional houses (Winistorfer et al. 2005). As building energy efficiency increases, then the material embodied energy and energy associated with demolition and disposal will assume greater importance.

However, on a global or national scale, the materials' embodied energy can be much higher than 30% of the total energy demand of the building sector, due to a growing population and hence a growing demand for buildings (Treloar et al. 2001).

The units used for reporting embodied energy are generally MJ per unit mass, or volume, or per defined functional unit, although some workers report this as kWh (=3.6 MJ). Transport of materials to the building site can have a major impact on the embodied energy of the construction materials. Morel et al. (2001) reported that the amount of energy used to manufacture and transport building materials represented nearly 8% of primary energy consumption in the UK. They showed that by using local materials, it was possible to reduce the embodied energy associated with a building by up to 215% and the impact of transportation by 435%.

In some LCAs, the energy used for the maintenance of the product is also included in the embodied energy analysis, although this should be reported separately as the recurring embodied energy. This is distinct from the initial embodied energy, which is constant once the product is manufactured and installed (Ramesh et al. 2010, Chau et al. 2015). Some workers also include the energy associated with the disposal of a product at the end of a life cycle, although (again) this should be reported separately.

Dixit et al. (2012) noted that some research workers do not include renewable energy in their definition of embodied energy and also found that the use of different information sources and failure to distinguish between primary or secondary energy could lead to errors as high as 40% when reporting embodied energy. They stated that there is a need to develop a common methodology to accurately determine the embodied energy associated with buildings and that there is a need to develop a complete and robust database of embodied energy information. There is also the widely-used University of Bath Inventory of Carbon and Energy database, which has just been updated (e.g., Lee et al. 2011). The current Bath ICE database has been shown to inaccurately report data for harvested wood products (Hill and Dibdiakova 2016). Cabeza et al. (2013) note that there is a relationship between embodied energy and GWP for primary production, for some building components and that there is a link between embodied energy and cost of buildings, which is related to the energy intensity per unit GDP for that country. An example is shown below for cement production, from research conducted by JCH Industrial Ecology Ltd (Figure 1).

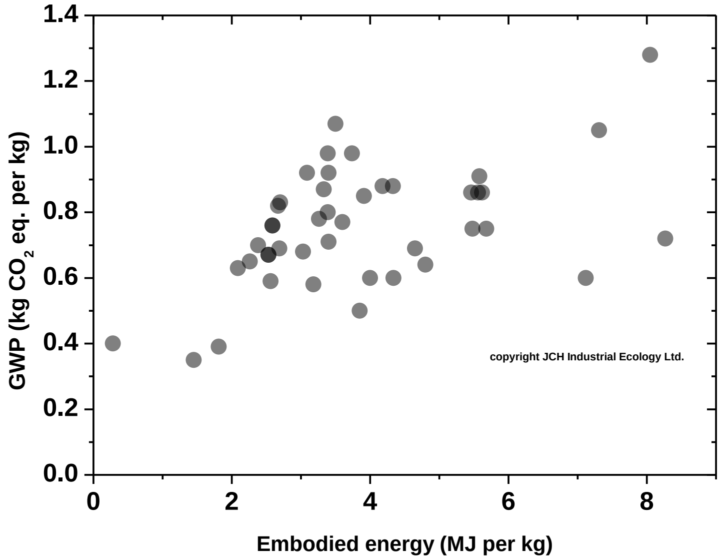

Figure 1: Relationship between embodied energy and GWP for cement products

It is necessary to define the meaning of primary energy, since it is not always clear that the primary energy has been used when the embodied energy is reported. The primary energy is defined as the energy measured at the natural resource level. This is the energy used to produce the end-use energy which includes the energy used in the extraction, transformation and distribution to the user (Fay et al. 2000). Measurements of embodied energy are only consistent if they are based upon primary energy but if delivered energy is used, the results are misleading. Unfortunately, there is a lack of clarity and incomparability in the reporting of embodied energy (Dixit et al. 2012) and the current standards do not provide clear guidance. For example, the European Standard EN 15804 does not mention embodied energy, although it does require the reporting of energy inputs as primary energy and requires the reporting of the following categories describing resource use:

- Use of renewable primary energy excluding renewable primary energy resources used as raw materials
- Use of non-renewable primary energy excluding non-renewable primary energy resources used as raw materials

It is important to distinguish between embodied energy, which is associated with the production of a good or service and the inherent (or embedded) energy, which is a physical property of the material. The terms embodied and embedded are sometimes (often) confused in the literature. As noted previously, the embodied energy of a material is the primary energy that is associated with the extraction, processing and transportation of that material from the cradle to the factory gate (and can include other life stages as well). In contrast, the embedded energy of a material is a property of that material and can be directly measured. For example, the inherent energy in a wood product can be recovered at the end of its life cycle by incineration, whereas the inherent energy of concrete is zero. The inherent (embedded) energy is reported in EN 15804 in the following categories:

- Use of renewable primary energy resources used as raw materials
- Use of non-renewable primary energy resources used as raw materials

However, different LCA practitioners report data for these categories in different ways. Because In addition, the inherent energy is reported as primary energy in these categories, which this does not necessarily represent the true value of the recoverable energy, which is usually more accurately reported for wood as the lower heating value (LHV).

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LIFE CYCLE ASSESSMENT

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 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 (Udo 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 (Tarantini 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) state 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. For example, 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 (Owens 1997).

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 gives the most accurate LCA, because this stage of a product life cycle involves the fewest assumptions and the data gathering process is relatively straightforward. However, 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

Goal and scope definition

The goal and scope stage involves the writing of a series of statements at the beginning of the process which tell the reader the reason why the LCA was performed, who is doing 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 (GWP_{100}) 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). Bueno et al. (2016) compared five different life cycle impact assessment methods (EDP 97/2003 (midpoint), CML 2001 (midpoint), Impact 2002+ (midpoint and endpoint), ReCiPe (midpoint and endpoint) and ILCD recommended practices for LCIA (midpoint)) for consistency of results. The two endpoint methods gave different answers. The midpoint methods gave consistent results for the impact categories: Aquatic and Freshwater Ecotoxicity, Ionising Radiation, Particulate Matter Formation, and Resource Depletion. Global Warming, Terrestrial Ecotoxicity, Human Toxicity (except for the Non-carcinogens impact category) and Land Use (except for Natural Land Transformation), but not for Ozone Layer Depletion, Photochemical Oxidant Formation, Acidification, Terrestrial and Aquatic Eutrophication, Marine Ecotoxicity and Water Depletion. Lasvaux et al. (2015) compared the Ecoinvent database and the Environmental Product Declaration (EPD) database in France. The environmental impacts of 28 building materials were compared using the Life Cycle Impact Assessment Indicators (LCIA) of EN 15804, for the cradle to gate part of the life cycle. The results obtained for the impact categories related to fossil fuel consumption, such as abiotic depletion potential, GWP and primary energy demand showed differences of less than 25% between the two databases, but other indicators showed much higher deviations (sometimes by more than 100%). They recommended that for some impact categories mixing LCA databases is not appropriate, but that for the main indicators used by the building sector (GWP and embodied energy) the information was reasonably comparable between the two databases.

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 (GWP_{100} 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).

Another topic that requires consideration is the use of databases and LCA software calculation tools. In principle, an LCA would have to include all of the aspects which are related to the system under consideration. For example, pallet use and transport thereof may form part of the overall analysis. There are a very large number of contributions that would have to be in turn subjected to an LCA, with the chain of cause and effect becoming very large and complex. In order to make an LCA manageable, considerable use is made of databases; the most common of which are Ecoinvent and GaBi (in Europe). From these, the practitioner can look up the environmental impacts associated with pallet use and transport and many other products and services. One of the questions to be asked is: does the database that is used affect the results? This was the question behind the study conducted by Herrmann and Moltesen (2014) on the LCA calculation tools SimaPro (which uses the Ecoinvent database) and GaBi. They found that in many cases the answers from these two databases were the same or closely similar, but in some situations very different answers were obtained. The unit processes for which different results were obtained were: ‘particles > 10 μm to air’, ‘sulphate to air’ ‘niobium 95 to air’, ‘thorium 230 to air’, ‘thorium 232 to air’ and ‘zirconium 95 to air’. When calculating the impacts based on the Soybean Brazil Farm unit process, differences were found in the results for all impact categories, except acidification and terrestrial ecotoxicity. A very large difference of 91.6 was found for the impact category photochemical ozone formation (human exposure) and 2.8 for global warming potential. In an LCA study of biodiesel production, differences were found in GWP and aquatic eutrophication and in a study of palm oil production there were differences in land-use change impact. This is a potential confounding factor when attempting to compare different LCA studies. The authors emphasise that they only examined about 1% of the full population of the data in the two LCA software systems and about 10% of the LCIA data was examined. There is a need for more work on the comparability of results when different databases are used to source secondary data.

