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Recycling wooden pallets: A case study of biochar production

Wednesday, 19th July - 14:00: Bio-Based Production

Mr. Pasindu Samarakkody [1]***, Dr. Thuy Nguyen*** [1]***, Mr. Ben Hetherington*** [1]***, Dr. Sazal Kundu*** [1]***, Mr. Jonas Bengtsson*** [1]***, Mr. Christian Keel*** [1]

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Abstract

Pallets play an important role in the supply chain, used in the various sectors of food and beverage, manufacturing, construction, retails, etc. The global market is predicted to reach USD 88.69 billion by 2026 (Fortune Business Insights™, 2019). Typically, pallets are manufactured from different materials including wood, plastic, wood composites, or metal. Wooden pallets account for 86.5% of the global market share, by material type (Fortune Business Insights™, 2019). Since the demand for using wooden pallets in the global supply chain is increasing, the fate of wooden pallets after the service lifetime should be taken into consideration. The end-of-life (EoL) scenarios in most studies demonstrated the benefits of reselling pallets in the secondary market or energy generation (Alanya-Rosenbaum et al., 2021; Bengtsson & Logie, 2015; Deviatkin et al., 2019; Khan et al., 2021). This study investigates the environmental impacts of three recycling scenarios for waste pallet including, (i) firewood, (ii) recycling waste pallets into mulch, and (iii) recycling waste pallets into biochar. The cost per unit of greenhouse gas emissions reduction will be analysed. This study provides the comparison of environmental performance among three selected recycling methods for waste wooden pallets.

Reference:

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Life Cycle Assessment of Biomass Co-firing in a Coal-fired Power Plant

Wednesday, 19th July - 14:00: Bio-Based Production

Ms. Anna Boyden [1], Mr. Fabiano Ximenes [2], Mr. Tim Grant [1]

1. Lifecycles, 2. NSW Department of Primary Industries

Australia's energy production system is heavily reliant on fossil fuels, contributing significantly to emissions of greenhouse gases nationally. Producing electricity by cofiring coal with biomass offers an opportunity to reduce the climate change impacts of electricity production. When this biomass is grown sustainably, the carbon dioxide emitted during its combustion is reabsorbed during the growth of new biomass, and hence has no net effect on climate change emissions. While the environmental savings associated with biomass cofiring have been characterised previously, studies have not focused on specific locations, and have only considered limited biomass feedstocks. The aim of this paper is to investigate the use of biomass-derived fuel for cofiring in an existing coal-fired power station in the Australian context. This involved quantifying the environmental savings associated with the introduction of a 5% fraction of biomass, as well as a comparison of these impacts to other electricity production systems presented as alternatives to coal-fired power plants. Life cycle assessment is used for both components of the analysis.

The results show that switching to cofiring with 5% biomass saves between 28.0 and 33.1 kgCO_2-eq per MWh of electricity generated. The source of biomass feedstock can affect the results, but all feedstock types were shown to create climate benefits. While land occupation increases when energy crops are utilised as a feedstock, this is typically marginal land with little competition. Other forms of renewable energy may create greater savings for the land use available, however if economics are viable and waste feedstocks or land for energy crops are available, the implementation of biomass cofiring in existing coal facilities reduces climate change impacts. The use of carbon capture and storage or carbon capture and use can further reduce impacts and has the potential for carbon sequestration.

Exploring pathways to decarbonise the electricity supply in Bangladesh

Wednesday, 19th July - 14:00: Bio-Based Production

***Dr. Nazmul Islam* [1], *Prof. Mohammad Mosharraf Hossain* [2], *Dr. Murray Hall* [3], *Dr. Nawshad Haque* [3]**

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Decarbonization of electricity supply is one of the priorities of many countries due to the global quest towards net-zero, and sustainable development goals (SDGs): affordable and clean energy (SDG7), sustainable cities and communities (SDG11), responsible consumption and production (SDG12), and climate action (SDG13). Bangladesh also wants to contribute towards global climate change mitigation with a commitment to reduce ~29% (unconditional) and ~58% (conditional) of greenhouse gas (GHG) from electricity generation compared to Business-as-usual (BAU) by 2030. Electricity contributes ~35% of industrial energy consumption after natural gas (43%); and for the residential sector, it is ~25% after biomass-based energy (55%). So, decarbonizing the electricity supply can be one of the keyways to ensure a nationwide reduction of GHG emission. This study focuses on Bangladesh, the world's second-biggest clothing exporter, after China. It explored the environmental implications of decarbonizing the electricity supply. Different scenarios are considered with different technology mixes and GHG reduction targets till 2050. This research in addition to GHG emissions also presents the life cycle impacts in terms of abiotic depletion, ozone layer depletion, acidification, eutrophication, photochemical oxidation, human, and ecotoxicity. The LCA analyses performed in this study enabled a comparative analysis of current and forecast energy systems by identification of the main sources of environmental impact. The results showed that transitioning to a higher contribution of renewables can be significant for the overall reduction of the life cycle impacts compared to BAU and enable the country to achieve the national commitment towards climate change mitigation.

Testing the greenhouse gas abatement of bio-based production from agricultural residues

Wednesday, 19th July - 14:00: Bio-Based Production

Dr. Marguerite Renouf [1], ***Prof. Peter G. Grace*** [1], ***Mr. Hakan Bakir*** [1], ***Dr. Naoya Takeda*** [1], ***Dr. Johannes Friedl*** [1]

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This paper compiles general learnings from a number of projects that have used LCA-based GHG accounting to test the GHG emissions reduction when agricultural residues are used as feedstock for bio-based production. In particular, it considers the potential 'leakages' (inadvertent consequences) when residues are removed from agricultural systems or existing uses, and whether this compromises the emission reduction status of the bio-based products.

The agricultural residues considered include those currently retained on, applied to, or burn in field, value added for animal fodder or bedding, or used as fertiliser. A consequential GHG accounting approach used in emission reduction crediting schemes was used to estimate changes the annual GHG emissions between the 'baseline' (existing fate of residues) and the 'project' (diversion of residues to a bio-based production process). The study also drew on agronomic modelling that simulates the changes in N_2O emissions and soil organic carbon (SOC) in cases where residues are removed from land.

Three sources of potential leakage were identified. For residues removed from land in some regions, increased CO_2 emissions from SOC loss can be greater than reduced N_2O emissions, resulting in an overall increase in on-farm GHG emissions. For residues with valuable nitrogen (N) content, urea-N replacement when they are removed also increases on-farm emissions. For residues that are already value-added into products, the lost displacement abatement when they are diverted away from these uses can also be a leakage. This challenges the assumption that agricultural residues, often considered to be 'wastes', come free of embodied or consequential impacts when used as inputs to production processes.

Ranking the environmental benefits and impacts of different biorefining options for food waste – a case study of citrus waste

Wednesday, 19th July - 14:00: Bio-Based Production

Ms. Roanna Jones **[1]*****, Dr. Marguerite Renouf*** **[2]*****, Dr. Robert Speight*** **[1]*****, Dr. Jo-Anne Blinco*** **[1]*****, Prof. Ian O'Hara*** **[1]**

1. *Queensland University of Technology,* ***2.*** *Lifecycles / QUT*

Managing food loss and waste (FLW) effectively is a top priority, according to the United Nations Sustainable Development Goal 12.3. Growing, storing and transporting food requires significant resources, such as energy, water, land and fertilisers. Consequently, the environmental impacts embedded in FLW are very large. Biorefining represents a significant opportunity to displace some of the environmental impacts associated with FLW, by processing FLW to extract or create new products. The goal of this study was to assess the life cycle environmental impacts of eight alternative management options for a case study food waste (citrus processing waste), to compare the relative environmental attributes and ranking of biorefining processes compared to traditional FLW management options.

A consequential, partial LCA was conducted, to assess the gate-to-gate environmental impacts of eight management options for citrus processing waste. The assessed scenarios included landfilling, composting, feeding to livestock, incineration with energy recovery, anaerobic digestion, solvent extraction of pectin, solvent free microwave extraction of essential oil and fermentation to produce lactic acid. The analysis also considered the displacement effects of the substituted products.

The LCA results were normalised, weighted and aggregated, to rank the resulting overall environmental score for the scenarios. The results suggested that feeding to livestock offered the best environmental outcome. Biorefining processes that produce energy products had low environmental impacts, while those that produced non-energy products tended to have higher environmental impacts. These results generally did not align with the priority order of the Waste Hierarchy.

This research is supported by an Australian Government Research Training Program (RTP) Scholarship. The work has been supported by the Fight Food Waste Cooperative Research Centre whose activities are funded by the Australian Government's Cooperative Research Centre Program.

Bio-Based Production Extended Abstracts

Life Cycle Assessment of Biomass Co-firing in a Coal-fired Power Plant

Anna Boyden[a*], Timothy Grant[a], Fabiano Ximenes[b]

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Main topic/key words: bioenergy, co-firing, life cycle assessment

1. Introduction

The use of biomass for bioenergy is one of the key elements of IPCC modelling to limit future global warming [1]. Bioenergy utilises living organic material to produce heat, electricity, biogas and liquid fuels, and unlike energy derived from fossil fuels, it is renewable. Furthermore, greenhouse gas emissions occurring from the conversion of biomass to energy are biogenic and hence do not lead to a net increase in carbon dioxide concentration in the atmosphere. In Europe, bioenergy is actively being considered in legislation such as the European Green Deal [2] and Directive (EU) 2018/2001 on the promotion of the use of energy from renewable sources [3]. In Australia, the use of biomass has the potential to generate a significant amount of electricity [4], greatly reducing the climate change impacts currently associated with this industry. However, Australia lags behind Europe in terms of the development of legislation promoting and regulating the use of bioenergy. To keep global warming to 1.5 degrees, in line with the goal of the Paris Agreement, Australia will need to keep 95% of its coal, 35% of its fossil methane gas, and 40% of its oil in the ground [5]. Given the nation's heavy reliance on these fuel sources for energy [6], it is a necessity that the use of alternative fuels is greatly expanded. While over 60% of Australia's coal-fired electricity generation will retire by 2040 [7], these closures must occur earlier in order to limit global warming to 1.5 degrees [8].

Co-firing biomass with coal provides a way of reducing the greenhouse gas emissions of coal-fired power plants in Australia in the case that the plants are not retired earlier than planned. This is an effective way of reducing greenhouse gas emissions when biomass is sustainably sourced, and can make use of current waste streams. It also provides further opportunities for combination with carbon capture and storage (BECCS) or use (BECCU), resulting in net negative greenhouse gas emissions [9]. The fuel sources used for co-firing can be obtained from forestry operations residues, sawdust and offcuts from sawmill operations, dedicated biomass crops, agricultural residues, or urban wood waste, among others. While several co-firing trials have been performed in Australia [10], only one plant is currently operating with biomass co-firing.

Existing studies on biomass co-firing have characterised its environmental benefits, showing reductions in climate change and other impact categories, both in Australia [9] and other locations [11-14]. Other co-firing studies have focused on the feasibility of implementing co-firing systems in specified locations [15, 16], based on economic feasibility, logistics, and biomass fuel availability. A 2011 study found that two key barriers to the adoption of co-firing are: resistance due to environmental concerns; and lack of information on biomass resource availability [17]. While biomass availability has since been further investigated [18, 19], the opportunity exists to address the barriers related to environmental concerns, especially for specific locations.

The overarching aim of this paper was to investigate the use of biomass-derived fuel for co-firing in an existing coal-fired power station in the central tablelands of New South Wales (NSW). This region was selected due to its high reliance on coal for electricity [6], and the existing biomass availability assessment for the region. A life cycle assessment (LCA) was performed to quantify the environmental benefits of co-firing with various biomass feedstocks compared to coal, and these impacts were compared to other electricity production systems.

2. Material and methods

This study follows the framework and principles of LCA as described in the international standard ISO 14040 [20].

2.1 Goal and scope

The goal of the study was to evaluate the environmental impacts and benefits associated with co-firing of biomass in a coal-fired power plant in NSW. This includes the identification of hotspots and leverage points for improving environmental outcomes. Mt Piper power station is used as a case study, with the incorporation of site-specific electricity generation and emissions data. The power station is located in the Central West region of NSW and it consists of 2x700 MWe coal fired units. It is currently fuelled using black coal sourced from mines in the local area, with a current planned closure of 2040.

A baseline scenario of electricity production through coal-fired generation at Mount Piper power station is compared to several co-firing scenarios, encompassing varying feedstocks and processing options. The results are presented as the difference between the two scenarios, representing the environmental effects of a shift from the current baseline scenario to a co-firing scenario.

2.1.2 Functional unit, impact categories and system boundary

The functional unit chosen is 1MWh of electricity production at Mt Piper power station. The impact categories assessed are: global warming potential (GWP100a, [21]), abiotic depletion (CML-IA V4.8, [22]), particulate matter (IMPACT World+, [23]), water scarcity (Pfister water depletion characterisation model, [24]), and land use (ReCiPe 2016 v1, [25]). The system boundary for this LCA is shown below in Figure 1. Forestry operations and production of forestry products are also excluded from the system boundary, as the residues from these processes are generally considered wastes and are not allocated any of the impacts of production.

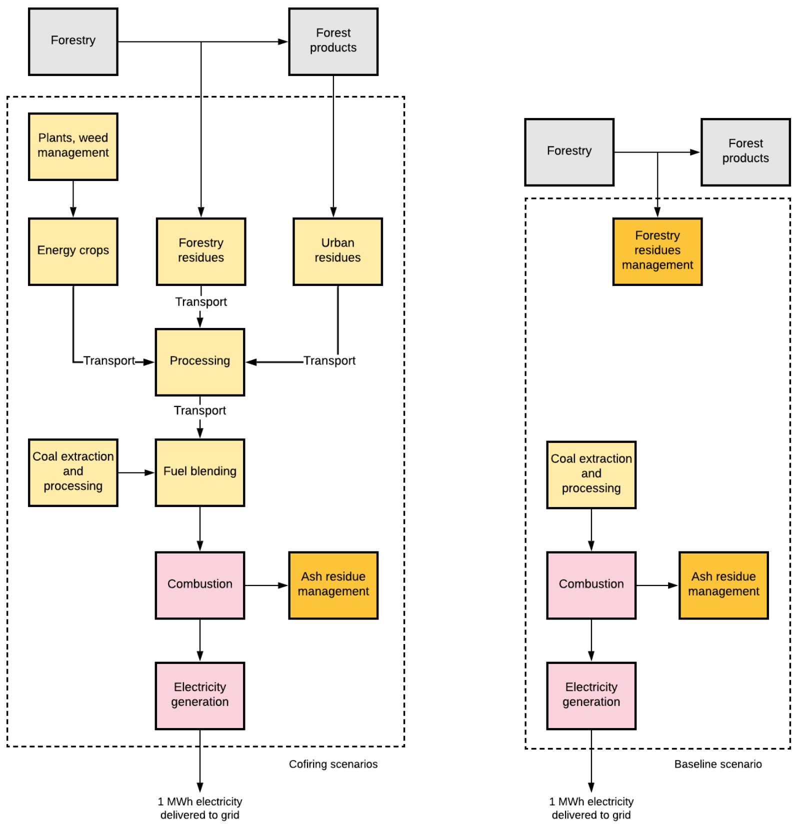

Figure 1. LCA system boundary

2.2 Scenarios, assumptions and data sources

Three different biomass feedstocks are assessed: dedicated energy crops, forestry residues, and urban residues. The dedicated energy crops are modelled using data from the species River red gum, since this species is currently performing particularly well in energy crop trials in NSW [26]. The forestry residues are modelled with data from the species Radiata pine, since there are large areas of Radiata pine plantations in the vicinity of the power station. The urban residues are assumed to be mixed woody waste acquired through municipal solid waste collection. The key characteristics of coal and the three biomass feedstocks are summarised in Table 1.

Table 1. Chemical properties of cofiring feedstocks

Scenario	**Species modelled**	**Energy content (MJ/kg)**	**Green density (kg/m^3)**	**Ash content (%, air dried basis)**	**Moisture content, as received (%)**
Baseline – 100% coal	n/a	23.1 [27]	n/a	n/a	n/a
Co-firing – 5% energy crops	River red gum (*Eucalyptus camaldulensis*)	17.14 (air-dried basis, *Eucalyptus globulus* used as proxy [28])	600 [29]	0.6 (*Eucalyptus globulus* used as proxy [28])	50 (assumption)
Co-firing – 5% forestry residues	Radiata pine (*Pinus radiata*)	18.62 (air-dried basis [28])	n/a	1 [28]	50 (assumption)
Co-firing – 5% urban residues	Mixed	17.18 (air-dried basis [28])	n/a	1 [28]	12 [28]

2.2.1 Baseline model

The baseline coal firing scenario is based on combustion at Mt Piper power station. It is assumed that Clarence Colliery supplies the power station with coal and transport of coal to the power station is 36km by rail. The model is obtained from the AusLCI 1.34 [30] database and modified with data specific to Mt Piper power station and Clarence Colliery. Foreground data for Clarence Colliery is obtained from their 2019 Annual Review [31] for the reported diesel consumption, electricity consumption, water consumption and fugitive methane emissions. Emissions at both Clarence Colliery and Mt Piper power station are sourced from the National Pollutant Inventory [32]. The energy density of coal is assumed to be 23.1MJ/kg [27].

2.2.2 Biomass cofiring model

Key data sources drawn upon in the biomass co-firing model are outlined in Table 2.

Table 2. Foreground data sources for biomass co-firing model

Process	**Data source**	**Notes**
Production of seedlings for energy crops	Morales et al. [33]	
Growth of energy crops	Morales et al. [33]	Inventory modified to include monthly mowing at crops site (determined through site visit to NSW energy crop trials)
Chipping of energy crops	Morales et al. [33]	
Biomass processing	Röder et al. [34] Bergman [35]	Inventory adjusted for relative moisture contents of different feedstocks

The following modelling choices and assumptions are applied to the model:

- A biomass fraction of 5% by mass.
- No impacts are allocated to the initial production of residue feedstocks, i.e. only the impacts associated with collecting and processing the residues are considered.
- The alternative fate of the forestry residues is 60% burning at forest, with the remaining portion left in forest.
- The degradation of forestry residues left in forest results in no net soil organic carbon change, and that the carbon dioxide emitted is biogenic.
- The production of energy crops occurs on land previously used for high-intensity grazing, and results in an increase in soil organic carbon. This increase is calculated assuming a baseline value of 55 t C ha^{-1} (warm temperate moist climate zone [36]) and a stock change factor of 0.9 (high intensity grazing [37]). It should be noted that calculations of stock changes in soil organic carbon contain high uncertainty. While it is more likely that unproductive land is targeted for crop production, this would result in larger carbon sequestration benefits and hence high-intensity grazing has been used as a conservative assumption.
- The alternative fate of urban residues is composting (i.e. through application of mulch), and 10% of the carbon is retained, resulting in a carbon sequestration effect [38].
- Carbon dioxide emitted during combustion of biomass is biogenic.
- The combustion of biomass is assumed to emit no sulphur oxides. Due to the low percentage of biomass in co-firing, it is assumed that all remaining emissions during combustion of the coal-biomass mix are approximately equal to the 100% coal baseline. This is supported by the underlying assumption that emissions at power plants (particularly nitrogen oxides and particulate matter) are more closely related to the equipment used than the fuel itself [39].
- The biomass feedstock burns with the same energy efficiency as coal, measured in primary energy terms to delivered electricity. The generation efficiency used for both scenarios is 37% [40] [41].

Four different processing scenarios are assessed, based on likely processing pathways. These are:

1. Drying of biomass by combustion of a portion of the biomass itself, grinding, and pelletisation
2. Drying of biomass with LPG, grinding, and pelletisation
3. Drying of biomass with waste heat at power plant, grinding into wood flour
4. Drying of biomass with LPG, grinding into wood flour

Processing scenario 1 is assumed for the main results of the study in section 3.1, and the other scenarios are explored in section 3.2.

2.2.3 Background data

Background data for electricity, transport, water, coal, diesel, wood chipping, fertilisers, pesticides, and fungicides are obtained from the AusLCI database v1.34 [30]. The emissions profile from burning of biomass at forest is obtained from the National Pollutant Inventory [42].

3. Results

The results of the study include the environmental impacts of a shift from the baseline scenario (100% coal) to three biomass co-firing scenarios. If this impact is negative, this represents a benefit – i.e. a saving in impacts when compared to the baseline. If the net impact is positive, this indicates that a switch from the baseline to co-firing results in net detrimental effects.

3.1. Main results and contribution analysis

The overall results are displayed in Table 3, showing that switching to co-firing with 5% biomass saves between 28.0 and 33.1 $kgCO_2$-eq per MWh electricity generated. Based on the annual generation at Mt Piper power station [43], this equates to savings between 193 and 228 $ktCO_2$-eq/year, depending on the biomass input. All biomass feedstock options result in overall benefits on climate change, particulate matter, fossil fuel depletion and water scarcity impacts. The energy crop biomass feedstock results in increased land use, while the two residues result in a decrease in land use. Other than the land use impact category, there is little variation in benefits across the three feedstock types.

Table 3. LCA results, relative to baseline scenario, per MWh

Impact category	Unit	Dedicated energy crop	Forestry residues	Urban wood residues
Climate change	kg CO2-eq	-29.6	-33.1	-28.0
	%	-3.3%	-3.7%	-3.1%
Particulate matter	g PM2.5	-4.4	-18.1	-11.2
	%	-1.2%	-5.1%	-3.1%
Fossil fuel depletion	MJ NCV	-314	-360	-334
	%	-3.2%	-3.6%	-3.4%
Water scarcity	L-eq	-1.6	-8.2	-8.6
	%	-0.1%	-0.7%	-0.7%
Land use	m2a crop-eq	9.61	-0.30	-0.29
	%	111.9%	-3.5%	-3.3%

Breaking down the climate change results by category shows that the majority of benefits occur as a result of the emissions savings from the reduced combustion of coal (Figure 2). Although the biomass processing causes a climate change impact of approximately 4 kg CO_2-eq/MWh, it is outweighed by the benefits from emissions savings. The remaining categories have minor effects on the impacts. The forestry residues option shows slightly larger benefits than the other feedstocks, due to the slightly higher energy content of pine wood. The urban residues have a slightly larger impact than the other options during the production phase. This is because urban residue composting – which has a carbon sequestering effect – is being avoided through the switch to biomass co-firing.

Although there is a 5% replacement with biomass, this only results in approximately 3.4% reduction in impacts. This is partly due to the added emissions associated with the biomass processing, but mostly due to the lower energy content of the biomass options when compared to coal.

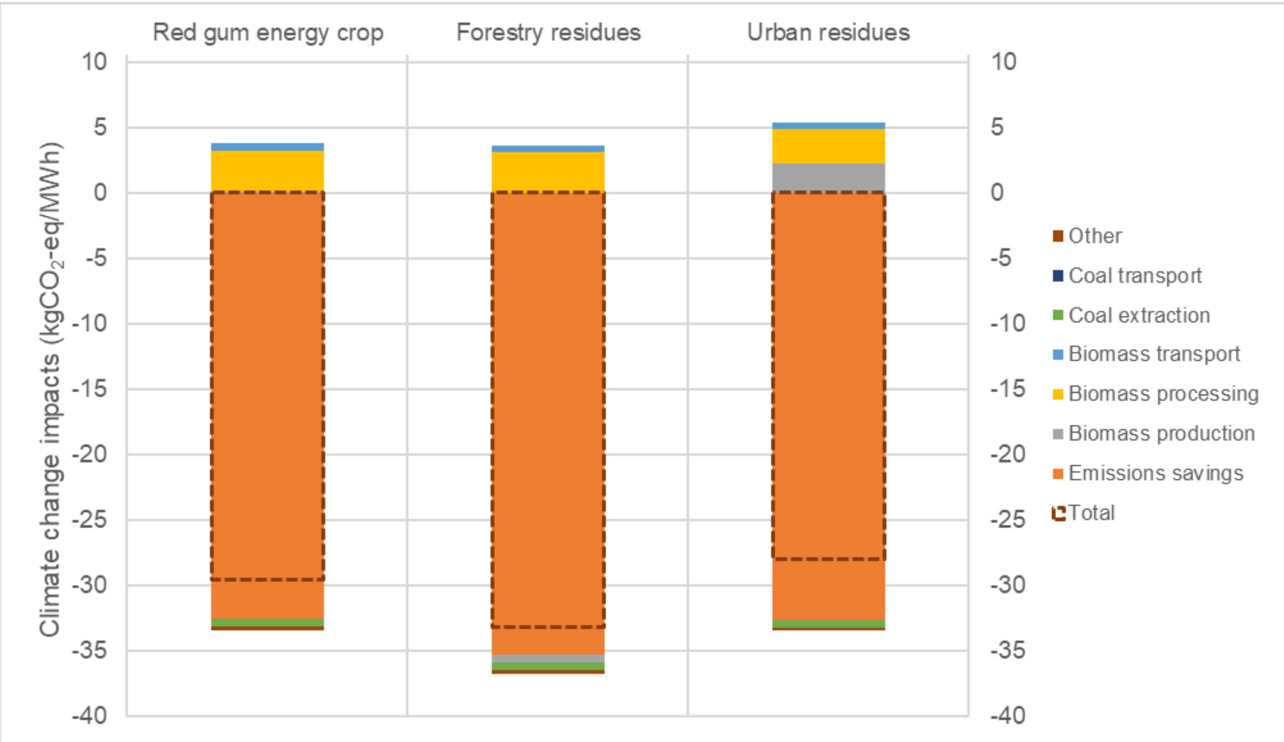

Figure 2. Net climate change impacts, hotspot analysis

3.2. Effect of biomass processing

The base assumption was that the biomass was processed into pellets using biomass as the fuel source for drying. Processing into wood pellets is required when the biomass must be transported longer distances, while grinding into wood flour is sufficient if feedstock is close to the power station itself. The three other processing options assessed were: pelletisation with LPG drying, wood flour processing with LPG drying, and wood flour processing with waste heat drying.

The results (Figure 3) show that for climate change, the largest benefits of biomass co-firing occur with wood flour processing and waste heat drying, closely followed by pelletisation with biomass drying. Drying with LPG reduces the climate change benefits by 2% for urban residues, and an average of 14% for the remaining biomass feedstocks, when compared to the alternative (biomass drying for pellets and waste heat drying for wood flour).

For particulate matter (Figure 4), there is more variation between the biomass options. The results show that pelletisation with biomass drying results in the least savings overall, with the remaining processing options showing comparable results. Of the biomass feedstock options, forestry residues result in the largest savings across all processing options, due to the savings in forestry residue burning.

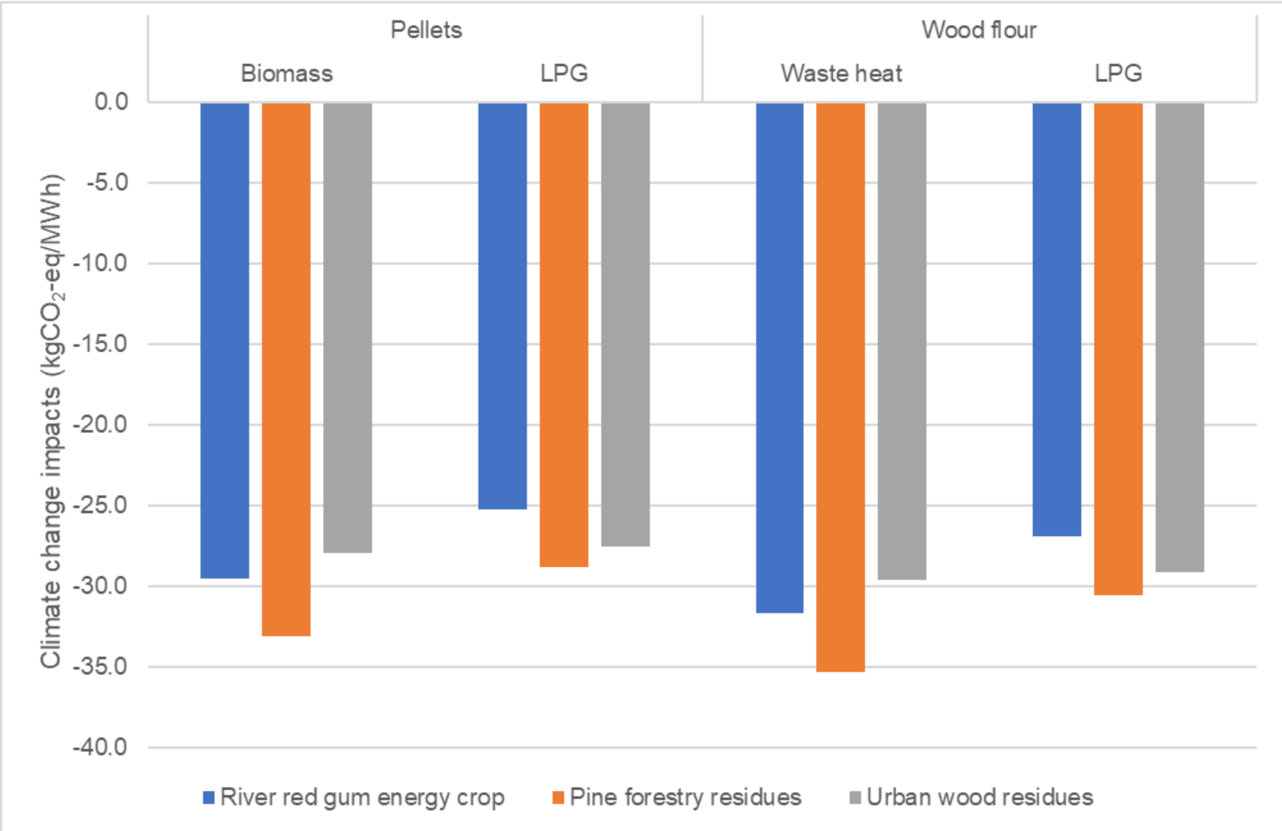

Figure 3. Net climate change impacts, effect of processing choice

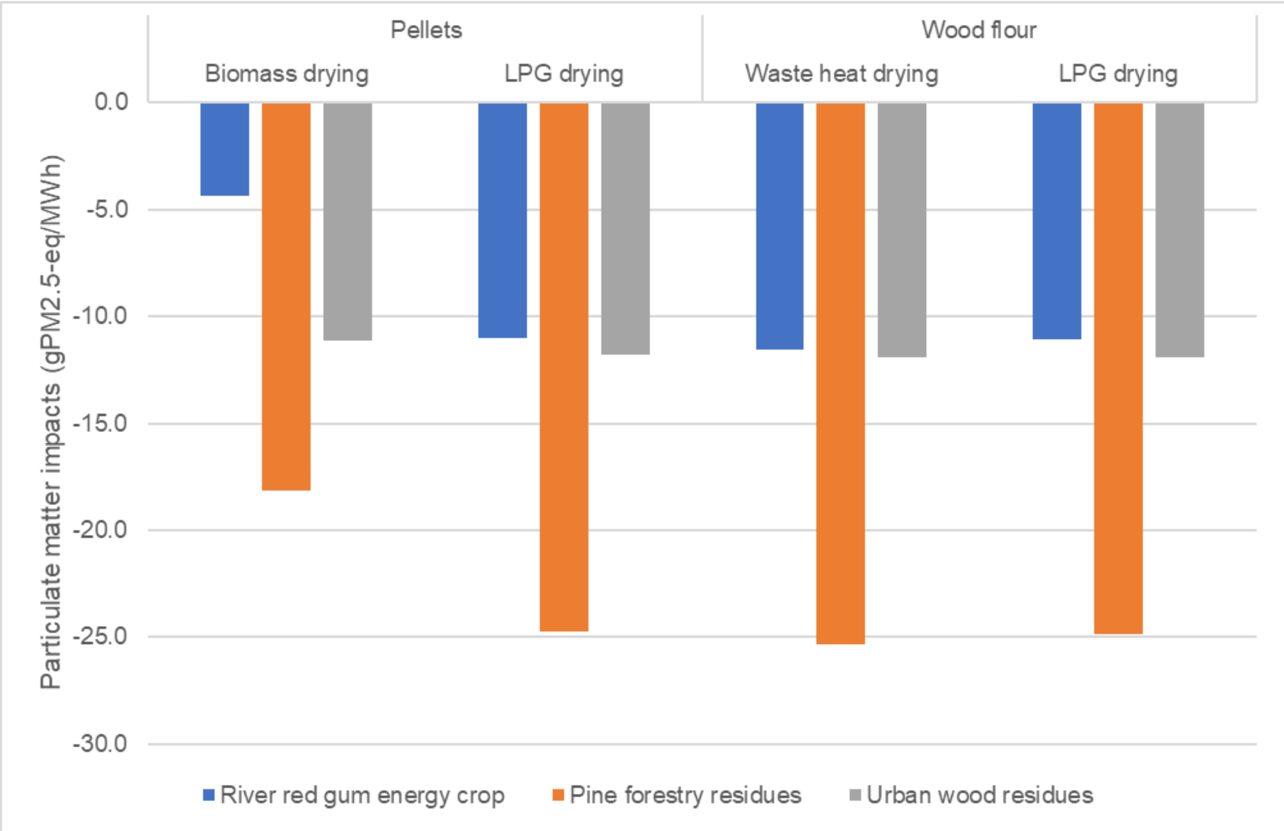

Figure 4. Net particulate matter impacts, effect of processing choice

3.3. Comparison to other electricity sources

In Table 4 the comparison of the LCA results to alternative dispatchable electricity generation options is shown. The models for coal and co-firing are based on the results of this study, while the remaining models are sourced from LCA databases: natural gas (AusLCI 1.34), 100% biomass (USLCI with small modifications), and concentrated solar thermal (Ecoinvent 3.6, RoW process). The results show that while co-firing has lower impacts when compared to 100% coal, further benefits can be seen from selecting 100% renewable energy technologies for dispatchable energy.

Table 4. LCA results, comparison of dispatchable energy sources, per MWh

Impact category	**Unit**	**Coal, Mt Piper**	**Co-firing, 5% biomass, Mt Piper**	**Natural gas, Australia**	**100% Biomass, Australia**	**CST, parabolic trough, Australia**
Climate change	kg CO2 eq	891	861	588	47	49
Particulate matter	kg PM2.5	0.35	0.35	0.02	0.03	0.02
Fossil fuel depletion	MJ NCV	9,891	9,577	10,183	59	689
Water scarcity	m3 eq	1.18	1.18	0.06	0.03	0.17
Land use	m2a crop eq	8.6	18.2	0.2	0.1	6.1

4. Discussion

The results of this study are in line with other studies which show environmental benefits through the application of biomass co-firing in coal-fired power plants – where benefits are generally in proportion to the fraction of biomass. While this reinforces the reasons to back up the implementation of co-firing, there are still other barriers which have significantly affected the uptake of the technology, such as the availability of cheap coal. The availability of sufficient, cost-effective biomass is also a challenge. Although the results of this study show that transport impacts are minimal in comparison to the savings, it is a key factor from a financial perspective. For this reason, co-firing tends to become economically feasible only when biomass resources are available in close proximity to a power station.

In 2019, the NSW Department of Primary Industries commissioned a report into the feasibility of using biomass-derived fuel in Mt. Piper Power Station [44]. The study found that all sawmill residues in the region are already consumed by existing wood processing facilities. However, the region within 100km of Mt. Piper Power Station could produce approximately 108 kilotonnes per year of 'upgraded, dried and milled biomass fuel' from unutilised tops and limbs from harvest operations. At 5% biomass, Mt Piper Power Station would require approximately 150 kilotonnes of feedstock for one year of operation (at 8% moisture content). This suggests that forestry residues alone could not supply Mt. Piper with sufficient feedstock for 5% co-firing (within 100km radius). Biomass availability is highly dependent on location, so further investigation would be needed to make similar comparisons for other power stations.

In terms of the feedstock types, little variation in impacts was found between energy crops, forestry residues and urban wood waste. All feedstock types result in environmental benefits for climate change, particulate matter, fossil fuel depletion and water scarcity. The only detrimental effect found was an increase in land use impacts when using energy crops for feedstock. It should be noted that land use impacts are measured in equivalent units, based on relative species loss. If biomass crops target marginal, unproductive land, then there is potential for a range of ecosystem benefits as well as biomass production.

In terms of maximising environmental benefits, co-firing provides a potential pathway to negative greenhouse gas emissions through BECCU or BECCS. If carbon capture technology is applied to a co-firing facility, the portion of captured emissions attributable to the biomass input are considered a negative emission, or extraction of carbon dioxide from the atmosphere. The larger the fraction of biomass in co-firing, the more carbon dioxide extracted from the atmosphere – creating a potential to reduce carbon emissions associated with coal-fired electricity. Co-firing may be able to act as an intermediate step towards BECCS with 100% biomass input. Similarly, further development of BECCS in Australia may in turn increase the uptake of co-firing. This area would need further research as the feasibility of converting coal-fired power plants to 100% biomass power plants depends heavily on the type of technology employed in the plant. Furthermore, biomass availability would need to be investigated to ensure a sustainable supply.

Like most LCA studies, the results of this work contain limitations. The data is obtained from a variety of credible sources, though no primary data is used. The data quality could be improved through obtaining first-hand data from coal-fired power stations using biomass co-firing – particularly in measurement of emissions. On the feedstock modelling, further modelling could be performed on the long-term average carbon storage effects of energy crops. In this study, the carbon storage effects of an increased soil organic carbon content are included, as the assumption is made that the energy crop production occurs on land previously used for high-intensity grazing. However, further carbon storage in roots and unharvested crops have not been included and may result in further benefits than have been estimated here.

5. Conclusion

Overall, the results indicate that when biomass is available and the economics work out, co-firing should be implemented in existing coal-fired power stations in Australia to reduce greenhouse gas emissions. The feedstock choice may depend on availability, though the alternative fate should always be considered. In this study, only feedstocks without current pathways into products were assessed. If further biomass feedstocks were to be assessed, their alternative fates would also need to be included.

Generally speaking, the additional impacts of a switch to co-firing with biomass are related to the biomass production (impacts on land use) and biomass processing (remaining impact categories), and hence these areas could be a focus for further increasing the environmental benefits of biomass co-firing.

The results of this study add to the literature supporting the environmental benefits of co-firing, and zoom in to a specific location to show benefits for a real case. This highlights the need for further research into why co-firing has not yet been implemented on a significant scale in Australia. Further research could investigate current attitudes towards bioenergy, or evaluate biomass requirements and availability on a larger scale.

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Exploring Pathways to Decarbonise the electricity supply in Bangladesh

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1. Introduction

Decarbonising the electricity grid is a top priority for many countries in alignment with the global goal of achieving net zero (Strbac et al., 2021; Wang et al., 2022). Besides, to score high in sustainable development goals like affordable and clean energy (SDG7), sustainable cities and communities (SDG11), responsible consumption and production (SDG12), and climate action (SDG13) - countries are aiming for low carbon emitting electricity supply (Nastasi et al., 2022; Yetano Roche et al., 2020). Bangladesh, the second-largest economy in South Asia and the world's second-largest exporter of readymade garments, must intensify efforts to decarbonise as it vies to become a trillion-dollar economy by 2040 (Bloomberg, 2022). Accordingly, Bangladesh commits to global climate change mitigation by aiming to reduce ~29% (unconditional) and ~58% (conditional) of greenhouse gas (GHG) from electricity generation by 2030, targeting an emission reduction goal of ~44 MT CO_{2eq} (million tonnes of carbon-di-oxide equivalent) compared to Business-as-usual (BAU) (MoEFCC, 2021). The draft Integrated Energy and Power Master Plan (Integrated Power and Energy Master Plan (IEPMP) of Bangladesh also wants to establish a low/zero carbon energy demand/supply system (JICA, 2022). In Bangladesh, electricity contributes ~35% of industrial energy consumption after natural gas (43%) and ~25% for the residential sector after biomass-based energy (55%) (Abdulrazak et al., 2021). Therefore, decarbonising the electricity grid is the key to achieving GHG emissions to transition into a low-carbon economy. Several studies investigated energy security and CO_2 reduction of Bangladesh's power system, such as the impacts of CO_2 emission reduction on future technology selection (Mondal et al., 2010), application of the logarithmic mean division index (LMDI) method for future emissions scenarios (Hasan and Chongbo, 2020), causality between energy consumption, electricity consumption, carbon emissions, and economic growth (Alam et al., 2012). But none considered life cycle inventory data of the generation technologies to assess the decarbonization of electricity supply by viewing the interconnected national system from a cradle-to-grave perspective using the life cycle assessment (LCA) approach. This study aims to overcome this limitation and explore the decarbonisation pathways of electricity supply in Bangladesh.

2. Material and methods

2.1. Goal and scope

The goal of this study is a quantitative environmental performance assessment of the electricity generation and supply in Bangladesh from 2005 to 2022, and for projection from 2023 to 2050. The outcome may serve as the basis to assess the existing electricity mix to foster discussion on future strategies development for decarbonizing the electricity supply. Figure 1 shows the system boundaries of this study. The functional unit was '1 kWh of grid electricity supplied to the end user', after considering the electricity losses throughout the network. Environmental impacts of future electricity generation were assessed based on the scenario formulations as discussed in section 2.3.

2.2. Inventory data compilation and impact assessment

Figure 2 presents the time series on the installed capacity and generated electricity by fuel category in Bangladesh from 2005 to 2022. Table 1 shows the electricity generation technologies in the national electricity mix from 2005 to 2022. Seven types of power plants contribute to the country's grid electricity mix, including four fossil fuel-based (Furnace oil, Diesel, Natural Gas, and Coal) and three renewable energy-based (Hydro, Solar, and Wind). The life cycle inventory was developed using data from annual reports of the Bangladesh Power Development Board. Data was obtained for the generation of technologies from the different annual reports of the Bangladesh Power Development Board.

Figure 3 presents the fuel consumption, production, distribution, and electricity consumption in the country for 2022. Table 2 shows the power plant characteristics used in this study. Secondary data for the background system and power import from India was obtained from the Ecoinvent V3.8.

The LCA enabled a comparative analysis of current and forecast energy systems by identification of the primary sources of an environmental impact considering the Impact World+ method. Impact World+ was chosen for its regionalized methodology (Bulle et al., 2019). In addition to GHG emissions, this research also highlights other mid-point life cycle impact categories, including photochemical oxidant formation, particulate matter formation, ozone layer depletion, terrestrial acidification, and freshwater eutrophication.

2.3. Scenario definition, electricity generation projection, and assumptions

In addition to the business-as-usual baseline scenario of the existing electricity mix (Table 1), additional scenarios are considered in this study with different technology mixes and GHG reduction targets till 2050 (Table 3). The scenario analysis is not an attempt towards future prediction; rather, these are the future possibilities to weigh different routes of energy mix planning considering the potential futures considering the ongoing and emerging national and global contexts. The actual future situation may lie between the ranges shown in the single scenario or a combination of the formulated scenarios. The draft IEPMP of Bangladesh and a recent study conducted by Das et al. (2018) provided the future electricity demand and generation projection (JICA, 2022). IEPMP forecasted demand based on the National Perspective Plan 2041 (GED, 2020) and Gross Domestic Product (GDP) for in-between cases using variables including GDP, Population, Energy Prices, Previous Demand, Exchange Rates, and International Trade. In contrast, Das et al. (2018) forecasted the demand based on the Integrated MARKAL-EFOM System (TIMES) model that predicts future power systems by capturing the variation in load with a higher time resolution and intermittency of renewables. This study used the electricity demand under the GDP in-between cases for scenario analysis - the Perspective Plan 2041 value was the upper bound, and Das et al. (2018) reported value as the lower bound (Figure 4) to incorporate all the variables that may reasonably influence the future demand.

3. Results

3.1. Impacts of electricity generation

Figure 5 compares the temporal trends of environmental impacts from the electricity mix in Bangladesh from 2005 to 2022. From 2016 to 2017, Bangladesh's electricity mix showed marked variations in impacts. The life cycle GHG emissions (Climate change category) increased from 726.7 g CO_{2eq}/kWh in 2005 to 753.8 g CO_{2eq}/kWh in 2022, which is clearly due to the import of electricity from India since GHG emissions from national electricity generation came down from 726.7 to 605.2 g CO_{2eq}/kWh within the time frame. In the future, GHG emissions from coal and furnace oil-based electricity generation will continue to increase compared to the reduction from national gas-based generation. The life cycle photochemical oxidant formation, particulate matter formation, and terrestrial acidification showed a similar trend with increasing contribution from power import. The results for the life cycle ozone layer depletion and freshwater eutrophication reduced from 121.3 (2005) to 88.0 (2022) µg CFC-11eq /kWh, and 105.7 (2005) to 101.6 (2022) µg CFC-11eq /kWh, respectively.

3.2 Transition Pathway for decarbonising electricity generation

Figure 6 presents the projected GHG emissions under the five scenarios for the power sector of Bangladesh till 2050. Emissions grow significantly in all the scenarios driven by increased electricity demand. In the BAU scenario, emissions may grow by five times over the modeling period reaching 330.70 in 2050 from 2022's 61.91 MT CO_{2eq} with respective contributions from natural gas-based power plants (41%), furnace oil-based power plants (29%), electricity import (19%), and coal-based power plants (10%). The projected upper- and lower-bounds of GHG emission in 2050 is 428.83 MT CO_{2eq} and 310.26 MT CO_{2eq}. PSMP scenario, on the other hand, indicated higher GHG emissions in 2050 (374 MT CO_{2eq}) compared to BAU consisting of a higher contribution from import (45%) and coal-based electricity (33%). The mitigation scenarios, PA, REI, and ST, indicated the potential to achieve an emissions reduction by 150.82 (45%), 93.0 (28%), and 52.14 (~16%) MT CO_{2eq}, respectively, compared to the BAU. All three scenarios substitute the use of fossil fuel with varying degrees of renewables and CCS-enabled fossil fuel-based power generation. Under all the mitigation scenarios, power import from India is the highest contributor to GHG emissions.

4. Discussion

The depleting natural gas reserves of the country and the limited economic feasibility of increasing coal extraction or expanding hydro create challenges for the country to balance the growing demand for electricity with economic growth (Das et al., 2018), decarbonization targets, and achieving SDGs. This research compared a set of probable strategies that Bangladesh needs to explore to achieve a low-carbon power system that sustainably meets future demand. The analysis provides interesting insights into possible low-carbon futures for the power sector of Bangladesh. The scenario analysis reveals that coal and natural gas-based power sources, nuclear power, and power imports will remain the major electricity sources to support the base load. New and efficient fossil fuel-based generation technology may lower the emissions, but higher demand and generation will substantially increase the overall emissions under the BAU. Hence, a strong mitigation strategy, through the expansion of renewables, will be inevitable for higher emissions reduction.

Due to its availability, natural gas became the prime fuel for electricity generation in the country. However, the country is already importing LNG, as the existing stock may deplete by 2027 (Das et al., 2018). In addition importing coal, nuclear fuel, or electricity import may become a more economically feasible option (BPDB, 2022). Figure 6 presents the projected GHG emissions of the power sector of Bangladesh under the scenarios considered. Under the BAU, the emissions will be substantially higher as the current electricity mix is dominated by fossil fuels, natural gas, furnace oil, diesel, and coal (Table 1). In BAU and PSMP scenarios, the GHG emissions will increase from 61.91 MT CO_{2eq} in 2022 to 330.70 MT CO_{2eq} and 374 MT CO_{2eq} in 2050, respectively. Bangladesh is a highly climate-vulnerable country, and the power sector is the highest contributing sector to energy-related CO_2 emissions. Therefore, future power generation needs to be diversified by considering emission scenarios by combining base load plants using CCS-coal or CCS-gas plants, electricity import, and increasing renewables, as well as from a consumer perspective adopting energy efficiency programs to increase the electricity use efficiency and hence reduce the overall demand.

If Bangladesh wants to contribute towards the Paris Agreement goal, the necessity of the accelerated adoption of renewables is apparent from the analysis. It can be achieved through a combination of supporting policy, better grid infrastructure, a secure supply chain for renewable power plants, and a sound investment environment. Interestingly, the draft National Solar Energy Action Plan of the country outlines a plan for up to 40 GW of renewables in 2041 (SREDA, 2020). About 2845 MW of renewable energy-based power plants are in queue to be added to the national grid within the next few years (Figure 7). Three mitigation scenarios (PA, ST, and REI) suggest a significant investment in renewables, in addition to the base load power sources, to ensure a low-carbon power grid in the longer term. The country has the technical potential to add around 50 GW of solar and 4.6 GW of wind to the grid. In addition, BPDB plans to increase the hydropower capacity by 100 MW on top of the existing 230 MW. Besides, BPDB also identified two additional sites at Sangu and Matamuhuri with potential capacities of 140 MW and 75 MW (Mondal and Denich, 2010). The growing electricity demand pushes increasing power generation using fossil fuel for base load. Therefore, in addition to integrating renewables, it became urgent to invest in changing consumer behaviour and technology to push energy efficiency and reduce the overall baseload electricity demand.

5. Conclusion

This research evaluated possible transition pathways for decarbonising the electricity supply in Bangladesh. The business-as-usual situation was compared to four scenarios through attributional LCA considering cradle-to-grave system boundary to identify the potential transition pathways. The BAU and PSMP scenarios will lead to higher emissions in the future. The country can ensure low-carbon electricity by developing domestic renewable resources, as shown in three mitigation scenarios (PA, ST, and REI) combined with higher electricity imports. Increasing the share of renewable power sources offers multiple benefits, including lowering GHG emissions and air pollution, enhancing energy security by regulating fuel imports, and lowering electricity generation costs due to the rapidly declining cost of renewable technologies. Energy efficiency measures are inevitable to reduce the base demand as well. Achieving these depends on adopting a data-driven fiscal and electricity generation policy to attract private investment in the future of low-carbon energy.

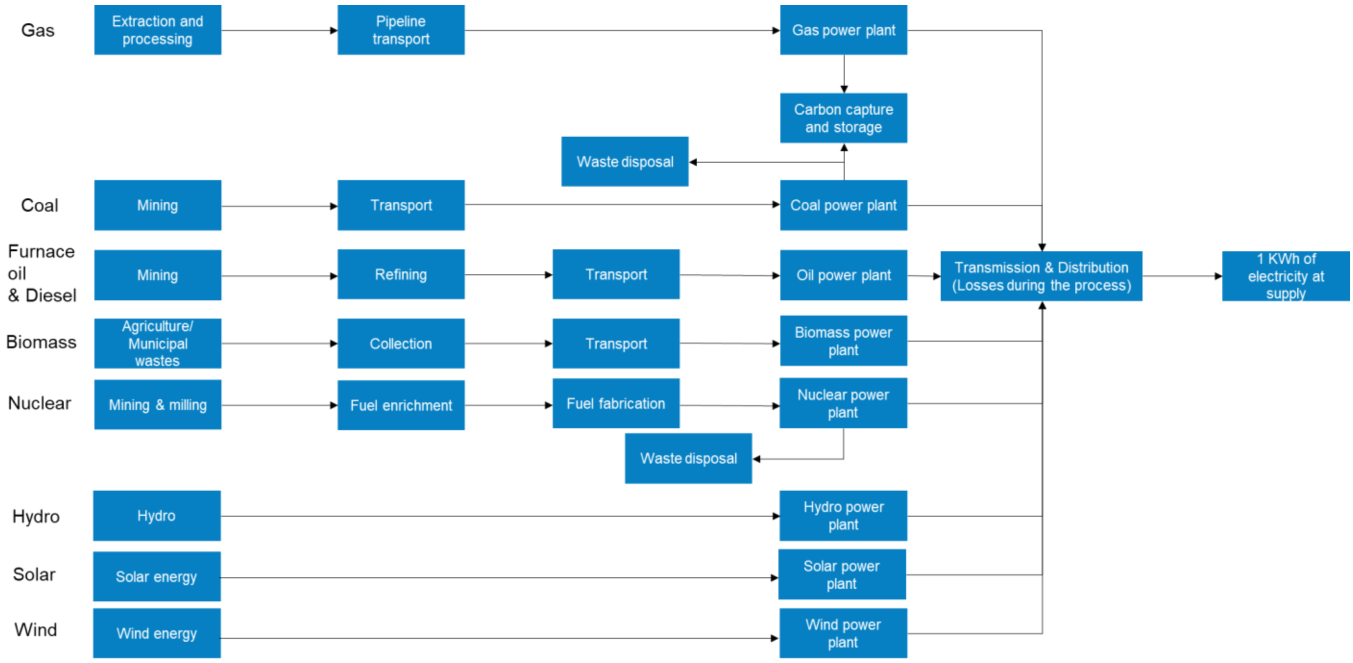

Figure 1: System boundary of the electricity generation, transmission, and distribution loss in Bangladesh.

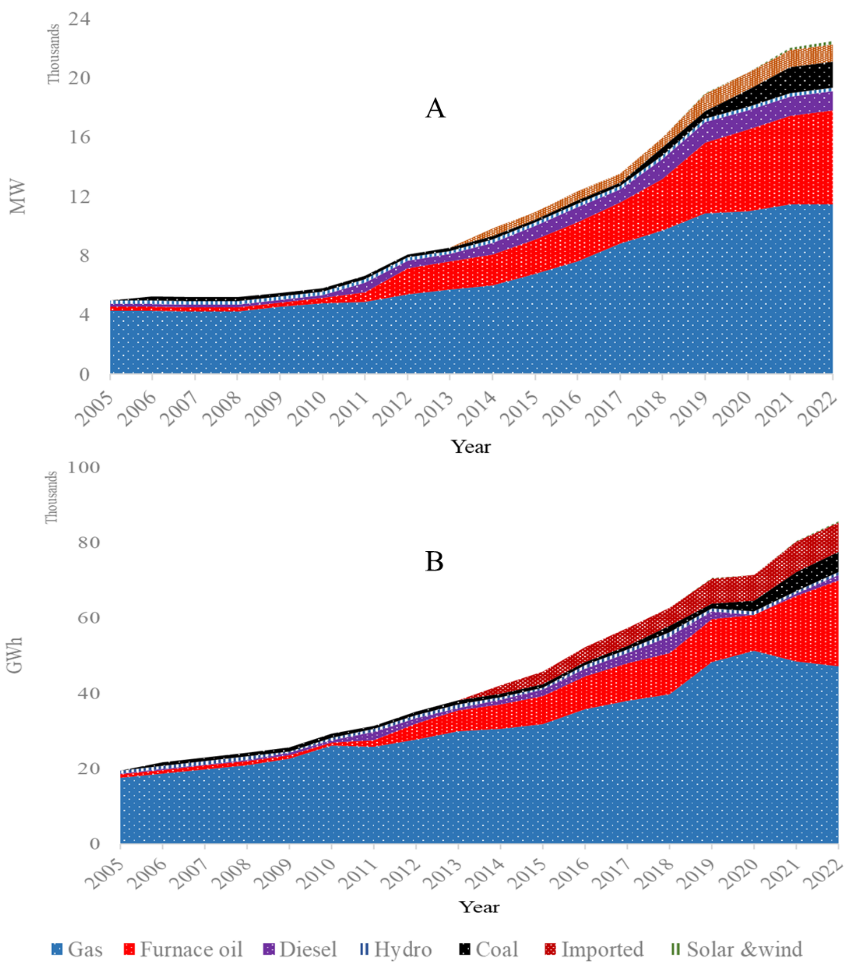

Figure 2: (A) Installed capacity and (B) Total electricity generation by fuel category for 2005 to 2022 based on data from the annual reports of the Bangladesh Power Development Board.

Table 1: Electricity generation technologies in the national electricity mix from 2005 to 2022.

Fuel/Energy source	**Generation technology**	**Contribution to national mix (%)**				
		2005	2009	2013	2017	2022
Public sector						
Furnace oil	Reciprocating engine	0.01		3.06	3.90	2.76
Diesel	Reciprocating engine	0.04	0.05	0.01	0.01	
Gas	Reciprocating engine	-	-	0.80	0.40	0.26
Coal	Steam power	4.64	4.41	3.52	1.99	1.79
Furnace oil	Steam power	1.66	1.04	0.10	-	-
Gas	Steam power	40.28	40.14	23.21	13.45	4.03
Gas	Combined cycle	2.47	2.68	5.12	19.17	21.40
Diesel	Combined cycle	-	-	-	1.68	0.19
Diesel	Gas turbine	1.02	0.94	0.21	0.50	0.55
Gas	Gas turbine	9.99	10.94	12.27	4.90	7.06
Hydropower	Hydropower Run-of-River	3.77	1.57	2.39	1.68	0.88
Wind	Wind power	-	-	-	-	0.001
Solar	Solar photovoltaic	-	-	-	-	0.02
Private sector						
Gas	Combined Cycle	21.47	21.95	35.38	25.94	21.67
Furnace oil	Reciprocating Engine	14.65	16.29	12.14	17.85	24.23
Diesel	Genset	-	-	1.78	0.60	1.04
Solar	Solar photovoltaic	-	-	-	-	0.36
Joint venture						
Coal	Ultra-supercritical pulverized coal-fired boilers	-	-	-	-	4.71
Import						
Gas	Reciprocating Engine	-	-	-	7.94	6.84
Coal	Supercritical coal-fired	-	-	-	-	2.24

Data source: Annual reports of Bangladesh Power Development Board.

