Environmental impacts of circularity in the built environment: How do system boundaries affect decision support?

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ARTICLE INFO

ABSTRACT

Life-Cycle Assessments (LCA) have become a common tool of decision support in the built environment regarding environmental impacts. The goal of this study is to investigate the influence of different LCA approaches and system boundaries on decision support regarding circular design strategies. Three different ways of applying attributional LCA (ALCA) and one consequential LCA (CLCA) are assessed using a case study of a wooden beam with either virgin or reclaimed timber. While the conclusions are not readily scalable, the case study's results indicate that using reclaimed material is environmentally beneficial when applying ALCA. However, when applying CLCA the reclaimed material performs worse than the virgin material. This highlights the potential pitfalls of scaling up solutions based on LCA results without considering the broader consequences. While the current ALCA approach is useful for declarations, it may not provide comprehensive decision support for driving a transition in the built environment.

1. Introduction

Due to the increasing focus on the environmental performance of buildings, the number of environmental performance assessments in the built environment (BE), published every year, is rising. Life Cycle Assessment (LCA) has become the most common method of making such assessments, and it has recently been included in many national regulations for new constructions in, e.g., Denmark, the Netherlands and Finland (SBST; Rijksdienst et al., 2023; Finnish Government - Ministry of the Environment, 2023).

Most LCA studies in the BE are performed by following European norms EN15978 for building-level assessments or EN15804 for product-level assessments (EN and 15804:2012+A2:2019, 2019; EN 15978, 2011). These assessments apply a clear-cut system boundary and use defined life-cycle modules covering extraction and production (A1-A3), transport (A4), construction (A5), the use stage, covering activities such as maintenance, replacements and operational energy and water consumption (B1–B7), and End-of-Life (EoL) phases such as demolition,

transport, and waste processing and disposal (C1–C4). Additionally, benefits beyond the system boundary covering potential avoided impacts through, e.g., reuse, recycling, or the recovery of materials, may be declared separately in module D. The current attributional approach in the BE is referred to as the BE LCA approach from here on.

While LCA is a method to assess the environmental impacts, many strategies to reduce these impacts in the BE exist, e.g., increasing the energy efficiency in the use phase, or using less carbon-intensive materials for construction. Another strategy is represented by the concept of the Circular Economy, where one of the aspects is the value retention of materials, defined as the 'R-strategies' (Kirchherr et al., 2017). Depending on the extent of strategies referred to 4-R or 9-R (Kirchherr et al., 2017), they include design actions such as Reduce, Reuse, Remanufacture, Recycle, Recover, etc.

Value retention strategies may be applied throughout the entire life cycle of a product or building. However, how this is represented and how the design/retention strategies are accounted for in an LCA, have been discussed extensively (e.g. (Joensuu et al., 2022; Van et al.,

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2022/12),). A particular focus is the attribution of impacts when multi-cycling a product or material (e.g. (Eberhardt et al., 2020; Eberhardt et al., 2020/11; De Wolf et al., 2020/10; Mirzaie et al., 2020; Garcia et al., 2020),): i.e. which life-cycle should bear the responsibility for which impacts? Recent developments also investigate the declaration of the potential climate benefits, represented as a 'carbon handprint' (Eberhardt et al., 2023).

However, LCA is not limited to the way it is described in current standards and regulations for the BE. It may utilise multiple approaches, including attributional LCA (ALCA) and consequential LCA (CLCA), as well as using different value choices, such as allocation methods and system boundaries.

When considering a transitional agenda, the relevant decisions may often have implications beyond the specific product system as modelled in an ALCA, as they affect changes in the value chain, which might affect other value chains. Making a design decision in one value chain may look environmentally beneficial for the product itself, but on a larger scale, e.g., in another value chain, the decision may induce increased impacts. Thus, the change may prove to be worse environmentally for society.