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SEQUESTERED CARBON

Atmospheric carbon dioxide is sequestered by plants during photosynthesis and remains stored in the material of the plant until the carbon is subsequently oxidised at the end of life. The question of how to account for the storage of carbon in biogenic products has been the subject of much debate and the issues that this presents have still not been fully reconciled. Between 2006 and 2009, the LCA community debated how to construct methods to calculate biogenic CO_2 and eventually they came to the conclusion that it was best not to calculate it at all (Vogtländer et al. 2014). The reason for this decision was that the stored biogenic carbon will eventually re-enter the atmosphere at the end of life of the product. The inbuilt algorithms that calculated biogenic carbon storage were consequently removed from computer programs, such as Simapro, which uses the Ecoinvent database. Biogenic carbon dioxide was removed from the GWP indicators of the IPCC and systems such as CML-2 and ReCiPe. However, more recently it has been realised that the storage of biogenic carbon does have a role to play in climate change mitigation and this needs to be recognised. Gustavsson and Sathre (2011) note that much of the methodology developed for determining the GWP impacts associated with the production and use of biofuels is also useful for the comparison of HWPs with alternative construction materials. The difference is that the harvested material is not immediately oxidised and the methodology has to take account of this ‘delayed emission’. The storage of atmospheric carbon in long-life timber products has a greater climate change mitigation benefit, compared with immediate oxidation for energy recovery (Stewart and Nakamura 2012). Furthermore, the use of bioenergy facilities to utilise the inherent energy of processing and harvesting residues provides additional climate benefits not included in the forestry chapter of national GHG inventories.

For the calculation of the GHG emissions and storage associated with products and services, there have been several methodologies published: The World Resources Institute (WRI) and World Business Council for Sustainable Development (WBCSD) Greenhouse Gas Protocol, publicly available specification (PAS) 2050, the ILCD method and ISO 14067.

The WRI/WBCSD GHG Protocol was developed as a product GHG accounting and reporting standard in 2001. It was subsequently revised in 2011 to provide a generalised methodology for GHG reporting and quantification.

When the benefits of carbon storage are considered in LCA, this requires consideration of the time of storage to determine what the GWP impact is. The question of the temporal nature of emissions of carbon into the atmosphere and considerations of the length of time that atmospheric carbon is held in storage are extremely important when biogenic carbon is considered (Cherubini et al. 2012). Unfortunately, there is no consensus regarding the methodology for measuring and accounting for carbon in biogenic products. Although the ILCD methodology is still current, there have not been any useful developments in standardisation. The 2008 version of PAS 2050 did include methods for calculating the temporal aspects of biogenic carbon storage in annex C, but by the time that the 2011 version had been published, this was no longer present. The European Standard EN 16485 giving product category rules (PCR) for round and sawn timber featured a temporal calculation method for determining the storage of biogenic carbon in the draft form, but in the final published version this had been removed.

ISO 14067: 'Carbon footprint of products - requirements and guidelines for quantification and communication' provides protocols for the transparent reporting of GHG results. Due to objections raised by some countries, ISO 14067 was published as a technical specification rather than an international standard in May 2013. Although the WRI/WBCSD GHG Protocol, PAS 2050 and ISO 14067 all have similarities, ISO 14067 gives details on the selection of appropriate system boundaries and it also gives guidance on the simulation of use- and end-of-life phases of the life cycle (Wu et al. 2014b, 2015). The development of a GHG reporting standard was first proposed by the ISO Technical Committee 207, Working Group 2 (ISO TC207/WG2) in April 2008 and development involved more than 100 experts from over 30 countries. ISO 14067 gives the methodology for the calculation and reporting of the carbon footprint of goods and services, due to the emissions and removals of GHG gases during the product lifetime. It was developed based upon previously published ISO standards on environmental labelling and environmental management:

- ISO 14020: Environmental labels and declarations - general principles.
- ISO 14024: Environmental labels and declarations - Type I environmental labelling - principles and procedures.
- ISO 14025: Environmental labels and declarations - Type III environmental declarations - principles and procedures.
- ISO 14040: Environmental management - Life cycle assessment - principles and framework.
- ISO 14044: Environmental management – Life cycle assessment – requirements and guidelines.

The methodology requires two main parts: a carbon footprint report and a critical review (which is not the same as a third-party verification). The purpose of the critical review is to ensure that the methods given in ISO 14067 are followed correctly and that they are scientifically valid. The data that is used and interpretations should be reasonable and appropriate, given the limitations identified and the goal and scope of the study. The study should also be transparent and consistent.

The current version of the PEF guidance document (Commission Recommendation 2013/179/EU) available from the European Commission, Section 5.4.9 deals with the issue of temporary carbon storage with the statement 'Credits associated with the temporary (carbon) storage or delayed emissions shall not be considered in the calculation of the default EF impact categories. However, these may be included as "additional environmental information".

Conventional LCA methods do not assign any benefits to the temporary storage of atmospheric carbon, because the timing of emissions relative to removals is not considered (Pinsonnault et al. 2014). Although there are benefits to be gained from using timber products in long-life products as a store of atmospheric carbon, there is still no agreed way of accounting for this (Brandão et al. 2013).

The advantages of using timber and other bio-derived materials as a means of storing sequestered atmospheric carbon in the built environment has received considerable attention in the scientific literature (e.g. Pilli et al. 2015, Jasinevičius et al. 2015, Brunet-Navarro et al. 2016). Werner et al. (2005) analysed the consequences of the increased use of timber in construction on the carbon storage and emissions associated with substitution for more energy-intensive building materials. The study showed that the effects due to carbon storage were of minor importance compared with those due to substitution for more energy-intensive materials and the use of timber residues and post-consumer waste wood as an alternative energy source compared to fossil fuels. Werner et al. (2006) argued that the climate change mitigation consequences of using increasing amounts of timber in long-life products depended upon a number of factors:

- Up to twice as much biomass is removed from the forest compared to the amount of wood being used in the built environment and it is important to consider what happens to this wood.
- What happens to the surplus wood in the forest after harvest is important with respect to the effect on CO_2 emissions associated with harvesting. Using this residue as an energy source to substitute for fossil fuels appears to be the most promising strategy.
- In the alternative scenario, where no extra timber is used in construction, the results obtained depend upon what happens to the unused forest biomass.
- The overall effect is highly dependent upon the ratio between the growth rates of the trees and the lifetime of the HWPs.
- Overall negative pool effects (where emissions exceeded storage) could potentially result from the increased use of wood in some situations, but when the energetic and material substitution effects were included then the increased use of wood was always justified (from the perspective of a pool a negative value is viewed as an emission, or loss, whereas from an LCA perspective a negative GWP impact represents sequestered atmopsheric carbon dioxide).

At the product, or building level the claim that carbon storage benefits arise from the use of wood products in the built environment depends upon the embodied emissions being lower than the amount of atmospheric carbon stored in the wood product itself. Many studies have shown that this is indeed the case and that there is a measurable carbon storage benefit. Pingoud and Lehtilä (2002) studied wood products in a Finnish context and estimated that the associated GHG emissions were only 7% of the CO_2 eq. stored in sawn wood products. These percentages rise with the amount of processing that is required for the wood product and are highest for virgin paper products (30-60%), but even in these extreme cases the amount of CO_2 eq. released is lower than the amount stored in the product. Hill and Dibdiakova (2016) conducted an analysis of published EPDs of different wood products and plotted the GHG emissions associated with the processing (modules A1 to A3) compared with the amount of biogenic carbon stored in the material. The results (reproduced in Figure 1) showed that the carbon stored in the timber product always exceeds the embodied GHG emissions.