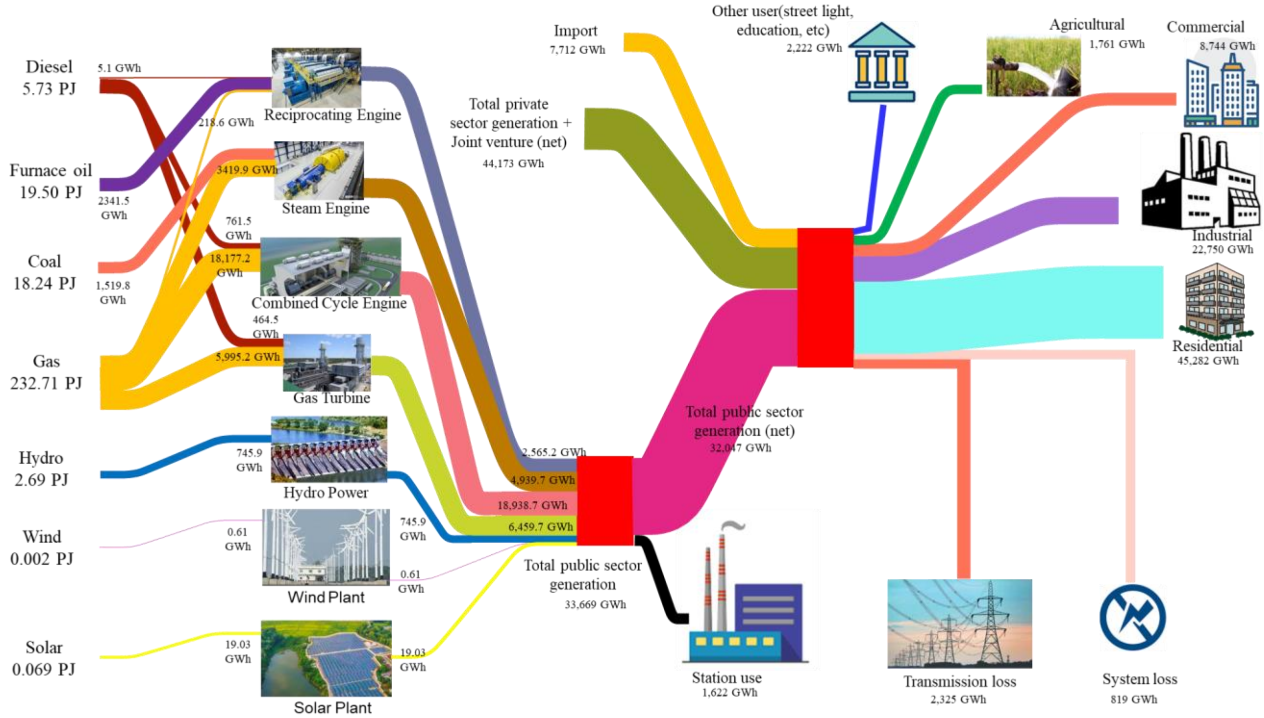

Figure 3: Electricity flow chart of Bangladesh for the year 2022. The figure is developed from the extracted data from the annual report of the Bangladesh Power Development Board.

Table 2: Power plant characteristics used in this study.

Fuel source	Power plant technology	Description
Gas	Combined cycle	500 MW combined cycle power plant same as current technology.
Furnace oil	Combined cycle	110 MW combined cycle power plant same as current technology.
Diesel	Combined cycle	110 MW combined cycle power plant same as current technology.
Coal	Ultra-supercritical (USC) pulverized combustion.	660 MW ultra-supercritical power generating unit same as current technology.
Solar	Crystalline silicon thin film	Conversion efficiency is 22% and the composition is multi-crystalline silicon (mc-Si) and Cadmium Telluride (CdTe)[a].
Hydro	Turbine running with water	Large (dam-reservoir) and small (run-of-river) hydropower plants same as current technology.
Nuclear	Pressurized water reactor (PWR).	Same as under-construction technology.
Biomass	Municipal waste and rice husk-based combustion steam turbine (ST).	The feedstock will be rice husk and municipal waste [a].
Wind	Wind turbine	50 MW. Hub height 300 ft; rotor diameter: 164m [a].
Gas CCS	Combined cycle	500 MW combined cycle power plant same as current technology. After combustion CO_2 capture, transport, and storage to the old gas reservoir with a removal efficiency of 90%[b].
Coal CCS	Ultra-supercritical (USC) pulverized combustion.	660 MW ultra-supercritical power generating unit same as current technology. After combustion CO_2 capture, transport, and storage to the old gas reservoir with a removal efficiency of 90%[b].

Notes:

(i) Power plant technology of the existing/similar generation facilities in the country is assumed from the annual reports of the Bangladesh Power Development Board.

(ii) [a](IRENA, 2019); [b] (Schenler et al., 2008).

Table 3: Scenarios on electricity generation in Bangladesh: Drivers, assumptions & characteristics.

	Scenario	Scenario source	GHG reduction/low carbon electricity generation target in 2050	Scenario description
1.	Business as usual (BAU)	Existing electricity mix	None	The existing electricity mix is presented in Table 1.
2.	Power sector master plan (PSMP)	Power sector master plan (2016) (MoPEM, 2016)	Power sector master plan (2016) (MoPEM, 2016) electricity generation target.	Power sector master plan (2016) (MoPEM, 2016) supporting the optimum electricity generation mix from gas (25%), coal (25%), furnace oil (5%), power import (25%), nuclear (10%), renewables (solar, wind, municipal waste incineration, & hydro) (10%) by 2050.
3.	Paris Agreement (PA)	Based on the Paris agreement	As per the Paris Agreement to keep the temperature rise “well below 2°C”, global renewable electricity generation should be 65% by 2050, compared to 15% in 2017 (IRENA, 2017).	The assumption is that technology transfer and financial assistance from developed economies creating a favorable enabling environment resulting in the development of renewable electricity (solar, wind, municipal waste incineration & hydro) contributing 65% of the total electricity mix by 2050. Other sources such as gas (5%), coal (10%), import (15%), and nuclear (5%) together contribute 35% of the total electricity mix by 2050.
4.	Stabilization (ST)	This study	No increase in GHG emission.	Climate change mitigation and energy policy promoting the diversification of electricity supply, and investment for (i) low-carbon technology from fossil fuels: gas (20%), and coal (20%) with carbon capture and storage (CCS) contributing together 40%, as well as gas (5%) and coal (10%) without CCS of the total electricity by 2050. Renewable energies from solar, wind, municipal waste incineration, & hydro contribute 15%, import 20%, and nuclear energy contributes 10% of the total electricity mix by 2050.
5.	Renewables and import (REI)	This study	50% renewables, and 50% non-renewables	Climate change mitigation and energy policy promoting the diversification of electricity supply, and investment for (i) renewable energies, and (ii) fossil fuel-based low-carbon options; resulting in solar, wind, municipal waste incineration, & hydropower together renewables contributing 50% of the total electricity mix by 2050; and remaining contribution from gas with CCS (5%), coal with CCS (5%), gas without CCS (5%), coal without CCS (5%), import (25%) and nuclear (5%) to the total electricity.

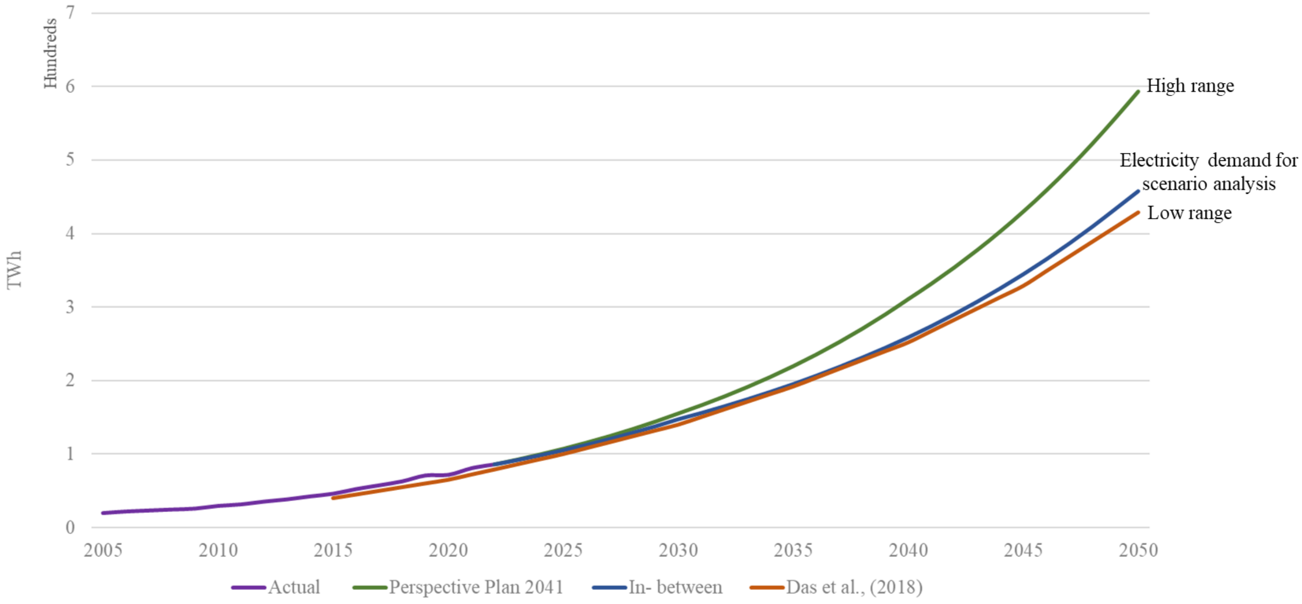

Figure 4: Projected electricity demand used for scenario analysis with high and low ranges in this study.

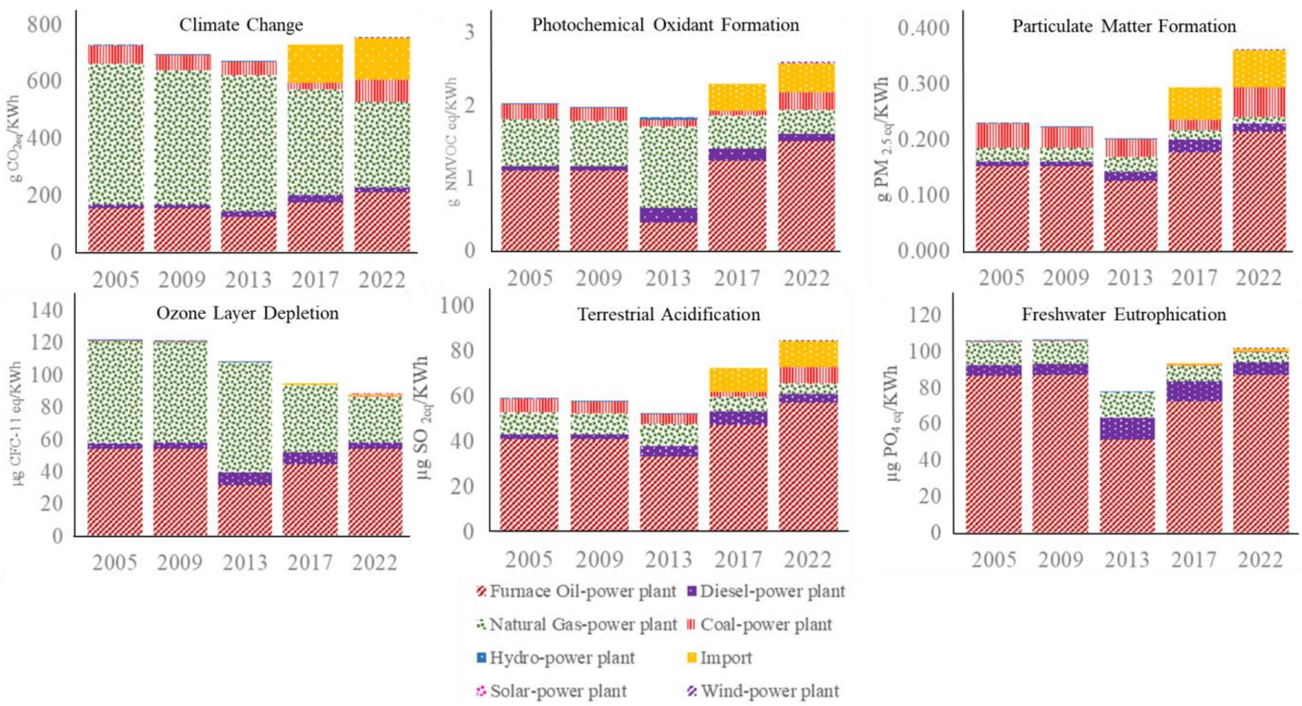

Figure 5: Characterization impact results per kWh (functional unit) for the selected impact categories from 2005–2022.

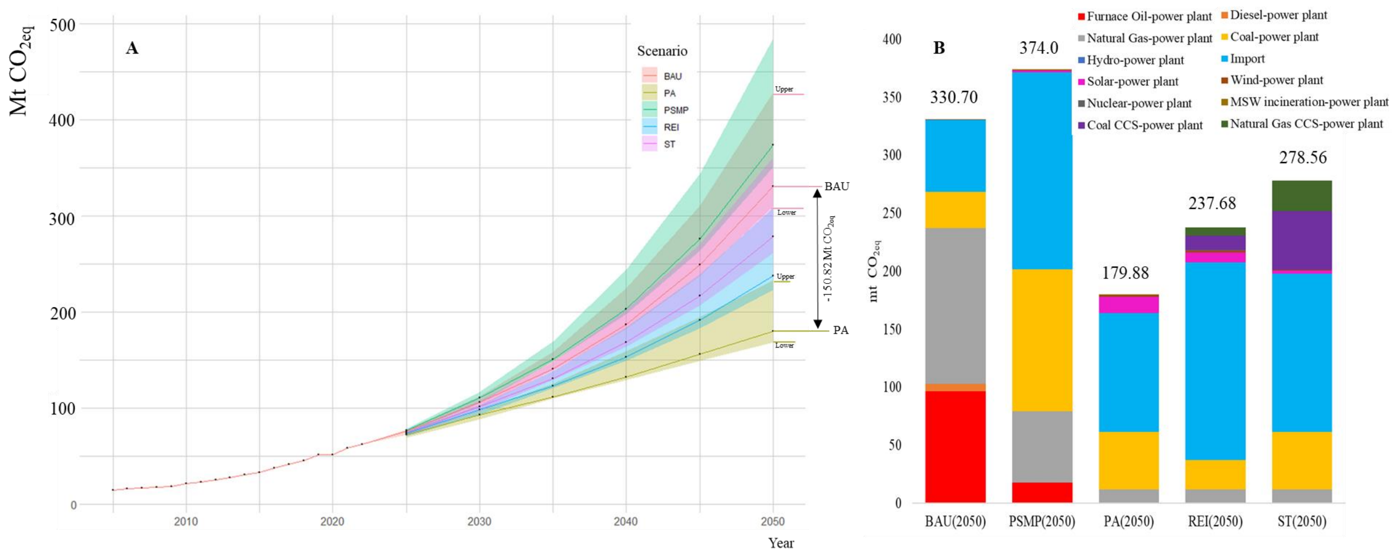

Figure 6: (A) Projected greenhouse gas emissions under different scenarios, and (B) total greenhouse gas emissions under different scenarios contributed by different electricity generation sources.

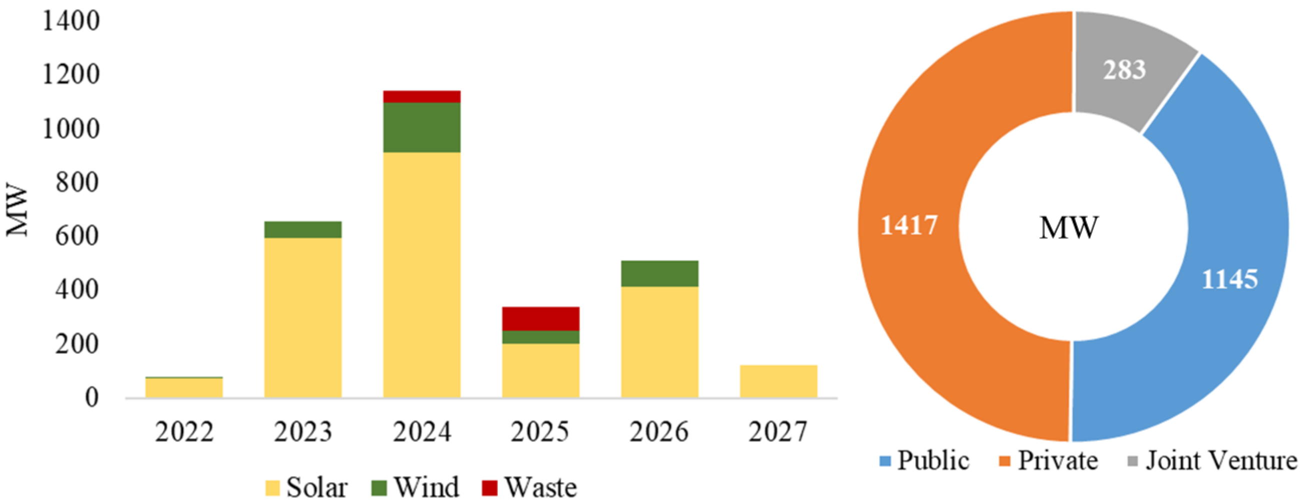

Figure 7: Renewable power plants to be commissioned in Bangladesh. Data source: (BPDB, 2022)

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Greenhouse Gases Abstracts

Metrics for Net Zero

Wednesday, 19th July - 14:00: Greenhouse Gases

Dr. Annette Cowie [1]

1. *NSW Department of Primary Industries*

Under the Paris Agreement, nations are pursuing the goal to reach net zero GHG emissions, at global level, in the second half of this century. At March 2023, 135 countries had declared net zero targets. Despite these declarations, many details of these national targets remain unclear, including which gases are covered, how net zero is defined, and whether offsets are permitted. These details determine the ease by which targets can be reached, their effectiveness in supporting climate stabilisation, and their implications for land use, and supply of agricultural and forest products.

One hotly debated issue is the climate metric to quantify net zero. Conventionally in LCA climate impact is assessed using GWP100, and this is the metric adopted for national greenhouse gas inventories under the UNFCCC. However, there is growing concern over the adequacy of GWP100, and countries with a large fraction of emissions from the livestock sector – such as Ireland and New Zealand - have pushed for different metrics to be considered.

A recent study investigated the implications of using different climate metrics to assess achievement of net zero, for the agriculture sector of Ireland, which is dominated by methane emissions from the beef and dairy industries. 3000 randomised scenarios of future agricultural production and land use combinations were screened using ten assessment methods, based on alternative applications of GWP100 and GWP*.

Different methods led to 1% - 99% of scenarios being categorised as achieving Net Zero. Several actions were identified as consistent with achieving Net Zero across all methods: high rates of afforestation, re-wetting of organic soils, and cattle destocking. Beef and especially milk production was reduced substantially in net zero scenarios, under all methods.

The choice of climate metric has strong implications for land use change, livelihoods, perceived fairness, food security, and risk of GHG leakage.

How to consider carbon neutrality in LCA? Challenges of offsetting accounting and possible solutions

Wednesday, 19th July - 14:00: Greenhouse Gases

Ms. Elena Huber [1]

1. *Technische Universität Berlin*

Achieving carbon neutrality by 2050 is one of the essential parts of the Paris Agreement and can only be reached through immense decarbonization efforts supported by offsetting through carbon dioxide removals (CDRs).
Emission reductions of decarbonization measures can be accounted for in life cycle assessments (LCA) according to ISO 14040/14044. For individual CDR technologies, LCAs can be performed. But there still exist several challenges, e.g. due to temporary storage of CO2 through nature-based solutions (NBS). The biggest challenge, however, is that CDRs cannot be accounted in a methodologically consistent way as offsetting in LCAs to balance remaining emissions for reaching carbon neutrality.
A comprehensive overview of existing challenges will be presented and first solutions for a consistent accounting of emissions and offsetting will be introduced.
A review of LCA methodologies (ISO 14040/14044), offsetting methodologies as defined e.g. within CDM, as well as scientific literature and case studies, has shown that the main issues are regarding e.g. functional unit, system boundaries, life cycle stages, and time difference of emissions and removals (offsets).
Possible solutions to integrate offsetting through CDRs in LCAs to achieve carbon neutrality are discussed, including approaches to align the system boundaries that allow balancing of the results. This alignment can be done by either harmonizing the methods used for emission and removal accounting respectively, or by developing a single method to cover both in a consistent way.
Further, regardless of which approach is followed, all emissions, timing issues, leakage risks, and potential problem shifting have to be considered in the chosen accounting approach.

Opportunities and challenges of assessing contribution to the reduction of GHG emissions through life cycle

Wednesday, 19th July - 14:00: Greenhouse Gases

Dr. Masaharu Motoshita [1]

1. *Research Institute of Science for Safety and Sustainability, National Institute of Advanced Industrial Science and Technology*

Toward the achievement of decarbonized society, innovative technologies take a crucial role to make a drastic change of GHG emissions in our society. Some of innovative technologies used for the intermediate goods like materials and parts will contribute to the reduction of GHG emissions through the introduction of final products incorporated the intermediate goods. For the industries that manufacture such innovative intermediate products indirectly reduce GHG emissions of the final product through its life cycle, whereas GHG emissions from their production sites may increase with the introduction of the final product. Many industries are facing the dilemma of contradiction between reduction of GHG emissions in the society and increase of those from their business. In this context, Life Cycle Assessment (LCA) has the potential to quantitatively assess their contribution to the reduction of GHG emissions through life cycle of products. This presentation will introduce the opportunities and challenges of assessing the contribution to the avoided GHG emission through life cycle of products based on LCA from the following perspectives: technological challenges, communication challenges, how to use the results for promoting the deployment of innovative technologies and avoiding miscommunication.

The greenhouse gas accounting interpretation and comparison challenges in the higher education sector: a university-based case study

Wednesday, 19th July - 14:00: Greenhouse Gases

Dr. Chalaka Fernando [1]

1. *Australian National University*

Higher education institutions, including universities, have recently prioritised their interest in Net Zero goals and quantitative reporting. The above developments are further motivated by the sustainability ranking systems of the higher education sector.

However, compared to corporate, no sector-specific greenhouse gas (GHG) accounting systems are developed for the higher education sector. Nevertheless, environmentally committed universities have been reporting their GHG footprint based on either international standards or country-specific GHG accounting principles, which are often generic. This work explores the university sector GHG accounting constraints and opportunities to improve them.

We identified three major challenges regarding the above issue. Firstly, the typical functional unit expressions for universities are challenged based on the unique built-environment applications of the universities. The industry–led Tertiary Education Facility Management Association uses gross/usable floor area to express the specific GHG impacts. However, an industry alignment does not exist for allocating shared space, such as shared laboratories with other research bodies/universities. The grey areas in operational control-based GHG inventory calculation of residential halls and other facilities also increase the uncertainties in interpreting the GHG impacts. Secondly, recent developments, such as transitioning from in-class to virtual learning platforms, challenge the GHG impact calculation. While it reduces the Scope 2 emissions, an industry-based consensus does not exist on capturing the associated Scope 3 emissions. Thirdly, the industry does not have a common alignment on reporting to/from commuter-based GHG emissions. The above issues create uncertainties in interpreting the GHG impacts and, eventually, challenge comparison.

In conclusion, a detailed disclosure statement on the above elements is recommended as the initial step, along with the GHG inventories. A higher-education sector-specific LCA-based GHG accounting framework is proposed in the medium term to reduce the interoperability and transparency challenges in the university sector Net Zero disclosures.

Circular Economy Abstracts

Sustainability at ResMed - The Role of LCA

Wednesday, 19th July - 14:00: Circular Economy

Ms. Amanda Chancellor [1], ***Dr. Mana Sitthiracha*** [1]

1. *ResMed*

ResMed is a global leader in sleep health that has its origins right here in Australia. From sleep apnea to respiratory conditions including COPD, ResMed has innovative solutions to help people live a healthier life. At ResMed we are designing in sustainability from the start and embedding it into the entire product development process to deliver the necessary change our planet deserves, without forcing our users to compromise on performance. To this end, ResMed is actively using Life Cycle Assessments to make data driven decisions and are target the right areas. To date, we have conducted two formal LCAs on our sleep apnea products and have two more in progress. The results of these LCAs have had a significant impact on our future design directions. We have also integrated LCA into our product development process, with our engineers required to present LCA results during design reviews and at each phase gates. By using LCAs we can ensure that we are making sustainable design decisions and incrementally improving the environmental impact of our products.

Comparison of reusable cup systems to single use cups

Wednesday, 19th July - 14:00: Circular Economy

Ms. Cathy Jiang [1]
1. *Lifecycles*

Each year in Melbourne over 8000 public events are held, and the Melbourne Cricket Ground (MGC) alone regularly attracts crowds of over 100,000 people to concerts and sporting matches. Beverages offered at these events are typically provided in single use containers, resulting in over 900 million plastic cups consumed annually, 790 million of which end up in landfill each year. WOSUP Australia is a start-up supporting increased circularity to such events by providing a reusable beverage container that can be washed and reused after each event.
This study compares the WOSUP cup, manufactured using aluminium, to 4 alternatives:

- Three single use products - a polyethylene terephthalate (PET) cup, polylactic acid (PLA) cup, and an aluminium can, and
- One alternative reusable cup system manufactured from polypropylene (PP).

Impact categories considered in this study include climate change, mineral depletion, fossil fuel depletion, water scarcity, and land use (ecosystem services). Recycling, composting, landfilling, and reuse (including washing cycles) were considered as end-of-life scenarios.
The results of the study indicate that beverage container choice can result in 32-244 gCO_2e per drink, demonstrating that beverage container choices can significantly impact the quantity of carbon emissions an event produces.
Increased awareness regarding the impacts of beverage containers over their entire life-cycle – particularly at high patronage events with significant demand for drinks – can help to reduce anthropogenic global warming and other environmental impacts. This is particularly important as single-use plastic bans are becoming more prevalent across the globe, and alternative materials are becoming a mandatory solution.

Cement Coprocessing - a Solution for Circular Economy Commitments of Corporates and supporting Nationally Determined Commitments: A Sri Lankan case study

Wednesday, 19th July - 14:00: Circular Economy

Mr. Sanjeewa Chulakumara [1], Mr. Rohan Lakmal [2], Dr. Chalaka Fernando [3], Ms. Navodini Daniel [4]

1. *University of Kelaniya,* ***2.*** *INSEE CEMENT LANKA,* ***3.*** *Australian National University,* ***4.*** *SLTC Research University*

Cement Coprocessing - a Solution for Circular Economy Commitments of Corporates and supporting Nationally Determined Commitments: A Sri Lankan case study

The circular economy is a potential application that can assist developing countries, like Sri Lanka, in meeting their Nationally Determined Commitments (NDCs) by reducing waste-related greenhouse gas emissions. Though corporate is inspired to integrate circularity principles in their businesses, only a few practical solutions exist in Sri Lanka with technological, economic, institutional, and cultural fit.

Many Sri Lankan corporates rely on cement kiln coprocessing as the preferred circular economy application for industrial and post-consumer waste. Currently, more than 1,000 corporate and 100 local government agencies are being serviced by local cement companies for their waste management requirements (Siam City Cement Lanka, 2022). However, the above volumes are significantly low compared to the total industrial waste generation. This work explores local cement coprocessing contributions to support NDCs.

The Sri Lankan cement industry co-processed 75,000 Mt of industrial waste in 2021, substituting 41 % of thermal energy for clinkerization and resulting in a 147,600 tCO2 reduction, calculated according to the Global Cement & Concrete Association's protocol. If the coprocessing-compatible waste generation is 50% of the total industrial waste, which is 1.25 million t/annum, the associated greenhouse gas (GHG) emission reduction can increase to 2.16 million tCO2 /year. The above potential to reduce emissions is twenty folds higher than the waste-related emissions from 2000- 2010 in Sri Lanka (Ministry of Environment Sri Lanka, 2022)

Therefore, cement co-processing can be identified as a solution for both corporate commitments in the circular economy and achieving local NDCs. Integrating GHG emission reduction through secondary materials as alternative raw materials and substitute for clinker is identified as an area to explore further.

Implications of transitioning from product selling to a product service system

Wednesday, 19th July - 14:00: Circular Economy

Dr. Mayuri Wijayasundara [1]

1. *Anvarta*

In transitioning to a circular economy, often product service systems gain increasing attention due to key advantages associated with them. One of the main advantages come from their ability to continuously regenerate the product, while providing the associated service. For a profit-oriented manufacturing firm, some of the product service systems provide the advantage to decouple revenue growth from production output, also while increasing firm's and product's environmental performance.

This presentation highlights how converting from a business model that focuses on product selling to new types of business models to provide 1) repair, 2) design for remanufacturing another product and 3) selling a service unit works, taking a pallet manufacturing company as an example. A high-level environmental impact comparison is presented by identifying how the three types of new business models has implications on the organisational footprint and the product life cycle assessments with the adoption of new product-service systems that support a circular economy.

Using Life Cycle and Systems Thinking Methods to Support Decarbonisation Policy Design in Australia: A Review

Wednesday, 19th July - 14:00: Circular Economy

Ms. Yoshinari Fukuzawa [1], ***Dr. Anthony Halog*** [1]

1. *University of Queensland, School of Earth and Environmental Sciences*

Australia, where its first bioenergy roadmap has been published only in 2021, is lagging behind European and North American countries, where there are dedicated support systems for their respective bioeconomy developments. Australia's Bioenergy Roadmap identifies various support mechanisms, including life cycle assessments (LCA), to formulate sustainability frameworks. Under the assumption that the academic discourse around LCA and systems thinking (ST) has grown leading up to and following the Roadmap's publication, the systematic review probed for any recent Australian studies, as well as North American and European ones, that were published in 2020-2023 to synthesise key findings, barriers, and solutions, regarding the use of LCA and ST on decarbonisation policy designs aiming to expand the bioenergy sector and transition to a circular bioeconomy. Results suggested that Australia did not publish any peer-reviewed literature on the relevant topic in the examined period, whereas the United States, the United Kingdom, and Italy dominated the research output. The first step for Australia, therefore, is to incorporate the following insights from the more-researched countries in its steps to transition to a circular bioeconomy. First, the second-generation feedstock should be preferred over the first-generation feedstock for its emissions reduction capability. Second, more investment can be put into renewable diesel, which is a more efficient, less carbon-intensive biofuel than conventional bioethanol or biodiesel. Third, in LCA, if impact categories are appropriately chosen, then the insights generated can be comprehensive and useful for policy design. Lastly, financial issues are often presented as the biggest obstacle to a bioeconomy transition, and thus, decarbonisation policies need to tackle this problem to be effective.

Measuring Circularity in the Water Industry

Wednesday, 19th July - 14:00: Circular Economy

Mr. Tim Grant [1]

1. *Lifecycles*

The application of circular economy principles differs for every sector of the economy. Circularity has a special meaning for the water sector, given the large volume material flows managed by that sector, water and nutrients, dissolved in or carried by wastewater systems, being the two most prominent ones.

Yarra Valley Water (YVW) has begun to integrate the circular economy into its business as an integral part of its sustainability strategy. YVW is committed to a regenerative approach which requires a holistic assessment of impacts and opportunities.

YVW engaged Lifecycles to conduct a circular baseline analysis. This was carried out by applying material flows analysis, life cycle assessment and the material circularity indicator to three main materials streams: water, nitrogen and construction.

The results show that the largest mass flows are water and that these also have the highest circularity results measured against the Material Circularity Index. Nitrogen flows are only partly controlled by YVW, with large losses being associated with agricultural application. For materials there are very large mass flows of spoil and low value materials, however, the main greenhouse gas emissions from infrastructure are derived from the smaller volumes of engineered products, mainly pipes.

The work managed to contextualise these different flows and helped YVW better conceive what regenerative design might look like and how they can start increasing the circularity of the water system while reducing climate change impacts at the same time.

Energy and Transport Abstracts

Life Cycle Assessment of various Hydrogen Production Technologies

Wednesday, 19th July - 16:00: Energy and Transport

Dr. Tara Hosseini [1]***, Dr. Mutah Musa*** [1]***, Mr. Tim Lai*** [1]***, Dr. Nawshad Haque*** [1]

1. *CSIRO*

The transition from depleting fossil fuel energy sources to carbon-free and renewable energy sources is among the most significant challenges facing humanity today. In Australia, the development of a low-carbon energy system is prioritised through a technology-based approach, aimed at attaining a net-zero economy, while protecting relevant industries, regions and jobs. Hydrogen as the most abundant element on earth has gained interest for the role it can play in the global clean energy transition and the reduction of greenhouse gas (GHG) emissions.

Hydrogen has found relevant applications in several sectors, such as fuel for transport or heating, and energy carrier to store electricity and raw material in industrial processes [1]. However, hydrogen is not freely available in nature, it has to be extracted from existing fuels or chemical compounds. The extraction process usually has high thermal or electrical energy requirements, which significantly increases the carbon footprint of the process. Several of the available hydrogen production pathways, have either not attained technological maturity or are embedded with significant CO_2 emissions.

Hydrogen is mainly produced through steam reforming of methane in natural gas but also from coal gasification with both methods attributed to high CO_2 emissions [2]. Production of hydrogen from fossil fuels produces a large amount of carbon emissions that need to be captured and stored. This implies a necessity for the development of new technologies for carbon capture and storage (CCS) on a large scale. Currently, only about 4% of global hydrogen production is via carbon-free technologies such as water electrolysis using renewable electricity. Therefore, there is a need to increase global energy inputs from low-carbon footprint hydrogen technologies to meet increasing energy demands and attain net zero emission targets required to combat climate change.

This work aims to understand the carbon footprint associated with different hydrogen production methods.

A Simplified Sustainable Circular Economy Evaluation for End-of-Life Photovoltaic

Wednesday, 19th July - 16:00: Energy and Transport

***Ms. Emily Suyanto** [1], **Mr. Massoud Sofi** [1], **Ms. Elisa Lumantarna** [1], **Prof. Lu Aye** [1]*

***1.** The University of Melbourne*

Sustainability and circular economy in the photovoltaic (PV) industry has been gaining increasing traction. Yet, it is still in its infancy. Sustainability is not synonymous to circularity. The correlations between the two paradigms vary case-by-case. PV panel waste has become one of the fastest growing electronic waste. The potential economic and environmental benefits through recycling and other recircularity initatives have been confirmed. However, there is still an urgency to delineate End-of-Life (EoL) PV management practice that is both sustainable and circular.

Private PV stakeholders play a prominent role in achieving best practice. However, the exhaustive nature of life cycle impact studies and their data gathering may deter PV producers and recyclers to consider sustainability and circularity performance in their decision-making.

This work aims to propose a framework to evaluate sustainability and circular economy performance of discarded PV processing in an integrated manner for private sector users. Sustainability will be assessed through life cycle assessment for environmental impacts, life cycle cost for economic impacts, and industry stakeholder survey to compensate for the lack of social impacts data. Circularity will be evaluated using selected sets of existing product-level circularity indicators. The two paradigms will be reconciled through a joint analysis via multi-criteria decision making.

Understanding the role of renewable diesel in decarbonising public transport: a case study from New Zealand

Wednesday, 19th July - 16:00: Energy and Transport

Dr. Chanjief Chandrakumar [1], ***Dr. Gayathri Gamage*** [1], ***Dr. Manoj Pokhrel*** [2]

1. *thinkstep-anz,* ***2.*** *Auckland Transport*

Auckland Transport is committed to providing low emission transport choices which will mitigate greenhouse gas (GHG) emissions, improve air quality, and reduce the city's reliance on fossil fuels in a transition toward a low emissions economy. To that end, Auckland Transport is aiming to electrify its bus fleet in future. Auckland Transport is also exploring using renewable diesel in some of their bus fleet. Auckland Transport therefore seeks to understand the GHG emissions of the life cycle of renewable diesel produced overseas, shipped to, and used in New Zealand.

The study investigated six different feedstock types: used cooking oil (UCO); animal fat / tallow; palm fatty acid distillate (PFAD); palm oil; rapeseed (canola) oil; and soybean oil.

Results showed that the carbon footprint (excl. biogenic) for renewable diesels produced in Singapore using UCO, animal fat / tallow, PFAD and virgin vegetable oils range from 15.3 to 391 g CO_2 eq. / MJ. When compared with fossil diesel, use of renewable diesel in Auckland Transport bus fleet has potential to mitigate GHG emissions in New Zealand – ranging between 7.8 and 82 %. However, carbon footprints of soybean-, PFAD-, and palm oil-derived renewable diesels are largely driven by land use change impacts and are respectively 2.1, 3.4, and 4.6 times worse than fossil diesel.

Overall, the results of this study indicate that production and use of renewable diesel in Auckland Transport's bus fleet has some potential to mitigate GHG emissions in New Zealand. Specifically, renewable diesels derived from wastes or by-products (such as UCO and animal fats) have the greatest potential compared to renewable diesels derived from virgin vegetable oils. Among renewable diesels derived from virgin vegetable oils, rapeseed oil shows better potential.

Circular economy certification of used electric vehicle batteries: a review

Wednesday, 19th July - 16:00: Energy and Transport

Mr. Arif Anugraha [1]***, Dr. Anthony Halog*** [2]

1. *The University of Queensland,* ***2.*** *University of Queensland, School of Earth and Environmental Sciences*

The exponential growth of the Electric Vehicle (EV) market is a promising sign, but it has also led to a new environmental challenge. EV batteries have a lifespan of 8-10 years, contributing to significant waste volumes. A circularity model for EV batteries can extend their lifespan and minimize waste through reuse and recycling. However, to ensure the success of this model, it is crucial to address the quality and reliability of EV batteries' second life. This can be achieved through certification. Existing research focuses on the technological and economic aspects of the circular economy in EV batteries, with a limited focus on certification. This paper aims to fill this gap by exploring the importance of certification for used EV batteries. It also explores its impact on the prevailing circular economy model. The current state of research on the circular economy in EV batteries is examined through a systematic review. Then, we analyse existing successful certification implementations of related emerging technologies to identify the importance of certification for used EV batteries. Lastly, we propose a conceptual model framework for the circular economy of EV batteries concerning a certification program. The paper explains the need for a certification program and how it can increase the sustainability of EV batteries.

Energy and Transport Extended Abstracts

A Simplified Sustainable Circular Economy Evaluation for End-of-Life Photo-voltaic

Emily Ruth Suyanto[(1)], Massoud Sofi[(1)], Elisa Lumantarna[(1)], Lu Aye[(1)]*

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Keywords: Life cycle sustainability analysis, circular economy, End-of-Life photovoltaic recycling

1. Introduction

First-generation solar photovoltaic (PV) modules around the world are reaching the end of their useful lifetime. After 20 to 30 years operational life, first generation PV installed will be consigned to the waste system (Bilbao et al. 2021; Giacchetta, Leporini & Marchetti 2013). Globally, 78 million tonnes of PV panel waste is expected by 2050, which represents 10% of all e-waste (Chaplin, Florin & Dominish 2018). Solar PV is deemed the fastest growing electronic waste (e-waste) in Australia (Sustainability Victoria 2022). Bontinck PA and Bricout J (2022) estimated 3118 tonnes of solar PV and battery-derived e-waste, having potential material value of $5.2 million, entered Australian waste system in 2019 with only $0.4 million recovered.

In spite of this mounting urgency, there is yet to be developed a standardised approach to assess the sustainability and circular economy performance of End-of-Life (EoL) PV panel. The interplay between the two interrelated but different notions are still unclear. Studies have delineated that sustainability and circular economy assessment can complement each other. End-users and product manufacturers are often left perplexed by the numerous tools available in the market without a clear comprehensive framework to ensure that all sustainability and circularity aspects are met considering potential trade-offs. The forefront of circular economy adoption lies within the PV industry. The proposed work will prove useful to instil and facilitate circular life cycle thinking,

Existing tools to assess environmental, economic, and social impacts, as well as circularity are fragmented. Life cycle assessment (LCA) is the only standardised method for environmental assessment to date. It demands life cycle expertise and intensive resources. A comprehensive tool that assesses all facets in a simple and integrated manner may reduce the barriers for private sector to adopt circular thinking. This original study aims to propose an integrated framework for private PV users to assess the environmental, economic, and social impacts of product EoL PV processing considering circular economy measures. Moreover, there is still a substantial data gap to model PV recycling in LCA (Lunardi et al. 2021). This study contributes to understanding how sustainability and circular economy can both be satisfied in the context of EoL phase of PV module.

2. Material and methods

2.1. Sustainable circular economy framework

At the forefront, it may seem intuitive that circularity improvement could contribute to the preservation of the environment and material criticality reduction. However, this is not always the case. Studies argued that circularity evaluation should not substitute sustainability evaluation. Presently LCA cannot directly measure how circular a system is. It does not advocate for linear or circular economy specifically but focuses on environmental implications throughout the life cycle.

On the other hand, circular economy often prioritises keeping individual resources within the economy (Saidani et al. 2022c). LCA and circularity indicators can complement each other in generating the most sustainable solution. Mannan and Al-Ghamdi (2022) suggested that LCA can improve different stages of circular economy evaluation in real-life scenarios and therefore can ensure proven benefits for the environment and society.

The study takes a step further by evaluating resulting sustainability and circular economy scores together. Potential trade-offs and complementary effects are investigated. Results are combined for joint analysis in an integrated manner. Multi-criteria decision making (MCDM) will be used in aggregation by normalisation and weighting.

Figure 1 exemplifies the conceptual relationship between the two broad themes of sustainability and circular economy. The study begins with a comprehensive literature review of state-of-the-art PV module recycling as well as its sustainability and circular economy practices. Initially, the two overarching concepts are treated as separate blocks of study. Before being combined for joint analysis and interpretation considering private PV stakeholder's perspective where available. The triangular radar diagram symbolises results interpretation to infer the relationship between circularity and sustainability indicators.

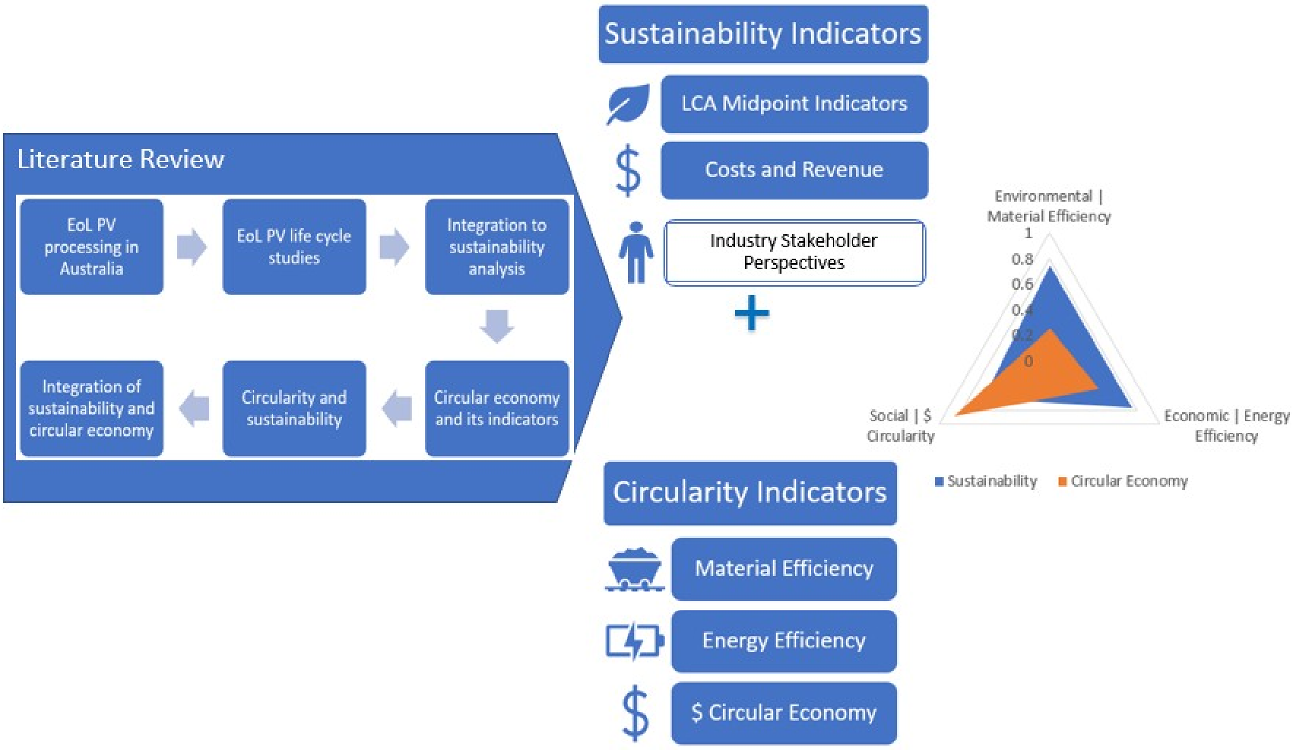

Figure 1. Graphical depiction of overarching key concepts.

2.2. Sustainability front

Within the sustainability aspect of the framework, life cycle sustainability assessment considers three pillar interpretation of sustainability as deducted from the Brundtland report (1987) which encompasses environmental, economic, and social equity (Klöpffer 2008). It is also considered as an ideal tool to assess circular economy strategies objectively to prevent burden shifting between stakeholders within the value chain (Niero & Hauschild 2017).

Environmental and economic input is taken from the first part of the study (Suyanto et al. 2023). The simplified analysis modified semi-quantitative Material, Energy, Chemical, and Other (MECO) method from Wenzel, Hauschild and Alting (1997) and Pommer et al. (2003). It streamlines conventional LCA and life cycle costing (LCC) without the need for an LCA software.

A chance to include social perspective is made possible through MCDM to combine the three pillars of sustainability as well as circular economy quantitative results. The lack of social LCA data for PV waste stream processes can be partially substituted by stakeholder survey with private PV industry participants such as PV panel producers, distributors, and recyclers. Weighting system in this paper are for mere demonstration. Social survey for MCDM weighting factor selection is out of the scope of this paper.

2.2. Circular economy front

On the other hand, within the circularity aspect of the framework, circular economy is defined as an economic and industrial model that is restorative and regenerative by design (Ellen MacArthur Foundation 2013). While there is no standardised definition of the circular economy as of now, literatures agree that this concept stands opposed to the linear "make-take-waste" model (Saidani et al. 2017). Circular economy initiatives can be realised in macro level as regional or national, meso level such as eco-industrial parks, and micro level at products, companies, and consumers level applications (Ghisellini, Cialani & Ulgiati 2016). There is still a gap in research focusing on individual product and company level circular economy indicators (Elia, Gnoni & Tornese 2017). In alignment to this knowledge gap, this work only focuses on circularity indicators at micro level within products, components, and materials operation.

The aim of this section is to identify existing quantitative micro circularity indicators that are suitable for EoL PV application that cover prominent facets of circular economy paradigm. A taxonomy of 55 sets of existing circularity indicators (C-Indicator) by Saidani et al. (2019a) was used as a starting point. The micro circularity indicator screening process is depicted in Figure 2. The twelve shortlisted indicators are further categorised into six facets based on how circular economy performance is derived.

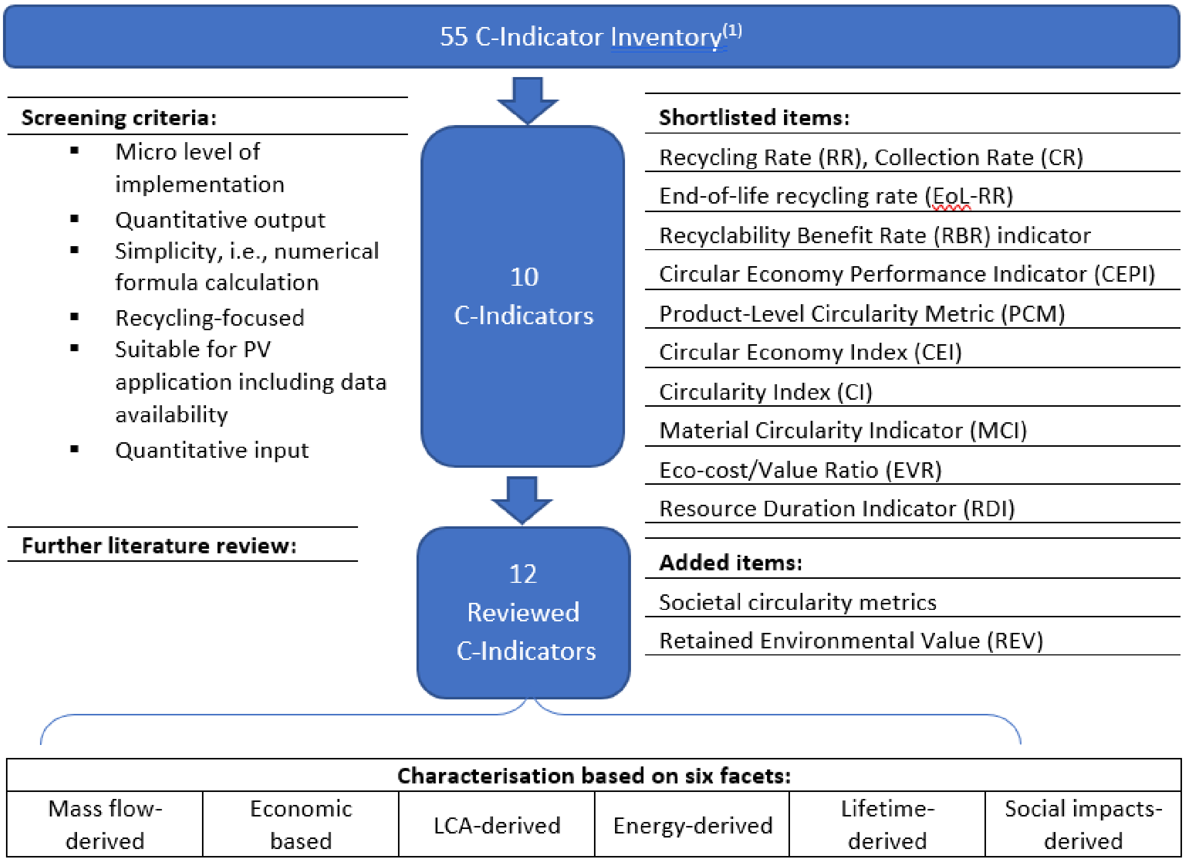

Figure 2. Micro-level circularity indicators screening process for EoL PV application.
(1) Saidani et al. (2019a)

The six facets categorisation are explained hereafter:

(i) Mass flow-derived circularity metrics

A well-established quantitative measure of circularity is Material Circularity Indicator (MCI) by Ellen MacArthur Foundation (2019). It combines mass and temporal units. Other metrics reviewed are recycling rate (RR), EoL metal recycling rates (EOL-RR; i.e., the percentage of a metal in discards that is actually recycled), recycled content (RC), and old scrap ratios (OSRs; i.e., the share of old scrap in the total scrap flow). Similarly, Haupt, Vadenbo and Hellweg (2017) utilised closed- and open-loop collection rate (CR) and RR to measure the available secondary resources produced from municipal solid waste in Switzerland.

(ii) Economic-based circularity metrics

Di Maio and Rem (2015) proposed circular economy index (CEI) which represents how effectively a recycling facility processes a product. A similar index is developed by Linder, Sarasini and van Loon (2017) to quantify the degree of recirculation of a product. Product-level circularity metric (PCM) is expressed as a ratio of economic value of recirculated product parts to economic value of all parts.

(iii) LCA-derived circularity metrics

EC-JRC (European Commission – Joint Research Centre) (2012) developed product reusability/ recyclability/ recoverability (RRR) parameters including recyclability benefit rate (RBR). RBR is the ratio of potential environmental benefits from recycling divided by burdens related to virgin materials production and disposal. It takes a step beyond RR by incorporating product components' LCA impacts from selected category.

Similarly, Huysman et al. (2017) focuses on natural resources impacts in the form of exergy to quantify post-industrial plastic industry's circularity using Circular Economy Performance Indicator (CEPI). Moreover, an LCA-based model called Eco-cost/Value Ratio (EVR) is a single indicator for sustainability that demonstrates how circular economy strategies such as reuse, remanufacturing, and recycling can fulfil eco-efficient objectives. Retained Environmental Value (REV) was proposed by Haupt and Hellweg (2019) to compare the net surplus from product reuse or recycling to lifetime environmental impacts.

(iv) Energy-based circularity metrics

Cullen (2017) considered the combination of recovered EoL material quantity compared to total demand and the energy required to recover them compared to primary production in circularity index (CI).

(v) Lifetime-based circularity metrics

Franklin-Johnson, Figge and Canning (2016) introduced resource duration indicator (RDI) which utilises the length of time of material retention in a product system as a measure of its contribution to circular economy. It is computed as the sum of three main longevity drivers, i.e. initial usage, refurbishment, and recycling lifetime.

(vi) Societal circularity metrics

In addition to the circularity indicators recommended by Saidani et al. (2019a) tool, a metric proposed by Reich et al. (2023) was reviewed. It took a more holistic approach to circular economy measurements in policy making. It considers not only material flow and environmental impacts as most circularity indicators do. But also considers socio-economic impacts, linking macro and micro indicators to the assessed system.

Finally, at the end of the screening process, three of the reviewed tools are selected based on their simplicity and suitability for the purpose of this EoL PV evaluation. The maximum value representing full circularity in all three indices is equal to one.

1) Circular Economy Index (Di Maio & Rem 2015)

$$\mathrm{CEI} = \frac{\text{market value of recycled product materials (\$)}}{\text{material value of EoL product entering recyclers gate (\$)}} \qquad \text{Equation 1}$$

In this work, recycled EoL PV module material sales revenue serves as the numerator and virgin material market value as the denominator.

2) Circularity Index (Cullen 2017)

$$CI = \alpha\beta$$

$$\alpha = \frac{\text{recovered EOL material (kg)}}{\text{total material demand (kg)}}\ ;\ \beta = 1 - \frac{\text{energy required to recover material (MJ)}}{\text{energy required for primary production (MJ)}} \qquad \text{Equation 2}$$

3) Material Circularity Indicator (Ellen MacArthur Foundation 2013)

$$MCI = 1 - LFI*F(X) \qquad \text{Equation 3}$$

$$\text{Linear Flow Index } LFI = \frac{V+W}{2M+\frac{Wf-Wc}{2}}\ ;\ \text{utility Factor } F(X) = \frac{0.9}{\frac{L}{Lav}\frac{U}{Uav}}$$

$V = mass\ of\ virgin\ material;\ W = total\ waste;\ M = mass\ of\ product;$
$Wf = waste\ from\ processing\ recycled\ content\ ;\ Wc = waste\ from\ recycling;$
$L = lifetime;\ Lav = industry\ average\ lifetime;\ U = use;\ Uav = industry\ average\ use$

In this study, MCI calculation assumes 100% virgin PV module production feedstock in all scenarios. Moreover, it only considers recycling and no reuse nor refurbishment as an alternative circular economy initiative. Lifetime and use of PV module in all scenarios remain to be the same as industry-average.