Instead of trying to develop the BE LCA approach further, e.g., by adding multi-cycling approaches, it may be useful to consider other existing LCA approaches or modelling of value choices. This could take the form of alternative arrangements of the system boundary and the burden-sharing principles, still using an ALCA, or could use a consequential approach, modelling the consequences of a design decision. Where the former may reflect on the burden-sharing of a material across product cycles, the latter may reflect indirect consequences beyond the assessed value chain. Both are relevant in assessing circular design strategies.

However, in the LCA literature there is a lack of comparative studies of how different methodological choices are applied in the BE. One of the few examples is a comparative study by Fauzi et al. (2021), who compared the methodological aspects of applying ALCA and CLCA. The authors found that the dominant contributor to life-cycle impacts in climate change shifted when using ALCA and CLCA respectively. Additionally, they argue that adding CLCA to ALCA as decision support can provide additional insights into the impacts and hidden consequences.

A recent review by Andersen et al. (2022) showed that, when discussing the Circular Economy in the BE, the information about LCA approaches used in answering questions regarding environmental performance are neither transparent nor consistent, nor do they follow the general recommendations (Joint Research Centre - Institute for Environment and Sustainability, 2011) on the appropriate scale of decision support derivable from the different types of LCA. Thus, although most LCA assessors are aware of the existence of recommendations regarding the valid decision-making context associated with the types of LCA (Joint Research Centre - Institute for Environment and Sustainability, 2011), it appears that most of the assessments stick to the BE LCA approach (Andersen et al., 2022). The results of these assessments are most often *correctly* used to judge the environmental performance of a circular design strategy seen from a single building or product perspective. However, they are often *incorrectly* used to make recommendations on how society benefits from it. Thus, they scale the conclusion from product to industry or societal recommendations but disregard the consequences of the affected value chains.

1.1. Hypothesis

The main hypothesis of this study is that the currently predominantly used BE LCA approach is not sufficient for evaluating the environmental impacts of circular product-design strategies. Circular value chains are highly dependent on surrounding markets and material availability, whereas, in the traditional take-make-waste economy, an inherent underlying assumption is the possibility of unlimited consumption. When used for decision support, the BE LCA approach may overlook the environmental implications beyond the specific value chain. Thus, conclusions may not apply to decision support on a larger scale.

The study aims to investigate how the selection of the LCA approach, system-model value choices, and system boundaries of the LCA study can affect the decision support and magnitude of the potential impacts. By introducing a case study, the hypothesis is tested by investigating if, and how much, a suggested circular (i.e. reclaimed material) design approach is environmentally beneficial compared to the virgin alternative, and whether this changes when shifting the applied LCA approach and model.

In the context of this paper, 'approach' refers to either ALCA or CLCA; 'system model' covers value choices, such as how multifunctionality is handled (i.e. using allocation or system expansion); and 'system boundaries' show the included processes and stages of the life-cycle.

In this paper, the circular economy is 'merely' seen as one of many tools used for framing a means to an end. Circular design strategies may thus cover many variations in the aim of high-value retention. However, for this study, the exemplification of decision support implications will be based on the practice of reuse.

2. Theory and background

LCA theory and methodology have been developed over many years and have been applied for about half a century (Bjørn et al., 2018; Guinée et al., 2011). Today, many interpretation variations and application methodologies have been developed and presented, but an overall distinction between the two approaches still applies to attributional and consequential LCA.

2.1. LCA approaches

A common definition is that ALCA aims to allocate a specific share of environmental burdens to an assessed product or service, whereas CLCA addresses the consequences or reactions in the global system or society (i.e. captures the market dynamics) caused by the production and/or use of a given product or service (Ekvall et al., 2019). Besides these two distinct areas of interest in assessment, the approaches may be supplemented by various ways of handling multifunctionality (how to handle the allocation of processes yielding more than one functional flow, e.g. through partitioning or system expansion). Furthermore, the type of data to be used varies: ALCA uses average data, whereas CLCA uses marginal data (Ekvall et al., 2019).