Kilpeläinen et al. (2014) noted that many studies of HWPs failed to take account of the dynamic nature of carbon exchanges with forests in the context of wood production. In their analysis they calculated the atmospheric impacts for timber production and utilisation in Finnish boreal forests, compared to a reference management regime. The study covered two 100-year rotation periods for a Scots pine (*Pinus sylvestris*) stand.

The model included carbon sequestration in the growing forest biomass (above and below ground), carbon emissions of humus and litter and emissions from the degradation of HWPs. In the managed forest regimes, the forest system acted as a net sink for atmospheric carbon, whereas in the unmanaged scenario the forest become a carbon source after 120 years. However, the wood management and production regimes emitted more carbon to the atmosphere during the first rotation. This situation reversed in the second rotation. Where biomass from forestry residues and end of life timber was examined, no consideration was given to the substitution effect where the biomass replaced fossil fuels.

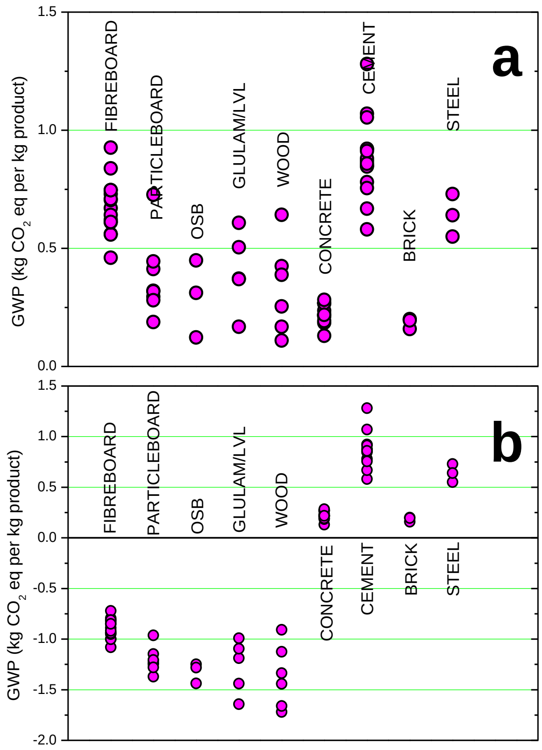

FIGURE 1: A COMPARISON OF THE GWP IMPACT OF DIFFERENT HARVESTED WOOD PRODUCTS WITH CONCRETE, BRICK, CEMENT AND STEEL. IN FIG. 1A THE SEQUESTERED CARBON IN THE WOOD PRODUCTS HAS NOT BEEN INCLUDED, WHEREAS IN FIG. 1B THE SEQUESTERED CARBON IN THE WOOD PRODUCTS IS INCLUDED (FROM: HILL AND DIBDIAKOVA 2016).

The impacts of storing atmospheric carbon dioxide are dependent upon the length of time for which the carbon is removed from the atmosphere (Levasseur et al. 2013). Clearly the reduction in radiative forcing due to the removal and storage of atmospheric carbon dioxide is only of benefit during the time that storage occurs. At end of life, oxidation of the timber products results in release of the stored carbon as carbon dioxide, whereas anaerobic degradation causes the production of methane, a much more potent greenhouse gas. End of life options are clearly important. Kirschbaum (2006) argued that the relatively limited time that carbon is stored in the HWP pool does not provide a significant benefit in terms of mitigating increases in GHGs in the atmosphere and that efforts to reduce energy consumption would provide more benefit. The reasoning behind Kirschbaum's argument is that a decrease in carbon dioxide levels in the atmosphere due to storage in wood products would reduce the concentration gradient between the air and the sea and hence reduce the rate of uptake of carbon dioxide by the ocean. At the end of this temporary storage period, the CO_2 is now released into an atmosphere which has a higher CO_2 concentration than it would be otherwise. The Kirschbaum paper has been criticised, because it disregards the cumulative climate impact and because the results of his analysis are an artefact of his particular perspective (Fearnside 2008, Dornburg and Marland 2008). Dornburg and Marland (2008) and Fearnside (2008) argue that even a temporary storage of carbon in the HWP pool is of benefit because it 'buys time' at a critical period, allowing for society to accumulate capital and develop technology to make a transition to a low carbon development path.

A much better approach to dealing with the carbon storage benefits of using biogenic materials in the built environment is to calculate the carbon storage in a product pool, which is an economic approach to the problem. If the harvested wood products pool (HWP) is considered from a carbon storage point of view, then there is a flow of biogenic carbon into the pool from the forest and there is a flow out of the pool as the timber is oxidized and the carbon is returned to the atmosphere. An increase in the HWP pool therefore provides a benefit during the time that the size of the pool is increasing, thereafter the pool will be in equilibrium with the environment. In other words, there is a sequestration benefit for as long as the carbon flow into the pool exceeds the carbon flow out of the pool. However, there are also other potential benefits arising from the use of timber as a construction material in place of a higher embodied energy alternative. Kirschbaum (2006) did not consider the potential benefits of materials' substitution and fossil fuel substitution if the wood is incinerated with energy recovery at end of life in his analysis. Gustavsson et al. (2006a) pointed out that the carbon stock in wooden buildings is actually less important than the reduction in CO_2 emissions associated with using timber as a construction material when compared to alternatives (substitution effect). They also concluded that there are sufficient timber resources in Europe to allow for a significant expansion of the use of wood for both materials and energy. Lundmark et al. (2014) also found that materials and energy substitution were more important contributions to climate change mitigation than stock change effects. Kirschbaum (2006) noted that there are benefits if the storage of carbon in HWPs is combined with landscape level policies which also include increases in biogenic carbon storage in forests. He further noted that expenditure in short term carbon storage projects is of little benefit without the obligation of the long-term maintenance of biogenic carbon stocks.

In LCA at the product level, the time of storage of atmospheric carbon dioxide has been taken into consideration in the UK PAS 2050 and the European Commission's International Reference Life Cycle Data (ILCD) Handbook. Both of these methods rely on the idea of a 'delayed pulse' of carbon dioxide into the atmosphere and use this to give a credit for the time that the atmospheric carbon is stored in the biogenic material. With PAS 2050, the benefits of carbon storage are calculated on the basis of a weighted time average approach for an assessment period of 100 years. For example, if a bio-derived product containing 1 kg of atmospheric carbon is used in a building for 50 years before disposal by incineration, then the benefit of carbon storage is calculated as (50/100) x 1 = 0.5 kg. A slightly different calculation method is used if the storage period is 25 years or less. The ILCD linear discounting methodology considers biogenic carbon sequestration as a negative value and emissions as a positive value. The carbon credits in biogenic materials arise from the effect of delayed emission over a 100-year assessment period. If emission of 1 kg carbon is delayed for a period of 50 years, this is calculated as (50/100) x 1 = 0.5 kg.

The benefits of atmospheric carbon storage in bio-derived products can only be accounted for if the material is derived from a sustainable production source. For the case of timber products, this means that there has to be regeneration of the forest after felling to produce the timber. If felling of the timber results in land use change (such as conversion to agriculture) then the benefits of atmospheric carbon storage in the HWPs are no longer present and according to the ILCD guidelines this biogenic carbon should be treated as if it were fossil carbon. ISO 14067 (2013) does not deal with the storage of atmospheric carbon in products containing photosynthetically-derived materials, other than stating that removals of atmospheric carbon associated with biogenic sinks should be treated separately in the carbon footprint report and that the storage time period should be reported if applicable.

Brandão et al. (2013) reviewed six methods (including PAS 2050 and ILCD) used for accounting for the impacts of carbon sequestration and the temporary storage and release of biogenic carbon. The paper identified that the benefits of carbon storage are highly dependent upon the time horizon adopted and that this is based upon value judgements rather than having any sound scientific basis. As such, the time frame adopted is informed by policy considerations and the commonly used 100-year period for GWP calculations is based upon the desire to bring about achievable change in a crucial period in the history of humanity. The intention is to change behaviour to a sustainable development trajectory.