2.3. Application on PV module waste

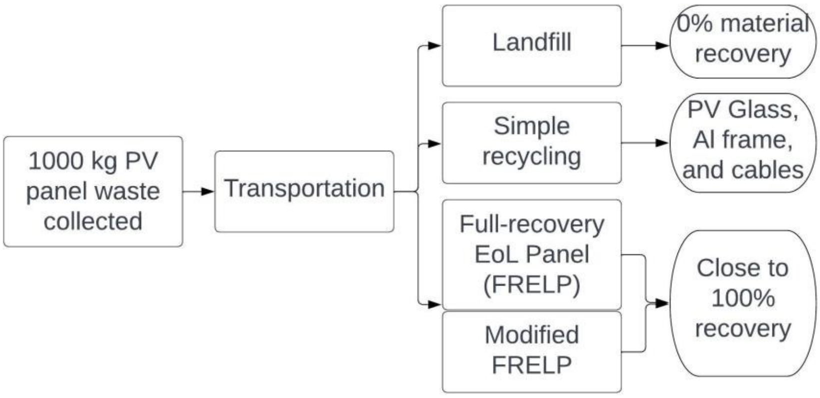

Figure 3. System boundary of EoL PV alternatives.

Four EoL PV processing alternatives in Figure 3 will be evaluated using this proposed framework. These options were previously studied in Suyanto et al. (2023). A mass-based functional unit of 1000 kg PV waste is selected for both sustainability and circular economy counterparts to ensure consistency. Landfill is the business-as-usual treatment of discarded PV modules in most countries. Simple recycling involves bulk material disassembly and glass separation. Full-recovery EoL Photovoltaic (FRELP) is a high value PV module recycling that was introduced by Latunussa C et al. (2016a). The fourth alternative is a modified version of Latunussa et al. (2016b) work. It deploys a different mechanical separation technology and focuses on solar-grade silicon recovery within the chemical separation techniques (Kang et al. 2012).

3. Results

3.1. Environmental and financial impacts of EoL PV

Following the studied framework, the sustainability aspect of EoL PV is first examined. Simplified life cycle assessment (LCA) and life cycle costing (LCC) results are taken from Suyanto et al. (2023) as summarised in Table 1. Two ecological contributions that are assessed are net primary energy impact and greenhouse gas emission. They are calculated from waste processing and transportation burden subtracted by avoided production of recovered materials and energy from recycling. In conventional LCA, these are the equivalent of cumulative energy demand and climate change impact indicators.

Table 1. Simplified environmental and financial analysis results for 1000 kg PV panel waste functional unit.

	Sustainability				
	Environment al		Financial		
Scenario	**Net Energy Impact (MJ)**	**Net GHG Emission (kgCO2-eq)**	**Processing Cost ($)**	**Revenue ($)**	**Net Cost ($)**
Landfill	-1590.83	-120.81	-263.26	0.00	-263.26
Simple recycling	30897.73	2646.31	-228.07	355.51	127.44
FRELP	38152.86	3208.49	-312.22	926.37	614.15
Modified FRELP	34424.92	2868.99	-321.86	1080.64	758.78

*Negative value signifies burden and positive value signifies surplus through avoided virgin material production

Direct landfill of PV module waste causes overall negative impact while all recycling alternatives incur positive net ecological gain. This is due to the consideration of avoided raw material production through recycling of key materials. For instance, aluminium frame and low-iron solar glass which comprises over

70% of PV panel by weight. FRELP method is found to be the most ecologically beneficial through energy and greenhouse gas impact avoidance through material recycling and incineration of polymers. Modified version of the separation technique garners over 10% less ecological benefits.

Simplified financial analysis proves the financial gain of all three recycling activities. Bearing in mind inherent assumptions in processing cost and revenue that focus on resource consumption and no fixed costs. Modified FRELP costs the most with over $300/ tonne of processed PV waste. It also attracts the highest revenue compared to the simple recycling and original FRELP methods. In terms of overall ranking, financial results favour modified FRELP method and environmental results favour original FRELP method. Whereas landfill and simple recycling routes remain in the same relative ranking positions for environmental and financial performance.

3.2. Circular economy of EoL PV

The second half of the framework examines the circularity of EoL phase of PV modules. Table 2 summarises the computation process of three shortlisted circularity indicators from the review process. Results favour FRELP and modified FRELP recycling techniques with slight variations. Simple recycling also performs considerably well compared to the two more sophisticated routes. Except for in CEI, in which the monetary values of recovered materials become prominent in the evaluation. FRELP method is deemed the most circular based on material and energy retention through CI and MCI tools. However economic value retention through CEI favours modified FRELP due to its higher revenue from harvested material.

Table 2. Shortlisted circularity indicator computation for three recycling scenarios.

	Simple Recycling	**FRELP**	**Modified FRELP**
Circular Economy Index (CEI)	**0.05**	**0.12**	**0.14**
Market value of recycled product materials (AUD)	347.60	874.08	1080.64
Material value of EoL product entering recyclers gate (AUD)	7461.31	7461.31	7461.31
	Simple Recycling	**FRELP**	**Modified FRELP**
Circularity Index (CI)	**0.84**	**0.88**	**0.87**
α	0.858	0.903	0.891
Recovered EoL material (kg)	858.11	902.90	890.56
Total material demand (kg)	1000.00	1000.00	1000.00
β	0.981	0.973	0.974
Energy required to recover material (MJ)	3255.20	4531.43	4321.87
Energy required for primary production (MJ)	167160.84	167160.84	167160.84
	Simple Recycling	**FRELP**	**Modified FRELP**
Material Circularity Indicator (MCI)	**0.50**	**0.52**	**0.51**
Utility Fraction $F(X)$		0.9	
Linear Flow Index LFI	0.555	0.537	0.542

3.3. Simplified Sustainable Circular Economy Evaluation of EoL PV

The most crucial part of the study is to couple the sustainability and circular economy counterparts. All resulting sustainability indicators are normalised and plotted on the same graph as circularity indices as depicted in Figure 4.

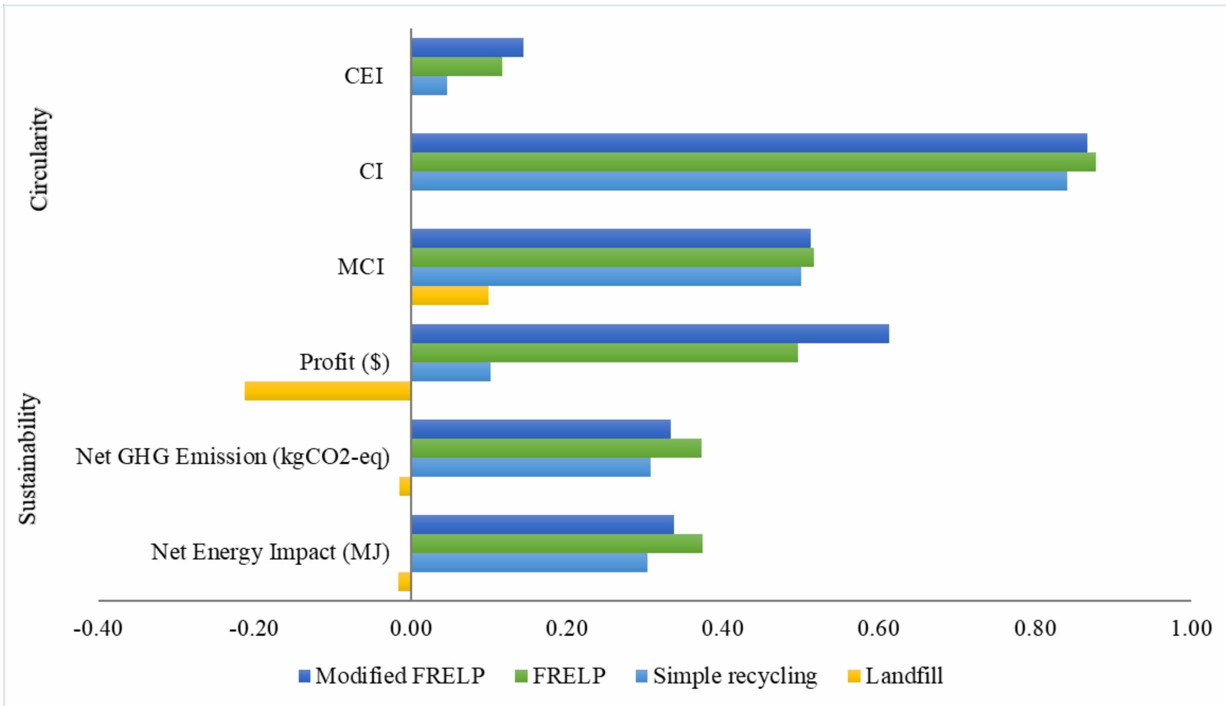

Figure 4. Normalised sustainability and circular economy indicator ranking for 1000 kg EoL PV scenarios.

General ranking of scenarios among selected simplified sustainability and circularity indicators are in agreement with each other despite variations in the degree of improvement shown by each indicator. CI results demonstrate high circularity for all recycling options due to high mass-based recovery rate of assessed recycling technologies.

The last step in the evaluation is to conduct a brief multi-criteria decision making (MCDM) based on designated weighting factors. This is designed as an opportunity for future works to incorporate social perspective through stakeholder involvement. A demonstration is presented in Table 3 with equal importance given to environmental and financial cause. Individual indicators still bear slight variations comparatively. But the resulting combined scores and rankings show an agreement between sustainability and circular economy domains. As was the case with individual indicators, negative values represent adverse impacts on sustainability and circular economy. For instance, in landfilling of EoL PV.

Table 3. Equal weighting applied on financial and environmental aspects of sustainability and circularity.

		Sustainability			Circularity		
		Environmental		Financial	Material	Energy	Value Retention
Social	Weighting	25%	25%	50%	25%	25%	50%
		Net Energy Impact	Net GHG Emission	Profit	MCI	CI	CEI

Scenario	Sustainability Score	Circularity Score	Sustainability Ranking	Circularity Ranking
Landfill	-0.11	0.03	4	4
Simple recycling	0.20	0.36	3	3
FRELP	0.44	0.407	2	2
Modified FRELP	0.47	0.417	1	1

4. Discussion

From an environmental perspective, energy and GHG emission savings are evident through recycling efforts. This finding that material recovery outweighs recycling burden is consistent with conventional gate-to-gate LCA of FRELP scenario by Mahmoudi, Huda and Behnia (2020) and many others. From a financial perspective, landfilling of PV waste cannot be considered as the cheapest end-of-use option when we consider the loss of material that can potentially be recirculated (Suyanto et al. 2023). Sustainability evaluation favours EoL pathway that generates less harmful impacts towards the environment and captures more revenue from waste recycling. However, it does not directly measure how well the PV modules are recirculated at the end of their useful lifetime.

Selected circularity metrics ensure that not only the selected EoL route is sustainable ecologically and socio-economically, but it also serves in closing the loop towards a fully circular economic system. They possess some inherent partialities towards material mass circularity. For instance, all the Circularity Index (CI) results obtained from simple recycling, FRELP, and modified FRELP are over 84% irrespective of the environmental and economic value of recovered materials.

Furthermore, it is inferred that each indicator cannot be treated as a standalone metric. Each have their own 'blind spots' or biases. For instance, simple recycling of EoL PV is scored less than 0.05 out of 1.00 in Circular Economy Index (CEI), more than 50% less than FRELP and its modified version due to its low material sales revenue, despite its relatively efficient energy performance compared to the other recycling methods. Maceno, Pilz and Oliveira (2022) reached similar conclusions from their examinations using Circular Economy Indicator Prototype (CEIP), Circular Economy Toolkit (CET), and MCI alongside LCA of PV module manufactured in Brazil. MCI proves to be complementary to LCA but should not be used in isolation to replace LCA in eco-design process. Products can yield excellent environmental performance while having a low degree of circularity.

Zubas et al. (2022) compared several circularity measures of PV silicon supply chain. Similar to this study, their LCA and MCI results mostly align. Slight variations were observed in their work as MCI favours scenarios with less virgin material usage despite longer lifetimes and higher recycling rates. Zubas et al. (2022) modelled FRELP as a closed-loop for silicon feedstock. They argued that the recovery of metallurgical grade silicon (>95% purity) can ensure the re-injection of secondary silicon to new PV module production. Their resulting MCI score was reported to be 0.80 out of 1. In an attempt to closely-represent exisiting technology, this work does not adopt the same assumption. Hence all recycling scenarios yield a more conservative average of 0.51 out of 1 for MCI scores. Additionally, they omitted impacts from recycling of discarded PV modules due to the lack of data. This study contributes to closing this gap by focusing on EoL phase in its assessments.

This study is unique in its pursuit of a streamlined framework to analyse sustainability and circular economy of EoL phase of PV modules through an array of selected key performance indicators. Some inherent limitations include the exclusions of other impact indicators such as land use and toxicology-related matters. Environmental and financial impact results can benefit from further refinement in a conventional life cycle assessment and life cycle costing. Net impacts are considered in this work. Notwithstanding that in a conventional LCA, it is always preferrable to assess ecological burden and gain separately.

5. Conclusion

Sustainability and circular economy have gained increasing traction over the years both from the academics and private PV sector. While the two paradigms are closely-related, circular economy is not synonymous to sustainability. This simplified evaluation framework strives for a balanced EoL PV alternative that ensures no burden shifting between circularity of materials and environmental, economic, or social impacts.

This work classifies six facets of circularity metrics for EoL phase of PV including material, energy, and value retention. They are heuristics tools that provide valid comparative insights to complement sustainability analysis such as LCA. Overall, environmental and financial indicators' comparative ranking are in agreement with selected circularity indicators. Landfill is the least beneficial disposal avenue from sustainability and circular economy perspective when material recovery benefits are considered. Whereas Modified FRELP is preferred from both sustainability and circular economy standpoints.

In conclusion the proposed framework is a simple tool suitable for initial comparative analysis. But should not be utilised to replace conventional life cycle assessment. Future research should focus on garnering more social impact data on EoL PV as incoming waste influx increases with time. Furthermore, the introduction of aggregation through normalisation, weighting, and linear addition compounds uncertainty within the analysis that should be quantified in future studies. Multi-criteria decision making can also be conducted alongside sensitivity analysis to ensure that numerical results are stable. In addition, computerised simulations can be developed based on the proposed framework, making use of existing life cycle thinking and circular economy tools.

Acknowledgements

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Understanding the role of renewable diesel in decarbonising public transport: a case study from New Zealand

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Abstract

Auckland Transport is committed to providing low emission transport choices which will mitigate greenhouse gas (GHG) emissions, improve air quality, and reduce the city's reliance on fossil fuels in a transition toward a low emissions economy. To that end, Auckland Transport is aiming to electrify its bus fleet in future. Auckland Transport is also exploring using renewable diesel in some of their bus fleet. Auckland Transport therefore seeks to understand the GHG emissions of the life cycle of renewable diesel produced overseas, shipped to, and used in New Zealand.

The study investigated six different feedstock types: used cooking oil (UCO); animal fat / tallow; palm fatty acid distillate (PFAD); palm oil; rapeseed (canola) oil; and soybean oil.

Results showed that the carbon footprint (excl. biogenic) for renewable diesels produced in Singapore using UCO, animal fat / tallow, PFAD and virgin vegetable oils range from 15.3 to 391 g CO_2 eq. / MJ. When compared with fossil diesel, use of renewable diesel in Auckland Transport bus fleet has potential to mitigate GHG emissions in New Zealand – ranging between 7.8 and 82 %. However, carbon footprints of soybean-, PFAD-, and palm oil-derived renewable diesels are largely driven by land use change impacts and are respectively 2.1, 3.4, and 4.6 times worse than fossil diesel.

Overall, the results of this study indicate that production and use of renewable diesel in Auckland Transport's bus fleet has some potential to mitigate GHG emissions in New Zealand. Specifically, renewable diesels derived from wastes or by-products (such as UCO and animal fats) have the greatest potential compared to renewable diesels derived from virgin vegetable oils. Among renewable diesels derived from virgin vegetable oils, rapeseed oil shows better potential.

Introduction

Transportation is crucial for enabling commerce, trade, and travel – however, it contributes ~20% of national greenhouse gas emissions (GHG) in Aotearoa New Zealand, mainly because fossil fuels are the dominant transportation energy sources (MfE, 2022).Mitigating GHG emissions from the transport sector is therefore crucial to stabilise the global climate within safe limits (Xu et al., 2022; Malik et al., 2021).

Auckland Transport (AT) is committed to providing low emission transport choices which will mitigate GHG emissions, improve air quality, and reduce the city's reliance on fossil fuels in a transition toward a low emissions economy (Auckland Transport, 2020). Intending to electrify their bus fleet in the future, AT explored other options to decarbonise their operations – using renewable diesel for some of its bus fleet is one solution. AT seeks to understand the true climate impacts and any market or regulatory implications on the horizon. AT commissioned this study to:

i. Provide evidence-based advice to support procurement decisions on renewable diesel.
ii. Ensure stakeholder buy-in for renewable diesel use in the bus fleet.
iii. Inform the public (primarily Aucklanders) about AT's commitment toward a low emissions economy.

This study was undertaken to provide scientifically robust understanding of the climate impacts associated with the life cycle of renewable diesel produced globally, shipped to, and used in Aotearoa New Zealand. Carbon footprint is the primary environmental indicator in this study as climate change is of high public and institutional interest. This study follows the guidance of international standards ISO14067:2018 (ISO, 2018) for product carbon footprinting and ISO14044 for Life Cycle Assessment (LCA) (ISO, 2006).

Carbon footprint is measured using Global Warming Potential (GWP) (excl. biogenic) and expressed as kilograms of carbon dioxide equivalent (kg CO_2 eq.) per functional unit, as specified in ISO 14067 (ISO, 2018). From a climate change perspective, carbon footprint (excl. biogenic) is more relevant, given the carbon sequestered from the atmosphere is re-emitted to it over a short period.

Literature Review - Summary

There is limited literature on the environmental impacts of renewable diesels since the relative novelty of the technology. Most of the existing studies focus on carbon footprint only and are largely based on Neste's renewable diesel production systems.

Using LCA methodology, Nikander (2008) calculated the carbon footprint of Neste's NExBTL renewable diesel produced at Neste Oil refinery in Kilpilahti, Finland. The study focused on three different feedstocks such as animal fats, palm oil and rapeseed oil. The carbon footprints (fossil only) for animal fat-, palm oil- and rapeseed oil-derived renewable diesels were 15.8, 33.4, 34.4 g CO_2 eq. / MJ, respectively.

For example, Xu et al. (2020) investigated the life cycle energy use and carbon footprint of palm fatty acid distillate (PFAD) derived renewable diesel produced in Singapore. They analysed the effects of using different material classification methods for PFAD sourced from Malaysia and Indonesia – such as co-product, by-product, and residue. The carbon footprints (excl. land use change impacts) for PFAD-derived renewable diesels were 13.5, 14.8, and 30.6 g CO_2 eq. / MJ, when PFAD is classified as a residue, by-product, and co-product, respectively.

In a recent study, Xu et al. (2022) estimated the carbon footprint of renewable diesels produced in the USA, using multiple feedstocks such as used cooking oil (UCO), animal tallow and virgin vegetable oils (soy, carinata and rapeseed), and compared the results with biodiesels produced using those vegetable oils. The study showed that the life cycle carbon footprint of the renewable diesels (excluding land use change impacts) varied notably – UCO (17.2 g CO_2 eq. / MJ), tallow (17.7), soybean (23.5), rapeseed (33.2), and carinata (28.8). When land use change impacts were accounted for, the carbon footprint of soybean and canola derived renewable diesels increased up to 53 g CO_2 eq. / MJ.

Similar studies exist for Finland (Nikander, 2008) and Brazil (Arguelles-Arguelles, Amezcua-Allieri, & Ramírez-Verduzco, 2021).

Data and Methods

An Attributional LCA (ALCA) approach was used in this study, which applies a backwards-looking, accounting approach – essentially aiming to divide up the impacts of human society and assign them to discrete products and services (Ekvall, et al., 2016). ALCA assumes that producing one additional unit of a product will have the same impact as the product that was produced before. It is an approach that relies on averages and linear scaling.

Given the relative novelty of renewable diesel production technology, limited data is publicly available (Xu et al., 2020; 2022). Most of the existing data is based on Neste's renewable diesel production systems (Nikander, 2008; Xu et al., 2022).

In this study, we focus on renewable diesel production in Singapore, using the published data available for Neste production systems in Finland and USA (Nikander, 2008; Xu et al., 2022).

Functional Unit

The functional unit for this study is 1 megajoule (MJ) of renewable diesel produced in Singapore, shipped to New Zealand, and used in the AT bus fleet in Auckland, New Zealand.

System Boundary

This is a cradle-to-grave analysis and covers the following processes:

1. Upstream processes (from cradle-to-gate) covering feedstock production/collection and transportation to renewable diesel production facility.
2. Core processes (from gate-to-gate) including feedstock purification and renewable diesel production via hydro-processing.
3. Downstream processes (from gate-to-grave) covering distribution to New Zealand and use in the AT bus fleet.

Figure 1 presents the overview of the system boundary considered in this study.

Raw materials production / collection and processing

The system boundary varies across six different feedstock types: used cooking oil (UCO); animal fat / tallow; palm fatty acid distillate (PFAD); palm oil; rapeseed (canola) oil; and soybean oil.

Key stages for the UCO and animal fat / tallow to renewable diesel pathways cover UCO collection and animal fat / tallow production in Asia. Upstream impacts associated with UCO are excluded as they are waste products from cooking. Likewise, upstream impacts associated with animal products (such as carcasses, animal head and feet) are excluded as they are by-products recovered from meat production. These modelling choices are consistent with previous work (Xu et al., 2022). UCO undergoes an additional process called grease/oil rendering. Animal products undergo animal fat rendering, yielding rendered tallow and meat bone meal.

Key stages for fresh palm fruit to renewable diesel pathway cover material and energy use, discharges and emissions associated with the cultivation of fresh palm fruit bunches, transport of palm fruit bunches to a conversion mill, and production of refined palm oil. Palm oil originates in Indonesia and Malaysia. Likewise, key stages for PFAD to renewable diesel pathway include all operations up to palm oil refining, since PFAD is treated as a co-product with refined palm oil (Xu et al., 2020).

Key stages for rapeseed and soybean to renewal diesel pathways cover material and energy use, discharges and emissions associated with rapeseed and soybean oil production (including rapeseed and soybean cultivation and transport to conversion mill). Rapeseed oil originates in Canada and Europe, and soybean oil in Brazil, USA, and Europe.

Raw materials transport

The feedstock (rendered fats and refined oils) is transported to the nearest port using trucks, shipped to Singapore where renewable diesel is produced, and stored in large storage tanks.

Renewable diesel production

Renewable diesel production involves two major processes:

i. **Pre-treatment of raw materials:** Once raw materials arrive at the renewable diesel production facility, impurities are removed using a pre-treatment process. Wastewater from the pre-treatment is treated in the refinery's wastewater treatment plant and solid waste is landfilled outside the production facility. Material and energy use and emissions related to the pre-treatment of raw materials are included.
ii. **Processing of renewable diesel using hydro-processing:** Pre-treated raw materials undergo the hydro-processing (aka hydrotreatment) process, where triglycerides of vegetable oils and animal fats are converted to saturated straight-chain hydrocarbons. The oxygen in triglycerides is converted to water, carbon monoxide, and carbon dioxide. The main product of this process is renewable diesel.

Distribution and use of renewable diesel

Renewable diesel is shipped to New Zealand (Auckland) and stored for distribution. The diesel will then be distributed to the fuel stations using tankers and combusted by AT bus fleet for public transport.

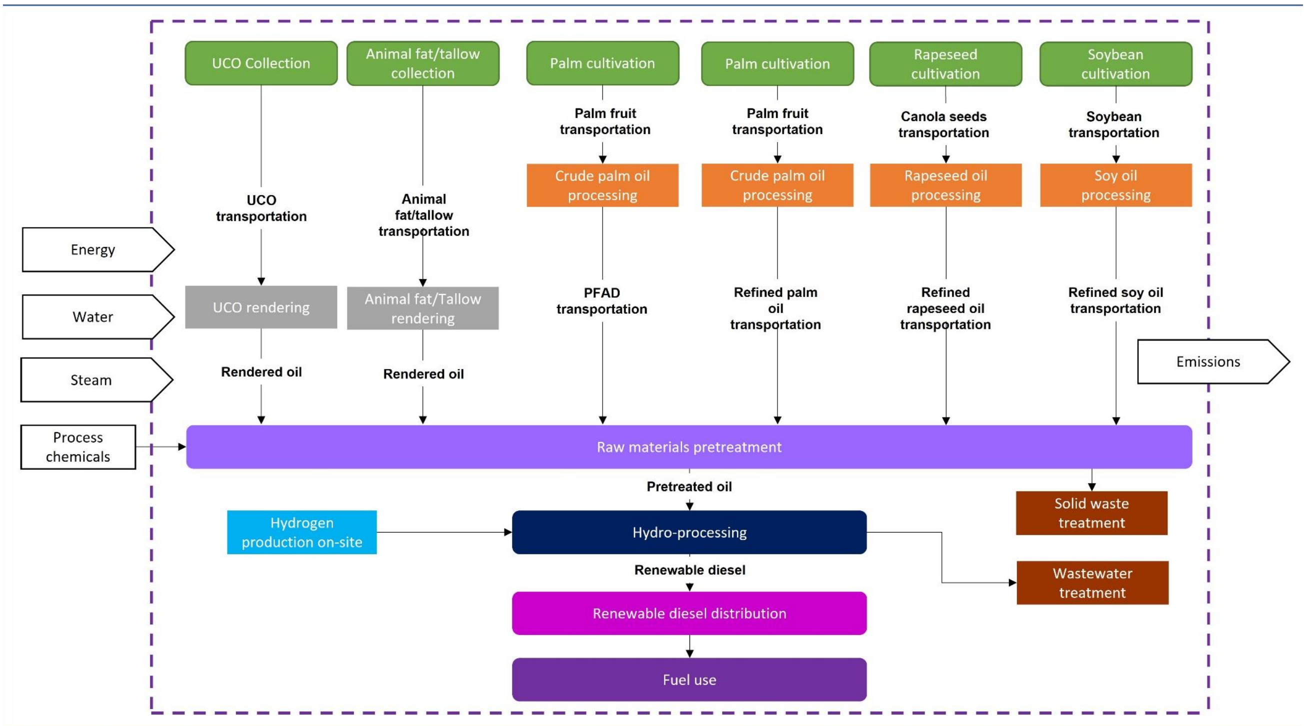

Figure 1: System boundary for renewable diesel LCA

Co-product Allocaton

Animal products (such as carcasses, heads, and feet) were treated as wastes from the slaughtering process. Therefore, upstream impacts related to animal breeding and slaughtering were excluded. Economic allocation was applied for the rendering process that produces rendered fat and meat bone meal, given there is economic value for both products, globally.

For oilseed crushing (palm, PFAD, soybean, and rapeseed) economic allocation was applied – but no allocation was used for the oil refining process, since there are no valuable co-products from the oil refining process. Similarly, no allocation was required for UCO, given there are no valuable co-products from the UCO rendering process.

Land Use and Land Use Change Modelling

We realise that the Sphera datasets had very low land use change impacts compared with published studies focusing on land use change associated with oil crops cultivation (palm, rapeseed, and soybean). We have addressed this by correcting the land use change impacts using the emissions estimated in the work commissioned by the European Commission (ECOFYS , 2015). However, we acknowledge that the land use change emissions presented in this study are the sum of direct and indirect emissions[1] (ECOFYS , 2015).

On the contrary, previous work suggests that palm oil certified by RSPO (Roundtable on Sustainable Palm Oil) has lower land use change impacts (Schmidt & De Rosa, 2019). This is not modelled in this study due to data limitations.

We also acknowledge that the effects of the land use modelling choice on the renewable diesel procurement decision would be minimal given Neste aims to reduce the share of conventional palm oil to 0% of its global renewable raw material inputs by the end of 2023 (NESTE, 2023).

[1] With an assumed 20-year amortisation

Results

Climate Change Impacts of Renewable Diesels

Carbon Footprint – Total (excl. biogenic)

Figure 1 shows the Carbon Footprint (CF)-Total (excl. biogenic) results for renewable diesels - range between15.3 and 391 g CO_2 eq. / MJ. From a climate change mitigation perspective, CF-Total (excl. biogenic) results are more relevant, since the carbon sequestered from the atmosphere is re-emitted over a short period.

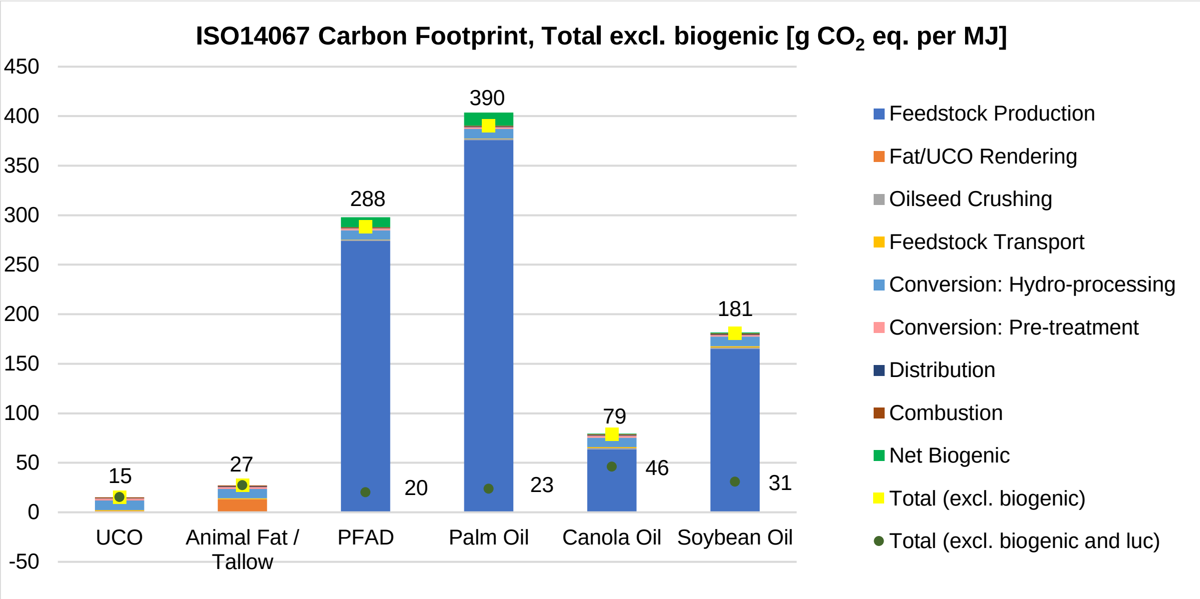

Figure 2: ISO 14067 Carbon Footprint (excl. biogenic) emission results for renewable diesel pathways

Palm oil derived renewable diesel shows the highest CF while UCO derived renewable diesel shows the lowest. Consistent with other studies (Nikander, 2008; Xu et al., 2022), vegetable oil (including PFAD) derived renewable diesels show higher impacts compared to renewable diesels derived from UCO and animal fat / tallow. This is mainly because of land use change impacts and how different feedstocks are modelled in this study. For example, all vegetable oil (including PFAD) derived renewable diesels include upstream impacts which include land use emissions whereas upstream impacts for UCO and animal fat / tallow are zero or negligible.

For vegetable oil derived renewable diesels (including PFAD), emissions across the feedstock and oil recovery vary considerably. These variations are due to the amount of feedstock, utilities required in each diesel production pathway, and net biogenic emissions. The variation in the feedstock transport emissions is explained by the differences in the distance between the location of feedstock production and the renewable diesel facility.

For processes such as conversion (pre-treatment and hydro-processing) and diesel distribution, emissions are the same across all renewable diesels. This is because the raw materials and utilities used in these processes are consistent across all renewable diesels.

Carbon footprint – Fossil

The CF-Fossil results for renewable diesels range from 15.3 to 45.8 g CO_2 eq. / MJ. Rapeseed oil derived renewable diesel shows the highest CF-Fossil results while UCO derived renewable diesel shows the lowest, which is different to the trend observed for CF-Total (excl. biogenic).

Carbon footprint – Land Use Change

Table 1 presents the land use change impacts associated with renewable diesel production in Singapore. Note that these emissions are modelled on the work commissioned by the European Commission (ECOFYS , 2015), and include both direct and indirect land use change emissions. Table 1 presents CF-Land Use Change for renewable diesels derived from vegetable oils (including PFAD) – ranging from 33 to 367 g CO_2 eq. / MJ. Palm oil derived renewal diesel showed the highest emissions while canola oil derived renewable diesel showed the lowest. These results clearly reflect the different agricultural land requirements for feedstock production, mainly in terms of agricultural expansion activities including deforestation. CF-Land Use Change for renewable diesels derived from UCO and animal fat / tallow are negligible. This is because the upstream impacts (including land use change) are modelled as zero or negligible.

Table 1: Detailed land use change emission results (g CO_2 eq. per MJ renewable diesel)

	Feedstock Production	Fat/UCO Rendering	Oilseed Crushing	Feedstock Transport	Conversion: Pre-treatment	Conversion: Hydro-processing	Diesel Distribution	Diesel Combustion	Total
UCO	0.000	0.001	0.000	0.005	0.000	0.000	0.001	0.000	0.006
Animal Fat / Tallow	0.000	0.003	0.000	0.010	0.000	0.000	0.001	0.000	0.014
PFAD	268	0.000	0.000	0.003	0.000	0.000	0.001	0.000	268
Palm Oil	367	0.000	0.000	0.004	0.000	0.000	0.001	0.000	367
Rapeseed oil	33	0.000	0.000	0.001	0.000	0.000	0.001	0.000	33
Soybean Oil	150	0.000	0.000	0.001	0.000	0.000	0.001	0.000	150

Carbon Footprint – Aviation

CF-Aviation results are negligible compared with CF-Fossil. CF-Aviation results for renewable diesels range from 1.96E-06 to 1.01E-05 g CO_2 eq. / MJ.

Hotspot Analysis

Most of the emissions come from the production stages (>90%). For UCO-derived renewable diesel, hydrogen production for hydro-processing is the largest contributor (60%). For animal fat / tallow derived renewable diesel, rendering is the largest contributor (49%), followed by hydrogen for hydro-processing (34%). For all other renewable diesels, feedstock cultivation is the largest contributor (81-96%) – mostly driven by land use change emissions, followed by hydrogen for hydro-processing (2.33-12%).

Comparison with fossil diesel

The CF-Total (excl. biogenic) of fossil diesel is 86 g CO_2 eq./MJ (Sphera, 2022), which is at least 1.08 times higher than the emissions of renewable diesels, except for soybean-, PFAD-, and palm oil-derived renewable diesels (see Table 2). Soybean-, PFAD-, and palm oil-derived renewable diesels are respectively 2.12, 3.37, and 4.56 times worse than fossil diesel. Overall, shifting to renewable diesel would emit at least 7.83% less greenhouse gases, except for soybean-, PFAD-, and palm oil-derived renewable diesels.

Table 2: Carbon Footprint-Total (excl. biogenic) results for renewable and fossil diesels (g CO_2 eq./MJ)

	UCO	Animal Fat / Tallow	PFAD	Palm Oil	Rapeseed Oil	Soybean Oil	Fossil Diesel
Feedstock	0.00	0.00	274	376	63.7	165	11.8
Fat/UCO Rendering	1.40	13.1	0.00	0.00	0.00	0.00	
Oilseed crushing	0.00	0.00	0.94	1.29	1.14	1.55	
Feedstock transport	1.00	1.01	0.28	0.33	1.12	1.30	
Pre-treatment	1.90	1.90	1.90	1.90	1.90	1.90	
Hydro-processing	9.43	9.43	9.43	9.43	9.43	9.43	
Distribution	0.54	0.54	0.54	0.54	0.54	0.54	0.928
Combustion	1.00	1.00	1.00	1.00	1.00	1.00	72.8
CF-Total excl. biogenic (g CO_2 eq./MJ)	15	27	289	390	79	181	86
Reduction potential (%)	82	68	-237	-356	7.83	-112	-

Discussion

The CF-Total (excl. biogenic) results for renewable diesels produced in Singapore range from 15 (UCO-derived) to 390 g CO_2 eq. / MJ (palm oil-derived). Virgin vegetable oil-derived (including PFAD) renewable diesels showed higher impacts compared to waste-derived diesel (UCO). This highlights the effects of land use change related to oil crops cultivation. This also highlights the influence of the allocation methods used for different feedstocks in the LCA model. For example, in this study, upstream feedstock production impacts for UCO are zero, because they were modelled as wastes.

Renewable diesels showed emission reduction potential, when compared with fossil diesel. UCO- and rapeseed oil-derived renewable diesels showed the highest (82%) and lowest (7.8%) reduction potential, respectively. In contrast, soybean-, PFAD-, and palm oil -derived renewable diesels are worse than fossil diesel, while PFAD-, and palm oil-derived renewable diesel emissions are largely driven by land use change emissions.

Conclusions and Recommendations

Overall, the results of this study indicate that production and use of renewable diesel in the AT bus fleet has some potential to mitigate GHG emissions in Aotearoa New Zealand. Specifically, renewable diesels derived from wastes (such as UCO and animal fats) have the greatest potential compared to renewable diesels derived from virgin vegetable oils. Note that rapeseed oil shows better potential, among renewable diesels derived from virgin vegetable oils.

Opportunities for reducing the overall GHG emissions of renewable diesels include:

- Mitigating land use change emissions from oil crop cultivation;
- Developing new or improving existing hydrogen production technologies for hydro-processing; and
- Identifying appropriate feedstock suppliers internationally.

To improve the accuracy of the results of the study, we recommend the following as future work:

- Source primary and latest data for renewable diesel production systems – primarily for Singapore;
- Perform scenario analysis to understand the effects of different modelling choices such as allocation, origins of feedstock, LCI and datasets; and
- Undertake further analysis to better understand the direct and indirect land use change impacts related to increased demand for renewable diesel in future – a Consequential LCA is an option (Ekvall, et al., 2016).

Acknowledgements

We thank Prof Sarah McLaren of Massey University for reviewing the technical content of this project.

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Environmental Product Declarations (EPDs) Abstracts

Holcim EPDs On-Demand Certification - An Australian first

Wednesday, 19th July - 16:00: Environmental Product Declarations (EPDs)

Mr. Evan Smith [1]

1. *Holcim*

Globally, the expectation to provide enhanced transparency and disclosure of environmental impacts, such as greenhouse gas (GHG) emissions has been growing. At the same time, the demand for construction materials is also growing because of worldwide population growth and an increase in urbanisation.

In an Australian first, Holcim achieved a Process Certification to provide Environmental Product Declarations (EPD) on-demand for its ready-mix concrete in 2021. This means Holcim can provide third-party verified environmental data for any project, any mix and at any stage of the project life cycle; giving customers insight into the real impacts of their projects for the first time.

After registering Australia's first EPD for ready-mix concrete, Holcim's Process EPD Certification was the next step in our transparency journey. This capability represents a significant step in the construction sectors' sustainability journey. Third-party verified data allows Holcim to work with its customers from tender through to design and construction to optimise ready-mix concrete mix designs and report on sustainability performance. Customers can now specify ready-mix concrete's environmental performance requirements in terms of kilogram-equivalent of carbon dioxide (kg CO2-e).

Certification was achieved by third party verification of Holcim Australia's EPD Process lifecycle model and internal organisational process by an ISO/IEC 17065 accredited certification body. Through the EPD Process Certification, Holcim was audited to verify compliance with the relevant standards and guidelines of the International EPD Programme and EPD Australasia.

The EPD Process Certification is a stamp of approval to produce compliant EPDs in-house, opening up significant capability and flexibility in producing and using life cycle impact data to inform our operations and our customers.

As a business, we have long led the way regarding transparency and third-party verification. Now, we plan to take the industry a step further and demonstrate how this transparency can translate into significant action.

Confessions of an EPD Verifier: How LCA Consultants Can Speed up the Verification Process of EPDs

Wednesday, 19th July - 16:00: Environmental Product Declarations (EPDs)

Mr. Andrew Moore [1]

1. *Life Cycle Logic*

This presentation provides recommendations on how LCA consultants can speed up the verification process of their EPDs. The importance of a well-defined scope, selection of appropriate data sources, consistent terminology, and accurate data is discussed. Practical recommendations include compliance with relevant standards, detailed documentation, transparent communication, and early engagement with verifiers. Efficient and robust verification is crucial for accurate and reliable EPDs.

By following the recommendations provided in this presentation, LCA consultants can speed up the verification process of their EPDs and enhance the credibility of their environmental performance claims.

A Global Overview of Capital Goods in Life Cycle Assessment

Wednesday, 19th July - 16:00: Environmental Product Declarations (EPDs)

Ms. Supriya Mahlan [1], ***Dr. Olubukola Tokede*** [1], ***Dr. Abdul-Manan Sadick*** [1]

1. Deakin University, School of Architecture and Built Environment

Capital goods form part of the Life Cycle Assessment (LCA) background process, crucial to performing the foreground functions. Several methodological issues exist in conducting life cycle studies, such as allocation, choice of impact assessment methods, limited development of some impact assessment indicators and the inclusion or exclusion of capital goods in life cycle inventory. Recent debates on the inclusion of capital goods in LCA-based studies have emerged primarily due to changes in the Product Category Rules of Construction Products. To date, no study has been conducted to systematically analyse the extant literature on the inclusion and implication of capital goods in the LCA of products and services. To identify the global research trends on capital goods, we systematically appraise the body of knowledge on capital goods to understand the relevance of capital goods in LCA.

Using the Scopus and Google Scholar databases and following the SALSA (search, appraisal, synthesis, and analysis) framework, 95 publications were reviewed and analysed. In addition, 23 papers were further analysed using content analysis. Findings suggest the limited availability of peer-reviewed literature on the relevance of capital goods in LCA. The majority of LCA-based studies on capital goods are from Europe. Sectors with a dominant share of research on capital goods include Agriculture, Energy and Waste management. Our review also suggests a lack of reliable data on capital goods and an absence of a comprehensive methodology to facilitate the collection and processing of data on capital goods. Issues with capital goods in LCA have been further exacerbated by notable inconsistencies in standards and a lack of consensus among the research community on the treatment of capital goods in LCA-based studies. It is recommended that future studies should investigate the relevance of capital goods in LCA studies to ensure data reliability and consistency in Environmental Product Declarations.

Should Capital Goods be included in Environmental Product Declarations (EPD)?

Wednesday, 19th July - 16:00: Environmental Product Declarations (EPDs)

***Dr. Olubukola Tokede* [1], *Mr. Rob Rouwette* [2]**

***1.** Deakin University, School of Architecture and Built Environment, **2.** Start2see Pty Ltd, Energetics*

Abstract

Over the last three decades, the inclusion of capital goods [A1] [A2] in Life Cycle Assessment (LCA) reports has been debated among scholars and practitioners in the LCA community. Investigating capital goods in Environmental Product Declaration (EPD) has become imperative because of changes in the Product Category Rules (PCR) for Construction Products. The updated PCR has led to capital goods data being implicitly required in the Life Cycle Inventory (LCI) of EPD as opposed to being explicitly excluded. Concerns regarding the inclusion or exclusion of capital goods in LCA-based studies have centred around the[A3] [A4] credibility of life cycle inventories in providing the required environmental information for comparing products and the significance of their influence on life cycle impact assessment (LCIA) indicators.

In our work, we assembled 38 construction products and extracted their unit processes based on the ecoinvent database, version 3.8, and analysed the impact of inclusion of capital goods based on the EN 15804+A2 impact assessment[A5] [A6] method. (Our research [A7] [A8] stressed that when capital goods are included based on currently available background LCI data, they mostly have a limited effect (<10% increase[A9] [A10]) on Climate change (GWP), but they can have a very significant effect (>100% increase) on Abiotic depletion,[A11] [A12] minerals and metals ($ADP_{minerals\&metals}$), land use (LUP), and/or human toxicity (HTP) indicators. Moreover, these results are driven by questionable LCI data. We conclude that the requirement for inclusion of capital goods leads to a major conundrum for LCA practitioners. We suggest that capital goods are excluded from EPD until there is better refinement and improvement of the quality of LCI datasets and EPD programs provide clearer guidance on dealing with capital goods.

Environmental Product Declaration of Insulated Concrete Form System

Wednesday, 19th July - 16:00: Environmental Product Declarations (EPDs)

Ms. Emma Green [1]
1. *ERM*

Climate change is the most pressing environmental issue of our time. It is estimated that 39% of global greenhouse energy related carbon emissions contributing to climate change arise because of the built environment (Ref). Insulated Concrete Formworks (ICFs) provide an alternative to traditional building methods, comprising hollow blocks joined by connectors to be stacked in place and filled with concrete and reinforcing steel. The use of ICF products in construction significantly reduces the operational carbon associated with buildings.
Tremco (NUDURA), a Canadian ICF manufacturer, commissioned ERM to conduct an LCA and associated EPD for the promotion of the environmental credentials of their products and gain materials credits with their LEED and BREEAM certifications. NUDURA's ICF system comprises 100% recycled expandable polystyrene, polypropylene and steel hinges, therefore reducing the environmental footprint further. The study was peer reviewed as part of the EPD verification with results to be published in the ISO 14025 format for Type III EPDs via the International EPD system.
This paper reviews the results of the NUDURA ICF system in the context of an EPD, showing the environmental footprint of the first ICF published through the International EPD system.
The Core environmental indicators for 1 piece of ICF system (A1-A3, C1-C4, D), as per EN15804+A2 shown below:
GWP – Total: 4.99 kg CO_2 eq.
ODP: 5.02E-07 kg CFC11 eq
AP: 1.23E-07 mol H+ eq
EP (terrestrial): 3.45E-02 mol N eq
POCP: 0.156 kg NMVOC eq
ADP (resources): 9.36E-06 kg Sb eq
ADP (fossil): 53.5 MJ
WDP: -9.11E-01 m3 depriv.

Robust LCA-based tools and data for meeting compliance obligations and guiding international trade policy

Wednesday, 19th July - 16:00: Environmental Product Declarations (EPDs)

Dr. Zhong Xiang Cheah [1]

1. Integrity Ag & Environment

The Environmental Footprint (EF) is a multi-criteria LCA-based initiative by the European Commission to guide product and organisational policies and investments towards environmental sustainability goals. It sets a precedent in mandating environmental performance credentials reporting for a collective of jurisdictions under a unified label.

We demonstrated how the Euro-centric methods currently applied in EF lack the resolution to properly represent the varied environments and climate of Australia's primary production regions. These methodological problems have material effects on some industries, and when confounded with subjective values and bias in weighting, elevated the impact of Australian products disproportionately to some competitors. The absence of some important impact categories such as microplastic leakage and plastic/solid waste skew scores in favour of synthetic materials production. The lack of a system to recognise the inherent value of natural/renewable production systems or regenerative efforts is a major limitation. Mandatory datasets required for LCA modelling have been shown to be very disadvantageous to the Australian industry. EF perverse outcomes by over-or-under emphasising environmental problems in global supply chains. Regionally relevant characterisation and weighting factors and representative datasets are urgently needed to mitigate these issues. We showcase how a representative official EF-compliant dataset was developed for an industry, reducing its reported impact by 11-fold.

The EF initiative is being closely followed by governments and corporates worldwide. Similar schemes might be mimicked by other jurisdictions, increasing the inherent compliance risk embedded in global trade. Exposure to such regulatory burdens presents a significant barrier and dire consequences to some industries. However, for some industries that are export-focused, misinformed decisions by entire economies based on an "erroneous" sustainability score could be an existential threat. Collaborative efforts by industry, academia, NGOs/NPOs, media, and government to advocate for a meaningful comparison of environmental performance are critical in levelling the playing field for Australia.

Environmental Product Declarations (EPDs) Extended Abstracts

A Global Overview of Capital Goods in Life Cycle Assessment

Supriya Mahlan, Olubukola Tokede, Sadick Abdul-Manan
Deakin University

Abstract

Capital goods form part of the Life Cycle Assessment (LCA) background process, crucial to performing the foreground functions. Several methodological issues exist in conducting life cycle studies, such as allocation, choice of impact assessment methods, limited development of some impact assessment indicators and the inclusion or exclusion of capital goods in life cycle inventory. Recent debates on the inclusion of capital goods in LCA-based studies have emerged primarily due to changes in the Product Category Rules of Construction Products. To date, no study has been conducted to systematically analyse the extant literature on the inclusion and implication of capital goods in the LCA of products and services. To identify the global research trends on capital goods, we systematically appraise the body of knowledge on capital goods to understand the relevance of capital goods in LCA.

Using the Scopus and Google Scholar databases and following the SALSA (search, appraisal, synthesis, and analysis) framework, 95 publications were reviewed and analysed. In addition, 23 papers were further analysed using content analysis. Findings suggest the limited availability of peer-reviewed literature on the relevance of capital goods in LCA. The majority of LCA-based studies on capital goods are from Europe. Sectors with a dominant share of research on capital goods include Agriculture, Energy and Waste management. Our review also suggests a lack of reliable data on capital goods and an absence of a comprehensive methodology to facilitate the collection and processing of data on capital goods. Issues with capital goods in LCA have been further exacerbated by notable inconsistencies in standards and a lack of consensus among the research community on the treatment of capital goods in LCA-based studies. It is recommended that future studies should investigate the relevance of capital goods in LCA studies to ensure data reliability and consistency in Environmental Product Declarations.

Keywords: Capital Goods, Environmental Product Declarations, Impact Assessment Indicator, Product Category Rules**.**

1. Introduction

Capital goods form part of the Life Cycle Assessment (LCA) background processes required in completing the data inventory for products and services. Despite the evolution of life cycle inventories across many sectors and regions, data on capital goods remain limited and obsolete (Brogaard et al., 2015; Silva et al., 2018). A capital good is a good used in the production of products and services that outlives this production process (Agez et al., 2022). Capital goods could be broadly categorised into buildings, machinery, energy infrastructure,(Brogaard et al. 2013) and transport infrastructure. While there is consensus that the inclusion of capital goods may be important in LCA studies under specific circumstances (Emami et al. 2019; Lasvaux et al. 2015), it is doubtful whether EPDs should mandate the inclusion of capital goods in all LCA-based assessments of products and services. The inclusion of capital goods remains a debated topic for industry and academia in the LCA community (Eickelkamp, 2015; Lasvaux et al., 2015). Inclusion of capital goods in LCAs and Environmental Product Declarions (EPDs) more specifically is a methodological choice, and there exist certain guidelines to support the inclusion or exclusion of capital.

Debates on capital goods can be traced back to the life cycle assessment workshop held in 1991 in Leiden, the Netherlands, where an agreement was held that capital goods should be included in comparative LCAs of two processes in which the number of investments would be clearly and significantly different (Huisingh, 1992). Subsequent interventions from the Society of Environmental Toxicology and Chemistry (SETAC) code of practice (SETAC, 1993) followed by the standards ISO 14040:2006 (ISO, 2006a) up until its latest amended version ISO 14040: 2006+A1 2020 (ISO, 2020), and EN15084 (CEN, 2019, 2013) have hinted on the relevance of capital goods in LCA contexts but have not established compelling basis to guide LCA practictioners in the development of EPDs.

However, recent changes in the current Product Category Rules (PCRs) for construction products PCR2019:14 section 4.3.1 (Environdec, 2019) state, “inventory flows from infrastructure, construction, and production equipment, and tools, that are not directly consumed in the production process can be excluded from the life cycle inventory if it is not known to have the potential to cause significant impact”. This subtle change to implicitly include capital goods in the PCR has created a connudrum for LCA practitioners given the limited data on capital goods in available Life Cycle Inventory (LCI) datasets. LCA practitioners rarely collect primary data on capital goods and even much fewer practitioners check whether the core process infrastructure data are representative of their Product system. In addition, out of the mainstream LCA software programs (i.e., GaBi, SimaPro, OpenLCA), only SimaPro seems to provide for the functionality to allow the inclusion or exclusion of capital goods leading to incomparability in EPDs developed using different software tools.

It is, however, noteworthy that the recent growth in the demand for Environmental Product Declarations (EPDs) has stemmed from the quest for transparency in achieving environmental comparisons of products (Andersen et al. 2019; Ibáñez-Forés et al. 2016; Passer et al. 2015). In light of these, this research aims to investigate how capital goods have been treated globally to better understand the relevance of capital goods in EPDs. The findings of this research will present the state-of-the-art of capital goods' inclusion in the LCA studies and will suggest future directions for overcoming the challenges associated with capital goods data in LCI.

2. Material and methods

A literature review is a transparent, rigorous and reproducible method for identifying, evaluating and interpreting the existing body of original works (Fink, 2019). A systematic review has been employed in this study, as it is considered a transparent and replicable method to harmonise the gaps in knowledge and evolution across the discipline (Chalmers, 1993). Furthermore, a systematic analysis of the literature on capital goods, will provide a compelling assessment of what remains unknown, and establish future research prospects (Grant and Booth, 2009). The SALSA (search, appraisal, synthesis and analysis) framework provides a descriptive /inferential method used in summarising data from numerous studies that can be integrated to reach a meaningful and coherent body of knowledge (Mengist et al., 2020) and is used in this study as presented in Figure 1.

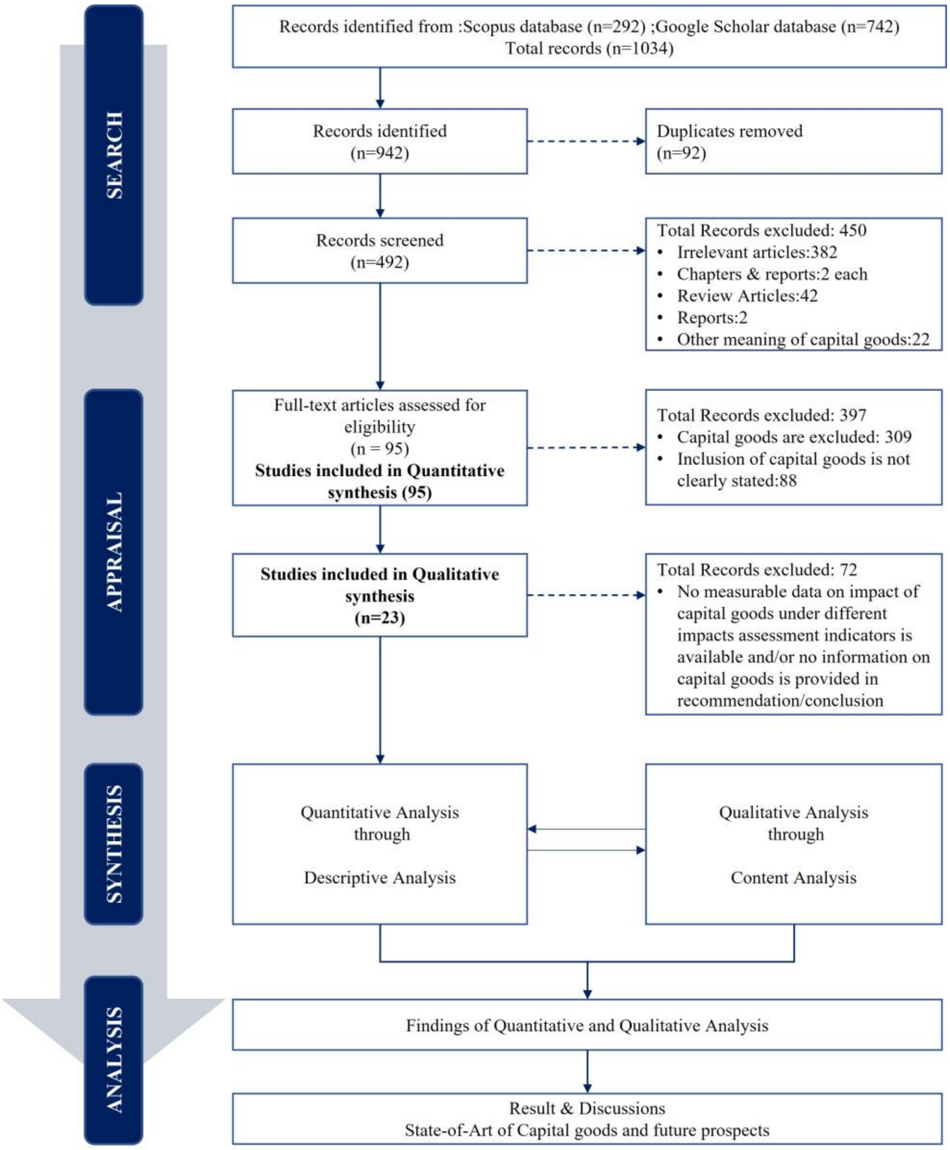

Figure 1: Literature Review Landscape

Two proprietary databases (Scopus, and Google Scholar) were used to identify studies on capital goods in life cycle assessment. The search criteria considered published peer-reviewed journal articles from the 1992 (i.e given the first discussion by Huisingh, 1992) until 10 May 2022. The first step to gaining knowledge about the role of capital goods in LCA was to focus on peer-reviewed articles to maintain the review quality (Grant and Booth, 2009). Using the Search criteria, “Life Cycle Assessment”, “Capital Goods”, and “Fixed Assests” in the “ALL FIELDS” in Scopus yielded 292 records of bibliometric data related to published studies . To broadly cover the existing literature on the role of capital goods in life cycle assessment, the same search terms were used in Google Scholar as it lists all available electronic publications on a subject (Falagas et al., 2008). The search resulted in 742 publications. Overall, all the articles were imported into Microsoft Excel, having 1034 articles altogether, followed by removing the duplicates (92 publications), leading to 942 articles. Afterwards, these articles were filtered to: exclude irrelevant publications, book chapters, reports, review articles, and articles having other meanings of capital goods. Altogether 492 papers were available, after following the above-stated screening steps, that mentioned capital goods anywhere in the article and used LCA methodology for their studies. However, by further examining the articles, it was observed that not all these publications had included capital goods in LCA. Based on this observation, selected 492 papers were reviewed and re-categorised under (a) capital goods are included, (b) capital goods are intentionally excluded, or the inclusion of capital goods is not clearly stated. As a result, the other 397 papers were considered irrelevant (falling in category b) for this review. Finally, the remaining 95 publications were integrated, appraised, synthesised, and analysed to assemble this review. All 95 selected publications were analysed through descriptive analysis. In addition, 23 peer-reviewed studies were narrowed down to conduct a content analysis having selection criteria as the articles should have (a) measurable extractable data on the impact of the inclusion of capital goods in the LCA study under different impact assessment indicators and (b) conclusion or recommendation related to the inclusion of capital goods. To further enhance the review's integrity, the relevant LCA standards related to capital goods were reviewed.