When looking at theoretical correctness, some will argue that CLCA is more accurate. However, ALCA might be more precise, and thus practitioners are encouraged to always make an informed choice of which approach to apply. This might also be the reason why, over time, many environmental declaration systems have been interpreted as ALCAs with allocation rules. While these interpretations might make environmental data for decision support more accessible to a wider audience, they also lead to generic uses and use in legislation. In many cases, this results in the 'use' of LCA by 'practitioners' who are not necessarily able to make an informed choice of LCA approach with regard to the decision-making context. They might not even be aware of the need or possibility to make this choice.

2.2. System models and data

A system model is a collection of modelling choices applied to a database, defined by several distinctions. The term 'system models' is most familiar through the Life-Cycle Inventory (LCI) database ecoinvent v3 (Wernet et al., 2016). Three characteristic distinctions define a

system model.

Firstly, the way by-products (multifunctionality) are handled can be either by allocation, as mainly done through ALCA, or through substitution, as mainly done through CLCA. Allocation can happen through the division of impacts of the different co-products of an activity, preferably based on the physical aspects, for example. Substitution is carried out by crediting other products or value chains substituted by the by-product of an activity.

Secondly, ways of considering the burdens of waste treatment and how they are attributed to different product systems differ. One way implies that the burdens be borne by the waste producer. This approach is modelled in the so-called 'cut-off' system models and the 'substitution, consequential' system model. In these models, the cut-off by classification also considers recyclable materials as burden-free, i.e. recyclable material will not cause a burden for the producer. However, this is not applicable for the cut-off in EN15804, where end-of-waste criteria apply (ecoinventecoinvent, 2021). Another way is sharing the burdens between the product in the value chains or activity, which is the method applied in APOS (At Point Of Substitution).

Thirdly, suppliers and the composition of consumption and market mixes differ. In cut-off and APOS (attributional models), all suppliers within the boundary of a market are fed into the consumption mix. In contrast, only the marginal suppliers (i.e., suppliers who can adapt to changes in demand) are fed into the consumption mix in substitution (consequential).

All system models in ecoinvent are based on a multi-output, unallocated, unlinked, gate-to-gate unit process representing the closest possible version of the actual physical flows. This undefined database ('baseline') then uses different 'linking rules' to reflect the different modelling approaches. 'Linking rules' represent value choices. In this way, the ecoinvent system models called cut off by classification, cut-off by EN15804, APOS and substitution (consequential, long term) are derived from the undefined database (Wernet et al., 2016; ecoinventecoinvent, 2022; ecoinventecoinvent, 2020a; ecoinventecoinvent, 2020b).

2.3. System-model value choices

Depending on the applied modelling choices, the results for 'the same unit process' can vary, underlining the importance of choosing system models with awareness. The differences are caused by the different assumptions applied throughout a system model. This applies not only to the single process assessed but also to the process assumptions regarding the upstream and surrounding value chains, which add up to significant variations in some studies (ecoinventecoinvent, 2022).

The cut-off model (by classification (Wernet et al., 2016) and EN15804 (Ioannidou et al., 2021; Ciroth, 2022)) incentivizes the use of secondary materials, i.e. the secondary use or life cycles. It does not incentivize the first cycle or waste producers to maximize the recycling or reuse of waste produced, as no credit is given to the first cycle (the polluter pays' principle). This attribution is determined by categorization within the activities, as flows are allocatable, whether waste or recyclables: i.e., all flows/products are visible in the activities, but the classification decides how they are accounted for. In EN15804, the cut-off point is set where a waste flow reaches the end-of-waste (EoW) criteria. However, through the introduction of module D, crediting substitution benefits at the EoL is also possible.

The cut-off system model, by EN15804, was released with the ecoinvent database v 3.8 (ecoinventa), to enable modelling with aligned data for Environmental Product Declarations (EPDs). It enables having a harmonized coupled calculation of inventory indicators (according to EN15804+A2), with a focus on the resource indicators as prescribed in the EPD standard. The cut-off EN15804 system model is attributional and uses allocation and average suppliers.