Although many studies of carbon storage in harvested wood products have been conducted, there are no commonly recognised methods for determining and reporting this in bio-derived products from a time perspective. PAS 2050 and ILCD give two methods for dealing with the temporal factor, but other approaches have been suggested. The method of Moura-Costa and Wilson (2000) calculates a sequestration-based equivalence factor called the ***Absolute Global Warming Potential (AGWP)***. The AGWP is defined as the cumulative radiative forcing potential for CO_2 of unit mass over a specified time horizon. This is calculated from the following relationship:

$$\boldsymbol{AGWP} = \int_{\boldsymbol{0}}^{\boldsymbol{TH}} \boldsymbol{a_x}\,.\,[\boldsymbol{C}(\boldsymbol{t})]\,\boldsymbol{dt} \qquad \text{(Eq. 1)}$$

Where TH is the time horizon under consideration, t is time, a_x is the radiative forcing due to the presence of unit mass of CO_2 in the atmosphere and C(t) is the concentration of a pulse of CO_2, decaying as a function of time, which is usually expressed in terms of the Bern model. Based upon these considerations, they found that removing 1 tonne of CO_2 eq. from the atmosphere and storing it for 55 years counteracts the effect of releasing a pulse of CO_2 into the atmosphere with a residence time of 100 years. This method allows for benefits greater than 100% if the 55-year storage period is exceeded. Another approach, referred to as the Lashof method, assumes that the storage of atmospheric CO_2 is equivalent to a delayed emission of fossil CO_2, but the carbon tracking is performed in the atmosphere rather than the biosphere (Fearnside 2002). Vogtländer et al. (2014) noted that the ILCD and the PAS 2050 methods both overestimate the benefits of the temporary fixation of carbon dioxide. The PAS 2050 method is a linearisation of the Lashof curve for the first 25 years, but thereafter follows the ILCD linear discounting approach.

Pingoud et al. (2012) developed a method for determining the GWP impact of forest biomass life cycles compared with functionally equivalent alternatives based upon fossil fuels and non-renewable materials, using an extension of that described by Cherubini et al. (2011b,c). In order to determine the change in radiative forcing with time, an impulse model based upon the Bern Carbon Cycle is used. An integral of this decay curve gives the cumulative radiative forcing over a desired time frame; this is called the Absolute Global Warming Potential, as with the Moura-Costa approach. The same behaviour can be assigned to any pulse of CO_2 entering the atmosphere, irrespective of whether it is fossil or biogenic in origin. However, with a pulse of biogenically-derived CO_2, the uptake of CO_2 from new growth in the forest also has to be taken into account; but this is highly dependent upon the biomass re-growth rate. Other factors, such as release of N_2O from fertiliser use, or change in albedo can also be included. In addition to this, the effect of substitution of the harvested biomass, either directly for a fossil fuel, or indirectly by the use of fossil fuels for processing a non-renewable equivalent material, also has to be taken into account (these are the displaced emissions), as well as any GHG emissions due to processing of the biomass to make the functional unit. When all of these effects are taken into account it is possible to arrive at a cumulative global warming payback time of the biomass compared with the functionally equivalent non-renewable-based reference service.

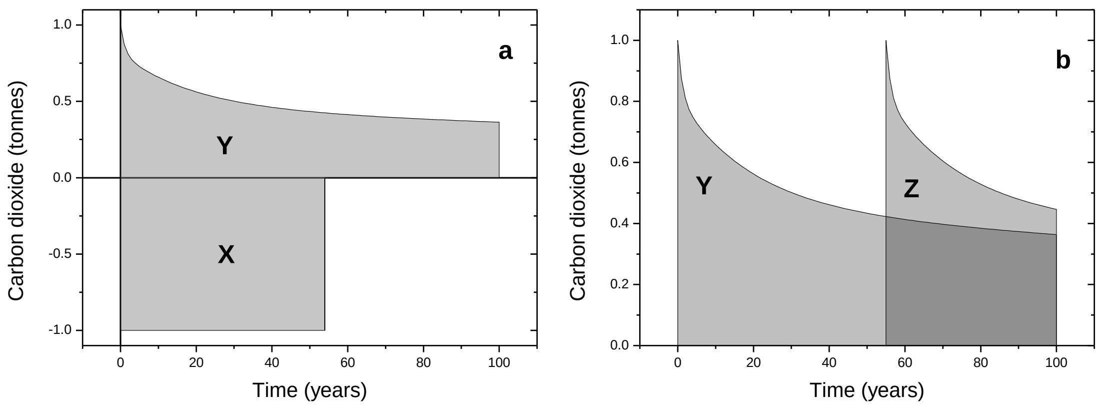

FIGURE 2: ILLUSTRATION OF THE MOURA-COSTA (A) AND LASHOF (B) METHODS FOR CALCULATING THE BENEFIT OF CARBON STORAGE. IN (A) A PULSE OF 1 TONNE OF CARBON DIOXIDE IS RELEASED INTO THE ATMOSPHERE AND THIS DECAYS ACCORDING TO THE BERN MECHANISM. THE TOTAL GLOBAL WARMING POTENTIAL (GWP) OVER 100 YEARS IS REPRESENTED BY THE AREA UNDER THE CURVE. THE SAME TOTAL GWP IS REPRESENTED BY STORAGE OF 1 TONNE OF CO2 FOR 55 YEARS (X=Y). IN (B) THE CARBON IS STORED FOR 55 YEARS AND THEN RELEASED AS A PULSE OF CO2. THE TOTAL GWP IS THE AREA UNDER THE CURVE Z, THE BENEFIT OF STORAGE IS GIVEN BY SUBTRACTING Y FROM Z.

Kendall (2012) noted that the commonly used method of measuring all GHG emissions to the environment and then calculating the GWP based upon IPCC values could introduce distortions, because the timing of the emissions was not taken into account. Kendall describes a calculation called the time adjusted warming potential (TAWP) method, which introduced a weighting value depending upon when the emission occurred during the period of analysis.

Levasseur et al. (2013) examined the problem of GWP impact using a traditional LCA approach without including sequestered carbon, as well as a traditional approach including sequestered carbon, PAS 2050, ILCD and dynamic LCA methodologies. Each approach gave different results, with there being dramatic differences in some cases. It was concluded that the dynamic LCA approach was the preferred method for providing reliable data, although the results obtained were heavily dependent upon the assumptions made and the time horizon considered. The study also examined the problem using a functional unit of a wooden chair, which can give different results compared with studying temporal carbon storage of a pool of harvested wood products (HWPs). A pool of biogenic carbon products does not release carbon to the atmosphere in a pulse, as is the case with a single product, but in a way that is better modelled as a probability distribution (Shirley et al. 2011). Many studies investigating the release of carbon from HWP pools have modelled this behaviour as a single exponential decay (as in the IPCC guidelines) (Pingoud and Wagner 2006), but this does not adequately consider the fact that the probability of a product being taken out of service is related to the age of that item (Shirley et al. 2011). This problem was dealt with by the development of a distributed decay model (Marland and Marland 2003, Marland et al. 2010), which uses a probability distribution to determine how much of production from a particular year decays in any given time interval. This type of model is analogous to the approach adopted by the life assurance industry in actuarial mathematics. This form of modelling is very useful when attempting to adopt a realistic methodology for pricing carbon and assigning a value to the cost of emissions from the HWP pool in the future.

Røyne et al. (2016) reviewed 101 papers which reported on the LCA of forest products. They found that most of the studies excluded the dynamic features of carbon uptake and storage and that climate change impacts from land use change, aerosols and changes in albedo were often not considered. They stated that this could have important implications for decision support. The time frame of the study was found to be a very important consideration affecting the outcome. Depending upon the time perspective of the study, the authors concluded that impact factors other than GWP might be more important. The authors also noted that LCA has to be limited in order to be valuable and that it is important that it provides the correct information in order to be useful to the decision-making process. They recommended that LCA practitioners should:

- Reflect on the decision-making context;
- Use a sensitivity analysis to estimate the influence of different aspects of the study;
- Consider which climate impact aspects are important for the decision-making.