3. Results, Analysis and Discussions

3.1 Descriptive Analysis

The overall trend of studies published till 2022 that were selected for this review is presented in Figure 1. The lowest number of articles were published in the year 2005 and 2008. Since 2014, more than four articles have been published annually. The highest number of publications have been published in the year 2020. As the latest literature for this study was searched on 10 May 2022, which was only till the half of the year 2022, more articles could have been published this year after this review based on the forecast trendline shown in Figure 1.

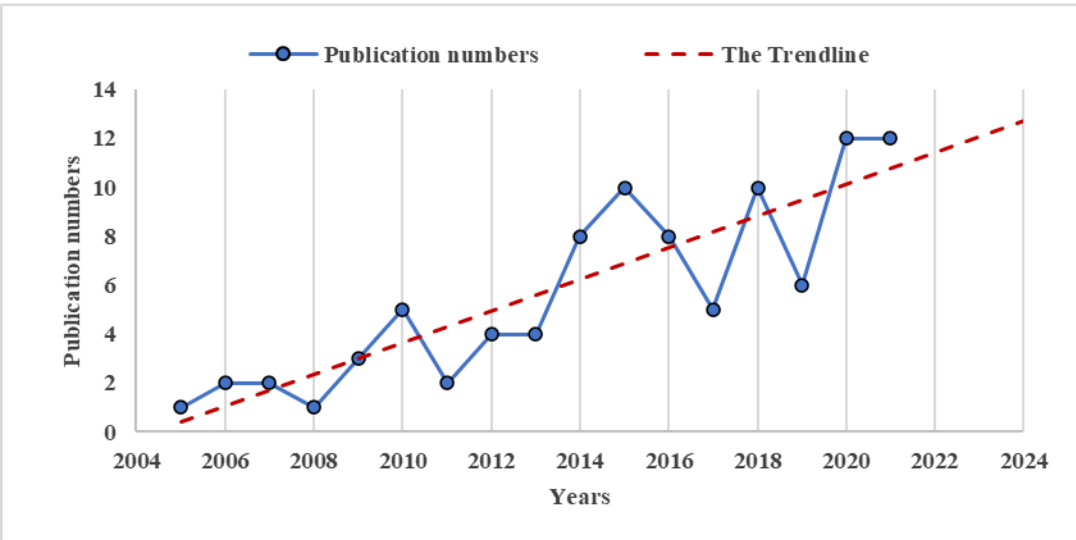

Figure 2: Yearly publications distribution related to Capital Goods

The academic publications from different countries towards the role of capital goods in LCA were investigated. The highest number of studies related to capital goods in LCA were from Europe, holding a significant share of 73% of the total literature sample, followed by The Unites States (18%), Malaysia (4%) and Australia (3%). Peru and Japan have only two and one publication, respectively, as presented in Figure 2.

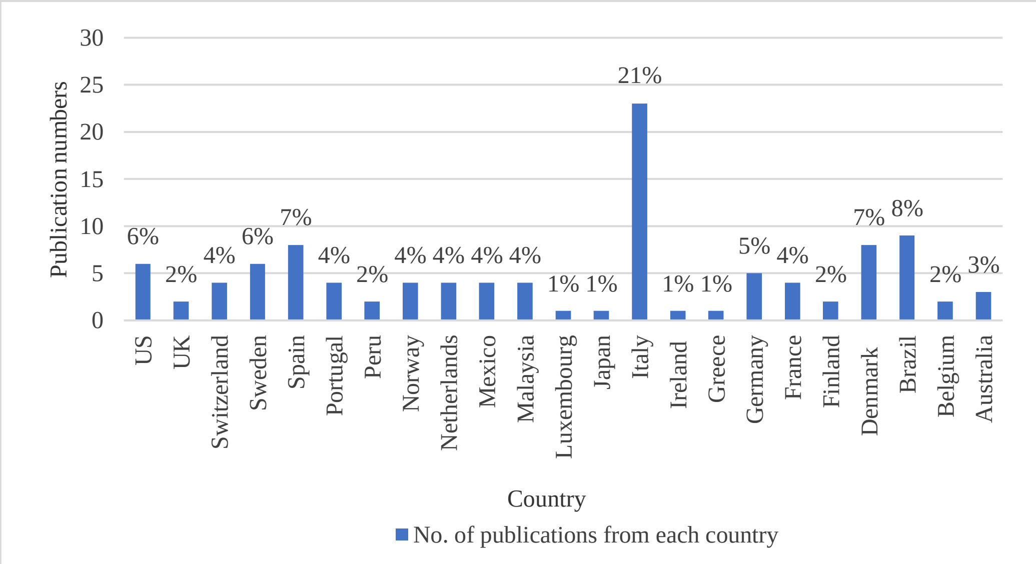

Figure 3: Geographical distribution of sample publications

From a sectoral perspective, as shown in Figure 3, the agriculture sector was dominant having 28 publications, followed by Energy, Waste Management, and Food sector with 21, 15 and 14 publications respectively. Construction, manufacturing, and studies having included multi-sectors had publications ranging between 5 and 10 papers while Utility, Electronics and Tourism sectors had less than five publications each related to capital goods.

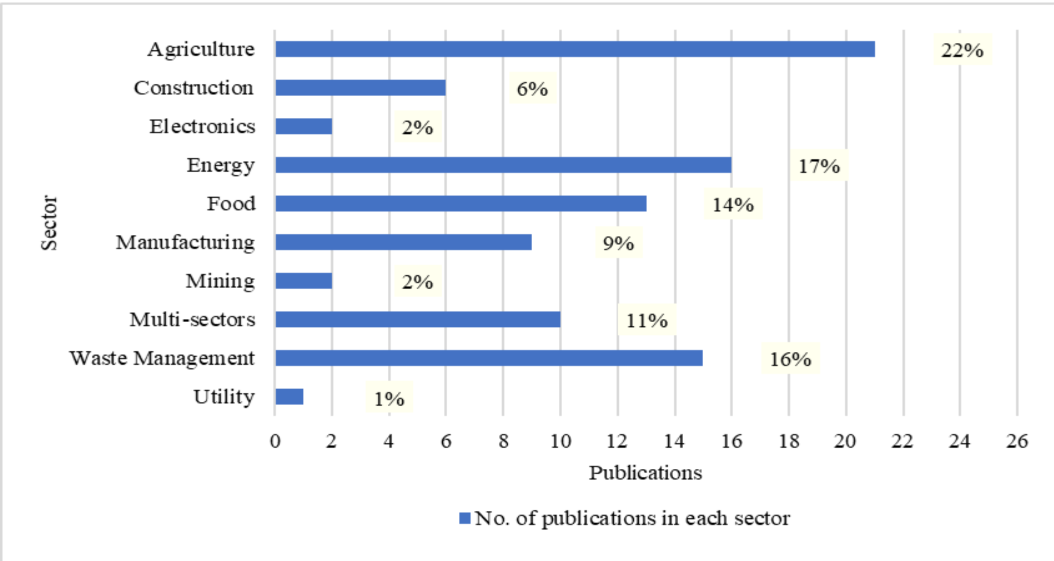

Figure 4: Sectoral distribution of sample publications

3.2 Content Analysis

Following the descriptive analysis, content analysis was conducted for the selected literature based on interpreting inductively developed categories to find the research gap. Content analysis allows the generation of salient concepts by treating the textual data into small pieces of information under different codes based on the meaning of the information. (Kleinheksel et al., 2020). Figure 4 provides an overview of the outcome of the research work and highlights the key issues related to capital goods as well as opportunities to mitigate challenges associated with inclusion of capital goods in LCA-context. The following sections provide an explanation of these key issues to support the proposed framework.

3.2.1 Lack of primary Data

The scarcity of primary data for LCIs in a LCA primarily hinders the inclusion of capital goods in LCAs. A lack of interest of the practitioners in collecting data for capital goods is a possible explanation for the scarcity of data. A significant factor for this indifference is the amount of time, cost and effort required to collect primary data on capital goods (Brogaard et al., 2015), leading to an increased workload and expenses for a practitioner and, in some cases, the data collection for capital goods becomes more rigorous and the benefit of inclusion may be minimal in developing EPDs of products (Silva et al., 2018). Moreover, no comprehensive method is available for collecting primary data on capital goods. With the increasing maturity of artificial intelligence techniques, machine learning has been implemented across several research domains through various algorithms such as data mining, image processing and predictive analysis (Kumar et al., 2020). There is scope to explore the application of artificial intelligence in collecting primary data on capital goods.

3.2.2 *Incorrect Databases and software*

In the absence of primary data to include capital goods in any LCA, practitioners either completely exclude capital goods from LCIs or rely on existing generic databases such as EcoInvent and GaBi. The data registered in the current database is only roughly estimated due to the complexity of quantifying capital goods (Wernet et al., 2016). For instance, in Ecoinvent, the round kiln is modelled by a "generic heavy industrial machine" in clinker manufacturing based on a rock crusher's specifications (Kellenberger et al., 2007). Moreover, the geographical, technological, and time-related representativeness of the existing generic databases is contentious (Weidema et al., 2013). For example, the buildings (building multi-storage and buildings hall-steel, construction hall-wooden) modelled in Ecoinvent as input for capital infrastructure are prototypical of 1927 and 1972. These buildings are not representative of today's infrastructure and need to be updated with current data on infrastructure (PRe Sustainability, 2021). Finally, even in the most comprehensive database like Ecoinvent (having more than 4000 datasets), not all processes are available, requiring proxy data to be used. Moreover, there are variations in methodologies, completeness, and transparency between different databases, hence LCA practitioners find inclusion of capital goods challenging and in some instances, impossible.

3.2.3 *Uncertainty in LCA outcomes due to the inclusion of capital goods*

The studies that have included capital goods have reported concern regarding the data quality, availability, and representativeness leading to uncertainty on existing datasets in life cycle inventories of capital goods. This indicates the need for a more robust life cycle inventory data to dispense reliable outcomes. According to Passer et al. (2015), one possible way to address data reliability is to develop more robust LCI data. In the 57th Life Cycle Assessment forum held in Dec 2014, Frischknecht et al. (2015) highlighted that background life cycle inventory data is one of the most urgent elements that need to be harmonised and recommended moving from generic data to actual data as far as possible.

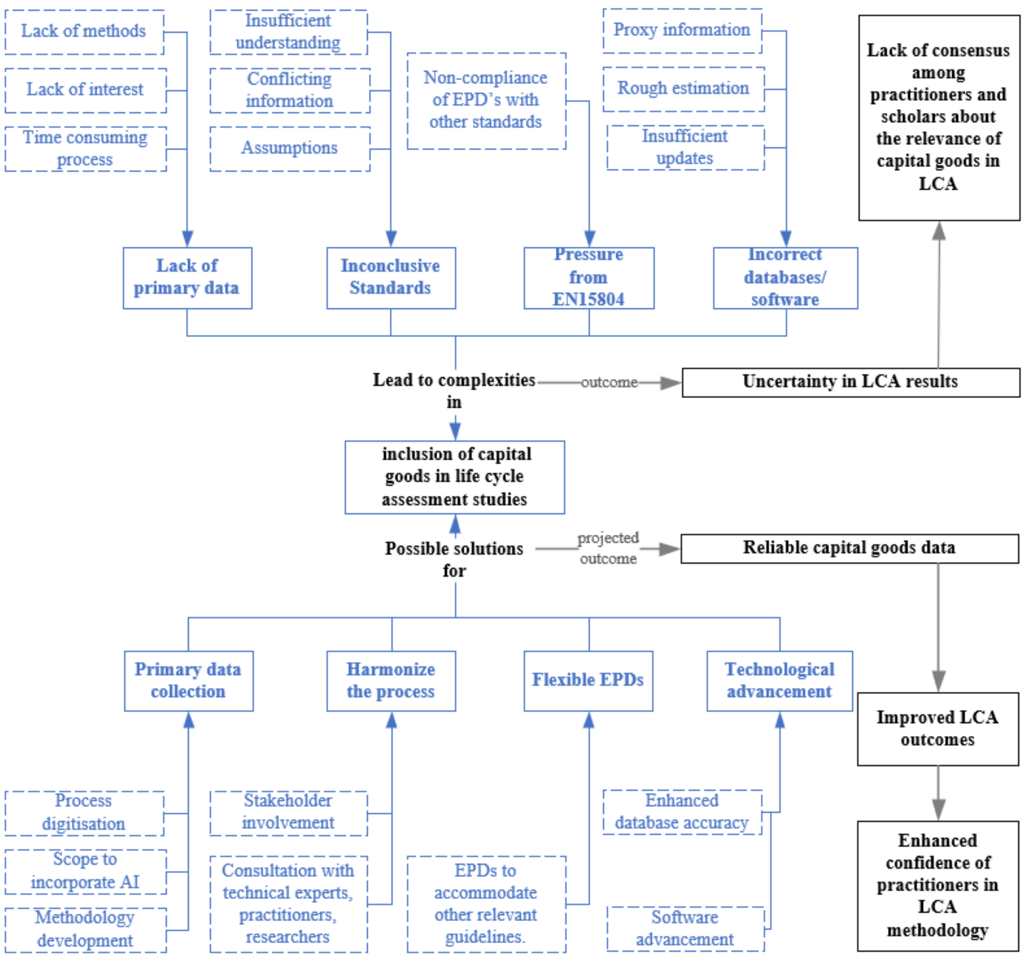

Figure 5: State-of-Art of Capital Goods and future prospects

3.2.4 Discord among authors' recommendations

There is a lack of consensus in the research community and practitioners about the relevance of capital goods in a life cycle study. Researchers and practitioners hold varying views on the inclusion of capital goods. Few researchers (Brogaard et al., 2013; Brogaard and Christensen, 2016; Corradini et al., 2019; Klint and Peters, 2021) acknowledge the importance of including capital goods, while others (Beausang et al., 2020; Kulak et al., 2016; Silva et al., 2018; Subramaniam et al., 2010) believed a limited value addition in including capital goods. Frischknecht et al. (2007) indicated a lack of clear understating of the importance of capital goods and raised the concern of neglecting capital goods in LCA. Finkbeiner (2009) pointed out the treatment of capital goods as a challenge and an underdeveloped research field. Eickelkamp (2015) noticed an insufficient investigation of the importance of fixed assets in life cycle studies.

3.3 Review of Standards/Reports related to capital goods.

It is apparent that different LCA Standards/Reports recommend the inclusion or exclusion of capital goods in LCAs, leading to a conundrum for LCA practitioners. There is an apparent lack of stringent guidelines on treating capital goods in the relevant standards related to including or excluding capital goods in assessing the environmental footprint of any product or service. SETAC Code of Practice (SETAC, 1993) and PAS 2050 (BSI, 2011) recommended an explicit exclusion of capital goods in evaluating the environmental impacts of any product or system under study; similarly, the GHG protocol (WBCSD, 2009) mentioned that the inclusion of capital goods is not required. Only ISO 14040 (ISO, 2006a) has roughly suggested considering capital goods without mandatory and clear guidelines. This disparity in treating capital goods among different standards exhibits an infirm understanding of the significance of capital goods in assessing a product's or service's environmental footprint from a life cycle perspective.

Surprisingly, no guidelines are available on treating capital goods in EN15084 (CEN, 2013). However, EN15084 is not a standard but an impact assessment method. Following the EN15084 impact assessment method is mandatory when producing Environmental Product Declarations (EPDs) for construction products. The guidelines related to capital goods for EPDs are available in Product Category Rules (Environdec, 2019, 2012). PCRs are a set of specific rules, requirements, and guidelines for developing Type III environmental declarations (ISO, 2006b). According to PCR 2012, presented in Table-2, impacts related to capital goods need not be accounted for in the life cycle inventory. However, the recent change in PCR 2019:14 (Environdec, 2019) to implicitly include capital goods, without any clarification on how and on what metrics "significant impact" can be quantified, has initiated a dialogue for the inclusion of capital goods. This latest radical change with limited information and understanding of capital goods necessitated an urgent need to better understand capital goods and its relevance in a life cycle study, especially for the construction sector.

Table 1: Changing requirements of PCRs

Product Category Rule	Stance on Capital Goods	Reference
PCR 2012:01	[7.5.4 (page 16/53)]: <u>Environmental impact</u> from infrastructure, construction, production equipment, and tools that are not directly consumed in the production process are not accounted for in LCI.	(Environdec, 2012)
PCR 2019:14	[4.3.1 (page 13/30)]: <u>Inventory flows</u> from infrastructure, construction, production equipment, and tools that are not directly consumed in the production process can be excluded from the LCI, if it is not known to have the potential to cause significant impact.	(Environdec, 2019)

4. Conclusion and Recommendation (Implications for future research)

The study has been instrumental in identifying global research trends on capital goods, with a view to understand the relevance of capital goods in LCA. It has also identified the challenges related to the inclusion of capital goods in LCA studies through a systematic literature review. Overall, the findings indicate that the amount of peer-reviewed literature on the role of capital goods in LCA is limited. Although an increasing trend from the last decade is observed, the distribution of literature on capital goods is concentrated in Europe, indicating the limited research on capital goods in other developed countries and, more acutely, in the developing world.

The results from content analysis highlighted significant issues leading to complexities in including capital goods in LCAs and EPDs in general. Firstly there are substantial challenges with availability of capital goods data in life cycle inventories. The primary data for capital goods are unavailable due to its complex and time-consuming estimation and lack of a well-established methodology to quantify them. Also, existing databases contain limited and in some cases obsolete data on capital goods. There is need to promote primary data collection on capital goods across different sectors and to improve the quality of datasets in proprietary LCI databases. It can be anticipated that data collection can be improved using advanced AI techniques.

Secondly, the provision of standards related to capital goods is ambiguous and invalidated, leading to varying practices among LCA practitioners. In the absence of stringent guidelines, the inclusion of capital goods has become a subjective choice leading to dilemmas for LCA practitioners on how to position capital goods in LCIs. Moreover, with the recent changes in PCR 2014 guidelines, it has become imperative to harmonise the standards through stakeholder involvement and to look for ways to provide clear guidelines on how to incorporate,or not, capital goods in any LCA study. Engagement of the stakeholders, experts, LCA practitioners, reviewers and researchers will support harmonising the standards pertaining to inclusion of capital goods in EPDs

Finally, uncertainty in LCA outcomes due to the inclusion of capital goods has potential to achieve inconsistent findings and recommendations. In light of these, addressing the issues with capital goods is fundamental to ensure that LCA results are comparable and verifiable. Despite continuous developments and improvement in LCA methodology since its advent, capital goods remain a less researched area. The study has outlined significant challenges concerning the inclusion of capital goods and suggested possible solutions (Figure 5). The challenges are complex and intricate and require support from the LCA community. An integrated outlook of the identified challenges and possible solutions to address the challenges on capital goods has been provided to highlight the limitations in background LCI data across multiple sectors. Thus, this study offers a roadmap and state-of-the-art assessment of capital goods research, and suggests some way forward to overcome the difficulties regarding the availability and reliability of capital goods data in LCA studies.

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Should Capital Goods be included in Environmental Product Declarations (EPDs)?

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1. Introduction

Capital goods are physical assets used in the manufacture of ‘products and/or services’ which could include the ‘operation’ aspects of the process chain as well as the accessories used to run the process of manufacturing the product and/or services (Frischknecht et al., 2007). Capital goods are distinctive in that they outlive the production process (Agez et al., 2022). The UNEP/SETAC (2011) recognised that capital goods could include cars, manufacturing machinery, factory halls, power plants, transmission lines, pipelines, roads and sewage systems. Capital goods can be broadly categorised into buildings, machinery, energy infrastructure, and transport infrastructure (Brogaard et al., 2013). While there is consensus that the inclusion of capital goods may be important in life cycle assessment (LCA) studies under specific circumstances (Emami et al., 2019; Lasvaux et al., 2015), it is doubtful whether Environmental Product Declaration (EPD) should mandate the inclusion of capital goods in all LCA-based assessments of products and services.

Across relevant literature, which has included capital goods in their studies, there are notable findings, and, in some cases, conflicting recommendations. For instance, both Frischknecht et al., (2007) and Eickelkamp (2015) advocate for the inclusion of capital goods in LCA, while Silva et al., (2018) recommend the exclusion of capital goods in LCA until more reliable data becomes available. It has also been postulated by Brogaard et al., (2013) and Eickelkamp (2015) that machinery and buildings are the most influential capital goods, and hence transport and energy infrastructure may have lower environmental impacts. In practice, however, LCA practitioners rarely collect primary data on capital goods and even much fewer practitioners check whether data on the core process infrastructure are representative of their product system.

Perhaps the earliest reference in the literature flowed on from the LCA workshop held in 1991 in Leiden, the Netherlands, where there was an agreement that “capital goods should be included in a comparative LCA of two processes in which the number of investments would be clearly and significantly different” (Huisingh and SETAC - SOetAc-E, 1991, p. 6). The justification in the 1991 workshop (i.e., including when investments differ) seems to have been lost in subsequent standards. Standards and guidelines addressing the inclusion of capital goods have barely evolved since then. From the LCA guidelines published by the Society of Environmental Toxicology and Chemistry (SETAC) in the 1990s, through the ISO 14040 series (ISO-14040, 2006; ISO-14044, 2006) and EN 15804 standards (CEN, 2019, 2013) and up until the current day Product Category Rules (PCR) (Environdec, 2021) there has been little or no justification for inclusion of capital goods and infrastructure elements provided. For example, there is only one reference to capital equipment in ISO-14040 (2006) (section 5.2.3 System boundaries), which mentions that the manufacture, maintenance, and decommissioning of capital equipment should be taken into consideration when setting the system boundary. There is no further mention, let alone guidance, of capital goods and infrastructure in ISO 14040 and ISO 14044, nor in EN 15804 standards.

For us, the more compelling basis for investigating the role of capital goods in EPD, and LCA in general, is the changing requirement within the PCR for Construction Products PCR2019:14 section 4.3.1 (Environdec, 2021). The wording has changed from “environmental impact from infrastructure, construction, and production equipment, and tools, that are not directly consumed in the production process, are not accounted for in the life cycle inventory" (Environdec, 2020) to “inventory flows from infrastructure, construction, production, equipment and tools…can be excluded from the life cycle inventory, if it is not known to have the potential to cause significant impact” (Environdec, 2021, p. 13). The available standard for EPD (ISO-14025, 2006) and accompanying documents such as the General Programme Instructions (GPI) have not provided clarity as to the justification for including or excluding capital goods, or thresholds to define “the potential to cause significant impact”.

Given the limited understanding of the treatment of capital goods in LCA settings, the research seeks to answer the question – what is the scope and impact of capital goods on EPD outcomes of construction products? In addressing this question, this research aims to investigate the impact of the inclusion of capital goods inventory data in construction products using ecoinvent version 3.8. Using 38 construction products from the background LCI database, we analysed the impact of inclusion/exclusion of capital goods and infrastructure based on the EN 15804+A2 impact assessment method.

2. Material and methods

In our work, we selected 38 common construction products – See Appendix 1 (that we believe show a reasonable cross-section of construction materials) and extracted their unit processes based on the ecoinvent database, version 3.8. Each product selected had a geographical scope of “Rest-of-the-World (RoW)” and an allocation system “cut-off by classification” to represent the allocation approach typically used in EPDs. Following this selection, each unit process was assessed against the EN 15804+A2 core plus additional indicators with the inclusion and exclusion of capital goods respectively. The LCIA method is mandated by EN 15804+A2. As our focus is on EPDs, this method - and only this method - is relevant to us. In SimaPro, there is a switch that allows the inclusion/exclusion of capital goods. A simple mathematical formula was then developed, in line with previous work by Frischknecht et al. (2007) to calculate the percentage increase of capital goods compared to impacts without capital goods for each impact category and all construction products. The percentage increases were categorised into four ranges, 0 < 10%, 10% ≤ 25%, 25% < 100%, and ≥ 100%, similar to previous work by Lasvaux et al. (2015). A share close to 0% indicates a negligible relevance of capital goods, while the converse reflects a significant contribution of capital goods (Frischknecht et al. 2007). The sensitivity analysis of the data was not reported in this paper, although a more comprehensive uncertainty analyses has been conducted in a more extensive version of the study.

3. Results and Discussion

3.1 Effect of Capital Goods on Life Cycle Impact Assessment (LCIA) Indicators

Based on our assessment of 38 selected construction products, we found that every single product experiences an increase caused by the inclusion of capital goods of over 25% in at least one LCIA indicator. Figure 1 shows that the three environmental impact indicators that are most likely to be heavily influenced (i.e., increase in impacts by over 100%) by the inclusion of capital goods are Abiotic Depletion Potential, minerals and metals ($ADP_{minerals\&metals}$) (79% of products), Land Use (SQP) (42%) and Human Toxicity Potential (HTP-c) (24%). Equally, Figure 1 shows the three environmental impact indicators that are least likely to be materially influenced (i.e., increase in impacts of less than 10%) by the inclusion of capital goods are Abiotic Depletion Potential, fossil (ADP_{fossil}) (87% of products), Global Warming Potential (GWP) (84%) and Water use (WDP) (79%). It is worth mentioning that our grouping of all results with an increase of more than 100% hides the fact that for almost half the analysed products we see an increase of over 1,000% in the $ADP_{minerals\&metals}$ indicator, with a maximum increase of 13,508% (!).

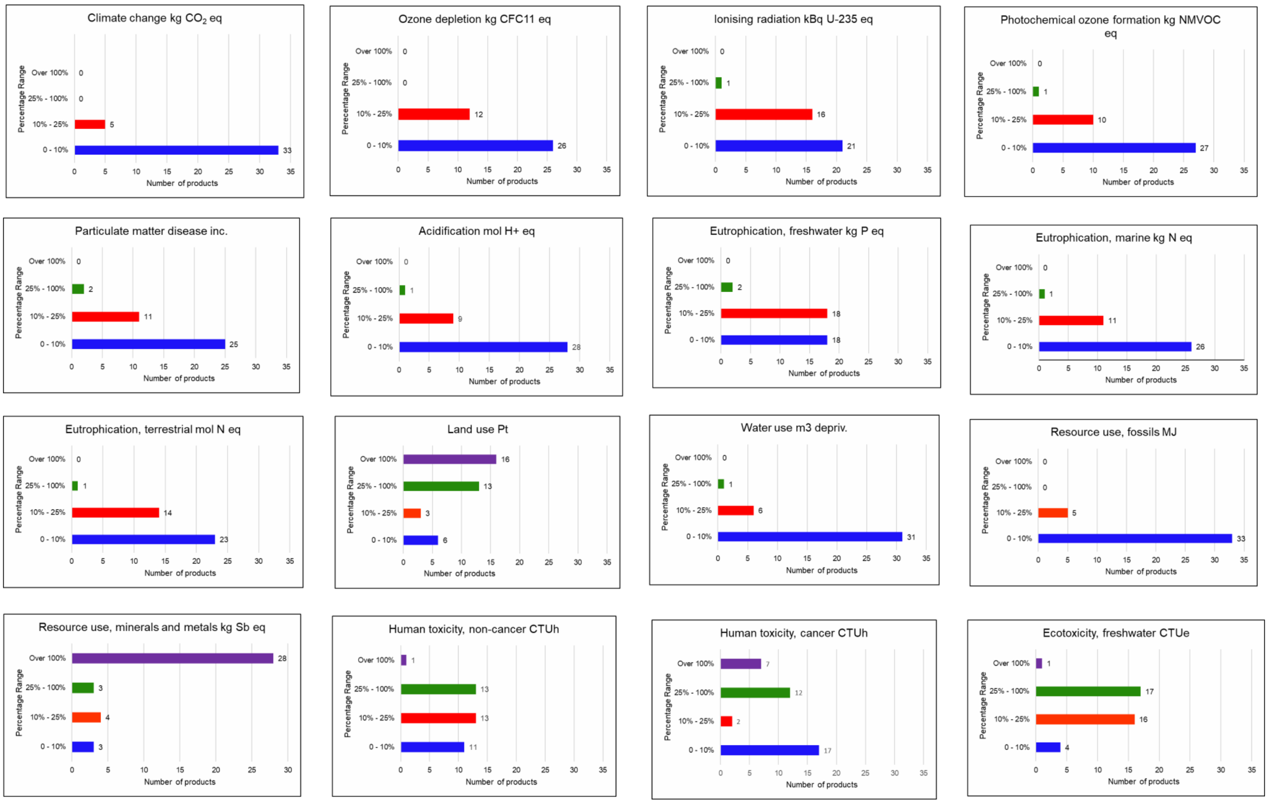

Fig 1: Number of products (out of 38) per range of increase in results for each LCIA indicator due to the inclusion of capital goods

3.2. Contributions to Capital Goods

Capital goods comprises of the contributions of buildings, equipment, transport and energy infrastructure used in the manufacturing processes of products. The ecoinvent data set uses only a limited number of building archetypes to represent all the manufacturing facilities associated with the 38 selected construction products, with the vast majority made up of building hall (steel construction), building hall (wood construction), building (multi-storey), and chemical factory. For example, a “concrete mixing factory” is built up from a building hall (steel construction) and building (multi-storey) and in this manner, all facilities lead back to these four building archetypes. The wood sector has a few additional factory building types, that combine a building hall with additional data. Similarly, there are a few archetypes for machinery and equipment: building machine, industrial machine (heavy), conveyor belt, and some machines used in the wood sector (such as forestry harvester, power saw and wood chipper). For transport infrastructure, we find a few infrastructure processes related to shipping (ships, ship maintenance, port facilities), road transport (vehicles, vehicle maintenance, roads, and road maintenance) and rail transport (rail rolling stock, rolling stock maintenance, railway tracks (including maintenance)). Lastly, ecoinvent includes various types of infrastructures associated with the energy sector (fuel extraction infrastructure, fuel processing plants, fuel distribution networks, power plants, electricity transmission network, electricity distribution network, industrial energy kilns).

It is quite complicated and time-consuming to analyse the exact contribution of various infrastructure components within SimaPro. In SimaPro, we need to analyse each indicator separately. So, a product may cause an increase of 1000% on ADPm&m, but only 25% on say EP. To find where the 25% comes from, we have to find and add up many smaller contributors. In essence, when the node cut-off for contributions using ecoinvent is set too high, small contributors go unnoticed. On the other hand, when the node cut-off is set too low it becomes more difficult and time-consuming to find the capital goods processes out of thousands of unit processes. Our high-level analysis reveals that buildings generally contribute most to the impact caused by capital goods. However, this finding is based on how capital goods are included in the current ecoinvent database. Rather than focusing on the results, we think it is a lot more interesting to focus on how the model influences the results. The inconsistencies and shortcomings are so significant that any EPD (and perhaps any LCA) that includes the effect of capital goods in the results are potentially invalid unless the practitioners have developed their own comprehensive and consistent model and data sources for capital goods. The following list of issues is by no means complete, but acutely critical to the validity of the results:

- Inconsistency in scope (1): the various types of capital goods do not capture a consistent set of materials (inputs). For example, a steel hall contains no copper input (even though any structural building would likely contain copper wiring), whereas a multi-storey building does contain copper. This creates arbitrary inconsistencies in the model.
- Inconsistency in scope (2): Not all processes have buildings, or a production plant input estimated. For example, mastic asphalt only has the industrial machine and conveyor belt inputs quantified. This creates arbitrary inconsistencies in the model.
- Poor technology matches: When including infrastructure in the results of concrete blocks, 31% of the $ADP_{minerals\&metals}$ result is caused by “packing”. We can ignore that the process refers to the packaging of clay in plastic bags, which is different from what happens to concrete blocks, and focus on the key contributor. The use of an industrial machine for packaging contributes 25% to the total $ADP_{minerals\&metals}$ of the concrete blocks. Compare this to the 3% contribution of the actual “concrete mixing factory” and it is clear that the outcome is quite implausible.
- Questionable causality: When analysing the $ADP_{minerals\&metals}$ of 1 kg of mastic asphalt, we find that 75% of the impact is caused by 0.002 kg of the lime sourced from zinc mining operations.
    - Let's assume the underlying data are correct, it is questionable whether a relevant link exists between a small but highly impactful amount of lime from the global zinc mining industry and lime use in the asphalt sector in each area (without a zinc mining industry nearby).
    - The $ADP_{minerals\&metals}$ that are attributed to the lime are coming from minerals (mainly lead, silver and zinc) that have been allocated to the various outputs of the multi-output process.

Whether there is a causal relationship between the lime and mineral depletion is not clear to us.

- Errors in quantification: The multi-storey building contains 8.5 kg of copper per m^3 of the building volume. We believe this is overestimating the amount of copper in a building by more than a factor of 100 (Copper Education facts, 2022).
  - Furthermore, we suspect this copper value is a data entry error ('typo') since the same value is used for aluminium in this building. The current data make copper in the building a material contributor to mineral abiotic depletion for a number of products, but nonetheless the potential error has survived for more than fifteen years in the ecoinvent database.
- High uncertainty: It is important to understand that the impacts of capital goods are not just determined by quantifying the relevant materials, but by indexing the materials to a unit of production by estimating the lifetime of the capital good and the average annual output over that lifetime. While it is difficult to establish the life time of capital goods as the use differs according to context, it is clear that significant uncertainties exist which can be modelled based on historical data on lifetime of capital goods.

In summary, current estimates of capital goods in LCI data are flawed to the point where their inclusion is likely to generate false results for several indicators.

Despite the insights from our analyses, it is acknowledged that considerable uncertainties and data gaps exist in LCI databases and more still needs to be done to better understand the data points for capital goods in LCA. Despite the realisation that the LCI flows for capital goods can be difficult to map out (Silva et al., 2018), the principle of conservatism, as highlighted in the guidance documents on EPD, would imply that the inclusion of capital goods is more in line with better understanding the impacts on products. Weidema et al., (2013) highlight that capital goods are based on rough estimates with high uncertainty. It is, therefore, advisable for more reliable data on capital goods processes to be collected within core processes to support the validity of environmental claims and to improve the quality of datasets needed in LCI. There is little indication that the difficulty regarding obtaining reliable data will abate given that there is no guidance on if and/or how LCA practitioners and scholars are expected to adjust background capital goods data in the LCA software. Without such guidance, any effort would lead to inconsistency in practices. Agez et al., (2022) have also cautioned that the inclusion of capital goods has the potential to shift attention from the supply chain to the background system. Interestingly, the validity of the capital goods background system is difficult to establish as by nature they are hidden from view (for practitioners and EPD owners alike) and relevancy for the system under study is almost impossible to establish for items such as buildings. In essence, it is unclear what buildings look like in the supply chain, and simplification of building models (in current capital goods LCI data) may not serve us well. It is questionable whether it is realistic and useful to reduce diverse capital goods to a few highly simplified archetypes. Corradini et al., (2019) add that data for each producer is different and the background system involves processes occurring in geographical areas not necessarily close to the foreground system and includes actions that cannot be directly controlled by the foreground companies.

4. Conclusion

The requirement for the inclusion of capital goods leads to a major conundrum for LCA practitioners. Based on our research, we suggest that capital goods are excluded until there is better refinement, consistency and improvement of the quality of LCI datasets. Furthermore, EPD Programme Operators should develop clear guidance for LCA practitioners, verifiers and databases on how to deal with capital goods. The opposing argument that making inclusion of capital goods mandatory will drive the improvement of LCI data regarding capital goods is in our view not supported by evidence. LCA practitioners have been including capital goods for more than twenty-five years, but we have not seen any widely adopted guidance or continual improvement of data. For now, EPDs should document transparently whether the results have been calculated including or excluding capital goods. We recommend that EPDs that do include capital goods indicate the contribution of capital goods on the reported LCIA indicators. We also note that our analysis does not cover the GaBi databases, although we do understand that the proprietary GaBi databases include estimates for capital goods in energy infrastructure and do not allow the user to select or investigate the impacts of other types of capital goods. As a result, for other software users, excluding capital goods when using ecoinvent minimises the discrepancy with LCAs performed using GaBi databases.

Given the issues around capital goods that have been identified in this paper, it is difficult to see how their inclusion would support the overall objectives of EPD. Currently, capital goods data are not verifiable (and to our knowledge are never part of an EPD third-party verification) as there is no appropriate procedure for the LCA practitioner, let alone the verifier, to follow. The PCR guidelines result in some EPDs including capital goods, while others exclude it. When capital goods are included, a lack of guidance means we would likely find further methodological inconsistencies within this group of EPDs. As a result, users cannot compare EPD results (for any of the indicators that are potentially heavily impacted by capital goods), unless they know exactly how capital goods and infrastructure were treated in the LCA, and that they were treated equally.

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Appendix 1 - Selection of Unit Processes of Construction Products

Product 1:	1 kg Autoclaved aerated concrete block {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 2:	1 kg Cement, Portland {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 3:	1 kg Concrete block {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 4:	1 m3 concrete, normal {RoW}| concrete, all types to generic market for concrete, normal strength | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 5:	1 kg Gravel, crushed {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 6:	1 kg Sand {RoW}| gravel and quarry operation | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 7:	1 kg cellulose fibre {RoW}| cellulose fibre production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 8:	1 kg Foam glass {GLO}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 9:	1 kg Glass wool mat {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 10:	1 kg Rock wool, packed {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 11:	1 m2 Cladding, crossbar-pole, aluminium {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 12:	1 kg Copper {AU}| production, primary | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 13:	1 kg Reinforcing steel {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 14:	1 kg Steel, unalloyed {RoW}| steel production, converter, unalloyed | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 15:	1 kg Zinc {RoW}| primary production from concentrate | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 16:	1 kg Steel, chromium steel 18/8 {RoW}| steel production, electric, chromium steel 18/8 | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 17:	1 kg Polyethylene, low density, granulate {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 18:	1 kg Polypropylene, granulate {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 19:	1 kg Polystyrene foam slab {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 20:	1 kg Polyurethane, rigid foam {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 21:	1 kg Polyvinylchloride, emulsion polymerised {RoW}| polyvinylchloride production, emulsion polymerisation | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 22:	1 m3 cross-laminated timber {RoW}| cross-laminated timber production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 23:	1 m3 glued laminated timber, average glue mix{RoW}| glued laminated timber production, average glue mix| Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 24:	1 m3 medium density fibreboard {RoW}| market for medium density fibreboard | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 25:	1 m3 oriented strand board {RoW}| market for oriented strand board | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 26:	1 m3 particleboard, uncoated {RoW}| particleboard production, uncoated, average glue mix| Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 27:	1 m3 plywood {RoW}| plywood production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 28:	1 p Air filter, central unit, 600 m3/h {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 29:	1 p Blower and heat exchange unit, central, 600-1200 m3/h {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 30:	1 m2 Photovoltaic panel, multi-Si wafer {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 31:	1 kg Acrylic varnish, without water, in 87.5% solution state {RoW}| acrylic varnish production, product in 87.5% solution state | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 32:	1 kg Mastic asphalt {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 33:	1 kg Bitumen adhesive compound, cold {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 34:	1 kg Clay brick {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 35:	1 kg Epoxy resin {RoW}| epoxy resin production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 36:	1 kg Fibre cement corrugated slab {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 37:	1 kg Gypsum plasterboard {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)
Product 38:	1 kg Synthetic rubber {RoW}| production | Cut-off, U (of project Ecoinvent 3 - allocation, cut-off by classification - unit)

Environmental Product Declaration of Insulated Concrete Form System

Emma Green, Environmental Resources Management

Key words: Environmental Product Declaration, Life Cycle Assessment, Insulated Concrete Form, Construction

1. Introduction

Climate change is one of the most pressing environmental issues of our time. According to the World Green Building Council, it is estimated that 39% of global greenhouse energy-related carbon emissions contributing to climate change arise because of the built environment (Adams et al., 2022). Insulated Concrete Formworks (ICFs) provide an alternative to traditional building methods, comprising hollow Expanded Polystyrene (EPS) blocks joined by polypropylene connectors to be stacked in place and filled with concrete and reinforcing steel.

Considering the significance of the environmental footprint of the construction sector, the need for transparency and communication of a product's environmental profile is more pertinent than ever. Environmental Product Declarations (EPDs) are often used to address this need.

An EPD is a voluntary environmental declaration that communicates reliable and accurate quantitative environmental data from the Life Cycle Assessment (LCA) of a product to users downstream. EPD documents are independently verified and registered, adhering to the principles inherent to the International Organization for Standardization (ISO) standard for Type III environmental declarations (ISO 14025), thus giving EPDs widespread international acceptance.

This paper reviews the results of the NUDURA ICF system in the context of an EPD, showing the environmental footprint of the first ICF published through the International EPD system (Environdec, 2023).

2. Material and methods

2.1 Product Details

NUDURA ICF products are used as stay-in-place permanent formworks for structural concrete, load-bearing and non-load bearing for below-grade and above-grade walls. The forms remain in place after the placement and curing of concrete. NUDURA ICFs consist of two uniform thickness panels of expanded polystyrene (EPS) which are cross-linked in parallel, with injection-moulded polystyrene inserts and hinges.

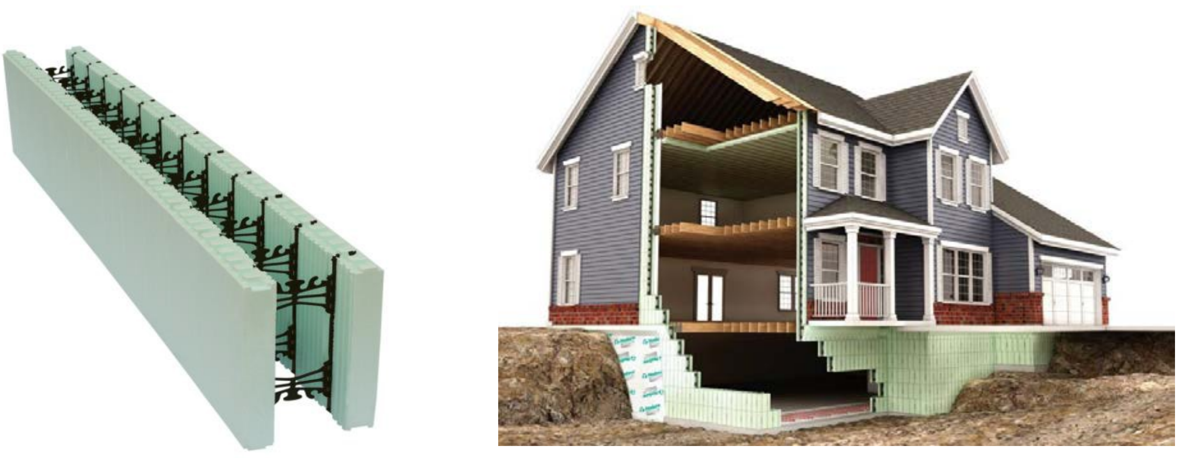

Figure 1: NUDURA ICF piece and in situ demonstration (NUDURA, 2022)

2.2 LCA Methodology

LCA is a method of systematically assessing the environmental burdens associated with a product, pro- cess, or activity over the whole of its life cycle. This includes the use of natural resources and environmental consequences of releases and emissions (ISO, 2006b). The technical framework for a life cycle assessment consists of four inter-related stages: goal and scope, definition, inventory analysis, impact assessment and in- terpretation. Products are appraised based on a functional or declared unit and considered within a defined system boundary, e.g., cradle-to-grave. Defining a unit and system boundary provides the information re- quired to assist comparison of similar products. However, it must be stated that there is limited comparability when different life cycle stages are included or different product category rules (PCRs) are used.

The NUDURA ICF product was assessed by the product stage (modules A1-A3) and end-of-life stage (modules C1-C4, D). Since the use phase was omitted from this study, a declared unit has been used instead of a functional unit. The declared unit for the study was defined as 1 panel of ICF system, weighing 6.91 kg. The methods used to conduct the LCA and resulting EPD are consistent with ISO 14025:2010, Sustainability of construction works - Environmental product declarations - Core rules for the product category of construc- tion products (EN 15804:2012+A2:2019) and the PCR for construction products (PCR 2019:14).

As per EN 15804 clause 6.3.5.1, the “Polluter Pays” principle has been assigned in this LCA to the prod- uct system that generates the waste until the end-of-waste stage has been reached. The end-of-waste state is determined by the economic cut-off method – the environmental impacts of processes that cause costs for the initial product, like waste processing, are allocated to the initial product’s life cycle. When processes raise the value of materials, which is for example the case in certain recycling processes, the environmental impact of the recycling process is allocated to the life cycle of the reclaimed materials.

Quantitative and qualitative specific and generic data were collected for each flow, for all unit processes within the system boundary of the product system and these data were used to compile the life cycle invento- ry (LCI). Specific data were sought as a preference; however, these could not be collected for upstream lifecycle stages. Specific data for all core processes were collected from Tremco to produce the standard form ICF product using data collection sheets via an iterative process and represent a time period of 12 months.

The LCA software SimaPro (version 9) was used to build a model for the product systems under investi- gation using specific and generic inventory data. The generic data was sourced from the LCI database ecoin- vent v3.8 (cut-off). (ecoinvent, 2021).

3. Results

The results were reported according to the impact categories specified by EN15804:2021+A2:2019. Fig- ure 3 shows an overview of all impact categories across the entire system boundary for one piece of ICF product. These environmental hotspot results show which life cycle stages contribute most (and least) to the cradle-to-gate plus end-of-life system boundary.

The Core environmental indicators for 1 piece of ICF system (A1-A3, C1-C4, D) are shown in Table 1.

Table 1. Core environmental indicators for 1 piece of ICF system

Parameter	A1-A3	C1-C4	D	Total (without D)	Unit
GWP - Total	17.3	6.62E-02	-12.4	17.4	kg CO_2 eq.
ODP	6.10E-07	1.48E-08	-1.20E-07	6.25E-07	kg CFC 11 eq.
AP	5.61E-02	4.86E-04	-4.43E-02	5.66E-02	mol H^+ eq.
EP_f	3.96E-04	2.17E--07	-2.19E-04	3.96E-04	kg P eq.
EP_m	1.14E-02	2.02E-04	-6.67E-03	1.16E-02	kg N eq.
EP_t	1.04E-01	2.21E-03	-7.19E-02	1.06E-01	mol N eq.
POCP	1.94E-01	6.05E-04	-3.85E-02	1.95E-01	kg NMVOC eq.
ADP - fossil	337	0.931	-284	338	MJ
ADP – minerals and metals	9.82E-06	3.12E-09	-4.70E-07	9.82E-06	kg Sb eq.
WDP	8.64	3.80E-04	-9.55	8.64	m^3 world eq. depriv.

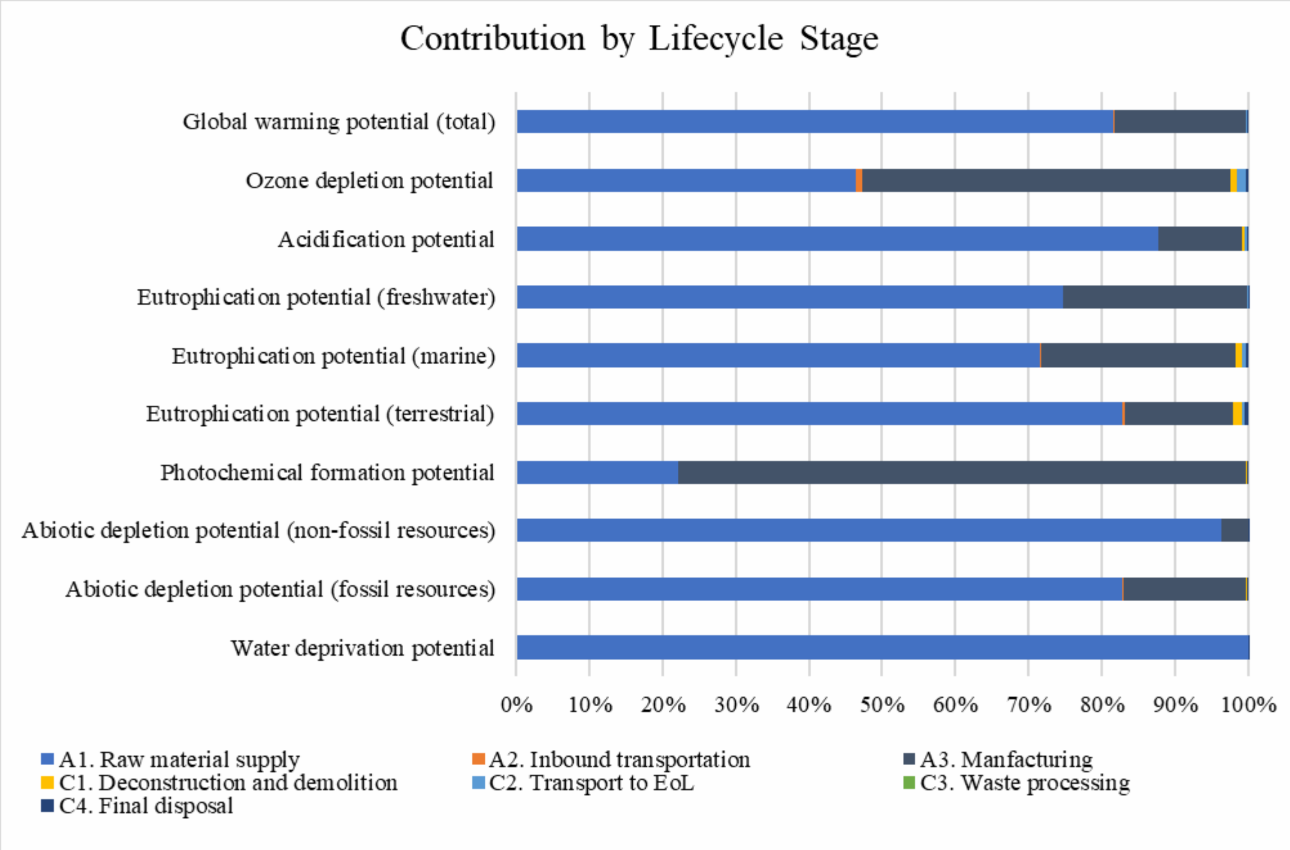

Figure 3: Summary of all impact categories for one piece ICF by % contribution for modules (A1-A3, C1-C4)

In summary, the results show that raw material supply (A1) and manufacturing (A3) have a significant contribution across all impact categories. Conversely, inbound transportation (A2), deconstruction and demolition (C1), end-of-life transport (C2), waste processing (C3) and final disposal (C4) have a minor or negligible contribution across all impact categories.

EN15804+A2 provides an additional module for the benefits and loads beyond the product system boundary (Module D) which covers the benefit of any recovery processes from net output flows leaving the system. Module D is not included in the total impact for the product, consistent with the Cut Off approach, this avoids doubling counting of the benefits of recycling.

4. Discussion

Figure 3 shows an overview of all impact categories across the system boundary for one piece of ICF product. These environmental hotspot results show which life cycle stages contribute most (and least) to the cradle-to-gate plus end-of-life system boundary.

The results show that raw material supply, or the ingredients of the product, is the highest contributor to all impact categories except for ozone depletion and photochemical formation potential. Specifically, the use of expandable polystyrene and recycled polypropylene dominates the impacts. This is to be expected, considering the high mass percentages in the final product.

The results of this EPD are consistent with previous studies comparing the lifecycle analysis of ICF homes versus wood frame houses, highlighting the extensive embodied carbon impact of ICF products (Marceau et al., 2006). However, since the update of EN15804+A2 now includes assessment of the loads and benefits beyond the system boundary (module D), it should be noted the benefits beyond the system boundary are significant.

This study did not assess the use phase of the lifecycle. However, existing literature highlights the operational savings of ICF versus wood- or steel-based construction. A study published by the Massachusetts Institute of Technology's Concrete Sustainability Hub compares construction methods in cold climates and warm climates across residential homes and commercial buildings (Ochsendorf, 2011). The study shows that over 90% of the life cycle carbon emissions are due to the operational phase, hence emissions from initial construction and raw materials could be compensated within a few years, dependent on the usage of the building.

The frequency of natural disasters has increased significantly over the past 50 years due to climate change (World Meteorological Organization, 2021). Extreme weather events expose the vulnerability of the built environment and associated socioeconomic impacts. Considering, resilient and seismic-resistant construction, including retrofitting must be priority. Globally, a widespread use of ICF has the potential to contribute to cost-effective solutions for population growth, housing scarcity and climate-related equality, whilst avoiding extensive carbon emissions.

Due to the ever-evolving materials and techniques used in construction, further work comparing the use of ICF systems and traditional building methods is required. As such, further work would involve an extension of the scope of this study to include the use phase, in addition to development of a comparative LCA assessing ICFs used in combination w ith green concrete versus timber or metal framed construction.

5. Conclusion

The purpose of this study was to enable transparency of environmental performance to customers of Tremco Canada, thus enabling the award of LEED or BREEAM materials credits. However, investigation of the product, building method and existing literature raises interesting opportunities for the future of mainstream construction and its impact on climate change.

Acknowledgements

The author is very grateful to Keven Rector and Vincent Roy at Tremco Canada for their dedication to this project, and to Katie Livesey for her supervision and guidance. The paper is an outcome from an EPD and associated LCA report published via the International EPD system.

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Life Cycle Impact Assessment (LCIA) Abstracts

Ecosystems Services in Australian Agriculture

Wednesday, 19th July - 16:00: Life Cycle Impact Assessment (LCIA)

Mr. Tim Grant [1]

1. *Lifecycles*

New metrics which measure ecosystem service impacts in LCA have been implemented by the European Union and more recently been proposed by UNEP Life cycle initiative. These indicators aim to represent the damage and benefits of different land uses on the value of ecosystem services. Ecosystem services are benefits derived directly or indirectly buy human beings from functioning ecosystems and are in contrast to the LCA endpoints of ecosystem quality which look at the impacts of land use and other flows on species.
This presentation will examine the effectiveness of these new indicators when assessing Australian land use in agriculture, electricity supply and paper production. The results show limitations on the use of national regionalisation provided by these metrics in a country as big as Australia. Its also contrasts the weighting approach used in the Product Environmental Footprint method with the economic valuation proposed by UNEP Life Cycle Initiative.

Proposed method from GLAM3 for assessing the impact of natural resource use at endpoint level

Wednesday, 19th July - 16:00: Life Cycle Impact Assessment (LCIA)

Dr. Masaharu Motoshita [1]

1. *Research Institute of Science for Safety and Sustainability, National Institute of Advanced Industrial Science and Technology*

Natural resources are incorporated and used in a wide range of product systems and organizational activities through supply chains, but various different assessment models have been proposed for assessing the potential environmental impacts associated with natural resource use. Within the UNEP Life Cycle Initiative's GLAM project, international consensus building on impact assessment in life cycle assessment (LCA) and the development of recommended methodologies are being carried out. In the GLAM3, the development of methodologies based on international consensus is being advanced for the impact assessment of natural resource use as one of the targets. This presentation reports on the progress and achievements in this regard.