The APOS incentivizes waste producers to assess the recycling or reusability of the waste, as the impacts of useful treatment products can be partially allocated to other product systems/activities. This also means that the use of secondary materials is not necessarily incentivized. In practice, the main differences between cut-off and APOS system models appear in waste treatment and recycling modelling (ecoinventecoinvent, 2020a). The point of substitution system boundary is, by ecoinvent, defined as being the first downstream activity after a treatment which produced a valuable product.

While in the foreground system of the study the approach and definition of ALCA is using allocation to avoid system expansion, by default the APOS system model challenges this, as it uses system expansion to avoid allocations within treatment systems. This contradiction is also touched on by ecoinvent, mentioning the scope of study causing the impossibility of following APOS in every foreground system (ecoinventb). In practice, the suppliers in APOS become diverse and complex, and thereby hard to read, with a variety of input and a mixture of supply chains. Therefore, the datasets available are averaged and weighted (ecoinventecoinvent, 2020b; ecoinventc).

In the substitution (consequential) system model, the reference product of the activity will be burdened with all the impacts of the activity. However, the benefits of by-products, substituting other products or supply chains, are credited, i.e., subtracted, from these impacts. Additionally, the consequential model only considers unconstrained suppliers, i.e., suppliers who can adapt and/or respond to changes in demand. This means that only unconstrained products are reference products, not by- or waste-products, as these are seen as constrained. Instead, by-products are credited with the marginal supplier of that product (i.e., a producer of the by-product having the product flow as a reference flow (primary production)).

If only constrained suppliers are available, the market is constrained. This means that the products cannot be replaced by an alternative production route, which is answered by the system model, by reducing a product's use in a marginal consumption activity. The marginal consumption activity is the activity that can change to an alternative means of production, i.e. supplier or product supplied (ecoinventecoinvent, 2021; ecoinventecoinvent, 2022).

3. Methods

3.1. Goal and scope

The goal of the study is to determine whether the decision support regarding the environmental benefits of a circular product design changes when applying other system boundaries than the BE LCA approach. The results are not to be compared with existing data, e.g., EPD data, nor are they to be misused in extrapolating the conclusions across assessment scales. The study uses the BE LCA approach, as prescribed by EN15804+A2 (EN and 15804:2012+A2:2019, 2019), as a benchmark for decision support. It compares these with potentially changed decision support when applying extended attributional and consequential approaches and modelling principles.

The scope of the study is a prototype of a timber beam in the Danish construction industry. The geographical context and market mechanisms both take their starting points in Danish contexts.