As noted earlier, the best approach is to deal with the issue of biogenic carbon storage at a product pool level, rather than within the LCA of a product. The GWP footprint of a product is reported separately to the sequestered atmospheric carbon that is stored in the product, which then allows for proper accounting of the flows of atmospheric carbon into and out of pools, such as the built environment.

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SUSTAINABILITY

The concept of sustainability originates from considerations of agricultural and forestry production (Hill 2011). A sustainable yield in this context is defined as the maximum amount of a commodity that can be harvested from a piece of land without compromising the ability of that land to provide the same harvest in the future. In terms of agricultural or forestry production, this is an easy concept to understand, although it is not always so easy to quantitatively define where this limit lies.

In the early 1970's concerns about the environment began to increase and the United Nations (UN) hosted a conference on the Human Environment in Stockholm in 1972. At the end of this conference, the Stockholm Declaration on the Human Environment was released, leading to the establishment of the United Nations Environment Programme (UNEP). One of the issues that had been considered at Stockholm was the polarisation of concerns regarding the environment between rich and poor countries. Richer nations put concern for the environment high on the agenda, while for the poorer countries, the over-riding issue was poverty alleviation. In 1974, the World Council of Churches held a conference on the subject of the use of science and technology for human development, where at which they proposed a definition of a sustainable human society and examined what sustainability meant (Dresner 2008). The key issues were:

- There should be an equitable distribution of physical resources between all of the peoples of the planet.
- All people should have the opportunity to participate in social decisions.
- The global capacity to supply food should exceed demand.
- Emissions of pollutants should not exceed the carrying capacities of ecosystems.
- The use of non-renewable resources should never exceed the increase in availability due to technological innovation.
- Human activities should not be negatively influenced by variations in global climate.

These statements show the necessity for a strong concern for human development and human concerns as an essential component in any definitions of sustainability. Without addressing these human needs, environmental concerns were seen as something of a luxury. Fritz Schumacher addressed this tension between the rich and poor nations in his book, *Small is Beautiful*. In this book, Schumacher pointed out that the development strategies that had been employed in developing countries did little to alleviate the problems of the poor in the countryside and were based upon complex imported technology. He coined the term 'appropriate technology' to describe small scale development projects where the technologies employed could be understood and controlled by local people.

Economic growth and sustainability

Some economic growth is necessary in order to allow for the development of the poor without creating social turmoil (Pirages 1977). The issue is one where economic development should be able to occur in developing countries in order to improve the living conditions of people trapped in poverty, yet simultaneously the wealthy should also be able to maintain their standard of living. It is highly unlikely those who already have high standards of living would be willing to sacrifice them. But, is it possible to achieve this? There certainly is potential to improve the conditions of developing countries by encouraging better and fairer trading conditions. In 1980, the International Union for the Conservation of Nature and Natural Resources (IUCN) produced a document entitled *The World Conservation Strategy*. This report emphasised the importance of population pressure, poverty, social inequity and importantly, trading agreements that were disadvantageous towards poorer countries, as being causes of environmental degradation.

In 1983, the UN set up the World Commission on the Environment and Development (WCED) chaired by the Norwegian Prime Minister Gro Harlem Brundtland. In April 1987, the WCED published the famous document *Our Common Future*, which is often referred to as the 'Brundtland Report'. In this publication, the meaning of the term sustainable development was defined as being:

'development which meets the needs of the present without compromising the ability of future generations to meet their own needs'

This is a simple and much quoted (and misquoted) definition, which means different things to different people. The published document contained the detail behind this definition to explain what the implications were:

- Environmental limits – these are not absolute limits, but there are limitations on the use of environmental resources due to the present state of technology and social organization and the ability of the biosphere to be able to deal with the wastes generated.
- Poverty – the elimination of poverty is essential to allowing all people to meet their aspirations for a fulfilling life. Participation – all of the people of the planet should be able to participate in decision- making at all levels.
- Equity – the way in which the resources of the planet are distributed should ensure that the poorer nations of the planet are able to undergo economic growth in order to improve the living conditions of their people.
- Those who lead more affluent lifestyles should modify their behaviour in order to ensure that they are living within the planet's ecological means.

All of the above statements indicate that the issue is not so much one of an insistence that there should be a fair apportioning of resources between generations (inter-generational equity) but rather, that the problems are here and now and therefore we should be concerned with intra-generational equity. It can also be understood that sustainability is about the environment, economics and society; the so-called 'three pillars' of sustainability. It is impossible to have any one without the other two. However, the problem then arises as to how it is possible to reconcile the needs of economic development for the poorer countries and economic growth for the rich countries with concerns about environmental limitations. Indeed, many developing countries often argue that concerns for environmental protection are a luxury for the rich. The Brundtland Report is largely supportive of economic growth and points out that in order to allow the poor and rich countries to achieve some sort of parity it would be necessary to expand the world economy by a factor of 5 to 10. Since this is deemed to be impossible given the resource and environmental limitations, the report insists that these growth requirements will have to be met by efficiency improvements (Factor 10).

Sustainability and capital

Capital is usually associated with money in most peoples' minds, but there are other ways of viewing capital. For the purposes of understanding sustainability, it is useful to define least four types of capital. These are:

- Human created capital – this is what we make, (e.g. buildings, machines and, infrastructure).
- Natural capital – (air, water, soils, forests, natural resources like oil and minerals, ecosystem functioning, etc.).
- Human capital – (investment in health, education, nutrition, etc.).
- Social capital – (the institutions and culture that make a society function).

The way that sustainability is defined considers how these four types of capital are managed. Crucial to this process is the degree to which substitution between these different types of capital is allowed. This gives rise to the concepts of weak, sensible, strong and absurdly strong sustainability (Serageldin and Steer 1994).

With weak sustainability, the total amount of capital remains the same, but no importance is placed on the composition of that capital. The assumption is that all forms of capital can substitute for one another equally. Sensible sustainability requires that, in addition to maintaining the total capital stock, the composition of the capital is important. This means it is necessary to define thresholds below which a capital stock should not fall. The problem lies in defining where these critical limits are. Strong sustainability requires that each of the capital assets should be kept intact. Loss of forest in one area should be compensated for by increasing forest area elsewhere. Profits from exploiting oil resources should be used to develop renewable energy technologies. This view assumes that natural and human-made capital cannot be substituted for one another. Finally, absurdly strong sustainability assumes that non-renewable resources could not be used at all, or at a rate no greater than their geological replenishment rates. Renewable resources can only be used at a rate that does not compromise future productivity.

Population growth and resources

Concerns regarding the optimum population of humans that can be supported by the land stretch back to antiquity. Aristotle was of the opinion that the ideal nation state should keep the size of its population in accord with that of its land. One of the most famous works concerning the consequences of populations outstripping their resources is *An Essay on the Principle of Population* written by the Reverend Thomas Robert Malthus in 1798. The essence of Malthus' argument was that human populations have a natural tendency to grow and that if such growth continues unchecked, then populations will always outstrip the ability of the environment to support them. To Malthus this inevitably meant that the 'lower orders' of society would be forever condemned to exist at a subsistence level. Furthermore, if any attempt was made to improve the conditions of the poor, the inevitable response would be an increase in population, with the result that the same miserable state of affairs would be rapidly reinstated. The essence of the argument was that populations have the property of increasing exponentially (geometrically), but it is only possible to increase the area of land to support that population in a linear manner (arithmetically). The inevitable result is that the population rapidly reaches the limit that can be supported by the land. This is what is known in ecology as the 'carrying capacity' of the ecosystem.

It is an inescapable mathematical property of exponential growth that it will always outstrip linear growth no matter what the linear multiplier is. This is called 'explosive growth' for a good reason. One of the principles underlying Malthus' arguments is that the productive capacity of the land remains unchanged and that any increase in yield can only result from an increase in area. An alternative viewpoint was adopted by David Ricardo who noted that as population increased, progressively lower quality and marginal land would be cultivated. However, the total available area of land is nonetheless, limited.