Biodiversity indicators and agriculture – application and development of methods.

Wednesday, 19th July - 16:00: Life Cycle Impact Assessment (LCIA)

Dr. Murray Hall [1]***, Dr. Tom Harwood*** [1]***, Dr. Simon Ferrier*** [1]***, Dr. Nazmul Islam*** [1]***, Dr. Javier Garcia Navarro*** [1]***, Dr. Maartje Sevenster*** [1]

1. *CSIRO*

Agriculture is a major driver of land use change and impact on biodiversity. The UN Food and Agriculture Organization (FAO) has recently recommended a method for biodiversity metrics which builds upon the UNEP-SETAC Life Cycle Initiative working group on land use change. The characterisation factors for this biodiversity indicator have now been published in leading LCA data bases. However, the application of the characterisation factors for Australian agriculture has not been explored. This will be considered with an application of the characterisation factors to the Australian Agricultural Life Cycle Inventory (AusAgLCI). This will highlight issues of terminology as well as concepts for land use which were developed after the AusAgLCI method was published. In addition, the method itself will be explored for refinements in land use and calculation of biodiversity metrics. This analysis will draw upon the CSIRO research which underpins the land use maps for the FAO recommended method and which has also been used to develop biodiversity indicators. The two approaches will be demonstrated for a test area to highlight any differences in the results. Possible research areas for facilitating the application of the FAO biodiversity method with AusAgLCi will be outlined as well as suggestions for improving the method itself.

Update of Best Practice Life Cycle Impact Assessment in Australia

Wednesday, 19th July - 16:00: Life Cycle Impact Assessment (LCIA)

Dr. Marguerite Renouf [1]

1. Lifecycles, Brisbane, Australia

One function of the Australian Life Cycle Assessment Society (ALCAS) is to inform the Australian LCA community about current best practices in the moving field of life cycle impact assessment (LCIA). It does this through the Best Practice Guide for LCIA in Australia, co-ordinated by the ALCAS Impact Assessment Committee and drawing on expertise from ALCAS' membership. The purpose of the guide is to provide up-to-date details about methods, and guidance for selecting appropriate methods for LCAs of Australia-centric products, services and processes. The original guide was released in 2008, with recommended methods and characterisation factors last updated in 2016 and 2018.

In light of the recent international efforts to develop a consistent global LCIA method, it is timely to review and update recommendations for best practice impact assessment in Australia. The third phase of the Global Guidance on LCIA Indicators and Methods (GLAM Phase 3), co-ordinated by the UNEP Life Cycle Initiative, has proposed a framework and method set which can be applied globally.

An important next step for best practice LCIA in Australia will be alignment with methods, characterisation factors and normalisation factors that are globally oriented. This recognises that process making up Australia-centric product and service systems are rarely solely based in Australia and usually linked to global materials flows and receiving environments. Therefore, impacts need to be assessed in the global context to ensure consistency.

This item is a panel discussion and open conversation about the next iteration of best practice LCIA recommendations for Australia, with session speakers invited to contribute to the discussion.

Panel Discussion: Use of globally consistent LCIA methods seems to be direction things are heading. Is this the right recommendation for best practice LCIA in Australia?

Food Abstracts

Nutritional LCA methods—a review of opportunities in a rapidly developing field

Thursday, 20th July - 09:00: Food

Prof. Jolieke van der Pols [1], ***Prof. Sarah McLaren*** [2]

1. Queensland University of Technology, 2. Massey University

Nutritional LCA studies (nLCA) consider the nutritional characteristics of foods or diets, as well as their environmental impacts. Interest in these methods has rapidly increased in recent years. They are well suited to inform front-of-pack labeling and other methods to inform consumer decision making. They can also support process optimisation and corporate reporting and can play an important role in global efforts towards more healthy and sustainable food systems. This presentation provides an overview of key nLCA methods currently used. It will discuss strengths and opportunities for further development and use of nLCA methods, with particular focus on nutrition and health related issues.

Life cycle-based environmental impacts of foods using the nutritional LCA method: a case study of New Zealand avocados and Cheddar Cheese.

Thursday, 20th July - 09:00: Food

Ms. SHREYASI MAJUMDAR [1], Prof. Sarah McLaren [1], Prof. Jolieke van der Pols [2], Dr. Carolyn Lister [3]

1. Massey University, 2. Queensland University of Technology, 3. Plant and Food Research

Nutritional LCAs can take two approaches: using nutrition as a function of the food system being studied, or as an impact category. When using the former approach, the Nutrient Rich Food (NRF) family of indices is the most common nutrient profiling method used in nutritional analyses and nLCA studies. However, the choice of NRF index will influence the nutritional scoring of a food. Case studies investigating the impact of this index choice in the context of nLCA studies is currently limited. In addition, nLCA studies

have mostly focussed on individual foods or diets However, a meal focus provides an alternative perspective that may be more meaningful, especially for consumers. To address these points, we asked the following research questions: "How does nutrient selection in index development influence: a) the assessment of nutritional quality, and b) the nLCA results for food items that are commonly consumed as alternatives in a single meal?"

To answer these questions, we quantified the nutritional quality and climate change impact (as kg CO2 equivalent) of Hass avocados and mild Cheddar cheese in the context of common toast toppings consumed in New Zealand. The nutritional quality of both foods was evaluated using the NRn, LIM and NRF indices for 9 and 20 beneficial nutrients and 3 nutrients to limit for both avocados and cheese, for three reference units (mass, energy, and serving size). These were then presented in an nLCA nutrition impact category along with the climate change impacts for both foods to demonstrate the nutritional quality of the food, in conjunction with the climate change impacts of the food per serving size.

The results of this study contribute to the fast-growing body of research in this area, by highlighting the importance of the choice of NRF index when conducting an nLCA in the meal context.

Sustainable diet in a highly dense population setting: The balance of water use and nutrition

Thursday, 20th July - 09:00: Food

Dr. Kamrul Islam [1]***, Dr. Ryosuke Yokoi*** [1]***, Dr. Amandine Pastor*** [2]***, Dr. Masaharu Motoshita*** [1]

1. *Research Institute of Science for Safety and Sustainability, National Institute of Advanced Industrial Science and Technology,* ***2.*** *Food, and Environment (INRAE), French National Institute for Agriculture*

Global crop production consumes a huge amount of freshwater due to which regional freshwater overconsumption is evident from major watersheds of the world. We as a global community are facing the dilemma of identifying ways to feed the billions of people and ensuring that our food production follows a sustainable practice. Bangladesh is a densely populated country with one of the lowest per capita agricultural lands of the world. In recent days, the country is gradually becoming self-reliant on food production. There are few issues regarding the limited freshwater resources of the country: seasonal freshwater overconsumption during the dry period, and water pollution due to agricultural intensification. In this study, we aim to assess the sustainability of the crop water use for supporting Bangladesh's diet and gain the implications for future sustainable national diet focusing on the aspect of water resources. The updated WaterGAP 2.2d model is used to calculate the freshwater overconsumption of Bangladesh's watershed, and the overconsumption associated with the crop production in the country is estimated corresponding to the total agricultural production for 1960–2019 period using updated crop evapotranspiration value. According to the results, the average overconsumption of freshwater from agricultural crop production during 2000–2016 period in Bangladesh was ~20,433 million cubic meter which was mainly due to own consumption of several crops e.g., rice, areca nuts, wheat, jute etc. Furthermore, we also seek to identify the relationship between the nutrient density of the food crops and induced freshwater overconsumption, which gives the implications towards the achievement of the sustainable national diet. The outcome from this study will provide strategies for sustainable crop production in the country with the identification of the key crops and the watersheds that need improvement in the context of sustainability of water use.

Adapting the Agribalyse Life Cycle Inventory database to Australia – a first step towards a comprehensive Australian food and agriculture model

Thursday, 20th July - 09:00: Food

Mr. Paul-Antoine Bontinck [1]

1. *Life Cycle Strategies Pty Ltd*

The food production and supply system is a major driver of human environmental impacts. In 2019, the IPCC reported that 21-37% of total anthropogenic greenhouse gas emissions are attributable to our global food system. Life Cycle Assessment is a method which can be used to drive effective decision-making aiming to reduce environmental impacts. A significant amount of work has been conducted in Australia to model the primary production of major crops including grains, oilseeds, some fruits, vegetables and nuts, as well as livestock. This data is published in the Australian National Life Cycle Inventory Database (AusLCI) and provides a detailed overview of the first stage of the food supply chain. However, subsequent stages are not currently represented in AusLCI.

Agribalyse is a French Life Cycle Inventory database, which has been developed to assess the supply chain of food products in France. It covers the supply chain from primary production, up to the final consumption of food commodities, and is used to represent the production of over 2,500 individual food products.

This study aims to use the architecture and nomenclature developed for Agribalyse to produce a model representative of the Australian food supply chain. To do so, existing AusLCI models for primary production of food commodities are used to replace their French equivalents in Agribalyse. A range of key aspects are also modified to represent the Australian conditions, including electricity and water supply, import fractions and domestic transport.

The resulting database is a live model of the Australian food supply chain, allowing for further improvements and updates over time. It may be used for activities ranging from research projects to mainstream applications such as food product labelling or as a basis to develop a simplified food environmental impact assessment tool.

Water, energy, and greenhouse gas footprint of city food system in Australia

Thursday, 20th July - 09:00: Food

Dr. Nazmul Islam [1], ***Dr. Marguerite Renouf*** [2], ***Prof. Steven J. Kenway*** [3], ***Prof. Thomas Wiedmann*** [4]

1. *Sustainability Assessment & Metrics, Commonwealth Scientific and Industrial Research Organisation (CSIRO), QLD, 4067, Australia,* ***2.*** *QUT Centre for Agriculture and the Bioeconomy (CAB), Queensland University of Technology, QLD, 4000, Australia,* ***3.*** *Australian Centre for Water and Environmental Biotechnology (formerly AWMC), University of Queensland, QLD, 4072, Australia,* ***4.*** *School of Civil and Environmental Engineering, UNSW, Sydney NSW 2052, Australia*

Information about where to direct eco-efficiency efforts is needed to ensure a sustainable city food system due to current interest in achieving net-zero, as well as sustainable consumption and production (SDG12). Therefore, the objective of this work is to assess the water, energy, and GHG footprint of an Australian city's food system. As a representation, this research modelled fresh vegetable production in Queensland, and home consumption in Southeast Queensland. In total, 30 types of fresh vegetables were evaluated for a field-to-plate system boundary. Inputs to all these processes were accounted for. The functional unit (FU) was 1 kg of fresh vegetables produced and consumed. The life cycle environmental impacts were estimated using SimaPro software 9.1.1.1, based on the Australian Life Cycle Assessment Society (ALCAS) Best Practice Guide for Life Cycle Impact Assessment (LCIA) in Australia V 2.04. Impact categories included were Climate change (kg CO_{2eq}), Resource depletion – fossil fuels (MJ) as an indication of primary energy demand, and Consumptive water use (L_{eq}), which derived by multiplying water use with water stress factors. The farm production and processing are the hotspots of beans, chilies, cucumbers, and eggplant. For most of the vegetables, downstream energy impacts of refrigerated retail, and household (through electricity consumption) are comparatively higher, such as for cabbage, carrots, cauliflower, celery, onions, and parsnips. Life cycle water-related energy use ranges from around 15% to 40% for different studied vegetables, in comparison with the supply chain fuel use (diesel use for transport and tractor) (~10% to 30%), agrochemicals (~5% to 15%), and packaging materials (~1% to 30%). This indicates the importance of directing the eco-efficiency programs towards energy saving through water management along the entire city food system compared to much-explored issues, such as fertilizer application, transport, and packaging efficiency improvement.

Food Extended Abstracts

Life cycle-based environmental impacts of foods using the nutritional LCA method: a case study of New Zealand avocados and Cheddar cheese

Shreyasi Majumdar[1], Sarah McLaren[1], Jolieke van der Pols[2], Carolyn Lister[3]*

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Main topic/key words: nutrition, nLCA, avocados, cheese, New Zealand, Life Cycle Assessment

1. Introduction

A main function of foods is to provide nutrition for healthy growth and development (Willett et al., 2019; McAuliffe et al., 2020). It is becoming increasingly common to consider the nutritional characteristics of foods in environmental analyses using the Life Cycle Assessment (LCA) methodology – either analysed separately or integrated in a combined nutritional LCA (nLCA) study (McLaren et al., 2021; Ridoutt, 2021). The nutritional value of food can be accounted for in an nLCA study, either by incorporating nutritional quality into the functional unit (nFU), or by addressing it in the impact assessment stage. In either case, the nutritional quality of the food item should be reported as comprehensively as possible (McLaren et al., 2021). Currently, most nLCA studies use nFUs based on individual nutrients like protein or calcium, or nutrient profiles that consider a number of different nutrients relevant to health (with beneficial and negative impacts on health). The latter allows for a more comprehensive assessment of the nutritional quality of a food. The Nutrient Rich Food (NRF) family of indices is the most common nutrient profiling method used in nutritional analyses and nLCA studies (Drewnowski, 2009, 2017; Fulgoni et al., 2009; Grigoriadis et al., 2021; Zhai et al., 2022). $NRF_{9.3}$ (with nine nutrients to encourage and three to limit) is the most widely used and validated NRF index. However, $NRF_{9.3}$ includes a limited number of nutrients, which may not reflect the full nutritional quality of a food. This is particularly relevant for food items with specific and unique characteristics where another index may be needed to better reflect their nutritional quality (Masset et al., 2015; Hallström et al., 2019; Grigoriadis et al., 2021). The choice of NRF index will influence the nutritional scoring of a food (Drewnowski et al., 2009a; Masset et al., 2015), as well as evaluation of the environmental impacts associated with that food if the NRF index is used to define the functional unit in nutritional LCA studies (Saarinen et al., 2017; Grigoriadis et al., 2021). Case studies investigating the impact of this index choice in the context of nLCA studies is currently limited (Bianchi et al., 2020). In addition, nLCA studies have mostly focussed on individual foods or diets (Bianchi et al., 2020; Grigoriadis et al., 2021; McLaren et al., 2021). However, a meal focus provides an alternative perspective that may be more meaningful, especially for consumers (Batlle-Bayer et al., 2021; Cooreman-Algoed et al., 2020; Mazac et al., 2023; Sonesson et al., 2017; Takacs et al., 2022).

To address these points, we asked the following research questions: *"How does nutrient selection in index development influence: a) the assessment of nutritional quality, and b) the nLCA results for food items that are commonly consumed as alternatives in a single meal?"*

2. Material and methods

We quantified the nutritional quality and climate change impact (as kg CO_2 equivalent) of Hass avocados and mild Cheddar cheese in the context of common toast toppings consumed in New Zealand, as follows:

2.1 Survey: An online survey was conducted to determine whether 'topping(s) on toast' is frequently consumed as a meal in New Zealand, and if so, what are the preferred toppings. Of 214 respondents, 96% indicated that they consume toast with toppings (usually breakfast or lunch), and 94% reported consuming avocado as a topping on toast. Of the 98% respondents who said they have other toppings as well, 45% said they prefer Cheddar cheese. Among the other toppings, preferences were towards nut butters (43%), eggs (38%), tomato, marmite and vegemite (35% each), dairy butter (34%), and jam (32%) (among others) – either as standalone toppings or in combination with others. Given that Cheddar cheese was the most commonly consumed (non-avocado) toast topping, and our research interest and prior work on avocados, it was decided to study avocados and Cheddar cheese in the case study.

2.2 **Environmental LCA scores**: The climate change (GWP_{100}) impact of New Zealand avocados was evaluated in a recent study at the national level for Hass avocados (Majumdar et al., 2022); the average value was 0.7 kg CO_2 eq./kg avocados at the retailer. For cheese, the average climate change (GWP_{100}) impact from recent studies was 12.7 kg CO_2 eq./kg at the retailer (Gosalvitr et al., 2019, 2021; Kim et al., 2013; Kristensen et al., 2015); two recent studies found that raw milk accounted for an average of 76% of the impacts in the cheese life cycle up to the manufacturing gate (Gosalvitr et al., 2019; Laca et al., 2020). New Zealand-specific climate change (GWP_{100}) values (without land use change) for pasture-fed raw milk production (cradle to farm gate), as well as the average protein and fat contents of milk (Ledgard et al., 2020; Mazzetto et al., 2022), were used to adapt the international Cheddar cheese climate change impact value mentioned above to reflect milk production in New Zealand. This gave a value of 11.7 kg CO_2 eq./kg cheese, assuming 1 kg cheese is manufactured from an average 10 L milk.

2.3 **Choice of index for nutritional profiling:** The $NRF_{9.3}$ index (Fulgoni et al., 2009) was chosen due to its widespread use in nLCA studies. Based on priority indicators of nutrient intake and their recommended daily intake amounts (Nutrient Reference Values (NRVs), this index considers 9 nutrients for which intake is encouraged (fibre, protein, vitamins A, C, and E, and minerals – calcium, iron, magnesium, and potassium) and 3 nutrients (saturated fat, sodium, added sugar) for which intake is discouraged. $NRF_{20.3}$ contains 20 nutrients to encourage (n-3 and n-6 fatty acids, monounsaturated Fatty Acids (MUFAs), vitamin D, several B-vitamins, and zinc in addition to the 9 of the $NRF_{9.3}$ index) and the same 3 to limit (Hallström et al., 2019) and was chosen for comparisons with $NRF_{9.3}$ results. Avocados are rich in unsaturated fats, specifically Monounsaturated Fatty Acids (MUFAs) – and the $NRF_{20.3}$ index is the only index in the literature that accounts for both MUFAs and Polyunsaturated Fatty Acids (PUFAs). Each NRF index consist of an NR_n score (quantifying the encouraged nutrients in relation to their recommended intake level) and a LIM score (quantifying the nutrients to limit in relation to the upper level of intake). The NRF index value is obtained by subtracting the LIM score from the NR_n score.

2.4 **Computing the nutritional and environmental performance:** The nutritional quality of avocado and Cheddar cheese was analysed separately using the NR_n and LIM indices (for nutrients to encourage and limit respectively) as well as the composite NRF score. In the nutritional analyses, all scores were calculated relative to three commonly used reference units (per 100 g, 100 kcal, and standardised serving size (Drewnowski et al., 2009b) obtained from the New Zealand Food Composition Database) and no additional weighting or capping was employed. In addition, the mean method was used to calculate the NR_n, LIM, and NRF indices in this study (where the beneficial and limiting nutrients are quantified in the indices by calculating their mean value). For the combined nutritional and environmental analysis (nLCA), environmental (climate change impact) and nutritional results were expressed per serving size, and the latter was assessed at impact assessment in a separate nutrition impact category (as specified in McLaren et al., 2021, Section 8.3.3, Figure 6 for comparing across more than one food item with the same function). The nutrition impact category therefore comprised the considered nutrition indicators (NR_n, LIM, NRF, and energy content) values which demonstrated the nutritional profile of the food and allowed for a comparison with the associated environmental impacts per serving size.

2.5 **Data on nutritional composition**: The nutritional composition of the avocado and Cheddar cheese and the respective standardised serving sizes were obtained from the New Zealand Food Composition Database (2022). The Nutrient Reference Values (NRVs) for macronutrients, micronutrients and fatty acids to calculate the NRF scores were mostly obtained from the National Medical Health and Research Council (2017). Because some NRVs are different for adult females compared to males, NRF scores were calculated separately for men and women, then their mean (based on the 50:50 ratio of men:women (as per Stats NZ, 2023) in the NZ population) was used in further analyses. The Upper Levels (ULs) of intake for sugar, saturated fats, and sodium were obtained from the Australia New Zealand Food Standards Code (FSANZ, 2021). The NRV value for MUFAs for a 2000 kcal (=8368 kJ) diet given in (Drewnowski et al., 2009b), was adapted to the standard New Zealand diet of 8700 kJ (FSANZ, 2021).

3. Results

3.1 Nutritional Profiling

Figure 1 shows the Nutrient Rich (NR_n) and LIM scores for avocado and Cheddar cheese, calculated on a per 100 g, 100 kcal, and serving size (85 g (half fruit) for avocado and 40 g (2 slices) for Cheddar cheese) basis. LIM scores are usually calculated as positive values, even though they represent potential detrimental impacts on human health. For the purposes of this study, the LIM scores are depicted as negative values to enable a more intuitive comparison with the beneficial nutrients represented by the NR_n scores.

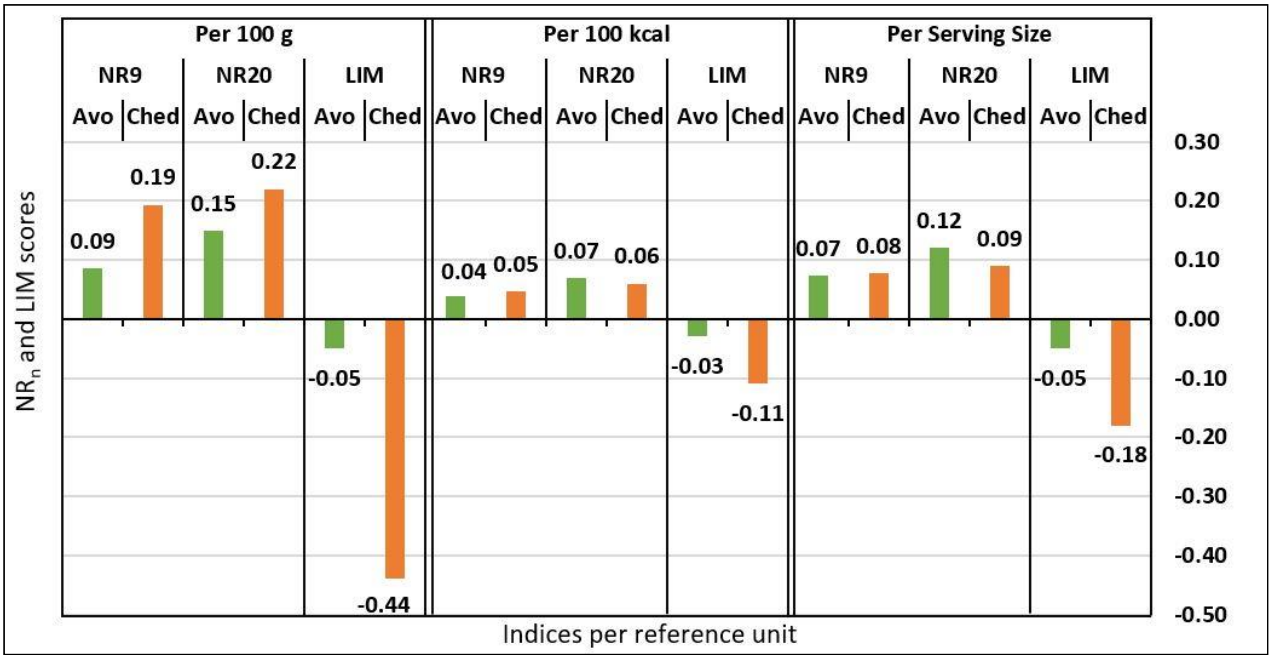

Figure 1 NR_n and LIM scores for avocado and Cheddar cheese, based on 9 and 20 nutrients to encourage and nutrients to limit, calculated by mass, energy, and serving size

When just considering the nutrients for which intake is encouraged (the NR_n scores), Cheddar cheese performs better (i.e. has higher scores) than avocado for the NR_9 index across all the reference units. For the NR_{20} index, Cheddar cheese performs better when using the 100 g reference unit but, for the energy and serving size-based units, avocado has slightly higher scores than Cheddar. This is partly due to the inclusion of MUFAs and PUFAs in the NR_{20} scores – which are present in higher quantities in avocado than Cheddar cheese.

When just considering nutrients for which intake should be limited (LIM scores), Figure 1 shows avocado performs better (i.e. has lower scores) than Cheddar cheese for all three reference units (note that the NR_9 and NR_{20} results are the same because the LIM is calculated using the same three nutrients in both indices). Moreover, the difference in the LIM scores for the two products is significant: Cheddar cheese has LIM values that are 780%, 266%, and 260% higher than avocado on a mass, energy and serving size basis respectively.

When the NR_n and LIM values are combined into a composite NRF score, avocado performs better nutritionally than Cheddar cheese for all three reference units (Figure 2). Avocado has higher NRF scores when considering 20 rather than 9 beneficial nutrients, but Cheddar cheese has similar scores for both indices.

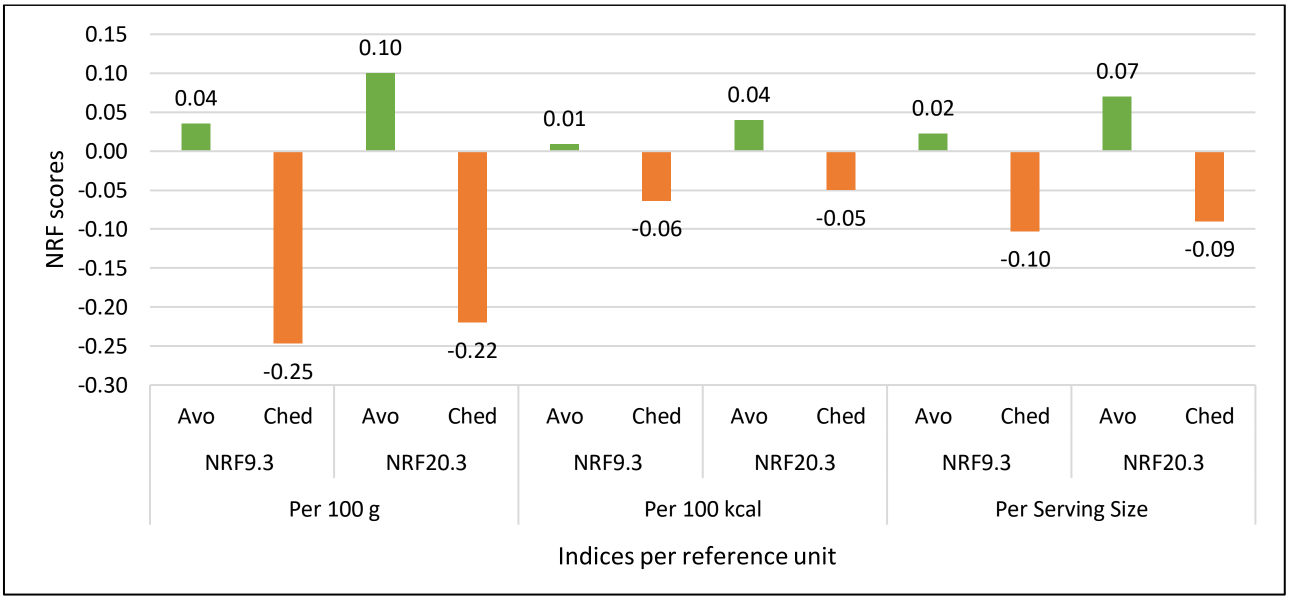

Figure 2 Combined NRF scores for avocado and Cheddar cheese for 9 and 20 nutrients to encourage and three nutrients to limit, by serving size, mass, and energy density.

The highest scores in terms of nutritional value (when considering the NR_n and LIM indices) were registered for the 100 g reference unit, followed by the serving size and then the 100 kcal unit. But for the NRF indices, the values are different for Cheddar cheese, which shows the best performance (lowest scores) with the 100 kcal unit, followed by serving size and then the 100 g unit.

3.2 Combined Nutritional and Environmental Analysis (nLCA)

The nLCA results for avocado and Cheddar cheese using serving size as the nFU are presented in Table 1. The climate change impacts of avocado and cheese are presented along with their respective nutritional scores (NR_n, LIM, and NRF indices) for 9 and 20 nutrients. The table also includes the energy content of each food. Apart from NR_9, in which the Cheddar cheese score is slightly higher, avocado performs better environmentally and nutritionally for the other results. This table effectively represents the new midpoint impact category for demonstrating the nutritional quality of the food, in conjunction with the environmental (in this case, climate change) impacts of the food per serving size.

Table 1 Environmental (climate change) impact and nutritional quality per serving size for avocado and Cheddar cheese (green and red boxes indicate better and worse performance of the food items respectively in the nutrition and environmental impact categories)

	Avocado (85 g)	**Cheddar Cheese (40 g)**
GWP (kg CO_2 eq.)	0.08	0.47
NR_9	0.07	0.08
NR_{20}	0.12	0.09
LIM	0.05	0.18
$NRF_{9.3}$	0.02	-0.10
$NRF_{20.3}$	0.07	-0.09
Energy Density (kcal)	186	168

4. Discussion

This study was conducted in two parts: a) analysis of the nutritional quality of avocado and Cheddar cheese considering nutrient composition in relation to recommended intake levels, and b) a combined nutrition-environment analysis (nLCA) that considered the environmental (climate change) impact of avocado and Cheddar cheese, with nutritional quality expressed at impact assessment in a separate nutrition impact category. The nutritional analyses show that when only nine beneficial nutrients are considered in the NR_n analysis, Cheddar cheese performs better than avocado across all reference units (per 100 grams, 100 kcal or per serving size). However, when additional nutrients are considered, including MUFAs and PUFAs, avocado has higher nutritional scores than Cheddar cheese for the serving size and 100 kcal reference units. At the same time, the analyses of nutrients to limit show avocado has a better nutritional profile than Cheddar cheese across all comparison groups. Analysis of the NR_n and LIM indices separately provides insights into the beneficial and detrimental nutritional qualities of these food items (Figure 1). When these two indices are combined into one NRF score, thus considering both nutrients to encourage and limit (as shown in Figure 2), avocado has higher NRF scores than Cheddar cheese for both the NR_9 and NR_{20} indices compared, and for all three reference units.

The nLCA results were presented per serving size with nutrition assessed as an impact category. This approach shows that avocado performs better than cheese for climate change impacts and nutritionally across all the nutritional quality indicators except for the NR_9 index (Table 1). The downside to this approach is that environmental impacts are not expressed relative to standardised nutritional quality of the food items, which is possible when incorporating the nutritional quality in the FU (nFU). On the other hand, when using an nFU, care must be taken because the methodology is still evolving and has inherent limitations. For example, the FU in LCA represents a (positive) service provided, and the NRF scores were negative (less than 0) for Cheddar cheese when the LIM scores were subtracted from the NR_n scores (Figure 2). Such negative values are difficult to interpret in the context of an LCA study (McLaren et al., 2021). This conceptual challenge in LCA has been identified as a fundamental problem when attempting to include potential harm to human health in an nFU (Green et al., 2020; Weidema & Stylianou, 2020). It may therefore be preferable in future studies to present the final results as the NR_n-based nLCA scores alongside the LIM scores (as suggested by Saarinen et al., 2017). Another way to account for the negative health impacts of some nutrients is by assessing the human health impacts of foods and diets in a separate impact assessment category in LCA, either directly using the LIM values or using the Global Burden of Disease study data to express health impacts in disability-adjusted life years (DALY/g) (Jolliet, 2022; Stylianou et al., 2021).

With respect to nutrient selection, it has been suggested that nutrition indices that include more nutrients are better suited for comparing foods within food groups (Bianchi et al., 2020). Cheese and avocado belong to two different food groups and, if analysing within food groups, they would not be studied as substitutes for each other. However, in a meal context people often consume foods across different food groups as substitutes for each other (as per avocado and Cheddar in this case study). This study demonstrates that consideration of a larger number of nutrients in indices can provide different rankings when comparing substituted food items located in different food groups. Care should especially be taken when assessing unique foods that are quite different from the other foods within their food groups based on specific nutritional characteristics; for example, avocado is the only fruit apart from olives that is high in unsaturated fats. Avocados are of much interest since national dietary guidelines often recommend switching from saturated and trans fats to unsaturated fats, often even citing avocados and avocado oil as an effective way to do that (National Health and Medical Research Council, 2013; Ministry of Health, 2020; U.S. Department of Agriculture & U.S. Department of Health and Human Services, 2020); therefore it is important to account for the unique nutritional quality of avocados (in terms of unsaturated fats) in indices in future studies.

Food-related LCA studies usually use a mass-based FU; however, such mass-based FUs can be unrealistic when considering the actual quantities of substituted food items in a meal. Thus, when considering substitution of food items in a meal, it is preferable to use serving size rather than mass- or energy-based units, as it reflects more realistic nutrient intakes at the meal or diet level (Grigoriadis et al., 2021; Jolliet, 2022; Masset et al., 2014). However, unlike foods within the same food group that are likely to have similar serving sizes (see, for example, Hallström et al. (2019) who studied seafood), substituted food items from different food groups may have quite different serving sizes, as was seen in our case study. The U.S. uses the Reference Amount Customarily Consumed (RACC) – a metric developed and mandated by the U.S. Food and Drug Administration (FDA) – to roughly standardise actual serving sizes for a food product (Drewnowski, 2017; Berardy et al., 2019; Grigoriadis et al., 2021) – but other countries may not have government-mandated harmonized, serving size values. In these cases, it may be desirable to use interim standardised serving sizes calculated for other countries whose populations have similar diets. For this case study, we used values from the New Zealand Food Composition Database (2022). It should also be noted that, while such standardised serving sizes are useful for comparative studies like ours, actual amounts consumed may vary substantially between people.

Finally, the nutrients in this study were not weighted according to their relative nutrient priority in the study population. Drewnowski (2017) notes that most indices exclude this weighting step due to a lack of scientific consensus on what criteria to use to weight nutrients. Grigoriadis et al. (2021) also pointed out in their review of several nutrient profiling methods that only the Nutrient Density Score (NDS) method incorporates weighting in its methodology. However, weighting according to the relative importance of different nutrients contained in a food is meaningful if directly relevant to the population of interest and should be considered in future research. This can be particularly relevant to regional variability in diets. For example, one population's diet may include high levels of calcium intake due to consumption of large amounts of dairy products. For this population, then, calcium might be weighted less than another population where the diet has lower calcium levels. Additionally, a limitation in this study was that our calculations of NRF values was based on adults and thus did not consider different nutrient requirements for children, pregnant/lactating women or older persons. Other limitations of the use of nutrient scores are discussed in McLaren et al. (2021); they include the role of bioactive components contained in foods (e.g. phytonutrients), bioavailability of different nutrients, incomplete food composition data, and food matrix and meal effects.

This case study was meal-focussed because we considered two commonly consumed toppings that people reported eating on a piece of toast as part of a meal. We thus assessed nutritional quality and environmental impacts (specifically climate change) in the realistic context of substituted food items used in a particular meal setting in New Zealand. However, a diet is characterised by complementarity and variety and meals do not reflect the quality of an entire diet (Van Kernebeek et al., 2014). This was clear from our survey, in which 71% of the respondents said they are likely to have avocados on toast as a meal less than once a week. As amply evident in literature, much effort has been made to assess nutritional and environmental efficiency in the context of whole diets (Bunge et al., 2021; Esteve-Llorens et al., 2020; Hallström et al., 2018; Masset et al., 2014; Sonesson et al., 2019; Strid et al., 2021; Van Kernebeek et al., 2014). Therefore, it is important to look at dietary patterns as a whole in future studies.

5. Conclusion

The results of this study highlight the significance of the choice of the nutrient index in calculating a nutrient score in nLCAs when comparing alternative food items in a meal. They also highlight the significance of choice of FU for a study when comparing alternative food items located in different food groups.
Future research should investigate how to account for the relative scarcity of nutrients in the New Zealand diets as opposed to assuming equal weighting of all nutrients considered in the analysis, including more detailed analysis of protein quality (Berardy et al., 2019; Moughan, 2021; McAuliffe et al., 2022), when undertaking nLCA studies. For meal-level analysis, it will also be important to expand the analysis to consideration of typical additional food items consumed alongside the avocado or cheese as a 'topping on toast' meal (e.g. tomatoes, olive oil). Finally, future studies should include additional environmental impact categories in order to provide a more comprehensive comparison of the environmental impacts associated with alternative food items.

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Adapting the Agribalyse Life Cycle Inventory database to Australia – a first step towards a comprehensive Australian food and agriculture model

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Main topic/key words: Agribalyse, food supply chain, life cycle inventory database

1. Introduction

The food production and supply system is a major driver of human environmental impacts. In 2019, the IPCC highlighted that 21-37% of total anthropogenic greenhouse gas emissions are attributable to our global food system (Mbow et al., 2019). This estimate covers the entire life cycle of food products: agriculture and land use, storage, transport, packaging, processing, retail and consumption.

In Australia, the effects of climate change are already felt through increases in temperature, more intense and frequent heatwaves and flooding events. There has also been a long-term increase in extreme fire weather, with an increasingly long fire season across the country (CSIRO, 2018).

Given the significant nature of food supply chains on global (and domestic) environmental impacts, there is a clear need to shift consumption patterns towards lower impact products. To do so, it is paramount to inform consumers using reliable, verifiable and high-quality information.

Life Cycle Assessment (LCA) is a method used to evaluate the full cradle-to-grave environmental impacts of products and services, by assessing environmental flows at each stage of a product’s life cycle and relating them to a set of environmental impact indicators. The method aims to inform decision-making with measurable data and scientifically robust models. LCA could become a key source of information for the public, supporting decisions geared towards lowering consumption impacts. Agribalyse is a Life Cycle Inventory database developed in France for this purpose (Asselin-Balençon et al., 2020). The database has been in development since 2010 and aims to be a database for the agriculture and food sector, representing the French food supply chain in its entirety. It includes over 2,500 food products, modelled using a set structure. This study uses the data infrastructure developed for Agribalyse, and strategically adapt key parts of the database to produce an Australian version.

2. Material and methods

The latest version of Agribalyse at the time of this study (v3) was modified by linking critical parts of the model to Australian-specific data. The general structure of the database was not altered, the overarching assumption being that the approach used to model supply chains in Agribalyse was applicable to Australia.

The modifications to Agribalyse focused on key aspects, including critical inputs, primary commodity production, domestic product transport and imports, as discussed in the following sub-sections. The resulting Australian inventories were compared to the original French models to assess the magnitude of the variation between the two models and identify aspects which could further researched. The analysis used the Environmental Footprint impact assessment method (Fazio et al., 2018), as recommended under the Product Environmental Footprint method (European Commission, 2012), weighted to a single score using the method developed by Sala et al. (2017).

2.1. Critical inputs

This analysis assumed that while the demand for commodities such as electricity or water could be deemed equivalent in French and Australian production systems, the impacts associated with the delivery of these commodities will vary significantly. For instance, the energy requirements of the retail sector, as modelled in France, was assumed to be applicable to Australia. However, the environmental effects associated with producing electricity vary significantly between France and Australia. Thus, an Australian-specific grid model was required.

Alongside electricity production, water inputs and road transport were identified as critical inputs warranting the use of an Australian-specific model. In each case, the model was sourced from AusLCI v1.36 (ALCAS, 2021). These initial modifications of some critical aspects of the supply chain helped develop a model more representative of the Australian context.

2.2. Primary food commodity production

Primary food commodity production systems vary significantly from one country to another. Since a large proportion of the food consumed in Australia is produced domestically, particularly fresh products, Australian models were sourced whenever possible to represent domestic production. Existing life cycle inventory models were sourced from AusLCI, re-modelled from existing scientific literature, or developed from publicly available data.

Data availability was a significant driver in selecting which primary food commodity would be modelled using Australian-specific data, and which would remain as originally modelled in Agribalyse. The potential variation in production systems was also considered.

AusLCI models were used to represent the primary production of livestock (beef and lamb), broadacre crops (wheat, canola, barley, oats, lupins, maize, sorghum, chickpea, lentil, field beans, faba beans, soybeans and sugar cane), and horticultural crops (avocado, almonds, banana, broccoli, capsicum, lettuce, sweet corn, strawberry tomato and potato). Livestock models unavailable in AusLCI were represented using the literature. This included chicken (Wiedemann et al., 2012), eggs (Wiedemann and McGahan, 2011) and pork ((Wiedemann et al., 2016) and (Wiedemann et al., 2018)). A model of Australian milk production was developed from a range of industry data (Dairy Australia, 2020g, Dairy Australia, 2020a, Dairy Australia, 2020b, Dairy Australia, 2020c, Dairy Australia, 2020d, Dairy Australia, 2020e, Dairy Australia, 2020f, Dairy Australia, 2010), which supplemented a model sourced from the literature (Gollnow et al., 2014). These livestock models were all modified to include information reported in Australia's most recent National Inventory Report (Commonwealth of Australia, 2021).

A range of horticultural crops were also modelled for the study. These commodities are sold in large volume on the Australian market. A large fraction of these commodities is produced domestically, and no existing Australian-specific LCA models could be identified. To model their production, agronomical data sourced from gross margin tools was used. These documents are guidelines provided to Australian farmers to estimate the typical economic margin associated with growing specific crops. They provide estimates of the requirements for different crops, including water, pesticides, fertilisers and machinery use, as well as typical yield. As such, they are useful to model farming practices. Gross margin tools have been successfully used in the development of life cycle inventories for AusLCI in the past, in particular through the AusAgLCI project (Grant et al., 2014, Eady et al., 2014).

2.3. Domestic transport of raw commodities

In Agribalyse, freight is represented in three steps (see Figure 1):

- import transport of raw commodity, considering both transport in the cultivation country and transport to the country of consumption;
- domestic transport of raw commodity; and
- transport of processed commodities along the supply chain

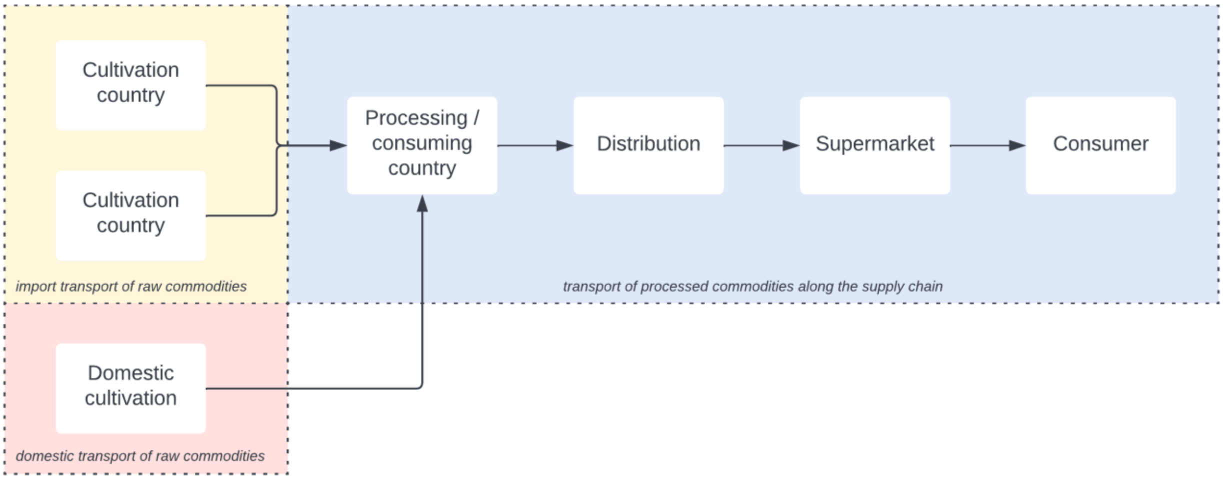

Figure 1 Freight modelling steps

Australia's sheer size means that domestic transport of food products will differ substantially from the freight effort required in the French market. Assumptions regarding freight of raw commodities were reviewed, considering intra- and inter-state transport.

Though statistics are available on the freight of all goods distributed in Australia, no comprehensive dataset is available for food commodities transport. The model used to represent the transport of raw commodities in Australia was therefore built on a series of assumptions, described in the following paragraphs, and summarised in Table 1.

Many food commodities are produced throughout the country, although some states produce larger fractions than others. To represent this, transport within the state as well as interstate should be considered. Tropical fruits are an exception, as they only grow in the Australian tropics, often in remote locations. As a result, these commodities will generally be transported over longer distances, mostly interstate.

Available statistics provide insights on the average split between modes of transport for non-bulk items in Australia (Bureau of Infrastructure, 2021). The modal split is only used for interstate transport, where rail or sea freight can occur. Intrastate transport is assumed to rely strictly on road freight. The figures used to represent raw commodity transport are reported in Table 1.

Table 1. Domestic transport assumptions

Agribalyse unit process	**Tropical fruits**	**All other commodities**	**Source**
Interstate fraction	**1.0**	**0.5**	Modelling assumption
Distance (km)	2000	1000	Modelling assumption
Modal split			
- road	0.80	0.80	(Bureau of Infrastructure, 2021)
- rail	0.16	0.16	(Bureau of Infrastructure, 2021)
- sea	0.04	0.04	(Bureau of Infrastructure, 2021)
Intrastate fraction	**0.0**	**0.5**	Modelling assumption
Distance (km)	*NA*	250	Modelling assumption
Modal split			
- road	*NA*	1	Modelling assumption

2.4. Import of raw commodities

Imports of raw food commodities to Australia are limited, apart from a range of specific commodities such as kiwifruit (78%) and garlic (76%) (Hort Innovation, 2019a, Hort Innovation, 2019b). International transport was considered if over 5% of a commodity consumed in Australia is imported. Data collected from the United Nations Statistics Division on the trade of food commodities to Australia (United Nations Statistics Division, 2020) as well as relevant data from industry bodies were used to estimate the fraction of imports.

A total of 68 countries of origin were considered. The model endeavours to include all countries representing at least 90% of total import for each commodity considered. Distances associated with imports were sourced from the CERDI sea-distance database (Bertoli et al., 2016), which provides a matrix of bilateral sea distances between 227 countries and territories. In addition, the database provides road transport within the country from the capital city to the nearest port. This was used as a proxy to estimate transport requirements from point of production to port in producing countries.

Whenever possible, the specific production system of imported food commodities was considered, though this was greatly limited by the amount of available life cycle inventories.

3. Results

Single score results were averaged at the product category level to allow comparing the French and Australian models. The results of the comparison are shown in Figure 2.

The analysis highlights that the environmental effects of food production in Australia is generally higher than in France, apart from sugar and confectionary, and fats and oils. The variation is most pronounced for baby food (+78%), non-alcoholic beverages (+45%) and miscellaneous (+43%), which includes condiments, sauces and cooking aids. On average, impacts were modelled as approximately 24% higher in Australia than in France.

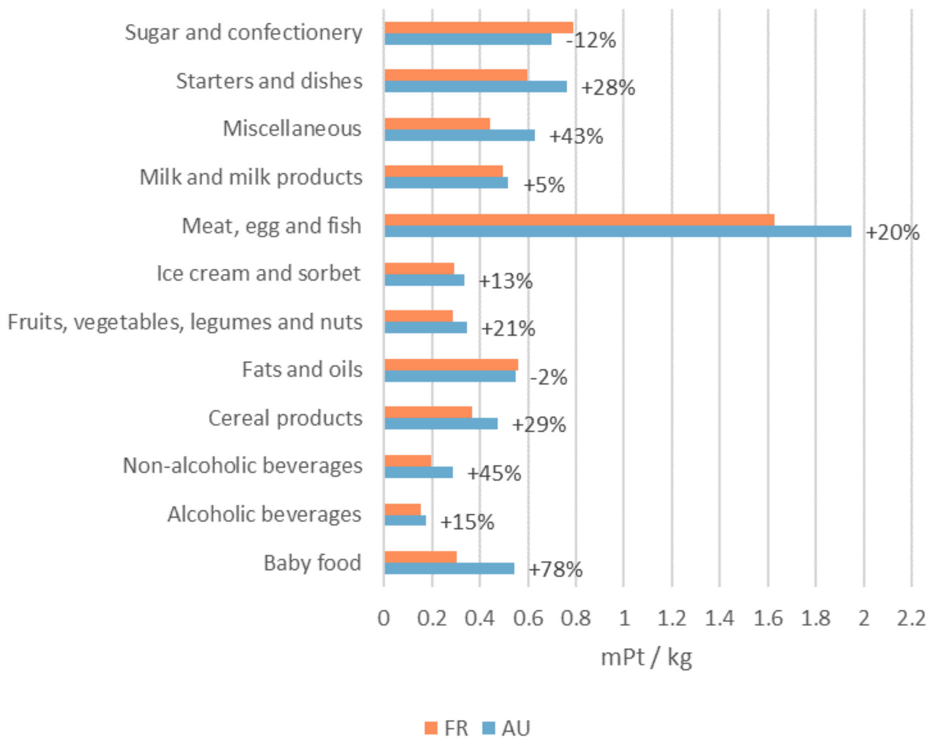

Figure 2 Comparing Australian single score results against the original Agribalyse models.

A further analysis highlights the significance of the electricity grid on the food supply chain. Replacing the French grid with the Australian grid is one of the drivers of the variation in impacts, which is particularly associated with the greater reliance on coal-fired power generators in Australia. Figure 3 below shows the fraction of the Australian single scores associated with electricity production and distribution, for the Australian model. It shows that importance of the electric grid on commodities which rely on chilling and freezing (e.g. ice cream and sorbet, 31%), or which include significant level of processing (e.g. miscellaneous, 34%). To an extent, the variation in electricity grid help explain the variation between the French and Australian models.

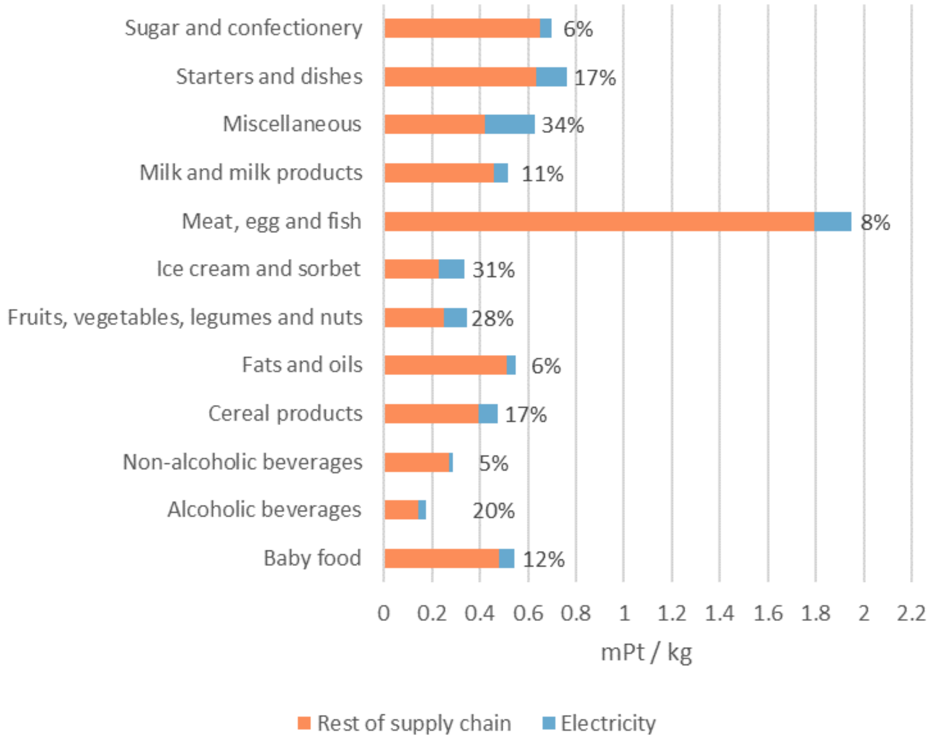

Figure 3 Fraction of single score associated with electricity production in the Australian supply chain.

Variations in freight assumptions do not result in significant variations in the model, apart from some specific commodities. For instance, a fraction of mangoes consumed in France are imported through air freight, while transport in Australia is typically done by road freight. When including air freight in the French supply chain, over 80% of the single score becomes associated transport the primary commodity. On the other hand, in the Australian model, the effects of transporting the primary commodity represent approximately 10% of the single score. The single score result is approximately five times larger in the French model compared to the Australian version.

Because Agribalyse aims to include a wide range of food commodities, proxy models are routinely used to represent commodities. Choosing a proxy model representative of the production system being modelled is therefore critical. Proxy model choices were reviewed, leading to significant variations in results. For instance, peanut production (a ground nut) was used in Agribalyse to represent tree nuts such as hazelnuts and pistachios. When available, specific models were used instead for this study. In that case, inventories sourced from the World Food LCA Database (Nemecek et al., 2019) were used.

When looking at specific product categories, certain aspects are worth discussing. In the grazing livestock models, the results against most impact categories are well aligned between the two models, with less than 20% variation in most case. The most significant variation is land use, where the Australian model result in impacts are 4.5 times larger than in Agribalyse. Australian grazing livestock are generally grown on extensive grazing land – in particular meat livestock. In fact, 75% of Australia's land mass is used as rangelands, with 58% of this area being occupied by pastoral enterprises (Australian Government Department of Agriculture, 2019). These are regions where rainfall is either too low or erratic for agricultural cropping or improved pasture. Thus, the amount of land use per animal is large, especially when compared with the French production system.

In the case of pork production, the variation in environmental effects is mostly related to direct emissions, particularly from manure management, which can represent up to 17% of the single score. Our model suggests that these emissions are significantly higher in Australia than in France, under current production conditions. Another significant aspect in the supply chain of pork is the source of feed, which represent approximately two thirds of the impacts. This is also the case in the poultry supply chain.

4. Discussion

This newly developed model has the potential to help inform decision-makers and consumers on the environmental effects of their food consumption. It provides a wealth of data and a set structure which can be leverage for future Life Cycle Assessment work. It is a first step, which, if adequately maintained and built upon, could become a great resource. The analysis highlighted several aspects which could be further developed, as well as a range of opportunities for application.

4.1. Future research

There are many opportunities to build on and refine the existing model using data that is already available. Agribalyse includes fractions of commodities lost at each step of the supply chain. These modelling assumptions could be reviewed and updated using food loss data developed for the updated Australian national food waste baseline, a model which was produced for the National Food Waste Strategy Feasibility Study (Food Innovation Australia, 2021). These could then be linked to Australian-specific waste management models, as published in AusLCI.

In this analysis, some large-volume horticultural commodities were modelled, using Gross Margin tools to represent cropping requirements and average yield. This approach could be extended to other horticultural commodities such as tree nuts, grapes or stone fruits, which were out of the initial scope of work. Conducting this work could help produce more specific cropping models, improving the breadth of cropping systems considered.

Other potential improvements would be worth investigating. Food processing steps could be reviewed in the Australian context, as they can represent a significant proportion of impacts in some case (e.g. drying processes) and technology may vary significant between France and Australia.

Although Australia produces most of its fresh food products domestically, there are significant flows of processed food commodities that are imported. Tracing the origin of processed commodities and matching their production systems with a representative life cycle inventory is currently impossible.

In Agribalyse, this issue is dealt with by assuming that all food processing takes place domestically. This keeps the model manageable and is assumed to have marginal effects on the results. For certain commodities, (e.g. tomatoes, strawberries, chicken and beef), Agribalyse identified a significant variation in the fresh and processed supply chain, with variation in import sources and raw commodity production systems. To produce a model that is representative of the supply chain, two raw commodity market mixes were developed in Agribalyse, one for commodities destined to the fresh food market, and the other for commodities destined to be processed.

This approach was replicated in the Australian model, but only for tomatoes and strawberries, as the vast majority of chicken and beef is produced domestically. This should be expanded to other commodities, such as pork. Indeed, a large proportion of processed pork consumed in Australia is imported, while most fresh pork meat is produced domestically. Thus, two separate supply chains could be built.

Finally, data could be collected directly from industry groups, producers and food manufacturers. This would allow to produce inventories which are more representative of the Australian production systems, while providing these stakeholders with relevant information on the environmental effect of the supply chain they operate in.

4.2. Opportunities for application of the model

In France, the Agribalyse database is used as an important source of data for the Eco-Score, an environmental label used on food products packaging at retail introduced in 2021 and expected to become mandatory by 2024. The model developed here could be used for a similar purpose, providing information directly to consumer on product packaging.

Decision-makers could also benefit from this information. Industry averages could be developed for specific end-products and commodities, allowing farmers and food manufacturers to benchmark their operations. The average inventories could also be used as a basis in Life Cycle Assessment work, tailored to specific clients, and further developed to include impact mitigation options. At the other end of the supply chain, the models could be used by supermarket to estimate the emissions associated with their entire range of food products. Restaurants could use it in the development of menus, and households in their meal planning.