The LCAs are modelled in openLCA v. 2.0.0 (GreenDelta) using background data from ecoinvent 3.9.1 (Wernet et al., 2016). The database used for EN15804 is an add-on for ecoinvent v3.9.1 and was developed by openLCA (Ciroth, 2022).
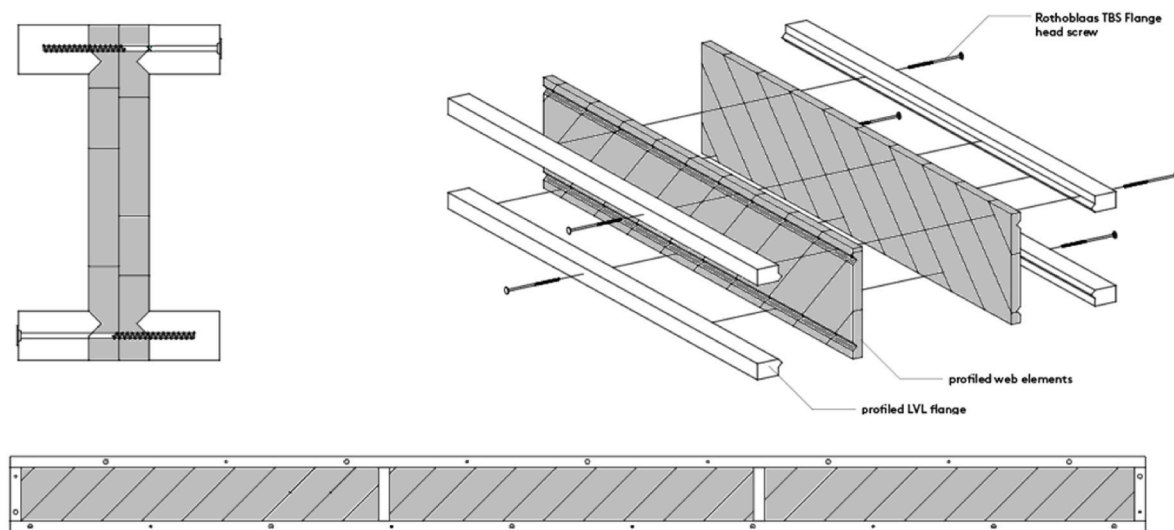
Fig. 1. Business-*as*-usual (BAU) beam section, isometric, and elevation. Elements marked in grey are replaced with reclaimed timber in the Re-use (RU) beam design.

3.1.1. Case study

The case study is a prototype of an I-section timber beam assembled from flanges and web elements as seen in Fig. 1. The beam consists of three main components: web elements (from primary or secondary solid wood), flanges (from primary, solid wood) and screws (galvanized carbon steel) (see Fig. 2).

The original beam design, referred to as a Business as Usual (BAU) beam, has been adapted to utilise reclaimed material and is referred to as Re-use (RU) beam. Each of the flanges is profiled from a regular rectilinear section into a notched geometry. A corresponding negative of this notch is milled in each of the flange elements, creating a precise 'locking' interface between the web and flange elements. Screw fastenings are added intermittently to keep the flanges together. This 'semi-standardizing' approach aims to use material that would otherwise not be considered suitable for producing load-bearing components.

The following information relates to the material required to fabricate two different versions of 4 m long beams. The RU beam's dimensions have been adjusted to match the BAU beam. The design differences of the beam are represented in the material feedstock for the web elements and the material efficiency.

Biogenic carbon sequestration has been omitted from the study. The rules and frameworks for the modelling of biogenic carbon are not within the scope of this study. An extended argument of the omission can be found in the supplementary material, Appendix A.

3.1.2. Declared unit

A declared unit is used, as the beam's final use context is not defined. The declared unit is a wooden beam mechanically assembled from profile elements and smaller pieces of wood, as shown in Fig. 1. At this stage the designs are not to be compared to other products or designs, e. g., an I-beam of steel, and the function of the designs is presumed to be equivalent.

3.1.3. System boundaries

The LCA models are divided into two overall product systems: upstream and core processes (UCP), and downstream processes (in the following models and scenarios abbreviated 'DP'). An overall scenario distinction is whether the beam is produced with web elements from either virgin wood (BAU Beam) or reclaimed wood (RU Beam).

The UCP of the case-study beam design considers the resources used, these being differently accounted for in the system boundaries (i.e., inclusions) and different market considerations. The DP consider the treatment of waste wood, modelled with two variations/scenarios: a conservative modelling considering 100% incineration and a statistical

modelling using the treatment distribution according to Eurostat. The latter considers the treatment distribution as ~81% recycling, ~19% incineration and <1% landfill (Eurostata; Eurostatb).

Additionally, considering the approach from EN15804, results both including and excluding crediting from module D have been generated. The End-of-Life formula from EN15804 has been applied, addressing the net impact where benefits cannot be counted twice. This means that, as secondary materials enter the system burden-free, no crediting is allowed in module D. See further in section 3.3.2.

The study only assesses the embodied impacts and excludes the use phase, as the latter is not described and assessed at this point.