At this time, the agrarian sector was the focus of almost all economic activity and, because of this, economic theories were based around issues linked to agricultural production. In the 1750's there developed in France a school of economic thought based on the principle that natural resources and especially fertile agricultural land were the basis of material wealth. This movement was known as Physiocracy and is considered to be the first organised scientific economic philosophy (Cleveland 1987). The Physiocrats argued that the economic process was ultimately subject to the laws of nature and not human free will and that if the operation of the natural laws could be understood then it would be possible to maximise social welfare.

The limitation that land area placed upon the size of populations and economies was changed forever by the advent of what Frederick Engels called the 'Industrial Revolution'. At the beginning of the Industrial Revolution, nearly every industrial raw material was supplied by agriculture or forestry. The single most important factor leading to the exponential growth of economies was the exploitation of fossil fuels, substituting for labour, water and wind energy.

Limits to growth – a brief history of the ideas of resource constrained economics

The predicted Malthusian collapse did not occur. The Industrial Revolution brought new technologies and ways of creating wealth that which did not rely on the harvesting of plants and animals. Improved farming methods and the use of fertilisers and, later on, pesticides increased crop yields enormously. Populations in the richest countries of the world began to stabilise. These were all things that Malthus could not have foreseen. However, the principle that exponentially increasing populations or exponentially increasing economies could exhaust their resource base has continued to occupy the thoughts of many. In 1966, the book *Environmental Quality in a Growing Economy* was published (Jarrett 1966). In this book it was argued in an essay '*The Economics of the Coming Spaceship Earth*' by Kenneth Boulding, that during much of the historical era, human impact on the environment has been insignificant due to a low population. Any problems that did occur were localised and could be dealt with by moving to other regions. There was always a frontier, beyond which lay the wilderness and where wastes could be disposed of with impunity. Boulding referred to this as a 'cowboy economy'. However, in the present contemporary era there was no frontier left and it was not possible to dispose of wastes without negative consequences.

In the 1968 book *The Population Bomb*, Paul Ehrlich argued that the exponentially increasing world population would result in catastrophic famines in the 1970's and 1980's. It was also controversially concluded that giving aid to countries that did not take steps to deal with population growth was pointless and would merely encourage further population increases. This is very reminiscent of the position that Malthus adopted and it attracted widespread criticism. Another book pointing out the resource implications associated with exponential growth was *Limits to Growth*, first published in 1972 and updated in 2004 (Meadows et al. 1972, 2004). In this book, the authors reported on the results from a computer model of the impact of population and economic growth upon the Earth's ecosystems. Under various scenarios, the model showed that the human population would overshoot the carrying capacity of the planet before the end of the 21st century. However, the Malthusian perspective that these and other books of this ilk represent has been criticised on a number of grounds over the years. A good overview of this debate is given by Hussen (2004).

Growth without limits? –- the role of technology and the arguments of classical economics

Socialist philosophers in Revolutionary France agreed with Malthus that there was a danger of overpopulation, but they considered that the solutions to this ever-present threat lay in proper social and economic organisation and also in the application of technology to give everyone an improved standard of living. Engels similarly believed that the impacts due a rising population could be compensated for by constant improvements arising from the application of science and technology. This idea has always been a theme within classical economics and although once associated more with the left of the political spectrum, it has become increasingly identified with the politics of the right. The economist Julian Simon has wrote extensively on the ability of human ingenuity to overcome any physical limitations on economic growth and argued that an increasing human population was a good thing, because it meant more brains and hence more ideas (Simon 1996).

Both John Locke and Adam Smith argued that economic self-interest could be the motive force for ensuring the common good and such opinions continue today in the free-market economic philosophy. Put simply, the free-market view of resources takes the following positions. The scarcity (or otherwise) of a resource is reflected in its price on the free market. If demand increases and supply does not, then the price increases. Price increases lead to reduced demand and the price will consequently fall; leading to a self-correcting mechanism. However, if the demand for that a resource does not fall and the price stays high, then this provides the incentive for:

- The exploitation of more marginal resources that which have a higher cost of extraction.
- Increased activity in exploration to go and look for new resources.
- Increased efficiency in the use of that resource through (for example) higher levels of recycling.
- Research into the use of alternative resources.

The argument then follows that it is unnecessary to take any action because the market itself provides the best possible mechanism for dealing with scarcity. This is the self-correcting 'invisible hand' of the market in operation as introduced by Adam Smith (Smith 1776). Hotelling (1931) was able to use classical economic methods to show that there was an optimum rate at which resources should be depleted. Traditional economics asserts that natural capital and manufactured capital are interchangeable (fungible) and that the loss of one can be compensated for by an increase in the other. It assumes that resources are substitutable and that any local constraints upon economic growth can be alleviated through inter-regional trade. As far as equity is concerned, the best way to alleviate poverty is to allow unfettered economic growth so that everyone will get rich ('trickle down' effect) which takes care of intra-generational equity. As for the future people of this planet, the classical economic view is that they are the rich ones and we the poor because of the abundant wealth produced through continual economic growth into the future. Based upon these ideas, there is no need to be concerned about running out of resources because the market is able to respond through price changes which encourage corrective actions. In a classic study of the economic costs of resources, Barnett and Morse (1963) were able to show that these prices of resources had actually declined in real terms between 1870 and 1957; and other studies performed since then, have come to similar conclusions. From the viewpoint of classical economics, this appears to account for the resource limitation argument, but what about wastes?

Externalities

Assuming that the relative abundance of resources is dealt with efficiently by market forces, can the same be said about impacts on the environment? How does the market deal with these? A few examples are given below where the classical economic position would seem to be at odds with that of common sense:

- A range of mountains in a water catchment is deforested in order to derive economic value from the timber. As a consequence, flooding, landslides and soil erosion occur, which inundate low-lying farmland causing huge economic damage.
- Heavy metal pollution from a mine seeps into an underground water body. This subsequently makes its way into the drinking water of a nearby town, but some years after the mine has closed. The public sector has to pay for expensive water treatment facilities. The mining company however which derived economic benefit from extracting the metal ore, but then has moved on, or perhaps gone out of business once the mine was exhausted.
- Coal-burning power stations produce sulphur dioxide which is an atmospheric pollutant causing respiratory disorders in the local population. The cheapest way to get around this problem is to build very high chimneys so that the pollutant is no longer deposited locally. However, the effect is felt hundreds of miles away in a neighbouring country where it causes damage to forests and fisheries at great economic cost.

These are all examples of negative externalities, where the costs associated with the problem are not borne by the originator. Negative externalities are examples of damage occurring to what is sometimes referred to as the 'global commons'. Or, in other words the environment.

The tragedy of the commons

This was the title of an article written by Garrett Hardin, published in the journal *Science* (Hardin 1968). In his essay, Hardin discussed the conflict arising between the desires of individuals to maximise their appropriation of goods or services from a common resource, which is in conflict with the requirement that correct management of this resource requires that each person should limit their exploitation. The example put forward by Hardin is that of a village with a common pasture, where each villager tries to keep as many animals as possible on this land. The rationale behind the argument is that each individual may seek to increase their own well-being by adding one more animal to their herd. This encourages the addition of more and more animals until the unintended consequence of land degradation, with an attendant collapse in both the animal and human populations. In the analysis of the behaviour of the villagers in degrading the commons, Hardin takes issue with the idea of the behaviour of individuals resulting in the common good through the mechanism of the 'invisible hand' as advocated by the Scottish essayist Adam Smith in *An Inquiry into the Nature and Causes of the Wealth of Nations* (1776). Hardin also considers the consequences of an increasing global human population and the burden that this puts upon the

Earth's resources, coming to the conclusion that in order to maximise population, it is necessary to minimise individual consumption, with every person ultimately having to rely upon a subsistence diet. This prognosis is clearly Malthusian in its philosophical foundations and in direct contrast with the no-limits Panglossian hypothesis advanced by Julian Simon, where an increase in population means more minds, more innovation and an increase in everything for all. Which of these two schools of thought is the more correct accurate – if either? One issue that which over-rides the ideas put forward by Simon is that relates to the energy required to process materials and provide goods and services, and this is the ultimate argument put forward by Hardin. Even if an infinite source of energy could be found, the problem would then become one of dissipation. The inescapable downgrading of any energy source to heat means that there is a limit to how much energy can ultimately be used. The emission of waste heat in an exponentially increasing economy would eventually result in the Earth glowing red-hot. There are limits to growth.