5. Conclusion

This analysis was a first step towards developing a complete food Life Cycle Inventory database specific to Australia, with a primary production to consumer boundary. It builds on the data infrastructure developed by Agribalyse, while adapting the data to produce a model that is representative of Australian production systems. There is significant scope for improvement, and a wealth of data which could be added to build on the existing work.

This initial database was developed so that it could be updated and built upon over time. This may be done as data becomes publicly available, or through direct collaboration with specific industry sectors. Though it was noted that some improvements could be made to the model, it already is a tremendous source of information.

The resulting model could have many applications in terms of informing consumers through product labelling, as well as decision makers through the ability use and tailor the model to specific supply chains, considering impact mitigation options and providing a benchmark tool to decision makers in the agriculture and food supply chain, thus participating in the transition towards a more sustainable consumption of food.

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Water, energy, and greenhouse gas footprint of the city food system in Australia

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1. Introduction

Food is one of the most water-energy and greenhouse gas (GHG) intensive consumable goods. The food sector currently consumes around 70% (2.8 trillion m^3) of water withdrawals (FAO, 2017b) and 30% (95EJ/yr) of energy around the world (FAO, 2011). In terms of water use, livestock production and food processing together account for more than 80% of the total food sector water consumption (FAO, 2017a). In terms of energy use, processing and transportation (~40%), crop (~10%), and livestock (~10%) production account for around 60% of annual food sector energy consumption in high-GDP countries, whilst in low-GDP countries preparation and cooking (~45%), and processing and transportation (35%) are more significant (FAO, 2011). The high fossil fuel-based energy dependency of food production is a matter of concern because, based on a recent estimate by Crippa et al. (2021), the food system contributes around one-third (range 25% to 42% with a 95% confidence interval) of global greenhouse gas (GHG) emissions.

Islam et al. (2021) reported outdoor vegetable growing in Australia and other sectors, like grain growing, has a comparatively higher water and energy footprint. While Frankowska et al. (2019) reported annual consumption of vegetables in the UK generates 20.3 Mt CO_{2eq} (~4.5% of national emissions) GHG emissions from 260.7 PJ energy consumption (~5% of national final energy consumption), and 253 Mt eq. of water. On the other hand, cities around the world are considerably targeting to reduce their environmental footprint through investment in sustainable initiatives (Koop and van Leeuwen, 2015), such as City Region Food Systems (CRFS) approach. Therefore, systematic water, energy, and GHG emissions evaluation and interpretation of the city food system are crucial for water and energy-sensitive, and low-carbon city planning to increase the resiliency against environmental shock (Vanham et al., 2019); and sustainable city development by reducing environmental footprint (Chini et al., 2017).

Information about where to direct eco-efficiency efforts is needed to ensure a sustainable city food system due to current interest in achieving net zero, as well as sustainable consumption and production (SDG12). Therefore, the objective of this work is to assess the water, energy, and GHG footprint of an Australian city's food system.

2. Material and methods

2.1. Goal, scope, and functional unit

As a representation, this research modelled fresh vegetable production in Queensland and home consumption in Southeast Queensland. In total, 30 types of fresh vegetables were evaluated for a field-to-plate system boundary. Inputs to all these processes were accounted for. The functional unit was 1 kg of fresh vegetables produced and consumed, and the associated system boundary of this research is presented in Figure 1.

2.2 Life cycle inventory and impact assessment

The inventory data for vegetable growing was sourced from the Australian AusLCI database (V1.34) wherever possible, where highly representative data for Queensland production systems were available (for 17 out of the 30 vegetables). Where AusLCI data was not available, inventories from the AGRIBALYSE (V3.0.1) (for 9 out of the 30 vegetables) and Ecoinvent (V3.6) (for 4 out of the 30 vegetables) databases were adapted for conventional growing practices and inputs for the climate conditions relevant to the major QLD growing areas (Lockyer Valley, Bundaberg, Stanthorpe, Buderim). The energy and water consumption throughout the life cycle stages are presented in Table 1 and Table 2. According to the 'Queensland fresh produce' report, most vegetable growers have on-farm facilities for washing, packing, and storing fresh produce before transporting it to Regional Distribution Centre (RDC) (QLDGOV, 2016). The recommended storage time of fresh vegetables in Australia is 5 to 10 days before consumption (A2EP, 2017). Two days of storage on-farm facility was assumed before transporting to RDC.

On-farm processing of fresh vegetables involves cooling and washing with chilled water, containing disinfectants which were assumed to be sodium hydroxide (0.198 kg), hydrochloric acid (0.63 kg), and sodium hypochlorite (0.49 kg) per tonne of vegetables (Frankowska et al., 2019; Moreno et al., 2018). For some of the vegetables, such as capsicums, broccoli, cabbage, chilies, eggplant, garlic, and onions washing process is not needed.

Most fresh vegetables were assumed to be sold loose without primary packaging but with secondary packaging (cardboard and styrofoam boxes). Some use of polyethylene terephthalate (PET) trays, punnets, and cling film for a portion of beetroot, tomatoes, beans, zucchini, capsicum, parsnips, chilies, corn, and cucumbers were accounted for. Minimally processed fresh vegetables (chopped, grated) were also considered, which have some plastic packaging (polyethylene bags). Since the exact proportion of loose and packed sold products is difficult to obtain, we assumed an equal proportion of loose packed with polyethylene, PET trays, and punnets. Packaging used for various fresh vegetables and processed products is presented in Table 3.

Most fresh vegetables in the retail were assumed to be sold whole (without peeling and cutting), with only some receiving minimal processing (e.g., pumpkin, zucchini, broccoli, cauliflower, carrot, potato chips, sweet potato chips, and green beans in a mixed, or, solo vegetable pack for stew and stir-fry packs). The average distance traveled by the consumer in Brisbane and the surrounding city to supermarkets is 7 km (round trip) (Nejad, 2016). We also assumed the same distance. The transport data is based on AusLCI. The energy and water consumption estimation approach for retail and household, as well as the waste generation and management approach, are presented in the supplementary materials.

The life cycle environmental impacts were estimated using SimaPro software 9.1.1.1, based on the Australian Life Cycle Assessment Society (ALCAS) Best Practice Guide for Life Cycle Impact Assessment (LCIA) in Australia V 2.04. Impact categories included were Climate change (kg CO2eq), Resource depletion –fossil fuels (MJ) as an indication of primary energy demand, and Consumptive water use (Leq), which derived by multiplying water use with water stress factors.

3. Results

3.1. Primary energy demand

The life cycle primary energy demand is presented in Figure 2. As can be seen, the life cycle primary energy demand ranges from about 13 MJ/kg for parsley and other fresh herbs and ginger to 77 MJ/kg for eggplant. Other vegetables like beans (66 MJ/kg), cucumbers (63 MJ/kg), potatoes (60 MJ/kg), and cabbage (55 MJ/kg) also have a relatively large life cycle primary energy demand (Figure 2a).

The farm production and processing are the hotspots of beans, chilies, cucumbers, and eggplant. For most of the vegetables, downstream energy impacts of refrigerated retail and household (through electricity consumption) are comparatively higher, such as for cabbage, carrots, cauliflower, celery, onions, and parsnips. Along with refrigerated retail, household consumption contributes significantly through refrigeration and cooking. (Figure 1a). In terms of resource input, electricity is the highest, which is presented as electricity (other) in Figure 2, to represent electricity used for slicing, peeling, HVAC, lighting, display cabinet operation, etc. Life cycle water-related energy includes energy used for irrigation, energy used for washing during the process and in the household, and cooking. Electricity (other) use ranges from around 20% for capsicum to around 45% for cabbage, cucumbers, eggplant, lettuce, leafy salad, etc. Life cycle water-related energy use ranges from around 15% to 40% for different studied vegetables, in comparison with the supply chain fuel use (diesel use for transport and tractor) (~10% to 30%), agrochemicals (~5% to 15%), and packaging materials (~1% to 30%) (Figure 2b).

3.2. Water use

To assess the life cycle water use impact of the studied fresh vegetable food system, consumptive water use (L_{eq}) (water use multiplied by water stress factors) was used as an indicator based on the ALCAS Best Practice Guide for LCIA in Australia V 2.04 (Renouf et al., 2018). This research only considered fresh, minimally processed, loose vegetables. Vegetables that are more highly processed (e.g., pre-chilled in plastic bags may have higher water use for processing) were not considered. As indicated in Figure 3, the volume of life cycle water use varies widely among the vegetables. Sweet corn (~86 Leq/kg) and broccoli (~82 Leq/kg) have the highest overall life cycle water use impact mainly due to high water consumption during the growing phase (4 and 3.5 ML/ha, respectively). At the lowest end of the range, parsley and other fresh herbs require around 11 L/kg of water. The life cycle water use impacts of the remaining vegetables range from ~14 to 68 L eq./kg. Farm production is the key contributor to the life cycle water use impacts for most vegetables, accounting for 50%–95% of the total, particularly due to irrigation water use. This is followed by the processing stage (5-45%). For beans, beetroot, carrots, peas, potato, and sweet potatoes, processing also requires significant water for washing and the life cycle embodied water of energy inputs. In the household, the life cycle water use impacts contribute between 2% and 11% (cabbage and potato) for cooking and food preparation due to the boiling water and life cycle embodied water of energy inputs.

3.3. GHG emissions

As indicated in Figure 4, the life cycle GHG emission also varies considerably among vegetables and mostly mirrors energy use. Hence potatoes and beans (~9 kg CO_{2eq}/kg) have the highest life cycle GHG emissions impact mainly because of high energy consumption during growing and processing. At the lowest end of the range, fennel, parsley, and other fresh herbs emit around 2 kg CO_{2eq}/kg. The life cycle GHG emissions impacts of the remaining vegetables range from ~3 to 8.5 kg CO_{2eq}/kg. Farm production, processing, and retail are the key contributors to the GHG emissions impacts for most vegetables, mainly because of energy (electricity and diesel) consumption. Life cycle GHG emissions from farm production account for ~10%–55% of the total, followed by processing (~5-35%) and retail (~4-40%). For beans (34%), beetroot (30%), cucumber (20%), eggplant (28%), peas (26%), potato (20%), and sweet potato (18%), processing requires significant energy compared to cultivation. The life cycle GHG emissions from retail are basically due to the electricity consumption from refrigerated storage, displaying cabinets, lighting, and HVAC. At the household level, the GHG emissions vary from 10% to 35% from energy consumption for refrigerated storage, cooking, and washing (Figure 4b).

4. Discussion

There exists a marked variation of impact methods and system boundaries (farmgate and factory gate) used in different LCA studies on vegetables. Comparatively fewer LCA studies have been conducted on vegetables considering field-to-plate system boundary. However, this study compared the assessed results with other LCA studies on vegetables though there exists marked variation in system boundary, cultivation practices, processing, and transportation. For example, the GHG emissions of Beans (8 kg CO_{2eq}/kg), Broccoli (2.5 kg CO_{2eq}/kg), Lettuce (3.8 kg CO_{2eq}/kg), Spinach (2 kg CO_{2eq}/kg), and Capsicum (3.5 kg CO_{2eq}/kg) in UK (Audsley et al., 2010) are close to the assessed values in this study 9.31, 5.6, 2.79, 2.23 and 2.85 kg CO_{2eq}/kg, respectively. Frankowska et al. (2019) estimated the GHG emissions of Beans (3 kg CO_{2eq}/kg), Broccoli (2 kg CO_{2eq}/kg), Lettuce (3 kg CO_{2eq}/kg), Spinach (1.9 kg CO_{2eq}/kg), and Capsicum (2.5 kg CO_{2eq}/kg) considering the farm to plate system boundary for the UK. On the other hand, Stoessel et al. (2012) analyzed the carbon footprint of fruits and vegetables considering field to retail system boundary in Switzerland and reported the GHG emissions of Broccoli (1 kg CO_{2eq}/kg), Lettuce (3.2 kg CO_{2eq}/kg), and Capsicum (1.5 kg CO_{2eq}/kg). Different farming practices and underlying assumptions related to the studied systems can affect the results. For instance, the GHG impacts of tomatoes cultivated in greenhouses in Spain and outdoor in Italy is around 30 and 78 times lower, respectively, than the same system in the UK (Frankowska et al., 2019). Due to the variation of underlying assumptions and cultivation practices in different countries (e.g., irrigation practices), the results obtained in this study compare reasonably well with other LCA studies.

This study recommends simultaneous analysis of water and energy efficiency enhancements, along with GHG emission reduction, for a sustainable city food system policy intervention. Such specific food system-oriented adoption of mitigation measures can be tested by aligning widescale national targets. For example, Wachsmuth and Duscha (2019) showed that sub-sectors-oriented efficiency and emission reduction targets for energy and carbon intensity reduction across different end-use sectors are more stringent than aggregated targets in the EU. Besides, Mundaca et al. (2019) and Creutzig et al. (2018) also recommend that sub-sector-oriented measures have the potential to contribute towards global environmental benefits, such as limiting global warming to 1.5°C.

5. Conclusion

This study investigated water, energy, and GHG emissions of city vegetable food systems considering field-to-plate system boundaries. It considered 30 fresh produce vegetables grown in QLD and consumed in SEQ. The life cycle GHG emissions from most of the studied vegetables ranged from ~3 to 8.5 kg CO_{2eq}/kg. Life cycle GHG emissions from electricity (other) use (e.g., slicing, peeling, storage, HVAC, lighting, and display cabinet) were highest and ranged from around 25% (capsicum) to around 55% (beans, carrots, celery, cucumbers, and garlic). Life cycle GHG emissions from water-related energy use ranged from around 12% to 35% for different studied vegetables compared to agrochemicals (~5% to 35%) and supply chain fuel use (diesel use for transport and tractor) (~2% to 26%). This indicates the importance of directing the eco-efficiency programs towards energy saving by water saving along the entire city food system compared to much-explored issues, such as fertilizer application, transport, and packaging efficiency improvement.

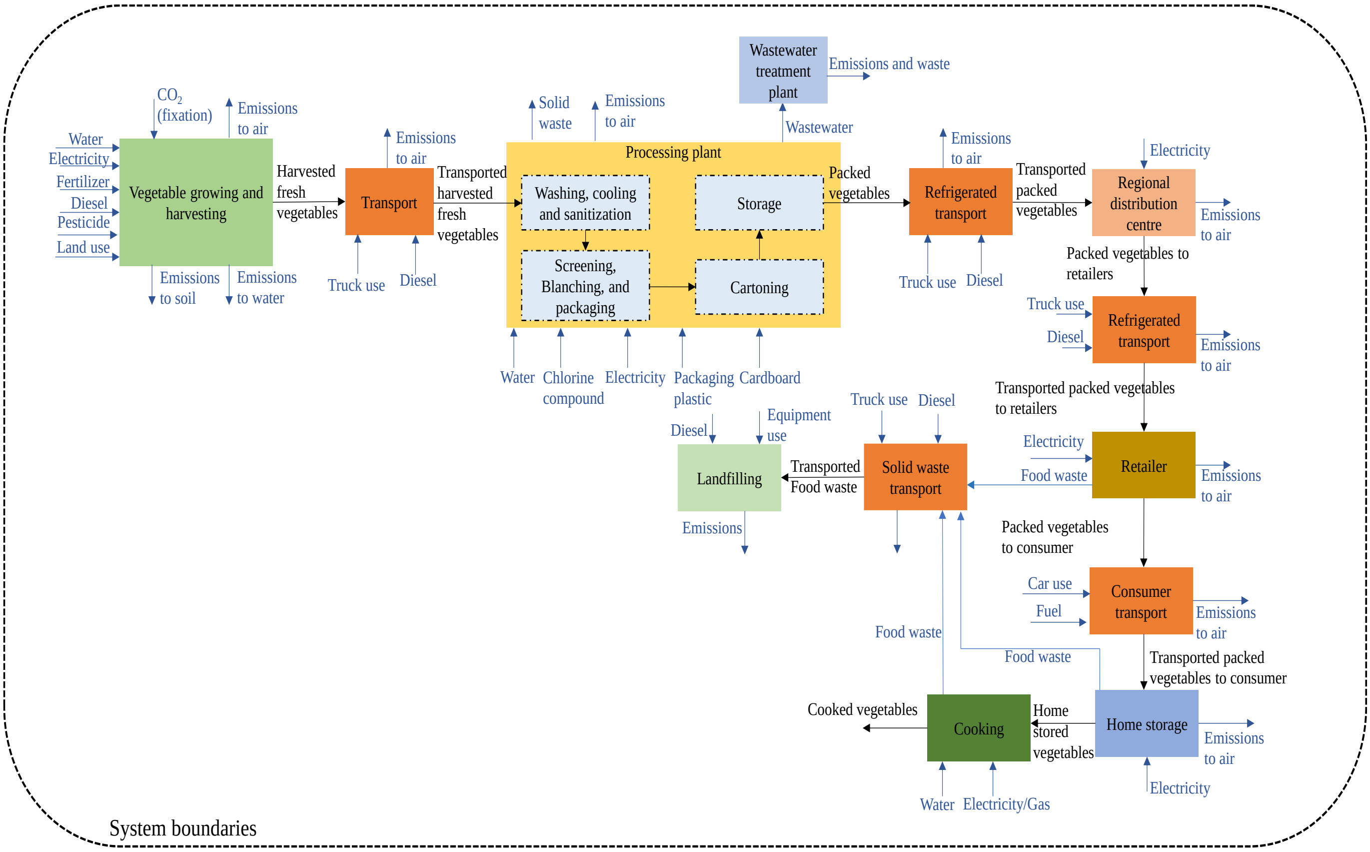

Figure 1: System boundaries used for the evaluation of fresh-packaged vegetables from the field to the plate.

Table 1: Summary of assumed energy use for vegetable growing and processing.

Production stage	Electricity	Refrigeration (Storage)	Steam (tonne/tonne of product)	Reference
Irrigation*	0.335 kWh/1000L of irrigated water	--	--	AusLCI
Irrigation**	0.206 kWh/1000L of irrigated water			
Washing	60 kWh/tonne of product	--	--	(Frankowska et al., 2019)
Slicing	8.9 kWh/tonne of product	--	--	
Peeling	3.7 kWh/tonne of product	--	0.9	
Blanching	8.7 kWh/tonne of product	--	0.137	
Freezing	180 kWh/tonne of product	--	--	
Storage	--	1.5 (fresh products) kWh/ tonne product/day ***	--	
Storage	--	0.75 (fresh products) kWh/ tonne product/day ****	--	
Display cabinet at retail	17 kWh/tonne product/day	--	--	(Canals et al., 2007)
Lighting and general at retail HVAC at retail	Electricity consumption at retail for the studied vegetables was allocated based on the approach mentioned in the retail stage.			

* Beetroot, Broccoli, Brussels Sprouts, Cabbage, Carrots, Cauliflower, Celery, Fennel, Garlic, Ginger, Herbs—Parsley and other Herbs, Leafy Asian Vegetables, Leafy Salad, Leeks, Lettuce, Onions, Parsnips, Potatoes, Pumpkins, Spinach, Sweet corn, Beans, Sweet potatoes, ** Capsicums, Tomatoes, Chillies, Cucumbers, Zucchini.
*** Celery, Sweetcorn, Beetroot, Carrots.
**** All other vegetables studied

Table 2: Summary of assumed water use for vegetable growing and processing.

Stages in production	Vegetables	KL/tonne	Reference
Irrigation	Beetroot, Broccoli, Brussels Sprouts, Cabbage, Cauliflower, Celery, and Pumpkins.	160	AusLCI
	Beans, and Sweet Corn.	152	
	Capsicums, Chillies, Cucumbers, Eggplant, and Zucchini.	79	
	Carrots, Leeks, and Parsnips.	53	
	Celery, Herbs—Parsley and other Herbs, Leafy Asian Vegetables, Leafy Salad, Lettuce, Spinach, and Fennel.	179	
	Tomatoes	83	
	Sweet corn	408	
	Potatoes, and Sweet potatoes	156	
	Garlic, Ginger, and Onions.	48	
Washing during fresh packaging and processed products	Beans, Brussels Sprouts, Cauliflower, Spinach, Fennel, Ginger, Lettuce, Leafy Asian Vegetables, Leafy Salad, Leeks, Peas, Sweet Corn,	3.85	(Frankowska et al., 2019)
	Beetroot, Carrots, Parsnips, Potatoes, Sweet potatoes,	5.7	
	Cucumbers, Celery, Pumpkins, Tomatoes, Zucchini	2	

Table 3: Summary of assumed packaging formats.

Packaging Type	Material								Reference
	A	B	C	D	E	F	G	H	
	kg/tonne of vegetables								
Plastic bag	3.9	-	-	-	-	-	-	-	(Canals et al., 2007; Frankowska et al., 2019)
Cardboard box (secondary packages)	-	-	-	22.8	-	-	-	-	
Plastic wrap (cucumber)	-	5	-	-	-	-	-	-	
Punnet with wrap	3.7	-	53	-	-	-	-	-	
Polystyrene boxes (Broccoli)	-	-	-	-	-	-	-	11	
Plastic tray with wrap	3.9	-	49.4	-	-	-	-	-	

Note: A. Polyethylene, B. Polypropylene, C. Polyethylene terephthalate, D. Cardboard, E. Glass, F. Aluminium, G. Steel, H. Polystyrene

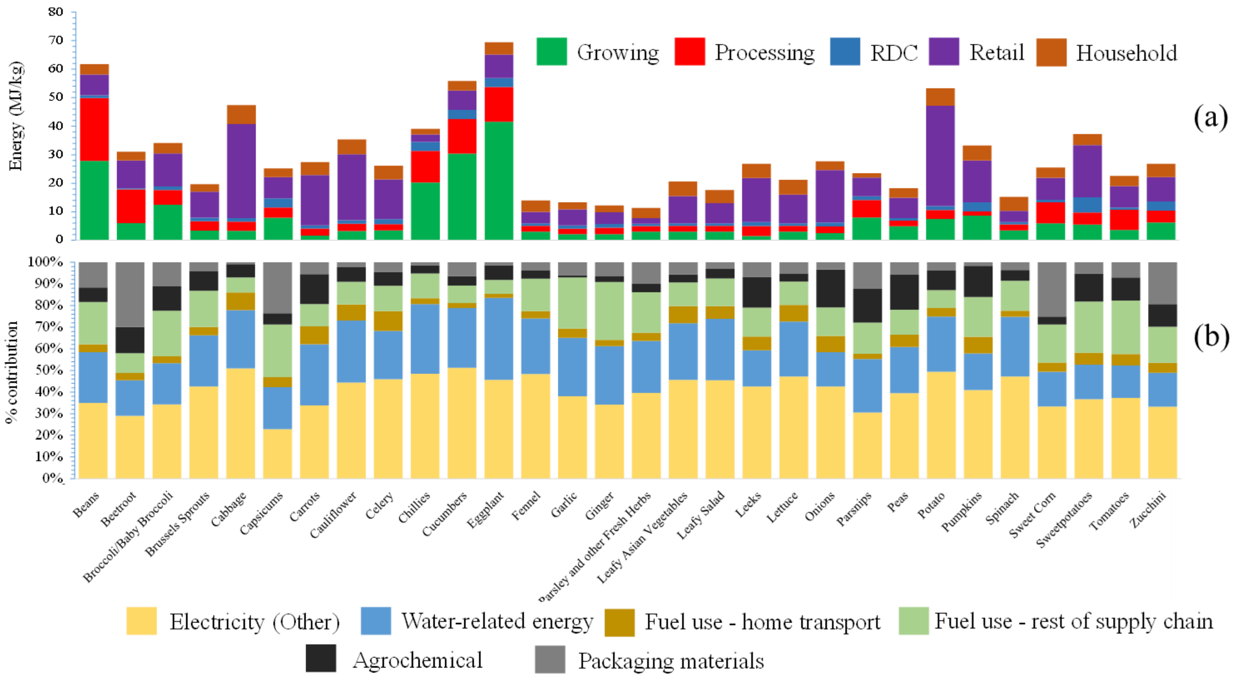

Figure 2: Life cycle (field to plate) energy used for vegetables produced in QLD and consumed in SEQ (MJ/kg), disaggregated by (a) life cycle stages, and (b) resource inputs.

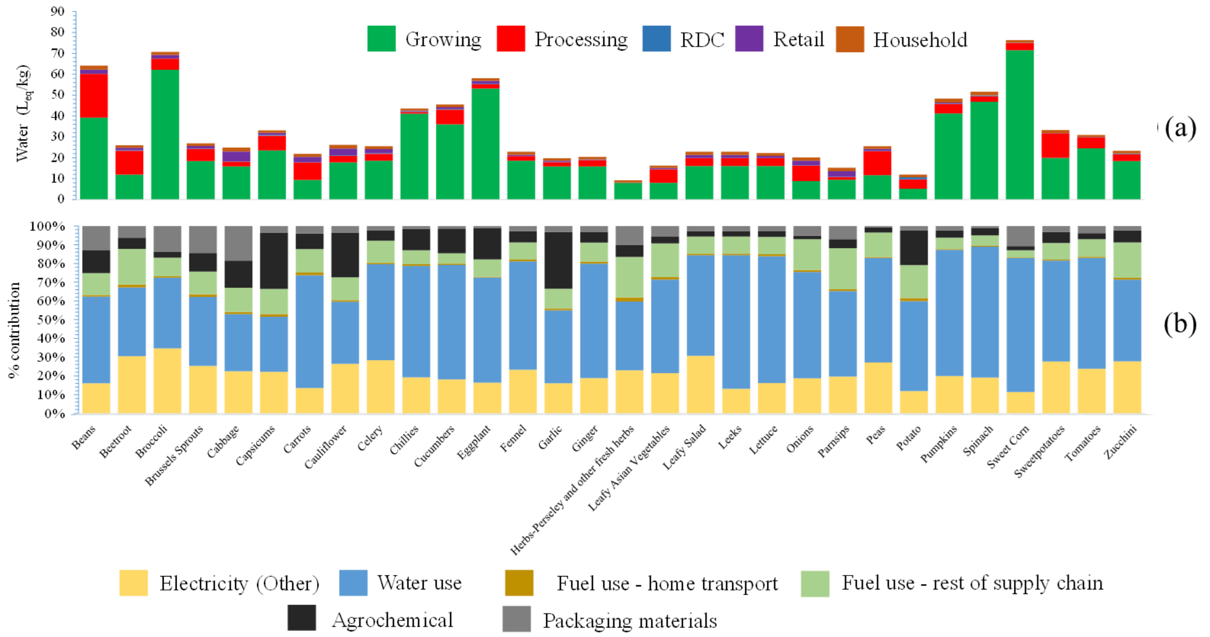

Figure 3: Life cycle (field to plate) water used for vegetables produced in QLD and consumed in SEQ (MJ/kg), disaggregated by (a) life cycle stages, and (b) resource inputs.

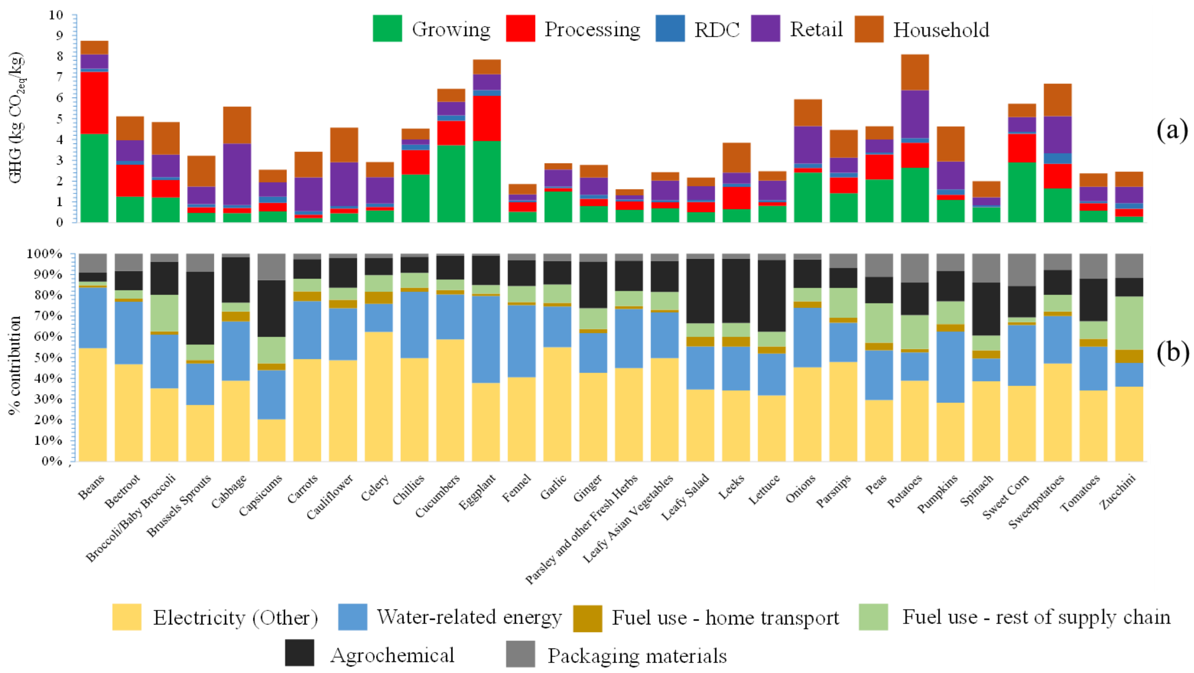

Figure 4: Life cycle (field to plate) GHG emissions for vegetables produced in QLD and consumed in SEQ (MJ/kg), disaggregated by (a) life cycle stages, and (b) resource inputs.

Acknowledgements

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Buildings Abstracts

Guiding early design using whole of life estimates: a building case study

Thursday, 20th July - 09:00: Buildings

Ms. Lucy Marsland [1]

1. *Atelier Ten*

Reducing the whole of life GHG emissions of buildings is a key design driver in 2023. However, the opportunity for greatest reduction potential is also the phase of lowest information availability: before schematic design. LCA practitioners face the uncomfortable challenge that 'all models and wrong, but some are useful' when supporting net zero/ neutrality pathways and certification responses early in building designs. The idea of this study is to mine publicly reported emissions data and estimate the whole of life emissions contributions of a cultural institute building before schematic design. This will include a literature review of publicly reported emissions data, analysis of the distribution of available data, and development of a reduction roadmap to guide early design decisions. The utility of this estimate model that is 'wrong but useful' is to support design decisions early, when the greatest impact can occur, and to communicate the importance of a whole of life approach to an integrated design team. This study found that there is some public reporting data for major emissions sources relevant to the cultural institute typology, however, the distribution of this data has a significant range. In estimating the contributions to whole of life emissions for the cultural institute design, a reduction roadmap was developed for the purpose of educating and guiding the design team towards facilitating emissions reductions. This case study demonstrates the value of mining public reporting data in early stages of building design and indicates that there is more work to do in making this data accessible and its assumptions transparent. Further, we are reminded that 'all models and wrong, but some are useful' and that the motivation of early life cycle accounting of buildings is to educate and to guide reductions in emissions towards net zero.

Tracking and analysis of GHG emissions for construction and infrastructure projects.

Thursday, 20th July - 09:00: Buildings

Dr. Umer Chaudhry [1], *Mrs. Jacqui Bonnitcha* [1]
1. LendLease

Lendlease has a Mission Zero target to reach net zero carbon by 2025 and absolute zero by 2040 with no offsets. It is estimated that embodied carbon (EC) is responsible for around 80% of Lendlease's greenhouse gas (GHG) emissions. In line with our Mission Zero Roadmap, we have developed several in-house tools to support the business with tracking and analysis of EC emissions across LendLease. One of our tools, known as the Asphalt-Concrete (AC) tool, is an asphalt and concrete GHG emissions calculator. The tool is tailor-made for the Lendlease Communities' business to track their progress and develop EC reduction roadmaps to reach mission zero targets. Concrete and asphalt are the largest contributors of scope 3 emissions in infrastructure projects therefore the AC tool is primarily focused on infrastructure assets made of concrete and asphalt. It takes inputs from project bills of quantities, engineering drawings and specifications data mapped against different asset types and sub-components of infrastructure projects. Using these various data sources, it then calculates the A1-A3 emissions of individual structural components as well as the asset type normalised per unit land/lot size. This study presents the tool features and the significance of LCA data collection at different stages of development. A case study will cover the emissions analysis of one of the Lendlease Communities' projects. This includes the engineering and emissions data extraction process, the analysis (A1-A3) and finally, the development of GHG emissions reduction roadmap. The analysis will contain different scenarios comparing virgin materials with alternative options. This will assist the project team to reduce their emissions during the project and provide a holistic overview of the EC reductions over the project timeline thus guiding sustainability strategies and decision-making.

Life cycle assessment considerations of prefabricated construction

Thursday, 20th July - 09:00: Buildings

Dr. Leela Kempton [1], ***Dr. Matthew Daly*** [1]

1. University of Wollongong

Prefabricated construction has been promoted as a potential method to improve sustainability in the construction industry, particularly through increasing production efficiency and reducing waste. However, the evidence base for substantiating these claims, particularly in the Australian context, is underrepresented. Life cycle assessment (LCA)provides a holistic approach to evaluating some of the sustainability benefits of prefabricated construction and quantifying these impacts in the context of the whole building life cycle. This paper explores how LCA can be used to assess and compare the sustainability impacts of prefabricated construction, conducted as part of the Australian Manufacturing Growth Centre's 'Prefabrication Innovation Hub'. A case study consisting of a prefabricated two bedroom holiday cabin in Australia was used. Waste audits undertaken in the factory during construction of the cabin were combined with material data collected to conduct a LCA focused on the manufacture of the cabin, including waste material end of life considerations. Although challenges were faced in obtaining the data required for the LCA and determining appropriate materials to use from available databases, LCA has been shown to be a useful tool in evaluating the embodied carbon and embodied energy within the case study building. From this basis, scenarios investigating the impact of material wastage rate and recycling options have been tested, demonstrating that current levels of waste produced are responsible for 6.4% of the embodied carbon in the case study building. Halving the level of waste generated just in the foundations of the building through leaner construction practices and increases material efficiency could reduce the embodied carbon in the foundations by 2.7%. This provides a framework to considering sustainability of prefab construction, however further work is needed to identify how other aspects such as future adaptability and reuse of the buildings could be considered in this framework.

Whole-life Baseline Carbon Assessment of Residential Building Stock - A Victorian Case Study

Thursday, 20th July - 09:00: Buildings

Ms. Maxine Chan [1], ***Prof. Greg Foliente*** [1], ***Dr. Seongwon Seo*** [2], ***Dr. Felix Kin Peng Hui*** [1], ***Prof. Lu Aye*** [1]
***1.** The University of Melbourne, **2.** Hobsons Bay City Council*

Assessing residential building decarbonisation opportunities requires a whole-life approach, given the increasing share of embodied carbon as housing becomes more energy efficient. Since most of the projected housing stock would consist of existing buildings, emissions from renovation should also be included in determining both embodied and operational carbon in the residential building sector.
A bottom-up typology framework was developed to estimate carbon emissions for existing and new housing up to 2050, scalable from local government area (LGA) to state-level jurisdiction which allows for granularity in testing scenarios for the future. Housing typologies were developed for existing, new, and renovation housing stock based on census data. Operating carbon was obtained using building energy simulation while embodied carbon data was accounted from localised life cycle construction datasets. The state of Victoria along with its corresponding LGAs was used as a case study for said framework.
Heating load comprised most of the operating energy demand for most typologies while external walls and floors contributed significant embodied carbon for new residential buildings, particularly for detached houses. For Victoria, detached houses built prior to 1991 contributed most of the operational carbon, however with high construction rates set for most LGAs, new housing may contribute more GHG emissions in 2050. Brick veneer housing yielded more embodied carbon from the external wall compared to timber homes while concrete slabs used in floors also incurred a large amount of embodied carbon for the residential building stock. Renovating existing housing has the potential to reduce operating energy demand while emitting less embodied carbon, thus policies on this should be considered in developing decarbonisation pathways.
Using the bottom-up typology whole-life carbon framework offers granularity in analysing individual-level carbon impact which can be expanded to LGA and state level.

Life Cycle Assessment of Different roofing materials

Thursday, 20th July - 09:00: Buildings

Ms. Anuradha Dinumgalage [1]***, Prof. Maheshi Danthurebandare*** [1]

1. *University of Peradeniya*

Asbestos is used as a common roofing material around the world due to its' commercially viable properties.However, many countries have banned or taken actions to ban asbestos due to its bad impact on the health and environment. The Sri Lankan Government also implemented a ban on asbestos use. Still, the chrysotile asbestos type is widely used in Sri Lanka, while encouraging the development of the clay roofing tile industry since clay roof tiles have been identified as a better alternative for asbestos sheets. Thus, both asbestos and clay roof tiles have a bad impact on the environment, a proper environmental assessment is needed. Consequently, for proper environmental analysis, a comparison Life Cycle Assessment (LCA) was conducted using SimaPro software for cradle to grave boundary. The results indicate that modern clay roof tiles have lesser impact than the chrysotile asbestos sheets in Sri Lanka. A significant burden identified from the extraction phase and the disposal phase of asbestos sheets.

Key Words: Life Cycle Analysis, Modern clay roof tiles manufacturing, Asbestos sheets manufacturing.

Life Cycle Thinking Frameworks Applied to Engineered Wood Products – Identifying A Need for Social Life Cycle Assessment

Thursday, 20th July - 09:00: Buildings

Ms. Shannon Preddy [1], ***Dr. Olubukola Tokede*** [1], ***Prof. Jane Matthews*** [1]

1. *Deakin University, School of Architecture and Built Environment*

An integral stakeholder in combatting the global environmental impacts of climate change is the Building and Construction industry. However, increased adoption of renewable building materials and the potential social and socio-economic impacts needs to be understood to better assist decision makers during the material selection process. The purpose of the study is to investigate the application of life cycle thinking frameworks to Engineered Wood Products (EWP's) used in Mass Timber Construction (MTC) and identify any existing barriers affecting its uptake in the Australian building and construction industry. Additionally, the study highlights a gap of the lack of social consideration given to EWP's particularly Cross Laminated Timber (CLT) especially from a life cycle perspective. A systematic literature review was conducted in accordance with the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRSIMA) 2020 process, focusing on studies which applied life cycle frameworks to EWP's published up until February 2023. The results revealed a Life Cycle Assessment (LCA) or Life Cycle Energy Analysis (LCEA) were predominantly applied across a cradle-to-grave life cycle phase. Moreover, despite the use of databases, case study buildings or modelling, primary data collection proved to be a challenge when evaluating EWP's. This challenge is further compounded when attempting to evaluate the social and socio-economic impacts of this material. Although some studies combined all three life cycle methodologies to conduct a Life Cycle Sustainability Assessment (LCSA), illustrating a progression towards assessing the social impacts of EWP's, the results from this study highlight a need for further empirical evidence to comprehensively understand its impacts.

Buildings Extended Abstracts

Life cycle assessment considerations of prefabricated construction

Leela Kempton and Matthew Daly
Sustainable Buildings Research Centre, University of Wollongong

Prefabricated construction has been promoted as a potential method to improve sustainability in the construction industry, particularly through increasing production efficiency and reducing waste. However, the evidence base for substantiating these claims, particularly in the Australian context, is underrepresented. Life cycle assessment (LCA)provides a holistic approach to evaluating some of the sustainability benefits of prefabricated construction and quantifying these impacts in the context of the whole building life cycle. This paper explores how LCA can be used to assess and compare the sustainability impacts of prefabricated construction, conducted as part of the Australian Manufacturing Growth Centre's 'Prefabrication Innovation Hub'. A case study consisting of a prefabricated two-bedroom holiday cabin in Australia was used. Waste audits undertaken in the factory during construction of the cabin were combined with material data collected to conduct a LCA focused on the manufacture of the cabin, including waste material end of life considerations. Although challenges were faced in obtaining the data required for the LCA and determining appropriate materials to use from available databases, LCA has been shown to be a useful tool in evaluating the embodied carbon and embodied energy within the case study building. From this basis, scenarios investigating the impact of material wastage rate and recycling options have been tested, demonstrating that current levels of waste produced are responsible for 6.4% of the embodied carbon in the case study building. Halving the level of waste generated just in the foundations of the building through leaner construction practices and increases material efficiency could reduce the embodied carbon in the foundations by 2.7%. This provides a framework to considering sustainability of prefab construction, however further work is needed to identify how other aspects such as future adaptability and reuse of the buildings could be considered in this framework.

Main topic/key words: Prefabrication, offsite construction, C&D waste, transportation

1. Introduction

The construction industry is a major consumer of resources, estimated to be responsible for almost 50% of the worldwide resource consumption, and projected to double over the next 50 years (OECD, 2019). Alongside this consumption, the construction industry is also one of the major contributors to greenhouse gas emissions. Buildings were estimated to contribute to 37% (11.7 gigatons) of global energy-related CO_2 emissions, through both the construction and operation of these buildings (United Nations Environment Programme, 2021). Although operational emissions currently dominate, it has been estimated that improving energy efficiency of buildings and decarbonizing the grid will result in embodied carbon emissions accounting for up to 85% of total emissions by 2050 (GBCA and Thinkstep-anz, 2021).

Prefabrication, otherwise known as off-site manufacture or industrialised building, has been proposed as a construction method offering potential sustainability benefits, particularly in the areas of improving materials efficiency and waste management (Kamali and Hewage, 2016; Attouri, Lafhaj, Ducoulombier, *et al.*, 2022; López-Guerrero, Vera and Carpio, 2022). Uptake of prefabrication in Australia has been limited, with the sector thought to account for less than 5% of the industry (AMGC, 2019), and largely dominated by offsite fabrication of roof trusses, wall frames and pre-case concrete elements (SBEnrc, 2015). However, in other parts of the world prefabrication has seen increased uptake and dominance (Steinhardt and Manley, 2016). The sustainability of prefabricated construction has not yet been fully explored in the context of Australia, however given the need for increased sustainability performance of buildings, capitalising on this could result in dual benefit.

Life cycle assessments (LCA), in the context of prefab construction, are a means of evaluating the performance of prefabricated construction over the various phases of a building's life. LCA provides a strong framework to understand and quantify potential sustainability benefits or potential negative impacts of prefabricated construction. Whilst there have been numerous studies and reviews on LCA of prefabricated construction (Chen, Zhou, Feng, *et al.*, 2022; López-Guerrero, Vera and Carpio, 2022), these studies often focus only on actual results, and do not adequately explore how the sustainability benefits of prefabricated construction can be quantified through LCA.

The aim of this paper is to identify challenges and opportunities of using LCA to assess the sustainability of prefabricated construction. This is achieved through two sections. Firstly, a comprehensive literature review of studies considering LCA of prefabricated construction, particularly in the Australian context is conducted. Secondly, a case study on a prefabricated building is used to explore how LCA can be used to evaluate the sustainability benefits, challenges and trade-offs of prefabricated construction.

2. Review of LCA of prefabricated construction

There have been numerous reviews of past research into life cycle environmental impacts of prefabricated construction, as shown in Table 1 which highlights the key findings from the four major reviews. Whilst there have been favourable outcomes from these reviews, with prefabrication shown to have improved performance across key environmental LCA indicators (particularly in results reviewed by López-Guerrero, et al. (2022)), there is significant variation in the results that has been recognised by the reviews. It has been noted that some influencing factors of embodied carbon quantification (namely building structure forms, level of prefabrication, data sources) have a major impact on the results and studies with variations in these factors should not be used for comparative or verification purposes (Chen, Zhou, Feng, *et al.*, 2022). Some of the limitations highlighted by the studies that restrict the application of the results include comparing buildings of different material types, including inconsistent scope items, and only focusing on one aspect of the study, e.g. the construction phase (Kamali and Hewage, 2016).

Table 1: Summary of key features and findings of past LCA reviews for prefabricated construction

Ref.	Case studies/ Location	Prefabrication rate	Dataset	Findings
López-Guerrero, *et al.* (2022)	67 papers, covering 86 case studies Location mostly China (24), Malaysia (11), Australia (4), Others (28)	Mostly high (>50%) – 56% of cases. Not defined in 67% of cases. Conflicting results between prefab rate and embodied carbon	Varied Some use of national databases, and imported databases	Majority of case studies (43 of 53) found better performance from prefabrication in terms of non-operational carbon. Average 14.85% percent reduction (of positive cases) Majority of cases (16 of 22) found a reduction in embodied energy from prefabrication. Average reduction of 8.6% for medium prefab rate, and 10.1% for high prefab rate.
Chen, *et al.* (2022)	43 papers, covering 96 case studies. Location mostly China (40%), plus 15 other countries	Prefabrication rate was note defined for half the cases.	Ecoinvent database was most common (56 cases), followed by Inventory of Carbon & Energy (ICE)	Embodied carbon of prefabricated construction varied from 26.6 to 1644.4 $kgCO_2e/m^2$. Carbon reduction measures of prefab found to include increasing productivity, supply chain design, lean manufacture techniques, alternative energy sources, adopting the use of low embodied carbon, local, reused or recycled materials.
Teng *et al.* (2018)	23 papers, covering 27 case studies. Mostly UK (6), China/Hong Kong (4), Italy (4).	Prefabrication rate was low for most cases (46%)	Not analaysed	On average, reductions in embodied (15.6%) and operational (3.2%) carbon in Prefab buildings. Significant variation - some cases of greater carbon consumption, and large range of results - embodied from 105 to 864 kg CO_2/m^2, and operational from 11 to 76 kg CO_2/m^2/ yr, respectively.
Kamali & Hewage (2016)	8 case studies considered, location mostly US (4), UK (2), also Austrlaia (1) and Canada (1)	Not discussed	SimaPro most often used.	Review found that no comprehensive study was available to compare prefabrication with conventional construction, and most studies has a narrow, ill-defined focus. No studies included refurbishment/replacement and most excluded end of life phase.

In the Australian context there have been a limited number of studies investigating LCA of prefabrication construction, using a variety of platforms and datasets, with a summary of these studies provided in Table 2. Similar to the large scale reviews, results from these studies have showed conflicting assessments of the sustainability of prefabricated construction, with some studies finding a favourable comparison (Minunno, O'Grady, Morrison, *et al.*, 2020a; Andersen, Sohn, Oldfield, *et al.*, 2022) and others finding that prefabricated construction was similar or worse than conventional construction methods (Aye, Ngo, Crawford, *et al.*, 2012; Ghafoor and Crawford, 2020).

There is difficulty with some comparisons where dissimilar construction materials or sizes are considered (Andersen, Sohn, Oldfield, *et al.*, 2022), or where no comparison with conventional construction is provided

(Mehrvarz, Barati and Shen, 2021). Studies which focused on the full life cycle impacts found results were dominated by operational energy consumption, whereas those focused on the construction only found that the choice of construction materials had a significant influence on the results.

Table 2: Summary of LCA studies within Australia considering prefabrication

Ref.	Location	Case study details	Dataset	LCA Scope and methodology	Findings
Anderson et al. (2022)	Australia and Denmark	Conventional and modular design of a single family home	OpenLCA, using Ecoinvent v3	Full life cycle, including waste, operational energy, land use and transportation	Results found modular housing perfomed better than conventioanl across all sustainability categories, but all exceeded the estimated environmental limits imposed.
Mehrvarz (2021)	Sydney, Australia	Modular residential building with floor area of 493m^2	BIM based LCA using the Inventory of Carbon & Energy (ICE) dataset	Only product (A1-A3) and construction (A4-A5) considered.	Embodied energy was mostly attributed to materials (87%) followed by transportation to factory (5%), transportation to site (4%) and energy for consutrction on site (2.2% and offsite 2.0%).
Ghafoor and Crawford (2020)	Melbourne, Australia	Detached dwelling, external walls of: cross-laimnated timber (CLT), structural insulated panels (SIP), prefabricated timber frame panels or original brick veneer	Path exchange hybrid technique used to produce embodied flow coefficients for materials.	Considers product stage (A1-A3), transportation of materials, components and equipment to construction site and construction/ installaion stage (A4-A5).	The timber framed panel was only option to show a significant reduction (7.4 tCO_{2eq} or 7% compared to base), while SIPs increased the embodied GHG emissions by around 7.13 tCO_{2eq} or 6% compared to base and CLT resulted in an increase of 1 tCO_{2eq} or around 1% compared to base.
Minunno (2020b)	Perth, Australia	Modular building designed for disassembly and resuse	SimaPro 9.0 with Ecoinvent 3.5 database	Full life cycle, but did not consider operational energy or transportation of materials, modules or waste	Modular building designed for reuse results in an overall impact of 5.4t CO_{2eq} vs 44.5t for the linear design. Saving is largely from the use of recycled components, as well as disassembly and reuse potential.
Aye *et al.*, (2012)	Melbourne, Australia	Compared prefabricated steel-frame and timber frame modular, with conventional concrete	SimaPro dataset (Australian version) & TRNSYS	Full life cycle Embodied and operational energy over 50 year lifespan.	Absolute embodied energy of steel prefab (14.4 GJ/m^2) was 50% more than the conventional concrete (9.6 GJ/m^2), and timber prefab 9% more (10.5 GJ/m^2).

3. Methodology

3.1. Goal and Scope of LCA study

The goal of the case study LCA undertaken in this project was to determine the impacts of various aspects of prefabricated construction that can have a positive or negative impact on the life cycle of the construction. As the focus of the study is on the construction methods only, the scope of the LCA includes the product - modules (A1-A3), as well as transportation to site (A4). In the case considered here of a volumetric prefabricated building, the on-site construction impacts (A5) are minimal and insufficient data was available to include this in the case study. Operational energy, maintenance and end of life are not considered in this study. Whilst these impacts will be significant in the life cycle of the building, the study is focused on improvements to the production stages (A1-A4) and the impact that prefabrication can have in this space. The LCA was undertaken using LCA for Experts (formerly GaBi) version 10.7.

When considered LCA of prefabricated construction, it differs from conventional construction in that the major labour and assembly of the buildings is considered as part of the product stage (A3) rather than the on-site construction stage (A5) (Kamali and Hewage, 2016).

3.2 Case study description

The case study investigated here has been drawn from a larger project investigating sustainability of prefabricated construction from a variety of angles. To investigate the challenges and opportunities of using LCA to assess environmental sustainability of prefabricated construction, a single case study has been used for simplicity. The case study building is a two bedroom holiday cabin, constructed as a volumetric prefabricated building built entirely offsite in a factory in NSW, Australia. The cabin has a total floor area of $84m^2$, constructed on a structural steel base. Details of the materials of construction are provided in Table 3. A detailed material and waste assessment of the case study building was undertaken to generate detailed information regarding the quantities, types and sources of materials used in the construction of the cabin.

Table 3: Case study construction details

Building component	Construction materials
Foundation structure	Structural steel
Flooring	Beige-tongued particleboard, covered with vinyl in living areas, tiles in wet areas and carpet in bedrooms
Walls	Lightweight steel frame, Weathertex timber external cladding, internal plasterboard lined, R2.0 glasswool insulation
Roof/ceiling	Plasterboard ceiling, Colorbond roofing with anticon blanket
External	Outdoor decking constructed of composite wood flooring inclusion of aluminium gutter guard on gutters for bushfire protection

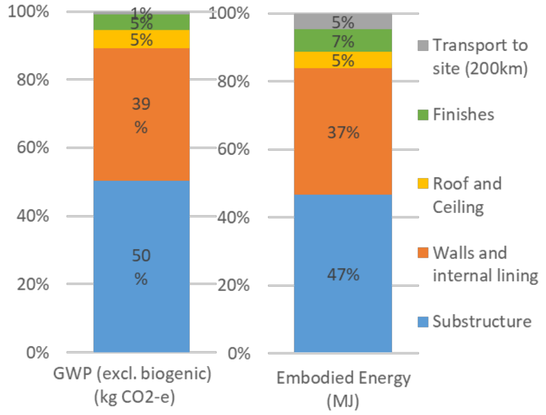

Figure 1: Contribution of major areas to GWP and Embodied energy in prefab case study building

4. Results and Discussion

At this point in the project, only preliminary results have been undertaken within the LCA model to develop the model and investigate the influence of various parameters. The LCA model with the major components has been established highlighting a split in contribution towards embodied carbon for each of the main areas as shown in Figure 1. Further analysis within the LCA model will enable the exploration of other scenarios of waste generation or recycling of wastes generated, as well as investigating the impact of transportation of the finished building on the overall life cycle impact.

One of the main advantages of prefabricated construction is the ability to measure and control waste generation compared to conventional construction. As part of this study, detailed waste audits were undertaken to measure the actual wastage rates across the project. This was combined with assumed details based on the materials ordered compared to the quantities required for the project to estimate average wastage values for materials used. Considering only the foundations of the cabin, as these contribute 29% of the embodied CO_2, actual material wastage within the LCA was responsible for 6.4% of the total embodied CO_2 for the foundations of case study cabin. Estimated construction waste rates on site are thought to be anywhere from 5-30% (Osmani, 2011). When a flat rate of 30% wastage was considered, the embodied CO_2 of the case study increased by 11.8%, highlighting the impact of considering actual wastage. Although prefabricated construction is associated with greater control of waste, the case study investigated still has areas of opportunity to increase the waste efficiency of the construction, with an average wasteage rate across the foundations of 11.6%. If this wastage rate could be reduced by half to 5% then this would result in a reduction of embodied CO_{2eq} of 2.73%. These values highlight the significance of an often overlooked aspect of construction.

5. Conclusion

Prefabrication construction offers advantages over conventional construction methods, particularly from a sustainability perspective, however there is a balance between perceived benefits and potential unintended consequences. Previous studies have attempted to identify some of these benefits, with LCA used to quantify the environmental impacts. However, these LCA studies have been limited by a lack of consistent methodological approach. Comparisons between dissimilar buildings, or inconsistent output values makes it difficult to quantitatively assess the benefits of prefabricated construction. In the Australian context few LCA studies have been undertaken considering prefabricated components or constructions to date, often with a specific focus or purpose in mind. These studies have not fully explored the impact of prefabricated construction from a sustainability perspective.

Through the use of a single case study, LCA has demonstrated usefulness in quantifying the impact of some of the potential sustainability benefits of prefabricated construction. Through the consideration of actual waste measured within the factory and structuring scenarios to consider alternate waste paths LCA can demonstrate the impact that these lesser considered aspects of construction can have on the overall embodied carbon within the building.

Acknowledgements

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Whole-life Baseline Carbon Assessment of Residential Building Stock – A Victorian Case Study

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Keywords: Life cycle assessment, Net Zero Pathways, Residential, Embodied Carbon

1. Introduction

The building and construction sector comprises around one-third of greenhouse gas (GHG) emissions. Globally it is set to rise with the trend of rapid urbanisation and increasing population growth. Most building carbon impact data as well as their corresponding policy strategies only incorporate the operational phase, however the need to incorporate the entire building life cycle has been growing and is emphasised through efforts such as the International Energy Agency's Energy and Buildings and Communities Programme (IEA EBC) Annex 72. Embodied energy and carbon come from processes outside the operational phase such as production of building materials, construction, renovation, until demolition. This contributes 20% to 50% in overall carbon emissions; increasing in share as more buildings adhere to energy efficient design regulations. Embodied carbon would become the main metric for building emission reduction over embodied energy as renewables increase within the energy supply mix. Though models that integrate embodied energy and carbon in buildings have been growing, further development in model granularity should be done to account for scalability across jurisdictions as well as the integration of renovation in housing stock analysis. Within the Australian context, the residential building sector makes up more than half of total energy use in the building and construction sector in 2018 and comprises more of the overall building carbon impact compared to non-residential buildings. Currently the Australian residential sector mainly relies on minimum energy performance standards for new buildings and voluntary energy efficiency schemes for existing buildings, however both only focus on operational energy.

A bottom-up typology framework was developed to estimate both operational and embodied carbon emissions for existing and new housing up to 2050, scalable from local government area (LGA) to state-level jurisdiction which allows for granularity in analysing global carbon targets. This would allow a more granular analysis of residential building emissions for both operational and embodied life cycle stages, which would lead to more localised and implementable policies. The state of Victoria was used as a case study, from which the housing typologies and local government area (LGA) data would be based on.

2. Material and methods

2.1. Setup of Housing Typologies

In developing the baseline projection for residential building GHG emissions, housing typologies were set for existing and new building stock. Attributes such as the Nationwide House Energy Rating System (NatHERS) rating, housing type, and construction wall type were used to create unique representative typologies. The NatHERS rating was used to account for the operational attributes through its overall effect on energy demand, particularly in heating and cooling loads. Although operational energy demand decreases with a higher NatHERS star rating, constructed floor area has grown over time for all housing types, which can have an impact in GHG emissions particularly for the embodied stages.