Energy, thermodynamics and economics (biophysical economics)

There continue to be arguments about where the material limitations are and how they could affect economic growth. However, there are fields of study which apply thermodynamic principles to economics (Cleveland 1987). The Ukrainian socialist Sergei Podolinsky was the first to attempt a description of economics from a thermodynamics perspective, performing energy analyses of society that were a century ahead of their time (Martinez Alier and Naredo 1982). In communications with Engels, Podolinsky stated that scientific socialism was flawed because it assumed that all resource limitations could be overcome by the application of technology. The German chemist Wilhelm Ostwald incorporated thermodynamics into a theory of economic and social development, noting that the history of civilisation was one of the ever-increasing control of energy for human purposes. Frederick Soddy, the Nobel Prize winning chemist, applied thermodynamic laws to economic systems and was a strong critic of standard economic theory. Soddy argued that economic progress had been made possible by a transition from a solar powered society to one that derived its energy from fossil fuels. He also maintained that while wealth was a real phenomenon that could be linked to physical things, debt was an entirely imaginary concept and it could be created or removed at will. Since wealth was real and linked to physical phenomena, it followed that it was subject to the same laws and therefore inherently limited. However, since debt was not subject to physical laws it could grow without limit and at some point, would outstrip wealth, causing a collapse of the banking system.

The idea of using energy analysis for examining social, economic and political systems was a central pillar of the ideas of the Technocracy movement in the USA during the 1930's. The Technocrats proposed replacing politicians with scientists and engineers who had the necessary knowledge to be able to manage a complex industrial society. Since energy was central to the functioning of society, the Technocrats proposed replacing money with energy certificates. However, interest in the ideas of the Technocrats waned with the advent of the Second World War.

Further developments in the examination of economics from a thermodynamics perspective followed with the work of Howard Odum. He developed the idea of using systems analysis to examine energy flows of human society and nature. Odum (1971) was very aware of the importance of energy quality in a society. He considered that societies with access to higher quality fuels were at an advantage. He also stressed the importance of using appropriate quality energy flows for specific tasks. For example, electricity is eminently suitable for the operation of a computer, but should not be used for heating. He also argued that energy flow was necessary for the creation of wealth and that money flowed in the opposite direction to energy in society. However, while he considered that money circulated in a closed loop in society, energy continuously flowed through it. Odum noted that the large energy flows associated with natural phenomena did not have any associated monetary flows and that consequently environmental services were often abused by the economic system.

Robert Costanza was also concerned with examining the relationship between energy use and economics. He adopted the term ‘embodied energy’ to describe the total energy cost of a product or service and argued that this was a better measure of the true price rather than the market price. He argued that a perfectly functioning free market would give prices that were proportional to the embodied energy content (Costanza 1980). This is a subject of active research by JCH Industrial Ecology.

The contribution to this area by the economist economics by Nicholas Georgescu-Roegen was the subject of a special issue of Ecological Economics (Cleveland and Ruth 1997). Georgescu-Roegen contended that conventional economics confused the concepts of funds and flows, resulting in a misunderstanding of the relationship between natural and manufactured capital, as expressed in what is known in economics as the Cobb-Douglas production function:

$$\mathbf{Q = S\alpha \times L\beta \times F\gamma}$$

Where Q is the quantity of output per unit time, S is the stock of capital, L is the amount of labour employed and F is the flow of natural resources, α, β and γ are fixed parameters. This equation implies that an infinite quantity of output can be obtained if capital (money), labour (energy) or resources are unlimited (Daly 1997). But since both energy and resources are constrained, the application of increasing amounts of capital will lead to the exhaustion of resources, even if energy is obtained from renewable sources (in the case of renewable resources, the limit is the flow, rather than the stock of energy). Others have expressed similar concerns that models of this type, or those invoking the concept of constant elasticity of consumption production functions (Simon 1996) take no account of the physical laws preventing infinite production or infinite substitutability. One of Georgescu-Roegen’s most famous arguments is that any use of matter will inevitably result in its degradation. This is seen as an inevitable consequence of the application of the Second Law of Thermodynamics to matter transformations. This idea is enshrined in Georgescu-Roegen’s ‘fourth law of thermodynamics’, stating that no mechanical work can be performed without the degradation of some matter. This ‘law’ appears to result from a misunderstanding of the concept of entropy and particularly the ‘Carnot efficiency’ of an engine. According to this ‘fourth law’, the result of continually engaging in material transformations is that more and more matter becomes unavailable and cannot be recovered, so that economic systems eventually run down in a manner analogous to the ‘heat death’ of the Universe (Georgescu-Roegen 1971; 1977). However, this idea is wrong, as has been pointed out by several critics (e.g. Ayres 1999).

Matter does not run down to a degraded form in the inevitable manner that energy does when there is a transformation. It is certainly true that in the real world there are almost always going to be losses through friction or any number of other dissipative mechanisms, but this is not inevitable. Furthermore, it is always possible, in principle at least, to recover every last piece of matter that is lost. However, what makes this ‘difficult’ (to understate the problem) is the expenditure of energy required to do this. If matter is highly dispersed, the energy expenditure required to recover it can be stupendous (Daly 1992). This does not mean to say that the issue of material quality is unimportant.

Resources occur in many forms, but it is a general rule that those which are the easiest to exploit are used up first. There can be many reasons for this, such as ease of access, but one very important factor is the material quality. This might be the amount of metal in an ore body, or the energy content of a fossil fuel deposit. The crucial thing is that it takes energy to extract and purify this resource and to transform it into a form that is useful to us. This is where we start to encounter physical limitations due to the universal nature of the Second Law of Thermodynamics.

Given that these limitations exist, is there any way that we can continue with indefinite economic growth without either exhausting resources or overwhelming planetary ecosystems? There are some who think that we can do this by decoupling the process of wealth creation from material and energy consumption. This argument invokes continual improvements in the efficiency with which we use resources as a compensation for increases in wealth. Others meanwhile contend that the exponential nature of economic growth will always overwhelm any efficiency gains and that it is growth itself that is the problem. This leads to the idea of the steady- state model for the economy as advocated by Herman Daly (Daly 1987; 1990). The arguments continually revolve around the issue of whether economic growth is ultimately resource-constrained and if so, can we avoid a catastrophic social and economic collapse?

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TIMBER IN THE BUILT ENVIRONMENT

The Intergovernmental Panel on Climate Change (IPCC) estimates that globally, the building sector is responsible for 40% of primary energy demand and 36% of energy related CO_2 emissions in the industrialised countries. However, this only takes account of operational energy, but does not include embodied energy and greenhouse gas (GHG) emissions associated with the materials used in construction. The built environment sector has significant potential for implementing climate change mitigation actions, through reductions in operational energy and by choosing construction materials with the lowest environmental impact. According to the International Energy Agency, the building sector has one of the lowest GHG mitigation costs. By increasing the use of timber in construction, it is possible to mitigate against climate change with no cost penalty.