Housing typology attributes such as housing type (detached, semi-detached, apartment units) and construction wall type (brick veneer, timber, concrete) for the external wall were included in developing the housing typologies. For housing type, detached houses made up most of the existing housing stock across most LGAs with around 80%, except for some inner city LGAs such as Melbourne and Port Phillip where the increase in apartment units can be seen within the residential building stock. For construction wall type, brick veneer and timber comprised most of the external wall building material used for detached and semi-detached houses with around 80% of housing stock, while concrete was used for external walls of apartment units. It is essential to account for the change in housing typologies for new housing stock given that there are upcoming mandatory policies such as an increase in minimum energy star rating for new homes.

2.2. Development of Model Framework

The set housing typologies corresponded to the following input shown in Figure 1. From this, the input was utilized by the embodied and operating carbon modules within the model framework. The output from the modules was integrated with other parameters such as constructed housing floor area and construction/demolitions rates across LGAs in Victoria to develop a residential building stock model. For the existing residential building stock, only the operational phase was considered in its carbon assessment since embodied carbon has already been incurred from when the houses were constructed.

Floor plans considering the average floor area of each typology alongside their corresponding building materials and assemblies was needed, particularly for the following building elements: exterior wall, windows, ceiling, and insulation since they influence direct energy use in houses. In line with this, only heating and cooling energy demand was considered in the study since aside from being most of the energy use for residential buildings within Australia, only heating and cooling energy consumption is influenced by both locational and jurisdictional differences such as climate and constructed floor area within an LGA. The type of construction materials used would greatly impact residential heating and cooling load compared to other end use appliances which stays constant regardless of building envelope.

To account for operational carbon, the Australian Zero Emission House (AusZEH) software tool was used since this would best reflect residential building energy consumption in Australia, given its pre-set end-use equipment and occupancy profiles. The tool has been tested and validated alongside an actual zero emissions house in Australia and a deviation of about 6.5% was observed, thus making AusZEH a good model for estimating residential energy consumption. All typologies were individually modelled with output of operational GHG emissions for each, which were then expanded to housing stock level for LGA and state level. To account for embodied carbon, the Environmental Performance in Construction (EPiC) database was used as it reflects the environmental performance of construction materials within Australia from production. Material quantities were obtained from the AusZEH Building Data sheet for each housing typology while the assemblies used were from the City of Melbourne database. Building elements such as external walls, windows, ceiling, roof, and floors were accounted for as they would impact building thermal performance. For renovation in particular, the embodied carbon from only the windows, wall insulation, and ceiling insulation was considered since these are the main building elements that when changed would influence building heating and cooling demand. In this study, embodied carbon considered material production and transportation as well as building construction.

The output of the operating carbon module was expressed in energy intensity (MJ/sqm) while for embodied carbon, this was in terms of carbon-dioxide equivalent (kgCO_2e) or global warming potential (GWP). The rationale for the initial mismatch in units would be that the fuel type used for heating across LGAs in Victoria would be either gas or electricity, thus the carbon emission factors would differ depending on the energy source type which would be reflected once the energy intensities per typology are integrated in the residential building stock model. Aside from this, LGA specific data such as climate zones and construction rates were considered in developing a housing stock level analysis for Victoria.

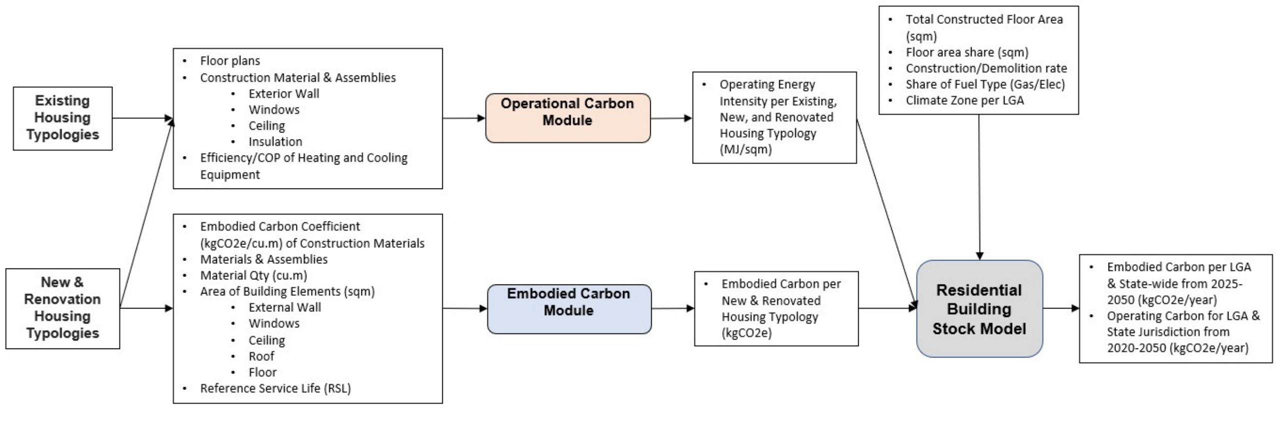

Figure 1. Model Framework for Whole-life Carbon Assessment

3. Results

3.1. Estimating Operational Carbon

Figure 2 illustrates the operational energy demand for the residential building stock in Victoria which considered the average energy intensity for houses across various NatHERS star ratings. Detached houses comprised most of the housing for existing residential buildings from which heating would have a higher energy demand compared to cooling. Timber construction wall type houses require more heating load compared to brick veneer homes for both detached and semi-detached housing types. For new housing stock, timber walled homes would have less or equal energy demand compared to brick veneer houses. Heating would still be the main source of energy demand across all typologies.

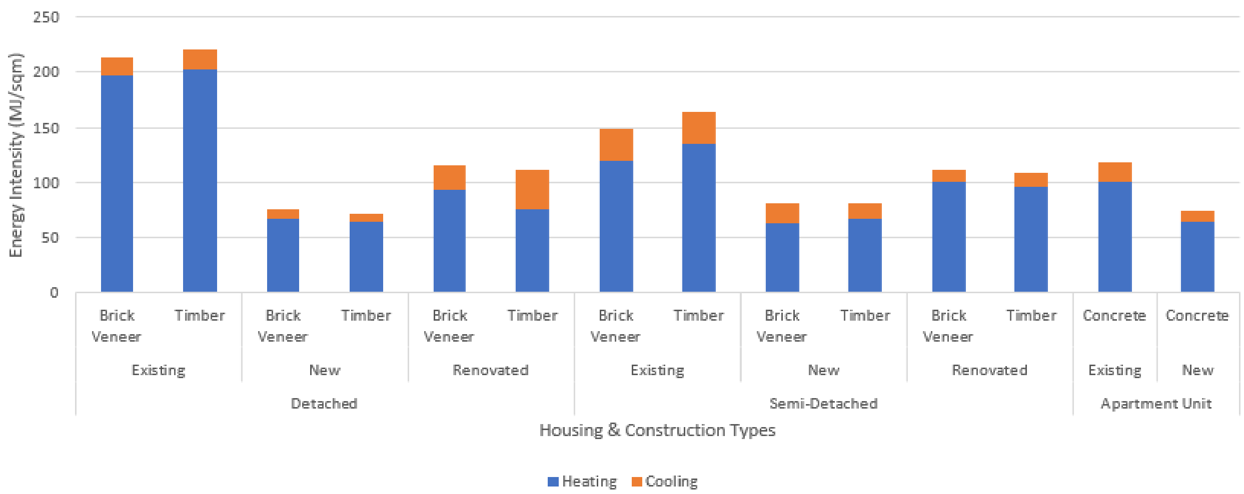

Figure 2. Housing typologies annual operational energy demand for residential building stock

Another aspect to consider would be the impact of major renovation in energy demand and how this would compare to newly constructed residential buildings. The set typologies for renovated housing in Figure 2 result from the existing housing typologies considering renovation changes done as shown in Table 1. It is assumed that renovation efforts would improve the existing housing typology up to a 6-star NatHERS rating, which is the current minimum standard for new housing built at present. For this study, a reference service life (RSL) of 40 years was used for residential buildings, and given that the baseline year used is 2020, only houses built prior to 2006 were considered for renovation.

Table 1. Existing Housing & Construction Types with corresponding renovation changes

Housing & Construction Types	Construction Year	Renovation Changes
Detached (Brick Veneer)	Pre-1991	Insulation (Ceiling): R3.0 Insulation (Wall): R3.0 Windows: Clear Double Glazed
Detached (Timber)	Pre-1991	Insulation (Ceiling): R3.0 Insulation (Wall): R2.0 Windows: Clear Double Glazed
Detached (Brick Veneer)	1992-2006	Insulation (Ceiling): R2.0 Insulation (Wall): R0.14 Windows: Clear Double Glazed
Detached (Timber)	1992-2006	Insulation (Ceiling): R4.0 Insulation (Wall): R3.0 Windows: Clear Double Glazed
Semi-Detached (Brick Veneer)	1992-2006	Insulation (Ceiling): R1.5 Insulation (Wall): R1.0 Windows: Clear Double Glazed
Semi-Detached (Timber)	1992-2006	Insulation (Ceiling): R1.5 Insulation (Wall): R1.5 Windows: Clear Double Glazed

It is observed that a higher R-value insulation is needed for brick veneer detached houses built prior to 1991 while the converse applies for houses with minimum insulation standards already set (1992 onwards). Semi-detached houses have lower R-values in all building element insulation, likely due to their smaller constructed floor area. It is assumed that the floor area for existing houses prior and after renovation would remain the same. Though operational energy demand reduction is higher in renovating older buildings, the energy use intensity for all renovation typologies is of similar values, though varying in end-use breakdown. The share of cooling energy demand has also increased in renovated houses likely due to the additional insulation along with the cooler climate zones within Victoria.

Expanding this into state level, Figure 3 highlights the operational GHG emissions for the Victorian housing stock from 2020 to 2050 (with year 2015 for validation), considering an emission factor of 0.92 $kgCO_2e/kWh$ for electricity and 0.05553 $kgCO_2e/MJ$ for gas. One of the key policies set to reduce energy consumption in the residential building sector is to increase the NatHERS minimum standard from 6-star to 7-star by 2022, thus this was considered in the baseline stock projection. Existing detached housing would incur most of the GHG emissions for the housing stock of Victoria, however the increase in construction rate and the larger floor areas for newer dwellings would contribute to operating emissions, and subsequently embodied carbon, in newer building stock.

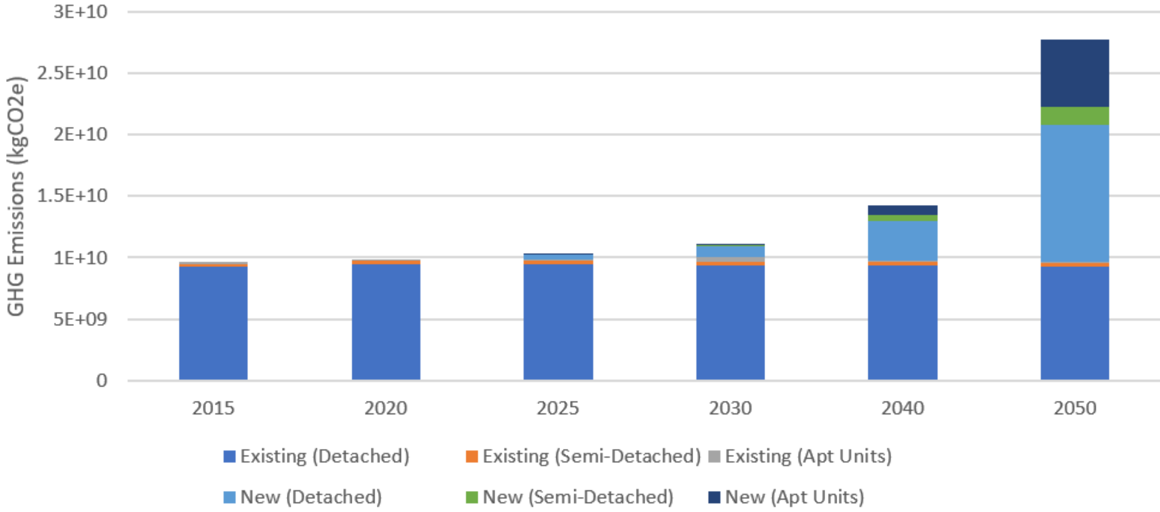

Figure 3. Operational carbon emissions for Victorian residential building stock (2015-2050)

3.2. Estimating Embodied Carbon

The breakdown of embodied carbon for each housing typology was illustrated in Figure 4, which would compare emissions from both new and renovated housing typologies. For all new housing typologies, most embodied carbon would stem from the floors, likely due to the concrete slab material used. Houses that use brick veneer for external walls comprises a significant share of embodied emissions. A smaller floor area would result in less embodied carbon, except for apartment units which yields a similar value with a brick veneer semi-detached house and a timber detached house despite its smaller average floor area, likely due to the concrete material which comprises most of an apartment unit. The emissions from the transportation stage would be based on the amount and weight of the construction materials needed for new construction and renovation. More embodied carbon is incurred in renovating 3-star detached homes compared to other renovated housing typologies due to having a larger constructed floor area. For renovated houses, the embodied carbon incurred from both brick veneer and timber houses are similar, however in building new homes, using timber for external walls would reduce embodied carbon by roughly 30%.

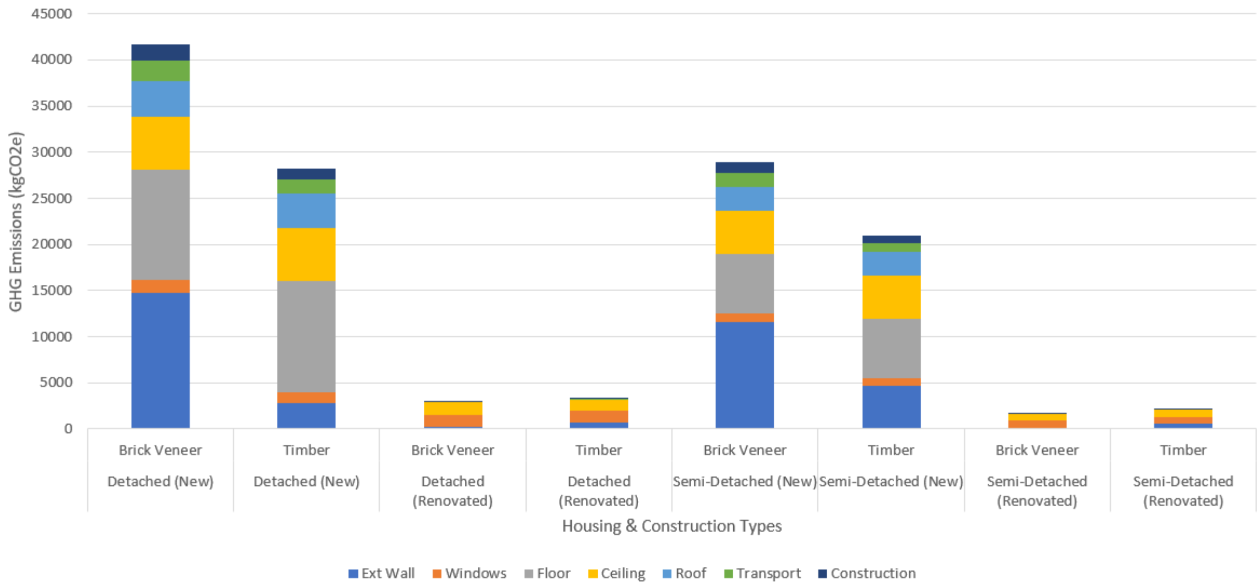

Figure 4. Housing typologies annual embodied GHG emissions from new construction and renovation

4. Discussion

4.1. Comparing Baseline Estimation

For this study, 2020 was used as the baseline year to set projected GHG emissions up to 2030 and 2050. To set the estimated construction rates per LGA, the growth between 2015 and 2020 was used and an average of 16% was observed across LGAs in Victoria with detached and semi-detached housing reaching 25%. Though most of the construction would stem from detached and semi-detached houses, some LGAs would reach 30% growth for apartment units such as in Melbourne and Port Phillip. According to the Housing Industry Association (HIA), the rising trend of new construction in detached housing can be observed in Victoria with roughly a 5% increase in commencements within a year. Considering the per LGA and state-wide average construction rate, both values have a less than 10% difference and highlight the significant impact from new construction if this trend continues.

A study conducted by the Commonwealth Scientific and Industrial Research Organisation (CSIRO) monitored the electricity consumption of sixty-nine (69) detached houses constructed between 2000-2010 in Melbourne. Houses below a NatHERS rating of 4.5 in Victoria do not use electricity for heating, mainly relying on gas, thus residential buildings constructed prior to 2006 were assumed to use only gas for heating and provided a separate air conditioning system for cooling. The study illustrated how increasing the NatHERS star rating minimum standards was able to reduce energy needed in building thermal performance, however the share of cooling load increased in higher-rated houses, which was shown in GHG emissions during the summer months.

A recent study that quantified the life cycle energy demand of the Victorian residential building stock highlighted that operational energy comprises more of the life cycle energy use, and heating would require the most demand within operational energy. With this, operating energy would have the more significant contribution to GHG emissions in housing, given that the rate of renewables integration in the energy supply grid remains constant. For embodied energy, concrete would account for the largest share, and this can also be seen from the materials and assemblies used in floors for new construction.

Key variables such as climate, building type, end-use fuel mix, and life cycle inventory calculation method were considered in a study that aims to develop whole-life net-zero pathways for the Australian built environment. In terms of energy efficiency in operational carbon, most of the GHG emission reduction potential was found to be in heating and cooling, with significant savings found in building envelope improvements such as glazing. For embodied carbon, strategies such as reducing impact of construction materials in the building process were proposed.

The embodied carbon component of the residential building stock model was tested with only process- based life cycle analysis (LCA) data from SimaPro simulations derived from the EPiC database (Figure 5). Comparing this with Figure 4, which uses a hybrid of process-based and I/O data, there is a reduction of GHG emissions in the external walls for both brick veneer and timber construction as well as for the win- dows, particularly in renovated housing. In terms of overall embodied carbon per housing typology, the new detached brick veneer housing would still yield the most emissions.

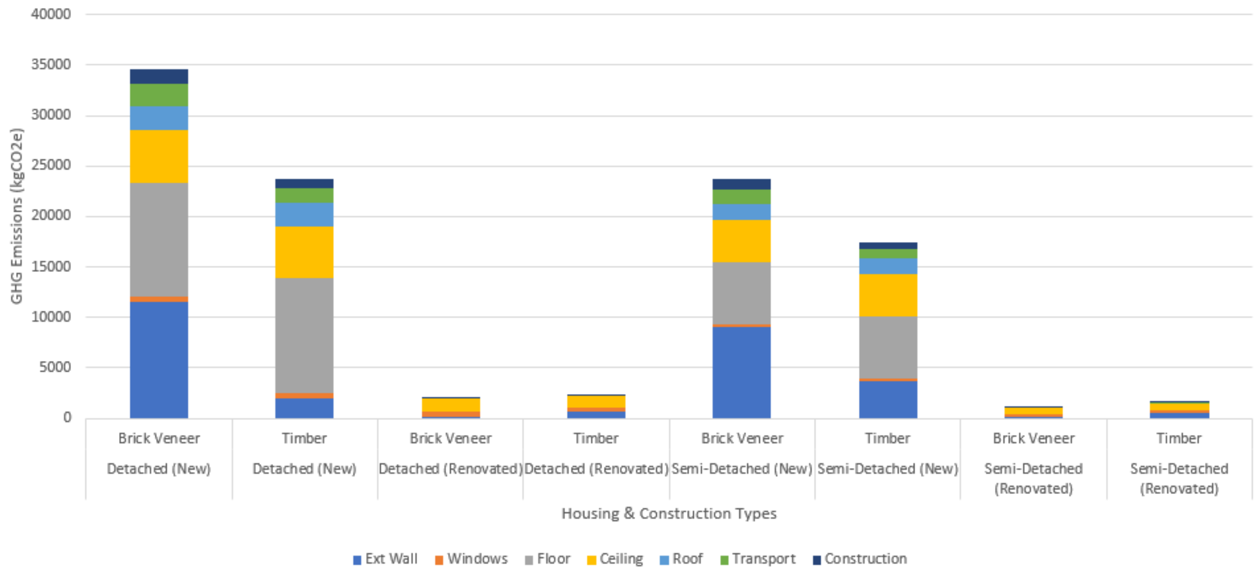

Figure 5. Housing typologies annual embodied GHG emissions from new construction and renovation – process-based LCA data

4.2 Limitations and Future Research

The operational carbon results from the study were derived from individual runs of the AusZEH software and though there is a detailed breakdown of building energy demand per housing typology, translating this to stock level results was based on a multiplicative factor and future studies can utilise batch processing to better capture the aggregated effects of building stock in residential building GHG emissions.

Future research can investigate various scenarios such as a quicker integration of renewables in the energy supply and how this would impact operational emissions. For embodied carbon, new detached brick veneer housing would incur the most emissions likely due to its larger constructed floor area and use of materials with high embodied emission coefficients such as concrete and brick. Analysing the LGA level breakdown of whole-life carbon, policies that focus on renovation for detached housing and the subsequent reduction of new construction and corresponding embodied emissions can be set for more LGAs outside Greater Melbourne. Likely other LGAs can explore a shift in building material use such as from brick veneer to timber, however the high new construction rates may offset significant reductions from material changes.

One of the key elements needed to better understand residential building decarbonisation would focus on how to include occupancy patterns at a more granular perspective. Future studies can use agent-based modelling (ABM) techniques and expand the housing typology method to capture household decisions and behaviour more realistically, leading to more effective decarbonisation policies.

5. Conclusion

The integration of both embodied and operational life cycle stages is essential in conducting a whole-life carbon assessment to develop more effective decarbonisation policies. For the residential sector, this would require a bottom-up approach due to its variability in energy demand estimation compared to non-residential buildings. Housing typologies were developed to estimate residential GHG emissions from both operational and embodied carbon more granularly. At an individual level, heating contributed most of the operational carbon for most typologies while the brick veneer external walls and concrete floor slabs incurred a significant share of embodied emissions for new housing stock. In renovation, there is a reduction for both operational and embodied carbon and so this should be considered as a key strategy in residential sector decarbonisation. Expanding this at building stock level, detached houses built prior to 1991 comprised most of the operational carbon in Victoria, however with the high construction rates set for most LGAs, new housing may contribute more GHG emissions in the future.

Expanding the typological analysis to building stock at state level, existing detached housing would initially comprise most of the operational GHG emissions however with the high construction rates of local government units (LGAs) within Victoria, operational and embodied emissions may stem more from newly built stock if current trends continue. Given this, there is a need to further analyse policies that aim to reduce the rate of new construction and focus of refurbishment of existing buildings. Considering both embodied and operational carbon, embodied emissions would contribute more as building stock is projected to 2050, especially given the effects of a high construction rate, which highlights the need to integrate embodied carbon in developing building decarbonisation policies. Future work can compare the developed baseline pro- jection with set carbon emission targets for 2050 to provide guidance for policy makers to accelerate decar- bonisation practices for the residential building sector.

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Life Cycle Thinking Frameworks Applied to Engineered Wood Products – Identifying A Need for Social Life Cycle Assessment

Preddy, S. Tokede, O. Matthews, J.

Main topic/key words: Engineered Wood Products, Life Cycle Frameworks, Social LCA

Abstract

An integral stakeholder in combatting the global environmental impacts of climate change is the Building and Construction industry. However, increased adoption of renewable building materials and the potential social and socio-economic impacts needs to be understood to better assist decision makers during the material selection process. The purpose of the study is to investigate the application of life cycle thinking frameworks to Engineered Wood Products (EWP's) used in Mass Timber Construction (MTC) and identify any existing barriers affecting its uptake in the Australian building and construction industry. Additionally, the study highlights a gap of the lack of social consideration given to EWP's particularly Cross Laminated Timber (CLT) especially from a life cycle perspective. A systematic literature review was conducted in accordance with the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRSIMA) 2020 process, focusing on studies which applied life cycle frameworks to EWP's published up until February 2023. The results revealed a Life Cycle Assessment (LCA) or Life Cycle Energy Analysis (LCEA) were predominantly applied across a cradle-to-grave life cycle phase. Moreover, despite the use of databases, case study buildings or modelling, primary data collection proved to be a challenge when evaluating EWP's. This challenge is further compounded when attempting to evaluate the social and socio-economic impacts of this material. Although some studies combined all three life cycle methodologies to conduct a Life Cycle Sustainability Assessment (LCSA), illustrating a progression towards assessing the social impacts of EWP's, the results from this study highlight a need for further empirical evidence to comprehensively understand its impacts.

1. Introduction

The Building and Construction industry is a key stakeholder in combating the global environmental impacts of climate change. The international initiative to reduce environmental degradation has led to the development of 'Green Buildings' (Jayalath et al., 2020) - buildings which have been designed to reduce environmental impacts, optimise occupant health conditions, generate a profit for the developer and the local community and consider the life cycle during the design and development process (Zuo and Zhao, 2014). An element of green buildings is the sustainable consumption of raw materials and the use of more environmentally friendly products. As a result, timber, more specifically Engineered Wood Products (EWP's) have emerged as a vital building material due to the reduced embodied energy output, high strength-to-weight ratio and carbon sequestration capabilities (Jayalath et al., 2020). EWP's such as Glue Laminated Timber (Glulam), Laminated Veneer Lumber (LVL) and Cross Laminated Timber (CLT) are commonly used in a construction method known as Mass Timber Construction (MTC)[1] (Kremer and Symmons, 2018).

Assessing a building's sustainability performance extends beyond operational outputs and occupant comfort levels (Zuo et al., 2017). Life Cycle Thinking addresses this void through the consideration of a product/services consequences to the environment, economy and society throughout its life cycle (Farjana et al., 2021). Due to its comprehensive, objective, and systematic framework, a Life Cycle Assessment (LCA) and the accompanying methodologies (i.e. economic LCA and social LCA) are regarded as an internationally accepted framework for evaluating the potential and actual impacts of products along their life cycle (Andrews et al., 2009; Pierobon et al., 2019).

[1] MTC does not mean the building has been entirely constructed from engineered timber, but that a significant portion of the structure is timber (Evison et al., 2018).

To date, impact assessments within the building and construction industry, have emphasised environmental and economic sustainability through the execution of Environmental Life Cycle Assessments (E-LCA) and Economic Life Cycle assessments (E-LCA) assessments or Life Cycle Costing (LCC) (Hafner and Schäfer, 2017; Lu et al., 2017; Pierobon et al., 2019). While there is no doubt how significant environmental sustainability is to economic prosperity, the social sustainability impacts cannot be ignored. Therefore, this study aims is to answer the question *"what level of consideration has been given to the social pillar in life cycle assessments of EWP's?"* through a systematic literature review. Furthermore, this work will contribute to an increased consideration of the social sustainability of EWP's.

Impact assessment in Social Life Cycle Assessment (S-LCA) is highly heterogenous requiring diverse approaches and multiple data sets across the life cycle (Arcese et al., 2017). For instance, social impacts during the production stage tend to be determined by the manufacturers and local government (Dong and Ng, 2015) while social impacts during disposal stage generally depend on the local or regional community's choice of waste management companies and technologies (Dreyer et al., 2006). Furthermore, Dreyer et al., (2006) advocated a share factor to ensure that social impacts are aggregated along the product chain. It may however be difficult to objectively distribute social impacts across the product S-LCA as most studies tend to sideline impacts in the use stage (Jorgensen et al., 2013). Even in instances, where the product life cycle is complete, measuring social impacts is complex and emphasis is often placed on negative impacts (Kühnen and Hahn, 2018). Impact pathways have been developed to represent linkages for demonstrating relationships between effects and causes (Jørgensen et al., 2010). However, many of the indicators used in appraising impacts are inaccurate (Arcese et al., 2013; Franze and Ciroth, 2011) and could be easily misinterpreted (Lehmann et al., 2013). For instance, a child working hardly says much about the impacts to their health which they may be experiencing. To overcome misrepresentation in impacts, (Jørgensen et al., 2010) suggested the need to distinguish between 'actual and perceived impacts'. Baumann et al. (2013) also canvassed for indicators to be unambiguously interpreted and meaningful in respective context across the life cycle.

The gap within the literature exists with the lack of social consideration of EWP's specifically CLT, especially from a life cycle perspective (Janjua et al., 2019). The consideration of CLT's social and socio-economic impacts in Australia are even more important as the vast majority of projects have been serviced from international suppliers, where Australian builders/developers exert the least amount of authority over the product supply chain. The limited social consideration of EWP's has mainly centred around the construction and forestry industry stakeholders perceptions, opinions and level of awareness (Adhikari et al., 2020). The purpose of the study is to investigate the application of life cycle thinking frameworks to EWP's used in MTC and identify any existing barriers affecting its uptake in the Australian building and construction industry.

2. Materials and Methods

Given the focus of this study, a systematic literature review was conducted to examine the applicability of a S-LCA to EWP's used in MTC. The research is considered important as there are no specific case studies which have addressed the social sustainability impacts that could occur as a result of the impending transition from traditional materials such as concrete and steel to more sustainable EWP's used in MTC and/or hybrid construction. The application of a S-LCA to an EWP such as CLT will therefore provide useful decision-making drivers and policy directions for the construction industry. To answer the overarching research question, the following sub-questions have been developed;

- (RQ1) What types of EWP's have been evaluated?
- (RQ2) What are the primary Life Cycle frameworks applied to EWP's published up until February 2023?
- (RQ3) What are the primary system boundaries evaluated?

- (RQ4) What are the positive impacts driving the increased use of EWP's?

The systematic literature review was executed in accordance with the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRSIMA) 2020 statement, checklist and flow diagram (Page et al., 2021). The review followed a series of steps, (1) identification of literature, (2) screening against the inclusion and exclusion criteria, (3) screening of titles and abstracts to only include life cycle framework studies (4) Textual narrative review of the identified literature. A fundamental criterion applied to the methodology was to assess only those papers which had applied a life cycle framework to EWP's. Grey literature was excluded to mitigate skewed results due to less standardised quality control measures which peer-reviewed publications are subjected to. The additional exclusion/inclusion criterion applied in each search engine has been provided in Table 1.

Table 1. Systematic literature review inclusion and exclusion criteria

Inclusion criteria	Exclusion criteria
• LCA/LCC/S-LCA case studies • Studies published until February 2023 • English only • Articles • Subject Area: Engineering, Environmental Science, Energy, Social Sciences, Agricultural and Biological Sciences, Material Science, Earth and Planetary Sciences, Economics, Engineering Multidisciplinary, Econometrics and Finance, Thermodynamics, Construction Building Technology, Engineering Civil, Environmental Sciences, Sustainable Science Technology, Engineering Environmental, Environmental Studies, Energy Fuels, Materials Science Paper Wood, Forestry, Materials Science Multidisciplinary, Architecture	• Non-English • Literature review • Conference paper • Book/Chapter of a book • No LCA framework application • No LCA framework application to a MTC product • Subject Area: Materials Science Textiles, Chemistry Multidisciplinary, Engineering Chemical, Marine Freshwater Biology, Mechanics, Meteorology Atmospheric Sciences, Nuclear Science Technology, Automation Control Systems, Biotechnology Applied Microbiology, Computer Science Artificial Intelligence, Computer Science Hardware Architecture, Computer Science Information Systems, Engineering Electrical Electronic, Engineering Industrial, Engineering Mechanical, Materials Science Ceramics, Material Science Composites, Parasitology, Physics Applied, Toxicology, Zoology

The analysed literature was sourced from three major search engines: Web of Science, Scopus and Engineering Village, using "AND/OR" Boolean operators. The search string terms listed in Table 2 were specifically searched in 'Topic' covering 'article title, abstract, author keywords and Keywords Plus'. The results from Table 2 were then exported into EndNote, where duplicates were removed by function and manually resulting in the removal of 137 papers. The titles and abstracts were screened against the inclusion and exclusion criteria which resulted in a further reduction of 85 papers. The remaining 74 papers were then categorised into an Excel sheet where key findings were populated under headings derived from the proposed five research questions. Finally, the contents of the studies were analysed through a 'Textual narrative synthesis' as it is a more rigorous and systematic process compared to a typical 'Narrative review' (Xiao and Watson, 2019).

Table 1. Systematic Literature review search string and Boolean operators

Search String and Boolean Operators		Web of Science	Scopus	Engineering Village
Topic	("LIFE CYCLE" OR "LIFE CYCLE ASSESSMENT" OR "LCA" OR "SOCIAL LIFE CYCLE" OR "SOCIAL LIFE CYCLE ASSESSMENT" OR "S-LCA" OR "SOCIAL SUSTAINABILITY")	114 papers	137 papers	99 papers
AND				
Topic	("MASS TIMBER" OR "MASS TIMBER CONSTRUCTION" OR "MTC" OR "CROSS LAMINATED TIMBER" OR "CLT")			
Date of search January 2023 – papers up until February 2023				
Inclusion criteria per Table 2		103 papers	133 papers	60 papers
Downloaded into EndNote – duplicates removed		76 papers	77 papers	6 papers
Screening Process – reading titles, abstracts and keywords		44 papers	26 papers	4 papers
Final number of papers for Textual narrative synthesis		74 papers		

3. Results and Discussion

From the 74 examined case studies it is evident the environment is the primary pillar considered with 72% of studies having evaluated environmental impacts such as Global Warming Potential (GWP), energy consumption, carbon emissions, acidification potential, ozone depletion potential and human toxicity potential (Lan et al., 2019; Passarelli and Koshihara, 2017; Robertson et al., 2012). Proceeding this was the combined assessment of the environment and economy with 25% of studies evaluating associated costs of EWP's compared to concrete and steel alternatives (Gu et al., 2020; Jayalath et al., 2020). Similarly, Amoruso and Schuetze (2022) highlighted reductions of GHG emissions when renovating buildings with hybrid-timber components and that the cost savings potential from the reduced energy use offset higher upfront costs of hybrid timber products. Only 3% of the studies considered all three pillars through the application of a Life Cycle Sustainability Assessment (LCSA)[2] by monetising the impacts of a reinforced concrete building and redesigning its structure in CLT revealing significant cost savings in the MTC redesign (Tokede et al., 2022). Neither the economic nor social pillars were considered as standalone assessments. The predominant EWP's assessed were CLT/CLT hybrids [3], Glulam/GLT and LVL with a LCA or Life Cycle Energy Analysis (LCEA) primarily being applied across a cradle-to-grave life cycle phase. Over 83% of studies referred to the ISO14040:2006[4] framework (Chen et al., 2019; Lan et al., 2019; Mitterpach et al., 2020; Morales-Vera et al., 2021; Vanova et al., 2021) and 13% used alternative frameworks. This highlighted a discrepancy amongst studies which either explicitly stated the goal and scope, functional unit [5] and system boundary as per the ISO1040:2006 framework.

As illustrated in Figure 1 there has been a steady increase of life cycle frameworks applied within the construction sector and more specifically on renewable materials such as timber/EWP's. Demonstrating a progression towards identifying alternative materials and processes to reduce environmental degradation whilst continuing to facilitate housing requirements, Gross Domestic Product (GDP) contribution and employment opportunities[6].

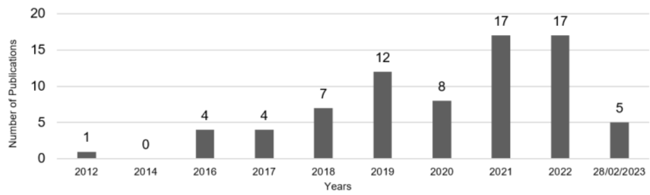

Figure 1 Number of publications each year from the literature search results

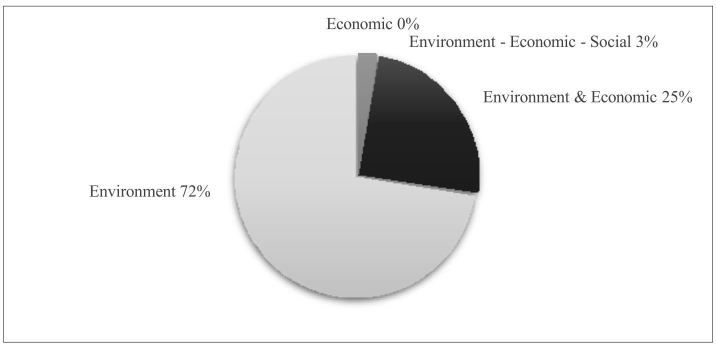

Figure 2 Percentage of Life Cycle frameworks applied to each pillar of Sustainable Development

[2] Combining the three life cycle methodologies to assess the environment, economy and social pillars leads to the application of Life Cycle Sustainability Assessment (LCSA) (Norris et al., 2020).
[3] Massiv-Holz-Mauer (MHM) is a German patented CLT system in which the cross-sectional panels are mechanically fastened together (Santi et al., 2016) opposed to the traditional method of using adhesives.
[4] Environmental management – Life cycle assessment – Principles and framework.
[5] Quantifiable description of a product serving as a reference for impact assessment calculations (Arzoumanidis et al., 2020).
[6] The Australian building and construction industry is the third largest industry in terms of the number of people employed (Labour Market Information Portal (LMIP), 2022).

Despite the varying objects of study, locations and nominated system boundary[7], a prevalent theme amongst the studies was limited data availability. For example, Dodoo et al. (2014) assumed that 90% of the demolished materials would be recovered at the disposal phase however, this potential is unknown for MTC as the demolition data of such buildings/EWP's is unavailable. The assumption of a best-case scenario of recovering such volumes of materials could be argued as not being reflective of current industry practices. Similarly, Connolly et al. (2018) evaluated the feasibility of MTC cores in lieu of concrete cores in an existing case study building, where data was sourced from both the literature and the steel used in the reinforced concrete (RC) core was assumed. This challenge is further compounded when conducting any type of life cycle assessment as data is required at each phase for multiple impact categories. Not only does this make it difficult to generalise findings across the construction sector, but on materials and methods of construction which may still be nuanced in certain countries. Fortunately, existing databases such as Ecoinvent, ReCiPe, SimaPro and Okobaudat render it possible to conduct environmental LCA's on hypothetical buildings/construction sites and materials. However, this is not the case when looking at the social and socio-economic impacts of the same objects of study, especially through the application of a Social Life Cycle Assessment[8] (S-LCA) framework.

With CLT consumption forecasted to reach a value of $1.1billion in 2023 (Schmidt and Griffin, 2013) (cited in Pierobon et al., 2019) and the availability of $300mil of funding for timber buildings in Australia (The Fifth Estate, 2022) this research into environmentally renewable building materials social and socio-economic impacts is vital. This evaluation will ensure the reduction of negative impacts in the environmental pillar, whilst attaining profitability in the economic pillar does not hinder the stakeholders of the social pillar i.e. Workers, Local Community, Consumers etc. Although limited data availability is not restricted to certain life cycle frameworks, it is further exacerbated when conducting a S-LCA due to the highly context-specific nature of the assessment, subjectivity of the social impact category and indicator selection[9] and requirement of data regarding sensitive topics such as wages, child labour, health and safety, discrimination etc. As an S-LCA is conducted on either a product or service provided by an organisation and their associated stakeholders along the life cycle, it is not a measure of a sectors performance but of the organisation/supply chains performance within that sector. S-LCA researchers tend to use the two common databases, (1) The Social Hotspots Database (SHDB) or (2) the Product Social Impact Life Cycle Assessment database (Psilca) to identify where the social hotspots of a product and/or service are located geographically and which stakeholders are potentially/actually impacted however, this information only forms part of the evaluation.

In addition to the structural challenges associated with obtaining social data, there are challenges with the social context of social data. Benoît-Norris et al. (2011), stated that many organisations are hesitant to reveal social data due to the reputational risk, and hence many social data are reported on a volunteer basis. Furthermore, social conditions tend to be dynamic and changes of social data are much faster than environmental data, which renders even more complexity (Wu et al., 2014). Majority of hotspots and social impacts might not be identifiable as user's experience may be lacking or occurring impacts are still un-known (Neugebauer et al., 2015) and in cases where hotspots are known, they could change over time (van der Velden and Vogtlander, 2017). Lastly, the indicator values of social data tend to be incomplete or difficult to define (Tokede and Traverso, 2020).

4. Conclusion

This study provides an investigation on the application of life cycle thinking frameworks to engineered wood products used in mass timber construction. The study highlights the dearth of

[7] Cradle-to-Grave (raw material extraction – manufacturing – on-site installation – disposal) or Cradle-to-Gate system boundary (raw material extraction – manufacturing – on-site installation).

[8] A S-LCA is a methodological framework established from the foundations of the ISO14040 to evaluate the potential/actual social/socio-economic impacts of a product/service across its life cycle. It adopts the same four step principals of (1) goal and scope definition, (2) inventory analysis, (3) impact assessment and (4) interpretation and communication of results (Norris et al., 2020).

[9] Per the 2020 Guidelines there are six social stakeholders which can be evaluated and a set of sub-impact categories per stakeholder group that can be measured by either a performance reference scale impact assessment or an impact pathway assessment (Norris et al., 2020).

studies on social life cycle assessment and argues for more research to support the impending transition from traditional materials of steel and concrete to more sustainable materials such as engineered woods in structural applications for buildings. The study analysed 74 papers and concluded that all the existing studies have assessed the environmental impact of EWP in buildings, while only 3% have focused on the potential social impacts of EWP.

In addition, to these, there is no specific industry framework on social LCA to support the assessment of social impacts of EWP buildings. There seems to be consensus that the lack of studies on S-LCA is due to lack of data availability on the social concerns of MTC and therefore EWP. Where there is some data the information available is not specific enough i.e. it may be country based/region based. It will be useful to collect data from studies. Some researchers have made effort to obtain S-LCA data, however the potential reputational damage and rebound effects creates limited access to data on sensitive issues as such on confidentiality, ethics, privacy and intellectual property to organisations. Although there are organisations who may be willing to provide information – they may only be small scale and therefore not be able to represent an industry. Therefore this is important for data to be collected on a larger scale so it can better represent industries and identify blind spots.

Furthermore, there is a need for empirical evidence to comprehensively understand EWPs social and socio-economic impacts, just as its environmental and economic impacts have been understood/assessed through E-LCA and LCC studies. Future work will provide a framework to support the development of a S-LCA framework applicable to EWP in the building and construction sector. It is believed that an understanding of the social impacts of EWP will provide a robust intervention in minimising the negative impacts of transitioning into a more sustainable built environment.

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Agriculture Abstracts

A Common Approach to Sector-Level GHG Accounting for Australian Agriculture.

Thursday, 20th July - 11:00: Agriculture

Dr. Maartje Sevenster [1], ***Dr. Marguerite Renouf*** [2], ***Dr. Annette Cowie*** [3], ***Prof. Richard Eckard*** [4], ***Dr. Murray Hall*** [1], ***Mr. Kieran Hirlam*** [5], ***Dr. Nazmul Islam*** [6], ***Ms. Alison Laing*** [1], ***Dr. Mardi Longbottom*** [5], ***Ms. Emma Longworth*** [7], ***Dr. Brad Ridoutt*** [1], ***Dr. Stephen Wiedemann*** [7]

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In December 2019, the need for a common approach to GHG accounting across agricultural sectors was identified in a workshop with representatives from a range of industry and government stakeholders. As sector-level reporting was starting to become important, the lack of clear methodological guidance for this type of GHG accounting was clear. A common approach for GHG accounting across agricultural sectors was seen as essential to enhance consistency, transparency and confidence in sector-level GHG reporting.

Existing national and international standards deal with GHG accounting at the level of products, corporate entities, projects, events and regions, but not for 'sectors'. Agricultural sectors are defined by the products they produce but involve an annually changing cross-section of entities and regions. This means that none of the existing standards have the required scope and some methodology choices need to be tailor-made for sector reporting. Consensus was a key focus of the collaborative project that resulted in a Common Approach, with very broad agreement from stakeholders.

Choices regarding system boundary, emission source categories, multifunctionality and inventory calculation were informed by an extensive screening of guidance frameworks relevant for agriculture as well as for Australian GHG inventory assessments. The key national and international overarching frameworks that the Common Approach draws on are the Australian National Greenhouse Gas Inventory (NGGI) and its approaches, ISO standards, guidance provided by FAO (Livestock Environmental Assessment and Performance Partnership) and the GHG Protocol.

The Common Approach consists of a "Methods and Data Guidance", that provides detailed guidance for sector-level accounting, and a "Common Terminology" companion document. The Methods and Data Guidance is written as a technical guidance for GHG accounting experts supporting agricultural sectors in their reporting activities. Some sectors have already committed to use the guidance for their GHG inventory and reporting.

Methodological issues with carbon accounting in agricultural supply chains

Thursday, 20th July - 11:00: Agriculture

Ms. Emma Longworth [1]***, Dr. Stephen Wiedemann*** [1]

1. *Integrity Ag*

Agricultural systems are complex; emissions arise from many biological processes, and some inputs and outputs are difficult to measure. Emission estimation relies on modelling approaches that are further complicated by a high degree of interannual variability, making it challenging to quantify the sources and removals of greenhouse gas emissions for agricultural organisations and products. Due to these complexities, agricultural emissions, emission reduction activities, land use and land use change emissions and removals have often been excluded from organisational carbon accounts or product carbon footprints. The following concepts and recommendations are raised to help improve the consistency and avoid material differences caused by different accounting methods for organisations or products that rely substantially on agricultural products, including food, beverages and textiles, and create alignment with best practice accounting methods. This presentation is an opinion piece and is designed to facilitate discussion.

GHG reporting and LCI databases: Australian wheat as a case study

Thursday, 20th July - 11:00: Agriculture

***Dr. Maartje Sevenster* [1], *Dr. Aaron Simmons* [2]**

1. CSIRO, 2. NSW Department of Primary Industries

There is increasing pressure on (large) companies to report supply-chain "Scope 3" greenhouse gas (GHG) emissions both as part of initiatives like the Science-Based Targets (SBTi) and as part of climate risk disclosures (Task Force on Climate-related Financial Disclosures; TCFD). A number of countries are already introducing legislation on GHG reporting for listed companies (e.g.TCFD, 2021). This is further increasing reliance on life cycle inventory (LCI) databases to provide data for embedded emissions of inputs, including agricultural commodities. For robust Scope-3 emissions reporting there is an urgent need for agricultural LCI data that has appropriate geographical coverage and excellent quality (e.g. representativeness).

The geographical coverage of agriculture LCI in databases has been increasing, but when it comes to emissions from agriculture, a better understanding is needed of exactly what constitutes consistency.

A case study for wheat shows that, of the main exporting countries, Australia is particularly misrepresented. The choice to apply IPCC Tier-1 emission factors for all countries, ignoring the availability of country-specific UNFCCC reporting, leads to overestimates of N_2O emissions for some countries and underestimates in others. There are also discrepancies in activity data, with e.g. nitrogen fertiliser rates for Australian wheat production set at 43 kg N/ha in one prominent database and 70 kg N/ha in others. Fertiliser mixes vary considerably, with some clearly not representative for Australian wheat production. With nitrogen input a key parameter for productivity, GHG emissions and soil depletion (Sevenster et al. 2022), such discrepancy is concerning.

All LCI processes purport to represent Australian wheat production but in reality, geographical and temporal coverage seem insufficient, and data quality ratings misleadingly. A better way to leverage country-specific knowledge while maintaining consistency and transparency needs to be established, e.g. by coordinating methodology choices between single-country LCI databases.

Challenges for reducing and reporting GHG emissions in the Australian grain supply chain and the role of the Cool Soil Initiative

Thursday, 20th July - 11:00: Agriculture

Mrs. Jocelyn Hordern-Smith [1], ***Dr. Cassandra Schefe*** [2], ***Dr. Alice Melland*** [3]

1. Collaborative Sustainability Systems, 2. AgriSci Pty Ltd, Rutherglen, Victoria, 3. University of Southern Queensland, Toowoomba

There is a growing requirement to quantify and report on greenhouse gas (GHG) emissions in the grain supply chain. Internal and external stakeholders, including governments, financial services, and business sectors, are also keen to quantify agricultural sustainability credentials to underpin their own environmental stewardship goals and targets. The practical challenges that exist are how to use existing methods to reliably estimate agricultural supply chain emissions over the long term while identifying and implementing realistic strategies to improve environmental outcomes.

Broadly, the grain supply chain is comprised of suppliers of agricultural inputs, farmers, grain aggregators and millers, manufacturers, and retailers. As the chain is non-linear and multiple inter-relationships exist, another challenge is how to practically quantify and equitably allocate any quantified agricultural emissions (and changes in emissions) amongst supply chain stakeholders.

The Cool Soil Initiative (CSI) is a science-based partnership across the grain supply chain that is implementing common metrics and processes to estimate and report farm-based emissions by working directly with farmers to create practical solutions to reduce their emissions intensities. Since 2018, the CSI has reported on-farm emissions using a globally relevant and credible farm emissions calculator (the Cool Farm Tool), and internationally recognised best practice protocols to apportion emissions throughout the supply chain partners. Direct farmer support to identify sustainable practice change provides both a value proposition to farmers, and a vehicle for real mitigation of on-farm emissions.

Key learnings to date are that good farmer engagement is critical for practice change, farm activity data collection processes need to be streamlined (requiring digital data platforms with individual support), and that as on-farm emission intensity reduction is a long-term process, ongoing support for implementation and accounting initiatives such as the CSI is imperative for success.

Customised LCA tool for viticulture (VitLCA) for identifying environmental improvement opportunities

Thursday, 20th July - 11:00: Agriculture

Dr. Marguerite Renouf [1], *Dr. Christel Renaud-Gentié* [2], *Anthony Rouault* [2], *Raphael Suire* [2], *Aurélie Perrin* [2], *Emmanuelle Garrigues-Quéré* [2], *Severine Julien* [2]

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Inventory development for LCA of wine grape growing (viticulture) is complex compared with other crop systems, which hampers the identification of priority environmental improvements for this sector. The aim of this work was to develop a customised LCA method and tool (VitLCA) for viticulture, which makes the evaluation of practice alternatives easier and more rapid than conventional approaches. This paper describes our systematic process for developing the methods in VitLCA, and demonstrates how it can enable better evaluation of viticulture practices. The method involved defining the practice variables that influence inputs and emissions in viticulture, identifying best practice methods for estimating them based on these variables, and operationalising these methods in an online tool. The resulting VitLCA tool generates a range of impact indicators, including those commonly assessed for agriculture (global warming, eutrophication, fossil fuel, mineral and water resource depletion), but also eco-toxicity, which is important for this sector. It enables a more com-prehensive assessment of viticulture practices than has been possible in past studies, considering all facets of viticulture (establishment and productive phases, trellis infrastructure, capital goods, irrigation as well as an-nual activities) for a wide range of impact categories. It also enables rapid assessment and speaks the language of viticulture professionals, thereby facilitating its use by the sector.

Building Materials Abstracts

Life Cycle Assessment of Prefabrication Construction: A Review

Thursday, 20th July - 11:00: Building Materials

Mr. Thang Tran [1], *Dr. Ziyad Abunada* [1], *Dr. Farzaneh Tahmoorian* [1]

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The construction industry plays a vital role in the global economy and has been viewed as one of the main industries that can promote global sustainability. The construction industry is responsible for around 40% of the energy consumed, 39% of the global CO2 emissions, and 35% of landfill wastes. It also consumes about 50% of the global materials and water resources.

Prefabrication construction has been fostered by many associated benefits including shorter construction times, higher quality, less cost and fewer on-site requirements, higher safety, and lower environmental impacts. Yet these benefits come with a price as they are normally challenged but the slow adoption of this technology in most countries, particularly developing countries.

Life Cycle Assessment (LCA) and its extensions, including life cycle assessment (LCA), life cycle cost analysis (LCCA), and social life cycle assessment (SLCA), are among the effective analytical tools that allow for a comprehensive sustainability analysis including environmental, economic and social impacts over the life cycle of the structure.

The current research undertakes a comprehensive review of the prevailing literature on prefabrication construction, identifies the gaps in knowledge, and establishes a fundamental understanding of the importance of LCA in comparing modular and conventional construction methods. In addition, this study aims to review the sustainability of modular construction and compare it with traditional construction.

Keywords: Prefabricated Construction, Conventional Construction, Life Cycle Assessment, Sustainability

Environmental Performance of Recycled Concrete Aggregates using Life Cycle Assessment : Comparing Business as Usual with 115 Hamilton, Western Australia.

Thursday, 20th July - 11:00: Building Materials

Mrs. Ugyen Lhachey [1]***, Dr. Martin Anda*** [2]***, Dr. Biji Kurup*** [3]***, Mrs. Naomi lawrance*** [4]

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Waste generated from the construction and demolition sector continues to increase over time, and this negligence poses a severe threat to the environment. Reusing and recycling construction and demolition waste-derived materials must be the priority for resource sustainability and achieving a Circular Economy (CE). Many studies demonstrate that recycled concrete aggregates from demolished buildings can be used in various engineering applications. Still, there is little research to assess the environmental impact and sustainability of the product using Life Cycle Assessment (LCA) techniques. Current specifications in Western Australia require a majority (80%-90%) of the recycled base in road construction to be concrete. LCA techniques are essential to evaluate the environmental performance and sustainability of producing recycled materials in the construction and demolition industry.

The study compared the environmental impacts of the processes involved in producing Recycled Concrete Aggregate (RCA) from demolition waste and Business as Usual (BAU) cases for road construction using LCA techniques for case-specific and primarily sourced data. The results indicate that with 100% RCA used in road bases, a significant carbon emission reduction of almost 95% was observed. While only a 55% reduction was achieved when replacing the BAU scenario with 70% RCA and 30 %NA. However, if the RCA processing involves significant transportation distances and more electricity consumption, then a reduction in environmental impacts from the project is unlikely.

The findings prove that data collection, location, in-situ utilization of the recycled materials, and backhaul plans are the ideal components for CE in the recycling industry. Further investigation on a similar topic is needed to understand recycled products' environmental impacts and sustainability so policymakers can be informed when legislating the necessary regulations and guidelines.

Building Materials
Extended Abstracts

Environmental Performance of Recycled Concrete Aggregates using Life Cycle Assessment: Comparing Business as Usual with 115 Hamilton Hill, Western Australia

Ugyen Lhachey [a], Biji Kurup [a], Martin Anda [a], Naomi Lawrance [b]

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Keywords: Recycling; Recycled Concrete Aggregate; Life cycle Assessment; Circular Economy; Environmental impacts

1 Introduction

Construction and Demolition Waste (CDW) are materials arising from construction, refurbishment, or demolition activities (DWER, 2019). Recycling demolished materials have become an essential attribute of modern society. According to the National Waste Report, Australia generated about 25.2 metric tonnes of CDW collected between 2020-2021 (Pickin et al., 2022). In Western Australia, 30-40% recycling was reliably achieved, and the Waste Authority aimed for 70% material recovery and a 30% reduction of CDW per capita by 2030 (Waste Authority, 2019). The European Union Waste Framework Directive 70% recovery of the CDW target by 2020 for reuse and recycling exceeded as most countries achieved > 80% beforehand (Directive 2008/98/EC, 2008).

To achieve this, recent innovative projects by DevelopmentWA, the WA State Government land development agency, in partnership with industry, have demonstrated that close to 100% of both construction and demolition wastes can be recovered directly for reuse and recycling. For example, the demolition project at 115 Hamilton Hill (115HH) has successfully achieved 86% reuse of materials on site and 10% recycling offsite, demonstrating the effectiveness of the Resource Recovery Strategy at 115HH and reducing 1000 truck movements with on-site processing of materials (DevelopmentWA, 2022, 2021). For this reason, 115HH was selected as the project site for its resource recovery potential and for determining the environmental benefits of on-site processing and use.

With the population increasing, natural resource consumption is a growing trend for the urban development of buildings, infrastructures, bridges, roads, airports, and harbours (Esguícero et al., 2021). The critical environmental burden of such construction is the waste generation and high demand for construction materials. Disposing CDW and extracting natural resources cause a problem for terrestrial and aquatic flora and fauna, thus destroying the environment (Balaguera et al., 2018). Akhtar and Sarmah (2018), Lawania and Biswas (2016), and Shooshtarian et al. (2022) all argue that the construction sector should prioritize sustainability and preserve natural environmental demands reducing carbon emissions and conserving essential natural resources.