Atmospheric carbon is accumulated in living biomass through the process of photosynthesis. Most of the carbon of the terrestrial biosphere is stored in forests, which contain about 86% of the above-ground biogenic carbon and 73% of the carbon stored in the soil. Boreal forests store about one third of the global terrestrial biogenic carbon (Pan et al. 2011, Hovi et al. 2016). Although carbon stocks in global forests are decreasing by 1.1 Gt per year, in most of the countries of Europe, the forest utilisation rate (fellings as a percentage of the annual increment) is less than 100%. This means that the carbon pool in European forests is increasing in size, with the sink being about 365 million tonnes of sequestered CO_2 per year; equivalent to 7% of annual EU emissions (Pilli et al. 2016). With the current rate of timber harvesting in Europe, managed forests will move into older age classes and the net increment of wood material will consequently decline (Nabuurs et al. 2002, Karjalainen et al. 2002). This provides an opportunity for increasing fellings in order to improve the carbon sequestration potential of these forests. This argument does not apply to old growth unmanaged forests and it has been shown that these forests continue to sequester carbon dioxide for a considerable time (Luyssaert et al. 2008). There is also evidence showing that ‘primary’ or ‘frontier’ forests hold more carbon in above- and below-ground biomass compared with plantation forests (Newell and Vos 2012). Continued, or even intensified, harvesting of managed forests will ensure that old growth forests can be dedicated to conservation measures. The best approach will prove to be a ‘mixed’ strategy, where some areas are optimised for timber production and others for biodiversity and amenity benefits (Triviño et al. 2016). Intensively managing forests for production has been shown to be the preferable option from the point of view of climate change mitigation. The biggest benefit from this management strategy comes from the use of the biomass to substitute for fossil fuels and more energy intensive materials (Poudel et al. 2012). The use of biomass in the built environment represents a stable and easily accountable way of storing atmospheric carbon for long periods of time, creating a new carbon pool. Furthermore, the substitution of other building materials which often have a higher carbon footprint brings additional benefits. If forests are not harvested, it is self-evident that no forest products will be produced, which consequently requires their replacement with more energy- and carbon-intensive materials. In addition, the potential for energy production from the by-products of harvesting and processing and wood waste at the end of life cycle, is lost.

The role of harvested wood products in mitigating greenhouse gas emissions has only recently been recognised by the Kyoto Protocol. In 2009, the 15th Conference of Parties of the UN Framework Convention on Climate Change, Copenhagen, it was agreed that HWPs could be included as an additional carbon pool. For the first commitment period (2008-2012), it was assumed that the amount of carbon leaving the harvested wood products' pool every year was equal to the annual inflow (instantaneous oxidation). This means that although a considerable quantity of atmospheric carbon may be stored in the wood products pool, this amount is assumed to be stable over time and there is therefore no net benefit in terms of mitigation potential. For the second commitment period (2013-2020) the carbon accounting included carbon stock changes in the HWP pool.

Although the IPCC recognises the importance of the built environment, its mitigation strategies listed in the fourth and fifth assessment reports are almost exclusively concerned with energy consumption. The use of wood as an example of a low embodied energy material is mentioned, but there is no consideration given to the potential for timber and other plant derived products to act as carbon stores in the built environment. Furthermore, the use of mitigation strategies associated with forestry is only concerned with bioenergy and does not discuss the carbon storage potential of timber products. However, the Conference of the Parties to the Kyoto Protocol in Copenhagen in 2009 did recognise the importance of including timber products as carbon sinks and the 2011 Durban and 2012 Doha conferences stated that carbon stored in wood products should be integrated into reporting procedures.

The utilisation of HWPs in long-life products allows for the carbon storage benefits of timber to be extended beyond the forest (Gustavsson et al. 2006, Liu and Han 2009, Lehmann and Fitzgerald 2012, Kremer and Symmons 2015). Furthermore, the benefits of using HWPs are not just limited to carbon storage but also because they can substitute for materials which have a higher embodied energy and/or global warming potential. These substitution effects may have a greater impact on climate change mitigation compared with the carbon storage benefits, but can be more difficult to determine (Marland et al. 1997, Miner and Perez-Garcia 2007, Matsumoto et al. 2016).

The question of how to account for the storage of carbon in biogenic products has been the subject of much debate and the issues that this present have still not been fully reconciled. Between 2006 and 2009, the LCA community discussed how to construct methods to calculate biogenic CO_2 and eventually came to the conclusion not to calculate it at all (Vogtländer et al. 2014). The reason for this decision was that the stored biogenic carbon will eventually re-enter the atmosphere at the end of life of the product. However, more recently it has been realised that the storage of biogenic carbon does have a role to play in climate change mitigation and this needs to be recognised, but there is no agreed way of accounting for this in LCA. Gustavsson and Sathre (2011) note that much of the methodology developed for determining the GWP impacts associated with the production and use of biofuels is also useful for the comparison of HWPs with alternative construction materials. The difference is that the harvested material is not immediately oxidised and the methodology has to take account of this 'delayed emission'.

The storage of atmospheric carbon in long life timber products has a greater climate change mitigation benefit, compared with immediate oxidation for energy recovery (Stewart and Nakamura 2012). Furthermore, the use of bioenergy facilities to utilise the inherent energy of processing and harvesting residues provides additional climate benefits not included in the forestry chapter of national GHG inventories. There are several proposed methods for accounting for the storage of biogenic carbon derived from atmospheric CO_2 in long-life timber products, but none has gained wide acceptance (Tellnes et al. 2017). The issues that need to be addressed are:

- the eventual loss of the stored atmospheric carbon dioxide back to the atmosphere,
- a way of properly accounting for the time of storage of the sequestered carbon,
- a way of differentiating between biogenic carbon released to the atmosphere and fossil carbon released to the atmosphere,
- comparing the effect of storage of the atmospheric carbon in HWPs compared with the fate of the same carbon if left in the trees in the forest for the same time period (the counterfactual).

There are considerable difficulties with determining the atmospheric carbon storage benefits in timber products in LCA and it is preferable to record the GWP and sequestered carbon separately. This allows for the flows of sequestered carbon into and out of the built environment carbon storage pool to be calculated separately from the substitution benefits of replacing higher impact building materials with timber equivalent functional units. The calculation of such flows requires the availability of accurate inventory data regarding the amount of timber used in buildings and the number of new-builds, as well as the quantity of material which is used in retrofits and refurbishment (Robson et al. 2014). Furthermore, accurate data is needed regarding the lifetime of buildings in order to model the flows of carbon out of the HWP pool. This data is not readily available.

The HWP pool is not a sink and eventually the sequestered carbon will exit the pool with different potential fates. Oxidation will return the sequestered carbon to the atmosphere as carbon dioxide, whereas disposal to landfill will result in a proportion of the biomass being converted to methane, with the residual carbon remaining in the ground. The amount of sequestered carbon residing in the HWP pool depends upon the product lifetime, which varies considerably depending upon application and ultimate fate. The amount of sequestered atmospheric carbon residing in the pool can be estimated by measuring the stocks in the pool or by measuring the inflows and outflows into and out of the pool (flow method). However, in practice, both methods are problematical to apply to the built environment. It is possible to determine the amount of timber used in different buildings and hence calculate what is in the new building stock.

The currently recommended procedures for determining sequestered carbon stocks are those outlined in the '*Revised Supplementary Methods and Good Practice Guidance Arising from the Kyoto Protocol*' (2013). This states that the emissions associated with the removal of HWPs shall be accounted for by the country and that imports cannot be counted. Although the national statistics for harvested wood products can be used for estimating the amount of sequestered carbon entering the wood products' pool, there is a lack of data regarding the rates at which HWPs exit the pool. The recommended methods for modelling the loss of carbon from the pools are:

- Tier 1 – instantaneous oxidation is assumed, which is applied to scenarios such as harvested wood used for energy.
- Tier 2 – first order decay, using default half-lives for three product categories (sawnwood 35 years, wood-based panels 25 years, paper and paper-board 2 years).
- Tier 3 – country specific methods, which can include flux data methods (where data is available), half-lives with country-specific data, other decay functions in combination with default values or country-specific data.

The two main approaches for modelling carbon loss from products pools are the single pool and distributed pool methods. In the single pool approach (as used in the Kyoto Protocol reporting methodology), the rate of loss of carbon from the pool is a function of the total amount of material in the pool, but does not depend on the product lifetimes, whereas with the distributed pool method has a separate pool in the product category for each year of production and the rate of loss is calculated for each year and summed over the period of study, with the rate of removal determined by product age rather than the total stock in the pool. A range of models have been developed to determine the rate of loss of carbon from product pools.

The most common approach to modelling the loss of carbon from the pool is to use assumed half-lives, but this use of an exponential function to model building lifetimes is not representative of reality and it is better modelled as a probability distribution (Shirley et al. 2011). Other approaches have included linear (Winjum et al. 1998), Weibull (Thompson and Matthews 1989, Karjalainen et al. 1994), normal (Muller et al. 2004), and gamma distributions (Klein et al. 2013).

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