There is an opportunity to consider alternatives to virgin materials, such as roadbase from post-consumer recycled aggregates from crushing and screening concrete CDW. Due to the adoption of green strategies in construction, many academics have concluded a potential use of Recycled Concrete Aggregate (RCA) in the lower layer of pavement construction and concrete structures due to its availability and material properties satisfying the specifications (Li et al., 2019; Martinez-Arguelles et al., 2019). Such coarse and fine RCA blends <19mm diameter are authorized as road bases in Western Australia (Waste Authority, 2021). The use of RCA as a substitute for natural aggregate is increasing due to increased landfill disposal charges, high transportation costs and the implementation of a waste hierarchy.

With emphases on sustainable development, circular economy, and green construction, more researchers are inclined to analyze the life cycle of the product and services (Guo et al., 2022, 2018; Xing et al., 2022). Life Cycle Assessment (LCA) methods are becoming more commonly used to measure environmental outcomes in CDW process and product development (Balaguera et al., 2018; M.D. Bovea and Powell, 2016; Marinković et al., 2010; Xing et al., 2022). In Europe countries, H and S, (2022) found that 78% of LCA research was on CDW. For example, much research on LCA for recycled aggregate in concrete investigates embodied energy, environmental impacts and resource use (Xing et al., 2022). Similarly, with growing interest in sustainable highways, many studies assessed the environmental performance of recycled aggregate in pavement construction through its entire life cycle (Li et al., 2019). However, the studies are unparalleled because of the variation in functional units, system boundaries, and type of recycled aggregate reused (Li et al., 2019; Xing et al., 2022).

For the Circular Economy (CE), LCA aspects avoided impacts, and recycled materials quality gained significance (Bayram and Greiff, 2023; Colangelo et al., 2018). Pomponi and Moncaster (2017); Van Stijn et al. (2021), recommend circular economies integrate LCA to measure environmental impacts.

Rosado et al., (2019) found that conventional CDW landfilling had the most significant environmental impacts and fewer economic advantages compared to processing it for reuse. Jain et al. (2020) found that recycling reduces primary material extraction as it preserves natural resources in the long term. They suggest it lessens greenhouse emissions of the whole product life cycle (Jain et al., 2020). Considering both perspectives, Guo et al., (2022) concluded that policymakers and stakeholders could make more informed and effective management decisions to promote sustainable practices.

1.1 Objectives of this study

This study examines the environmental outcomes of using Recycled Concrete Aggregate (RCA) from a demolished concrete State government school structure in Hamilton Hill (115HH) in Perth, Western Australia, as an on-site paving roadbase. It used data from a site managed by DevelopmentWA. It answers the following research questions.

1. What are the dominant environmental loads in each production phase? Is RCA likely to have a lower load than natural aggregates (NA)? Does recycling significantly change the total load?
2. How sensitive is the environmental performance to transport and diesel fuel use?
3. How does the LCA of RCA compare to the Business As Usual (BAU) and 7:3 RCA: NA scenario?

2 Materials and Method

2.1 LCA Application

The ISO 14040 2006 Standard defines LCA principles and frameworks for practitioners and stakeholders (Finkbeiner et al., 2006). The method determines environmental impacts from cradle to factory gate or grave (Finkbeiner et al., 2006). LCA studies require (1) Goal and Scope, (2) Life cycle inventory, (3) Life cycle impact assessment, and (4) Interpretation of results (Marinković et al., 2010).

2.1.1 Goal and Scope Definition

This study aims to assess and compare impacts from three scenarios. These were the production of 100% RCA, BAU natural aggregates (NA) and a 7 :3 RCA: NA mix as a road base material at 115HH.

Figure 2-1, Figure 2-2, and Figure 2-3 depict unit operations in system boundaries. The first stage of LCA involved determining overall objectives, system boundaries, data sources, and scope. The cradle-to-gate scope included all known inputs and outputs from resource extraction, demolition, scrap separation, grinding and screening to delivery. The functional unit per tonne of each scenario was used. The three scenarios used for the study are as follows.

Scenario I. 100% RCA

This LCA used data from a school demolition at 115HH. The RCA was processed and used on-site from a demolished concrete school structure at 115 Hamilton Hill (115HH) Perth, Western Australia. It encompassed demolition, transport of 0.2 Km to the Material Recovery Facility (MRF) from on-site crushing, and screening aggregate depicted in Figure 2-1.

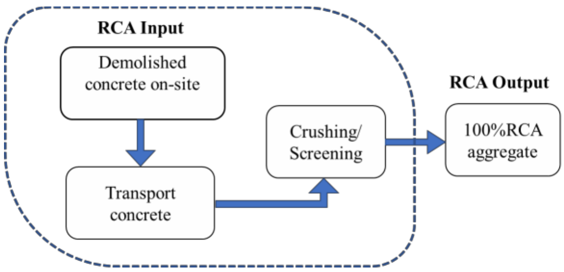

Figure 2-1: RCA System

Scenario II. Business as Usual (BAU)

The traditional approach of disposing of resources from demolition sites in landfills and using new virgin materials as road base materials are taken as the BAU scenario. Figure 2-2 depicts the BAU system boundary, including demolition, transport to landfill, natural resource extraction, transport to MRF and crushing/screening phase. This scenario considered MRF the same as the scenario I. However, emissions from landfill are excluded. The distance between the demolition site and Armadale Landfill and Recycling facility was 25 km. The quarry was assumed to be within a 25 km radius of the 115HH site, as many are within that range.

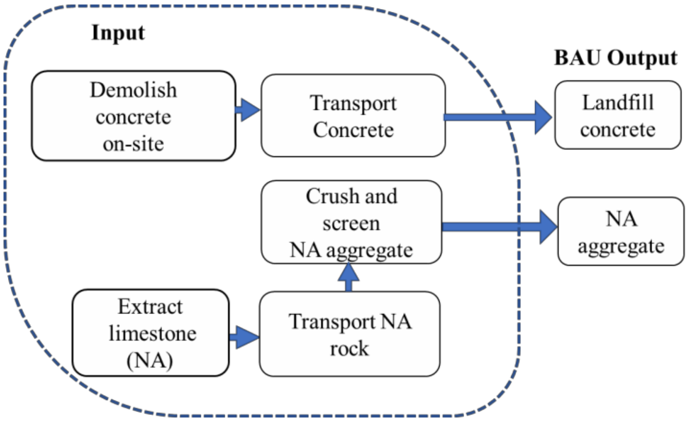

Figure 2-2: BAU System

Scenario III. RCA and NA

The third scenario uses a 70:30% RCA: NA blend, with RCA from the demolition materials as 100% RCA in the scenario I and the NA extraction process from scenario II. The novelty of this lies in its focus on assessing the potential environmental benefits of this mix compared to the BAU scenario. Figure 2-3 shows the system boundary for determining the environmental impacts of the blend.

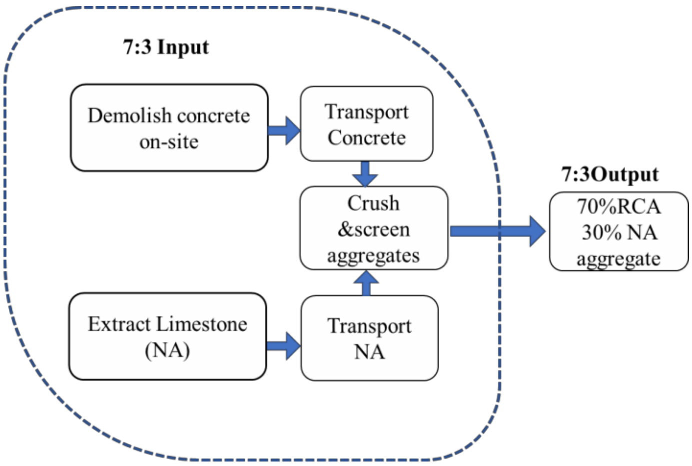

Figure 2-3: 70% RCA 30% NA Mix System

2.1.2 Life Cycle Inventory (LCI) Analysis

The next LCA stage quantifies flow charted input and outputs. It includes all energy flows, material inputs, emissions, and waste outputs along the life cycle. Flows are calculated according to the functional unit defined in the goal and scope. Primary data was collected from the DevelopmentWA demolition contractors. This study used background data for electricity, fuel and water supply from Eco -invent 3. 0 and the Australian Life Cycle Assessment Society (ALCAS). Limited ALCAS LCI models some critical local operations. Data from Eco-invent 3 defined was used to fill gaps. Eco-invent 3 is used worldwide in many regions.

2.1.3 Life Cycle Impact Assessment (LCIA)

The LCA's third stage involves the assessment of potential damages and loss. Processes involved in this study were modelled using SimaPro 9.0 software. LCIA begins with selecting impact categories and characterization factors and ensuring proper flow classification. Classification attributes flow to their respective impact categories to ensure all are noticed and double-counted. Then characterization involves applying up-to-date factors to weigh each flow's minor to major effect on damages and loss to calculate the sum equivalence potential for each operation. The study compares the environmental impact of three different aggregate materials for use as roadbase. ALCAS Australian Best Practice Recommendation method covers 13 midpoint impact categories and three end-point damage categories. Midpoint modelling is typically more confident, and end-points are less specific considering time, place and supply chain variations.

2.1.4 Interpretation of Results

LCA results are then tested considering the study goal scope and data sensitivity. Fit-for-purpose results are presented, interpreted to expose limitations and discussed to offer conclusions and recommendations.

3 Results

3.1Answering Question 1: Significance of dominant environmental load by phase and material?

The results are presented below in 3 sections and then discussed in part 4.

Table 1. LCIA result Comaprisn of 3 scenarios

Impact category	Unit	RCA (On-site)	BAU	RCA: NA
Global warming (GWP100a)	kg CO2 eq	1.13	22.03	9.94
Abiotic depletion (elem., econ. reserve)	kg SB eq	1.65E-5	1.24E-3	5.43E-4
Abiotic depletion (Fossil fuels)	MJ NCV	17.02	3.31E+2	1.49E+2
Ozone layer depletion (ODP)	kg CFC-11 eq	1.38E-7	2.84E-6	1.28E-6
Photochemical oxidation	kg C2H4 eq	1.3E-4	0.01	2.45E-3
Acidification	kg SO2 eq	2.43E-3	0.14	0.06
Eutrophication	kg PO4--- eq	3.54E-4	0.03	0.01
Particulate matter	kg PM2.5	1.99E-4	0.02	0.01
Human toxicity, cancer	CTUh	1.41E-9	7.04E-8	3.13E-8
Human toxicity, non-cancer	CTUh	2.62E-8	6.1E-7	2.73E-7
Freshwater ecotoxicity	CTUe	41.18	3.25E+4	1.44E+4
Ionizing radiation HH	kBq U235 eq	2.9E-5	3.92E-3	1.74E-3
Water Scarcity	m3 eq	0.01	0.25	0.1

Table 1 lists 13 environmental damage midpoint impact results per tonne aggregate for RCA, BAU and RCA: NA mix. It shows RCA production emits 21kgCO_2eq less than the BAU and 8.8 kgCO_2eq than the RCA: NA mix. Also, the RCA: NA mix emitted 12 kgCO_2eq less than BAU. RCA used 95% less fossil fuel than BAU and 7:3 RCA: NA mainly because it use far less fossil fuel in processing and transport. RCA was, however, worst in freshwater ecotoxicity.

3.1.1 Contribution Analysis

Table 2. RCA process contributions

Impact Categories	**Unit**	**Transport truck**	**Diesel Used**	**Water used**
Global Warming Potential	$kgCO_2eq$	0.04	1.09	0.01
Abiotic Depletion (Fossil fuel)	MJ NCV	0.029	16.3	0.128
Particulate Matter	kgPM 2.5	1.58E-5	0.00018	3.23E-6

Table 3. BAU Process Contributions

Impact Categories	**Unit**	**Transport truck**	**Diesel Used**	**NA Extraction**
Global Warming Potential	$kgCO_2eq$	4.93	0.433	16.7
Abiotic Depletion (Fossil Fuel)	MJ NCV	78.7	6.49	246
Particulate Matter	kgPM 2.5	0.00197	7.2E-5	0.02

Table 4. RCA: NA Process Contributions

Impact categories	**Unit**	**Transport truck**	**Diesel Used**	**NA Extraction**
Global Warming Potential	$kgCO_2eq$	5.95	1.08	2.91
Abiotic Depletion (Fossil Fuel)	MJ NCV	95.029	16.2	38.1
Particulate Matter	kgPM 2.5	0.0025	0.0018	0.00722

Table 2 to 4 list environmental loads from processes involved in each production scenario. Most damages in global warming, Fossil Fuel Accessibility and Fine Particulates arise from fossil fuel use in BAU transport and natural aggregate extraction. Fossil fuel combustion typically generates the most carbon dioxide and particulate emissions into the air, contributing significantly to the anthropogenic loss of climate and human health. The RCA avoided significant damages from global warming, fossil fuel depletion and fine particulate. In this scenario, most such damages arose from fossil fuel use in RCA crushing, loading, and screening, as on-site scrap crushing and screening limits its transport distance from demolition to reuse.

3.2 Answering research question 2: Environmental sensitivity to transport and diesel fuel use

BAU natural aggregate extraction and transport generate most GWP, fossil fuel, and PM2.5 impact primarily from the combustion of diesel fuel used in driving heavy mining and transport motors. Such damages are avoidable by switching to renewable energy supply and post-consumer recycled feedstock. Mines land use change reducing vegetation cover and its carbon drawdown contribute most significantly to global warming and biodiversity loss, but this study did not include that modelling. Results show that the 55% lower impact RCA: NA mix scenario was mainly influenced by transport which will increase with greater distance.

3.3 Answering research question 3; Benefits of RCA.

Valuation is an optional step in LCA. This study applied the hierarchies (H) valuation perspectives based on the scientific consensus on the time frame and impact mechanism. Figure 3-1 displays ReCiPe 2016 LCIA results from 3 scenarios of hierarchical end-point damages such as human health, ecosystem quality and resource access. Overall, RCA had the most minor damage, then 7:3 RCA: NA mix and BAU had the most damage.

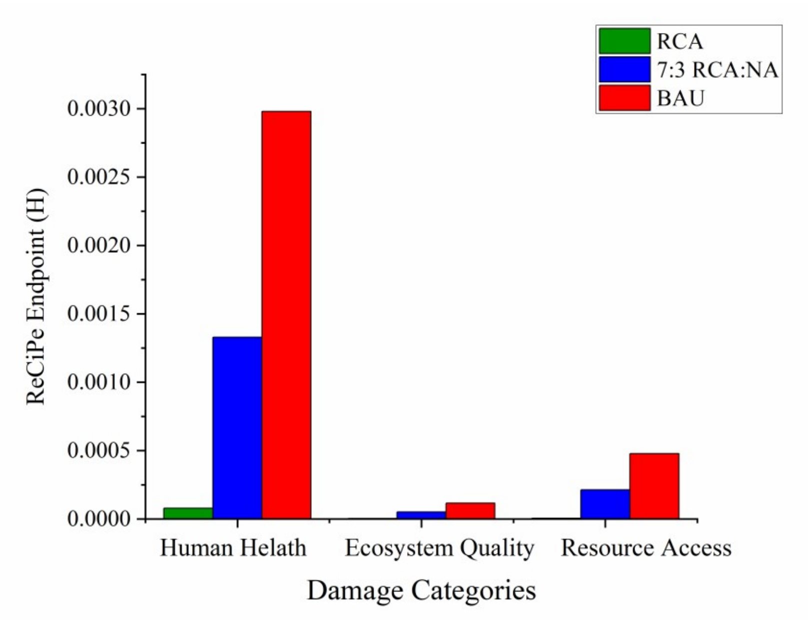

Figure 3-1. ReCiPe (H) Human Health, Ecosystem Quality and Resource Access End-point Damages

Figure **3-2** and Figure **3-2** depicts GWP damages as positive and avoided GWP as negative. As it uses neither scrap transport to landfill nor natural aggregate extraction, RCA avoids 20.9 $kgCO_2eq$/tonne compared to virgin aggregate processing, which needs both. Moreover, compared to the third scenario RCA avoids 8.81$kgCO_2eq$/tonne.

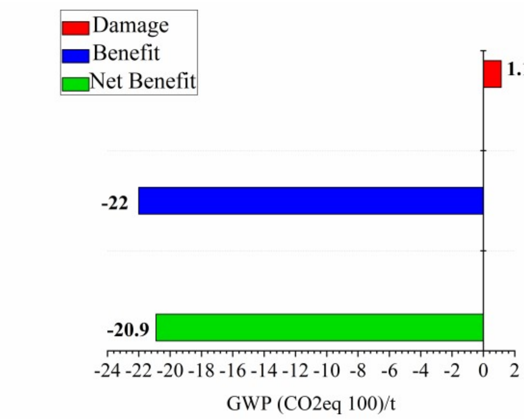

Figure 3-2 RCA avoided GWP

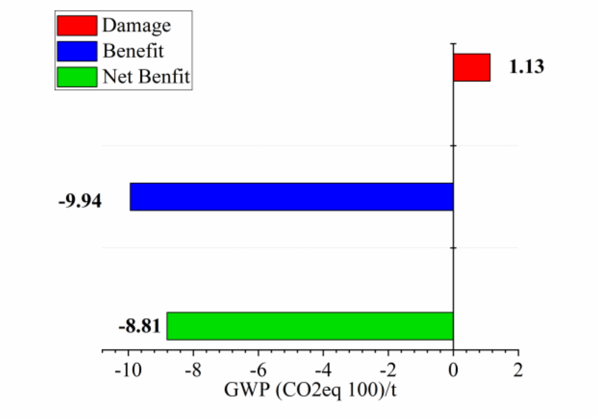

Figure 3-3 Natural Aggregate Extraction Avoided GWP

4 Discussion

The result suggests that RCA for roadbase and reduced damages are positively correlated. In the present study, concrete from demolition recycled on-site showed significantly less environmental damage than in the BAU case. RCA reduced the three climates forcing, ecosystem loss and resource depletion end-point damages and all 13 midpoint damages. These findings support our hypothesis that recycling aggregate can offer better environmental outcomes. However, this study only modelled a 0.2 Km transport distance as demolition, processing, and application occurred on-site outcomes will change with longer fossil fuelled transport distance.

The LCA used energy and emissions primarily from ALCAS LCI, with gaps from Ecoinvent 3.0 LCI. This study used ALCAS recommended Best Practice LCIA calculation method for impact, contribution, and benefit analyses. Valuation employed ReCiPe 2016 hierarchies end-point damage assemble for each scenario.

A CDW management system LCA must incorporate standardized procedures to establish and customize appropriate system boundaries, functional units, and inventory data sets (M. D. Bovea and Powell, 2016). LCA study environmental impact quantification accuracy and reliability depend on data quality and completeness, as well as selected indicators and characterization factors (Mesa et al., 2021). Primary data for this study were collected from a demolition contractor. However, before contractor data became unavailable, some values were assumed from previous study results.

This work's findings are consistent with the previous research showing the environmental benefits of recycling. For instance, similar studies by Martinez-Arguelles et al. (2019) in Colombia and by Colangelo et al. (2018) in Italy concluded that efforts to reduce environmental impacts could focus on minimizing diesel fuel use and transport distance. Zhang et al. (2019) also showed the findings from alternative production or transport strategies, using renewable energy sources, or improving supply chain efficiency.

According to Dahlbo et al. (2015); Di Maria et al. (2018); López Ruiz et al. (2020), sustainability supports the use of post-consumer recyclate from demolished buildings to minimize waste in landfills, saves energy, and conserves natural resources, which could explain these correlations.

The research also indicates that although recycling and reusing 100% of demolished resources as road base material may not be possible, incorporating a certain proportion of natural resources can still provide some environmental advantages, rather than resorting to new virgin materials. For example, substituting a portion of NA with RCA is considered an eco-friendly alternative with less environmental impact linked to the extraction of finite NA (Esa et al., 2021; Ghisellini et al., 2018), and our study proved that. However, as per the specification on recycled road base and drainage (Waste Authority, 2021), the benefits of recycling per tonne of recycled aggregates should not be impacted by the amount of recycled content as long as it remains within the technical limitations of the recycling production process.

The strength of recycled materials is greatly influenced by the composites of the parent materials, regardless of the type of the recycled material. The factors such as the crushing method, size, density, and compressive strength of recyclate need to be systematically investigated in this study. The RCA used as a road base at the 115HH site was tested according to recycled concrete specifications. The test results demonstrated that the RCA met the required standards. However, it is essential to note that during the crushing of demolished materials, the low-strength parent concrete tended to break faster than the recyclate produced from high-strength parent concrete. This results in the production of finer materials, which ultimately leads to lower strength recyclate. It is essential to consider this factor as it influences the overall strength characteristics of the recycled aggregates.

5 Conclusion

This research studied incorporating recycled aggregate from the demolished concrete building for road base as a potential strategy for reducing waste, conserving natural resources, and promoting sustainability. The study was prompted by the pressing concern of escalating construction and demolition waste generation and ensuring the imperative to mitigate the construction sector's environmental impacts. The LCA compared the original case study with the BAU and 7:3 RCA: NA. Its significant importance stems from the growing interest of national agencies in employing recycled aggregates for road infrastructure.

This work confirmed that demolished materials on-site processing generated less environmental damage than offsite processing. It showed that material recycling highly depends on assumed transport distance, especially for low-value materials like recycled concrete. LCA results indicate that substituting RCA for primary aggregate can save about 20.9 kgCO_2eq/tonne of aggregate produced. They also revealed that adopting RCA in construction could conserve up to 95% of typical primary nonrenewable resources. Overall, results indicated that such recycling activities generate environmental benefits across all impact categories studied.

However, although the RCA processing GWP is very low compared to other scenarios, its crushing and screening could lead to higher human toxicity from increased diesel use and crude oil extraction impact. Displacing such fossil fuel usage with renewable energy could reduce such impact.

Results revealed that the BAU scenario extraction process and haulage distance generate the most significant environmental stressors. In light of this, resource authorities and policymakers could explore promoting recycling and reuse to safeguard the environment in this rapidly transitioning world. The study highlights the significant demand for using recycled material in road construction. This demand could play a crucial role in promoting the adoption of circular economy practices in Australia. Crucial exclusions of the LCA study were also identified, such as a lack of cost-benefit and sensitivity analyses.

Acknowledgements

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Data and Databases Abstracts

Scaling LCA data usage in an evolving regulatory environment.

Thursday, 20th July - 13:30: Data and Databases

Dr. Nic Meyer [1]
1. *ecoinvent*

The ecoinvent association began as a series of Swiss research projects in the 1990's and spun off as an independent association in 2013. Approaching our 20 year anniversary of the release of version 1.0, and our 10 year anniversary as an independent organisation, we take this opportunity to reflect upon the history of ecoinvent and to share an outlook on how ecoinvent continues to create sustainable, mission driven impact.

During the 1990's, LCA data was created and maintained by individual research groups in the Swiss research community. Expertise was building amongst key working groups, but harmonisation was low and duplication of efforts slowed progress. By the 2000's key members of the community banded together in a series of scientific projects to create a common database, leveraging the core strengths of each group and harmonising the data, and so ecoinvent V1.0 was born.

10 year later, with the release of version 3, ecoinvent set a direction that laid its' foundation in the environmental data sector. Now containing > 19,000 data sets, more than 5000 products and providing background data to many of the worlds' databases and software tools, ecoinvent is committed to maintaining the robustness and quality of our database, developing more approachable interfaces, and continuing to add the latest data from around the world.

As a mission driven organisation we believe that each and every company needs to have access to the data and tools to make sustainable business decisions and play their part in the worlds sustainability endeavours. Subsequently, ecoinvent works actively to grow our partner network and internal capability to pursue this mission. Join ecoinvent CEO, Nic Meyer, on this exploration of ecoinvent, past, present and future.

Development of Embodied Emissions Database based on AusLCI

Thursday, 20th July - 13:30: Data and Databases

Mr. Tim Grant [1]

1. *Lifecycles*

AusLCI is the national LCA database developed by ALCAS over the past 15 years. The initial vision for the database has been as a publishing framework for Australian Life Cycle Data. The database has grown over the past 15 years with major investments in Agriculture sphere however the building products sector investment has gone into BPIC data inventory library of processes and environmental product declarations.

While EPDs are an excellent form of data for end users of data, they are less useful in the development of connected unit process libraries. However connected unit process libraries are essential for the development of EPDs. The outcome of this approach is and temporal and technical inconsistency in EPDs data. An approach to circumvent these problems is proposed which maintains both the integrity of the AusLCI database and the confidentiality requirements of the construction products industry.

Coffee LCA studies, how can results vary so much?

Thursday, 20th July - 13:30: Data and Databases

Dr. Cécile Chéron-Bessou [1], ***Dr. Ivonne Acosta Alba*** [2], ***Prof. Adisa Azapagic*** [3], ***Dr. Joachim Boissy*** [4], ***Dr. Sandra Payen*** [1], ***Mr. Nicolas Pouraiilly*** [5], ***Dr. Clement Rigal*** [1], ***Dr. Arief A. R. Setiawan*** [6], ***Dr. Maartje Sevenster*** [7], ***Dr. Thierry Tran*** [8]

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See extended abstract.

Understanding the impacts of adopting different system boundaries on the life cycle assessment of biosolids processing systems

Thursday, 20th July - 13:30: Data and Databases

Mr. Jingwen Luo [1], ***Dr. Ruth Fisher*** [1]

1. *School of Civil and Environmental Engineering, UNSW, Sydney NSW 2052, Australia*

Biosolids, produced from the wastewater treatment process, present an important opportunity for the sustainable development of the water industry. As the most widely adopted sustainability assessment tool, there is a lack of consistency in the selection of system boundaries in the past applications of life cycle assessment (LCA) in biosolids processing systems. This study conducted an LCA of a conventional biosolids processing pathway using anaerobic digestion and land application in the Australian context and, combining the information from a literature review, investigated the implications of adopting different system boundaries. This study included all processes from pumping the primary and waste-activated sludge into the processing system up to the long-term effects associated with the land application of biosolids. All the relevant inputs (energy, chemicals, transportation) and outputs (emissions and system credits) were considered, including the odour management system and the treatment of returned liquor. The results indicated that the adoption of different system boundaries could bring a large influence on the assessment results. For instance, some typical system boundaries adopted by previous studies can cover 31% to 97% of the total global warming potential (GWP) and 0% to 99% of the total human toxicity potential (HTP); the most widely adopted system boundary only captures 84% of the overall GWP. In addition, the inclusion of some key process inputs and outputs can also have large impacts. For example, the returned liquor stream from dewatering, which is not commonly included, can result in a 15% change in the GWP and a 114% change in the eutrophication potential (freshwater). The results of this study can provide a basis for the harmonisation of methodological practices for LCA studies aiming at assessing the environmental impact of biosolids processing systems.

Data and Databases
Extended Abstracts

Coffee LCA studies, how can results vary so much?

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Keywords: coffee, life cycle assessment, carbon footprint

1. Introduction

Coffee is a major agricultural product; it is one of the most widely consumed beverages and one of the most traded commodities in the world. Australia is the seventh greater importer worldwide and its vibrant coffee culture is likely to play an increasing role in the global market. Coffee global consumption and production keep growing rapidly (FAO, 2021). To meet the growing global demand, coffee production is expected to double by 2050 (Conservation International, 2020[1]), which has pushed a rapid development of sustainability initiatives among coffee sector stakeholders (Noponen, 2012). However, there is not yet a clear cut view on how these initiatives do help protect the environment in the field.

Coffee is grown in the tropics but consumed all around the world, and in particular in Europe (32.5%) and North America (19%) (International Coffee Organization, 2020/2021[2]). The great diversity of agricultural systems in the tropics and the various trade routes give rise to very diverse supply chains. The diversified practices lead to contrasted potentials and performances. Coffee can notably be grown in agroforestry plots, whose potential triggers interests in the application of Climate Smart Agriculture strategies to coffee production (FAO, 2021). On the other hand, several studies have shown the climate sensitivity of coffee and the variable impact of climate change on coffee suitability, yield, and farmers' livelihoods (Grüter et al. 2022; Alemu and Dufera, 2017; Läderach et al., 2017; Rahn et al., 2014). Both mitigation and adaptation strategies require quantifying the performances and improvement margins while accounting for the diversity of the production systems.

In this context, Life Cycle Assessment (LCA) studies of coffee products are needed to provide sound information on impact contributions and tracks to improve systems. LCA is the most widely used methodology, as its holistic approach covers the whole supply chain and several environmental impact indicators. However, LCA results are highly variable and there is still a lack of a comprehensive understanding on the relative impacts of the various management systems and trade-offs along the supply chains. This article presents an overview of the coffee LCA in the literature, investigating, first, the contrasted supply chains and system boundaries, then focusing on the agricultural stage and its specific challenges.

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[1] https://www.conservation.org/press-releases/2020/12/21/the-sustainable-coffee-challenge-sets-ambitious-2050-climate-goal viewed on 2023.2.1
[2] https://www.ico.org/trade_statistics.asp viewed on 2023.2.1

2. Material and methods

2.1. Literature review

We conducted a narrative review of coffee LCA studies in the literature. The searches yielded 144[3] and 162[4] outputs on the Web of Science and Google Scholar, respectively. We also added papers dedicated to carbon footprint analyses[5]. More than 75% of the studies were published in the last seven years. Primary checks on search errors[6] and duplicates[7], led to a consistent corpus of 205 papers and reports. Then, publications were first filtered according to their goal and scope, and studies eliciting no specific system boundaries or coffee LCA results were discarded (~65%). Most of those discarded studies did not display any LCA coffee results (40%), were out-of-the-scope (20%) or concerned recycling processes for coffee waste that entered the system with no environmental burden, i.e. not accounting for coffee production and processing (13%). The rest of the studies related to coffee machines and technologies (10%), chemical analyses (8%), socio-economic aspects including consumers' view on LCA results (6%), or were inaccessible (3%). Then, an in-depth review revealed a further bulk of LCA studies excluding the farm stage or flawed[8], which eventually led to a final corpus of 31 examined studies.

3. Results and discussion

3.1. Overview of the covered coffee systems and the LCA system boundaries

The great majority of reviewed studies investigated *Coffea arabica sp., n* = 21; four studies looked at *Coffea canephora sp.* Robusta; the remaining studies looked at both or did not specify (*n* = 6); none investigated *Coffea liberica sp.* Latin America was over-represented with 76% of all studied systems, including 21% for Brazil alone. Colombia, Costa Rica and Nicaragua were the next most investigated countries with 17%, 17% and 10%; respectively. Vietnam (7%) arrived fifth. Kenya, Indonesia, Uganda, Tanzania and a few more were studied in one or two studies each.

Almost half of the studies (42%) did not use primary data for the farm stage and relied on existing published datasets (mostly the ones on Brazil by Coltro et al., 2006, i.e. ~38% of studies based on secondary data[9], and from ecoinvent databases, i.e. 23%, which also greatly rely on Coltro et al., 2006). Finally, in terms of cropping system complexity, the great majority of primary datasets on the farm stage covered monoculture plantations without (48%) or with (30%) shade trees. The remaining 22% were more or less complex agroforestry systems (Figure 1).

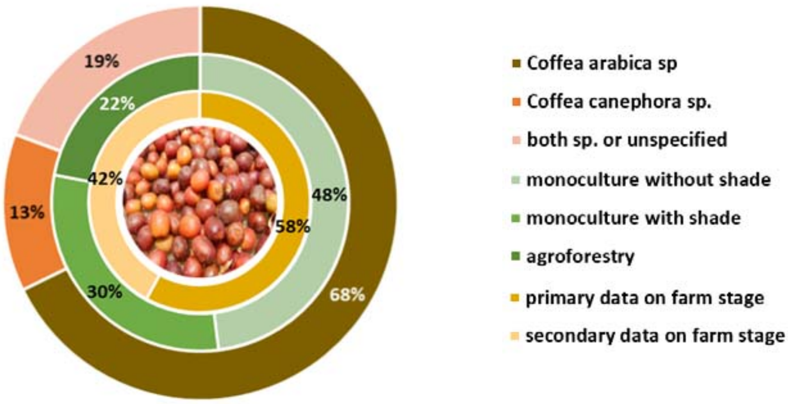

Figure 1: Overview of the reviewed coffee systems

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[3] coffee (Topic) AND lca OR "life cycle a*" (Topic): https://www.webofscience.com/wos/woscc/summary/add2e1e3-4e94-4f86-9f80-d5b001810b88-63f4f04f/relevance/1

[4] "coffee lca" OR "coffee life cycle assessment" OR "coffee life cycle analysis" OR "lca of coffee" OR "life cycle assessment of coffee" OR "life cycle of coffee" OR "life cycle analysis of coffee" OR "life cycle analyses of coffee" in English only and without including references. The least relevant pages, i.e. the second half of output pages, were filtered manually.

[5] "coffee carbon footprint" OR "carbon footprint of coffee" in Google Scholar, 88 outputs viewed on 2023.2.1.

[6] Errors in the title, key word interpretation, not English for the Scholar search, etc.

[7] Using a R script on DOI, with previous check on DOI record consistency.

[8] The most common source of error or uncertainty on the paper quality lies in the lack of explicit field emission modelling. In case of any doubt, we wrote to the authors to seek for clarification. When sufficient clarifications were given, studies were kept in the final corpus.

[9] Equivalent to 5/31 studies, with Coltro study itself apart.

In terms of system boundaries, about half the studied system included the consumption of the coffee drinks, mostly comparing at least three preparations (Table 1). Moreover, some studies presented results both at farm or processing-plant gate and after consumption, which provided results for 155 systems in total. Surprisingly, five studies defined the functional unit as kg green coffee, although those included secondary transformation and coffee consumption. Those results expressed per kg green coffee would be misleading, especially if extracted from the studied contexts and compared on the same functional unit basis with different system boundaries.

Likewise, at consumption level (cradle-to-grave), assumptions on coffee dilution and coffee waste varied across serving preparations and could lead to some confusion when comparing coffee drinks and their impacts. Some studies presented results with both functional units “per serve”, with various volumes, and similar adjusted “coffee volumes”, which made it possible to limit result differences strictly related to some dilution effect. However, despite focusing on differences across coffee preparations, none of these studies include organoleptic criteria within the coffee functional unit. For instance, espresso or filtered coffee are mostly compared on a drunk volume basis without any consideration of differences in strength or taste. As consumer taste preferences might be the main driver for the coffee preparation type, which in turn may influence the coffee final impact, it might be justified to considered some organolecptic properties (e.g. as it is done for fat and protein corrected milk, a common functional unit for LCA on milk). In this sense, some harmonisation could be done based on the actual coffee content (Table 1). Future studies could further investigate organoleptic properties associated to both the type of coffee and its preparation, and adjust the LCA calculation to the actual expected function (e.g. more focused on the taste or the cafeine effect).

Table 1: Numbers of studied systems for which at least one impact indicator is provided and overview of global warming impacts. Notes: LUC=Land Use Change; FU=Functional Unit

Total count = 155	Cradle-to-farm Gate without transformation	Cradle-to-1st transformation Gate (on- or off-farm)	Cradle-to-2nd transformation Gate	Cradle-to-Grave (=including coffee consumption)
Studied systems (count)	65	38	6	46
Studied system counts by functional unit	1ha.yr: 39 1acre.yr: 3 1kg coffee cherry: 23	1kg green coffee: 21 1kg parchment coffee: 14 1,000USD ha-outputs: 3	1kg non-packaged ground coffee: 1 1kg decaf blend coffee: 1 1MJ ethanol or electricity: 4	Drip/filter coffee: 22 Espresso coffee: 3 Instant coffee: 3 Pressed coffee: 4 Single-pod coffee: 7 Ground coffee: 2 1kg green coffee (although consumed as ground coffee): 5
Published global warming impact range: min-max ($kgCO_{2eq}$)	per ha.yr: -9,960 to 102,330 per kg fresh cherry: 0.145 to 1.82	per kg parchment coffee: 3.10 to 11.61 per kg green coffee: 0.015 to 10.52 per 1,000USD: 1,500 to 3,500	1kg non-packaged ground coffee: 0.56 1kg decaf blend coffee: 3.29 1MJ ethanol or electricity: -0.005 to 0.24	Drip/filter coffee (various FU): 0.013 to 0.796 Espresso coffee (various FU): 0.032 to 0.050 Instant coffee (various FU): 0.05 to 0.08 Pressed coffee (various FU): 0.01 to 0.06 Single pod coffee: 0.014 to 0.225 Ground coffee: 0.09 to 0.126 1kg green coffee (although consumed as ground coffee): 1.6 to 16 Overall per various FU: 0.01 to 16
Coffee-content harmonised global warming impact range (including differentiation between with or without LUC): min-max ($kgCO_{2eq}$)	per ha.yr (with LUC): -9,960 to 102,330 per ha.yr (without LUC): 109 to 10,220 per kg fresh cherry: 0.145 to 1.82	per kg parchment coffee (without LUC): 3.10 to 11.61 per kg green coffee (with LUC): 4.51 to 10.52 per kg green coffee (without LUC): 0.15 to 7.32 per 1,000USD: 1,500 to 3,500	1kg non-packaged ground coffee: 0.56 1kg decaf blend coffee: 3.29 1MJ ethanol or electricity: -0.005 to 0.24	Drip/filter coffee (per g coffee consumed): 0.002 to 0.02 Espresso coffee (per g coffee consumed): 0.002 to 0.007 Instant coffee (per g coffee consumed): 0.007 to 0.04 Pressed coffee (per g coffee consumed): 0.002 to 0.008 Single pod coffee (per g coffee consumed): 0.002 to 0.023 Ground coffee (per g coffee consumed: 0.013 to 0.018 Green coffee (per g coffee consumed): 0.002 to 0.016 Overall per g drunk coffee: 0.002 to 0.04

None of the studies included capital goods, which is in line with commonly used guidelines for agricultural production such as PAS2050 (BSI 2011), as justified by some authors. Capital goods are unlikely to contribute significantly to any impact unless focusing on the coffee machine production, which was the focus of a few LCA studies that were disregarded in this review as they did not include the coffee farm stage. It might be relevant to investigate those studies further in order to double check whether detailed LCI on machines and transformation infrastructures would lead to further discrepancies between coffee drinks prepared with various technologies.

A great majority of the studies (72%) did not consider or mention any co-product allocation. At farm level, residues from pruning or coffee pulp from wet processing may be recycled and used as mulch for instance. Among the remaining studies, there were three studies without primary data on the farm stage and relying on background database including system expansion for waste management and energy recovery; two studies from the same authors focusing on downstream energy production from cut stems by applying mass allocation between coffee and stems, then substitution; and one study applying economic allocation between coffee and pepper produced in the same plot. Given the diversity of systems, including agroforestry plots, and the potential diversity of coffee co-products, notably highlighted by numerous discarded studies only looking at coffee waste valuation (e.g. spent coffee ground), the lack of in-depth investigation on co-products stressed potential gaps in accounting for the specificities and discrepancies of coffee supply chains.

Finally, 60% of the studies covered more than just the global warming impact (or climate change). 48% were full LCA mostly relying on various versions of RECIPE LCIA method (20%), then CML2001, IMPACT 2002+ and ILCD (8% each), or TRACY (4%). The remaining 22% looked at both global warming and either water consumption or energy related impact indicators. Given the focus of the conference, we only further discussed results on the global warming impact category.

3.2. Overview of the global warming impact results

Global warming impact indicators varied greatly across studies depending on the system boundaries, but also within similar system boundaries. A summary of key results on global warmin impacts is given in Figure 2.

At cradle-to-grave level, when harmonised per g of coffee within the preparation, results varied from 0.002 to 0.04 kgCO_{2eq} (Table 1). Comparing on a similar volume basis with different coffee dilution ratios would not be suitable as long as the quality of the drink is not investigated.

About one third of the studies included import; 6 studies to European countries, 2 to North America, and 1 to Japan. In all cases but one (Vietnam), coffee was produced in Latin America. Across these studies, the global warming impact of the import transport varied significantly from a negligible impact in the case of ship transportation up to more than 73% in the case of export by aircraft. On average, when export was considered by ship the global warming impact was around a few percentage points (up to 15% but median around 3%) depending mostly on the relative contribution of the farm stage.

Among results without aircraft export, overall transport did not contribute significantly to the global warming impact. Main impact contributors were the production of green coffee (median 49%), brewing and washing accounting each for about 18%-20% (median), packaging (median 9 to 27%), and roasting (median 7%). The packaging contribution greatly differed in the case of single-pod or capsule use (27%) compared to various dripping systems (9%). Only one result provided the contribution of instant coffee processing, which was 14% in the exemplified supply chain. Despite the great diversity in the studied systems, there was only a 1-2 percentage point difference between the mean and median values across all contributions. The key role of the farm stage up to green coffee stressed the need to compare studies based on a similar coffee content. The contributions of brewing and washing stages were related to the amount of energy use and varied depending on assumptions related to coffee waste, including waste energy for keeping the coffee warm. When looking at energy or water use indicators, those stage contributions were even greater.

3.3. Detailed analysis of the coffee farm impacts and discrepancy sources

Focusing on the green coffee production, the discrepancy across results was even bigger especially when switching from positive to negative global warming impacts with carbon storage due to land use change (LUC). Overall, only three studies accounted for LUC and four others considered some biogenic carbon storage in the coffee plantations without modelling any LUC. Biogenic carbon stored in plantations, within coffee or other trees, should not be included in the carbon footprint as specified by the guidelines (e.g. PAS2050, GHG Protocol, etc.) unless considered within a proper long-term LUC modelling. Carbon storage in any stand may be accounted only in relative quantities compared to previous stands and providing a consistent time frame inline with a minimum time-averaged storage (at least over 20 years according to IPCC recommendations). In Noponen et al. (2013), carbon stocks were estimated and amortised over 9 years due to experimental constraints. The usual time frame for carbon estimates is at least 20 years, therefore the LUC modelling in Noponen et al. (2013) might be distorted compared to other studies and those results were not further discussed. According to the two other studies, the global warming impact of coffee following land use change to establish the plantations varied from 3.26 to 10.52 $kgCO_{2eq}$/kg green coffee (based on IPCC 2006 Tier 1). LUC contribution to the final global warming impact ranged from 39% to 74%, leading to an impact increase by a 1.3 to 3.9 fold. LUC change contribution is usually quite critical in agricultural LCA, particularly in the tropics where rainforest maybe converted to agricultural land. It can hence lead to significant differences between coffee systems given contrasted local development contexts and LUC history. Across the reviewed studies, negative global warming impacts might have resulted from distorted LUC modelling or inconsistent biogenic carbon accounting. Distortion might be due to varying choices across studies in terms of time frame and allocation parameters. Inconstistent biogenic carbon accounting might be due to flawn extrapolation in carbon stock changes or imbalanced accounting for carbon storage and release in LUC contexts.

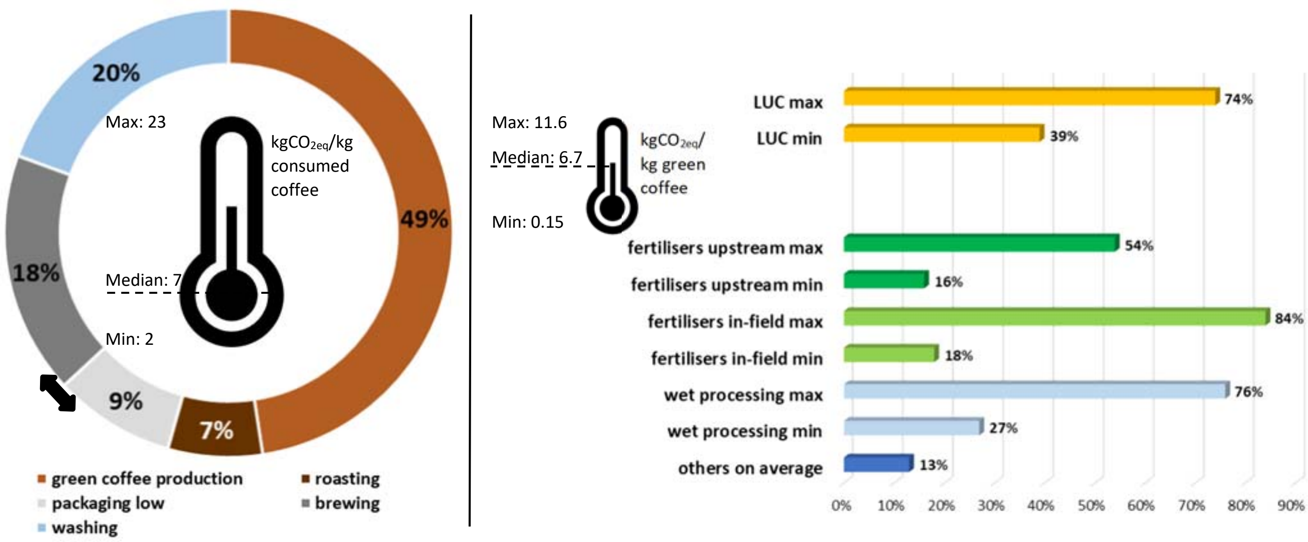

Figure 2: Overview of global warming impacts and main contributors; ***left***: cradle-to-grave and ***right***: cradle-to-gate
Notes: Export and instant coffee are excluded from the cradle-to-grave displayed contributors and max values, and the contribution of packaging would be higher in the case of single-served pod. Green coffee also encompasses parchment coffee here.

LUC apart, overall main contributors to the global warming impacts were synthetic fertilisers. In studies providing aggregated information, greenhouse gas emissions from both fertilisers upstream (manufacture) and downstream (field emissions) contributed to 16-78% of the impact. When disaggregated, fertilisers upstream emissions contributed to 16-54%, downstream ones to 18-84%. As commonly observed in agricultural LCA, the relative impacts of fertilisers were related to the intensification of the systems in terms of both inputs and outputs. Overviewing key agronomic parameters, cropping systems were highly contrasted (Table 2). In comparison, the respective contributions of fuel for transport or field operation, pesticides manufacture or irrigation were marginal. Only 8% of the studied systems included irrigation (in three studies), hence the reviewed contributions of irrigation were likely not representative of constrasted intensities in irrigation.

Table 2: Variations in key agronomic parameters across reviewed studies

	Mean	Median	Min	Max
Planting density (coffee trees/ha)	4,128 (±40%)	4,500	150	10,000
N fertilisers* (kg/ha)	205 (±64%)	177	0	842
Fresh coffee cherry yield (kg/ha)	5,696 (±46%)	6,081	628	13,605
Parchment coffee yield (kg/ha)	1,094 (±56%)	1,032	126	2,387
Green coffee yield (kg/ha)	1,505 (±77%)	1,245	225	5,386

*Not all studied displayed the detailed amount for each fertiliser types nor the N content of organic amendments applied. Total N fertiliser estimates are likely underestimated.
Standard deviations to the means are given into brackets

Although fertilisers-related field emissions were consistently modelled across the studies[10] (mostly based on IPCC 2006 and derivatives), there was a lack of transparency and consistency regarding the residues-related ones. Only Noponen et al. (2012, 2013) explicitly quantified emissions related to residue decomposition and those contributed between 9 to 42% of the global warming impacts across systems. Overall, there was a lack of details and transparency on the various types of residues or other organic amendments decomposed in the field. In connection with the above-mentioned lack of details on the generation and handling of co-products along the value chain, more attention should be paid in quantifying on-farm or off-farm residues decomposition and emission profiles, so as to make sure that the quantification of emissions is complete, as well as to check whether synthetic fertilisers were or could be substituted.

Emissions related to wet processing were not consistently modelled across studies, which raised a critical issue as they were quite significant contributors, from 27% to 76% of the global warming impacts (Killian et al., 2013; Maina et al., 2016; van Rikxoort et al., 2014; Van Rikxoort et al., 2013).

Finally, despite the description of quite complex coffee systems with shade trees or in agroforestry plots, little attention was paid to consider this complexity, and potential allocation issues, in the quantification of greenhouse gas emissions and global warming impacts. First of all, very few studies considered the perennial cycle of coffee trees in the modelling of agricultural practices and related emissions (which should be averaged along the whole cycle). Then, most studies focused on the shade trees to estimate carbon stock within the biomass but did not consider inputs/outputs allocations among crops and trees. There were a few exceptions, though, with some more systematic and holistic studies (e.g. Acosta-Alba et al., 2020) or at least studies eliciting allocation ratios among associated crops (e.g. Basavalingaiah et al., 2022).

4. Conclusion

Across the various coffee production systems, there is a range of global warming impact drivers whose relative importance depends on the system configurations. At the coffee cup level, the main impact drivers were the coffee production, then brewing and washing. At the farm level, main impact drivers were LUC and fertilisers. Biogenic carbon accounting and LUC modelling were not consistently applied. Likewise, there was a great variability in the modelling of emissions from wet processing.

Whereas the impacts of intensified monocropping coffee systems can be quite easily assess individually, those of more complex systems are more intricate and superficially addressed. There are still gaps in the

[10] It should be noted that it was a robustness criterion to actually retain a study.

understanding of all consequences, positive and negative, of functional biodiversity and species interaction within complex multi-cropping systems such as coffee agroforestry systems. Hence impact trade-offs across systems along a bio-complexification gradient cannot yet be fully deciphered. The system diversity extends beyond the plantation up to processing, and the whole continuum eventually affects the organoleptic properties of the coffee. More data is still needed to uncover the full spectrum of coffee peculiarities and impacts according to their origins.

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Mining Abstracts

Prospective, spatially-explicit LCA of global copper mining considering uncertainties in regional supply

Thursday, 20th July - 13:30: Mining

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Strategies to decarbonise the global economy through electrification and renewable energy deployment are contributing to rapid growth in copper demand. This creates a trade-off between the avoided impacts associated with copper's uses and the environmental impacts associated with copper mining. Prospective – or scenario – based life cycle assessment (LCA) is a valuable tool for understanding these trade-offs and identifying opportunities to mitigate impacts. However, there is considerable uncertainty inherent in the results of any prospective LCA. For instance, the regional supply mixes embedded in inventories can be highly influential over impact assessment results for some product categories. Future regional supply mixes for mined products are particularly uncertain because mineral deposits are depleted by the extraction process, leading to mine closures (or abandonment) and simultaneous development of new mines elsewhere to fill the supply gap. From a practical perspective, this can create a reluctance to use spatially-explicit impact characterisation models within prospective LCA of mineral products, despite these emerging as the state-of-the-art in some impact categories such as those related to water and land-use change.

This article and presentation provides an overview of extensions to the Primary Exploration, Mining and Metal Supply Scenario (PEMMSS) model that will allow prospective, spatially-explicit LCA to be conducted for mineral products. As an initial case study, the direct land-use and biodiversity impacts of future copper mining were modelled to demonstrate just one aspect of the complex environmental trade-offs and uncertainties associated with decarbonisation.

Future greenhouse gas emissions from metal production with implication for climate goals

Thursday, 20th July - 13:30: Mining

Dr. Ryosuke Yokoi [1]***, Dr. Takuma Watari*** [2]***, Dr. Masaharu Motoshita*** [1]

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Metals play an essential role in human life, while metal use is associated with not only metal depletion but also environmental concerns. To discuss strategies towards sustainable metal use with lower environmental impacts in line with climate goals, quantifying future environmental impacts from metal production and exploring effective measures for alleviating the environmental impacts are essential. Therefore, we estimated the global greenhouse gas (GHG) emissions from the future production of six typical metals (aluminum, copper, iron, lead, nickel, and zinc) under the five shared socioeconomic pathways (SSPs) for 2010-2100 and compared the results with a GHG emission reduction target (2°C target). In addition, we explored the influential parameters of metal cycles to reduce the environmental impacts by scenario analysis.

We show that trends for GHG emissions from metal production are significantly different among SSPs, while the 2°C target will not be achieved for the metal sector under any SSP, mainly due to the increase in GHG emissions in the early 21st century in middle-income countries. This suggests that substantial efforts to reduce GHG emissions are required in addition to the transition to the sustainable socioeconomic pathway. From a short-term perspective, lowering the per capita in-use metal stock level and GHG emission intensity of metal production is identified to be effective. From a long-term perspective, improving the recycling rate will also be an effective way. However, our analysis shows improving a single parameter is expected to be insufficient for achieving the 2°C target. Given that GHG emissions from metal production will increase mainly in the early century and improving parameters cannot be achieved promptly, implementing multiple measures immediately with international cooperation, as well as following the sustainable socioeconomic pathway, is essential for sustainable metal use in line with the climate goals.

The Initial Assessment of Social Life Cycle Value Based with Consideration of Empowering Locals by Integrating JBG's Community Empowerment Program with its Plant Conservation Initiative: the Sasirangan Eco-printed Handicrafts Case Study

Thursday, 20th July - 13:30: Mining

Mrs. Dewi Permatasari [1], ***Mr. Rizali Rakhman*** [2], ***Mr. Elisa Weber Siregar*** [2], ***Mr. Rasmat Riady*** [2], ***Mrs. Afifah Zabarij Mustafifah*** [2], ***Mr. I Gede Widiada*** [2]

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In carrying out its community empowerment program, PT Jorong Barutama Greston (JBG) places a heavy emphasis on a stakeholder engagement approach, collaborating closely with people living around its operational area in South Kalimantan. Taking the social problems faced by the local community as a jumping-off point to design its programs, JBG is attempting to lend its support to locals beyond the caritative approach, by also equipping locals with the skills and capacity they need, in alignment with the creating shared value (CSV) principle, seeking to spur a sustainable social innovation. In this paper, we will examine the company's community development program as part of starting point to conduct social life cycle assessment in the future.

In 2021, the program involved 20 craftswomen as its beneficiaries. In terms of environmental impact, the program supports a social life cycle principle and it has brought down the carbon emission from vehicles used to transport synthetic ink by 0.05 ton of CO_2-eq per annum (the organic ink is produced near the workshop of these craftswomen thus does not produce as much carbon emission during its transportation).

The value creation of this innovation lies in its transformation of the Sasirangan product value chain, to make it more sustainable and advanced. Now, the Sasirangan product has a competitive advantage thanks to its use of environmentally friendly organic material, emits less carbon in its production process, and contributes less toxic and hazardous waste for the local environment. This has impacted local people's health greatly, while at the same time also succeeding in boosting local people's knowledge and skills about environmentally friendly handicraft products. This innovation can also be replicated in other community development programs, involving other beneficiaries. This social innovation program also aligns with the sustainable development goals (SDGs) number 1, 5, 8, 12 and 17.

Adopting Social Life Cycle Principles to Program Implementation of a Cultural Tourism Approach to Social Empowerment in West Kutai: a Case Study of the Lamin Lou Bentian House

Thursday, 20th July - 13:30: Mining

Mrs. Dewi Permatasari [1], ***Mr. Jones Silas*** [2], ***Mr. Sony Herlambang*** [2], ***Mr. Lukman Malik*** [2], ***Mrs. Sri Handayani*** [2], ***Mr. Budhi Cahyono*** [2], ***Mr. Wahyu Harjanto*** [2]

1. *Environmental Professional & Sustainability Practitioner,* ***2.*** *Trubaindo Coal Mining*

Trubaindo Coal Mining is a subsidiary of the PT Indo Tambangraya Megah, Tbk., which works in the energy sector, specifically coal mining. As a response to challenges related to the national energy needs, which also intersects with the expansion of corporate social and economic responsibility to locals who live in the vicinity of its operational site especially at the final stage of the mining closure, TCM has been pioneering cultural programs based on local wisdom to support local people's socio-economic-autonomy, while at the same time advancing the region's economy. To accomplish the goal, TCM is carrying out Societal Empowerment initiatives to attain its mission of Creating Shared Value (CSV) which foundation lies on cultural values as the local community's social capital to attain autonomy in the future.

TCM has carried out a number of community empowerment programs which have been developed to answer to the socioeconomic challenges and vulnerabilities faced by the local community members. TCM has also adopted sustainability principle as the core of its program development process, so environmental preservation and welfare improvement become the foundation of each community empowerment program it implements on its direct beneficiaries but also the West Kutai community in general. This project specifically describes TCM's success in pushing a tourism development program to empower local community, using a plot of land spanning 1,600 square meters which has been granted by the West Kutai community.

In the long haul, the Lou Bentian house will be a center for tourism, cultural, economic and social activities, as well as education and community development. TCM sets its sights on turning Lamin Bentian into a mining site closure exit strategy best practice. We believe that the adoption of social life cycle principles plays an important role of the success of the program currently and in the future development.

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