3.2. The LCA method

The characteristics of the LCA approach (ALCA/CLCA) and how they conform with different modelling value choices such as multi-functionality have been questioned over time. However, for this study, the definitions given by Ekvall (Ekvall et al., 2019) are used, coupled with the system models presented in the Life Cycle Inventory (LCI) database ecoinvent v3 (Wernet et al., 2016). The system models in ecoinvent present different variations of the database, each adhering to specific methodological value choices and calculation rules. The basic theory of this is presented in sections 2.2 and 2.3, and the specifics on how they apply to the case study are presented in section 3.3.3.

3.2.1. Modelling approach and system models

The four system models are held separately in open LCA while modelling, using unit processes for disaggregation possibilities. As far as possible, the foreground systems in the models are the same, but system boundaries are adjusted to follow the modelling theory of the respective system model's value choices.

3.2.1.1. Baseline model.
The BE LCA approach, following EN15804, is the starting point for the modelling of the product using either primary or reclaimed wood. According to BE guidelines on LCA, complementary Product Category Rules (cPCR) should be followed for material-specific calculation rules. A cPCR from CEN on wood (EN 16485:2014) (DS/EN et al., 2014) is used as the reference for modelling decisions.

The CEN cPCR mainly refers to the main PCR, EN15804, with few specifications relevant to the case study at hand.

The system boundaries are modelled according to the cut-off principle, where all impacts regarding waste handling are attributed to the waste producer, i.e., first cycle, entailing secondary material to be 'provided burden-free'.

3.3. Life Cycle Inventory (LCI)

The full LCI of the study, assumptions and case-study descriptions are available in the supplementary material, Appendix B. Only key values and modelling differences are presented below. First, the model according to BE LCA guidelines and system boundaries is presented and divided into the UCP and DP. Then the system model adaptions and differences are presented.

3.3.1. Upstream and core processes (as modelled according to BE LCA)
The UCPs cover the life-cycle phases that would equal modules A1-A3 in the BE LCA approach. The UCP of the beam includes raw-material sawn wood as input for both web elements and flanges (A1, from background database); transport of the wood to the processing site (A2); and processing of the wood into suitable sizes for beam assembly (A3), which also produces off-cuts and sawdust waste (product by-products). The screws for assembly are modelled from the background system and transported to the assembly site. Table 1 and Table 2 summarize the key input values.

3.3.1.1. Energy. The energy used for processing the wood is based on measured processing minutes. The beam prototype is made manually. Therefore, the efficiency of both machinery and time consumption might have been higher in industrialized production.

3.3.1.2. Transport. Transport is assumed by EURO trucks, with an estimated market average of EURO4-6. Statistical details and derivation of these assumptions are described in the supplementary material, Appendix B.

Table 1
Inventory for web elements, used for input parameters variables, DU: 4 m beam.

		BAU beam	RU beam	Unit
Web elements	Material	Sawn timber	Reclaimed wood	–
	Density	450	450	kg/m3
	Jointing	18	0	minutes
	Planing	4	0	minutes
	Cutting	30	30	minutes
	Routing	48	144	minutes
	Material stock	0,063	0,038	m3
	Volume in one beam	0.052	0,028	m3
	Yield	*82,5%*	*73,7%*	*%*
	Small solid off-cuts	11,1%	18,4%	%
	Sawdust off-cut	6,4%	7,6%	%

Table 2
Inventory for flanges and screws, used for input parameters variables, DU: 4 m beam.

		BAU beam	RU beam	Unit
Flanges	Material	Solid timber	Solid timber	
	Density	450	450	kg/m3
	Cutting	5	5	minutes
	Profiling	40	0	minutes
	Material stock	0,033	0,033	m3
	Required volume	0.027	0,03	m3
	Yield	*81,8%*	*90.9%*	*%*
	Small solid off-cuts	4,76	4,76	%
	Sawdust off-cut	13,4%	4,44%	%
Screws	Type/Material	TBS 6140/Carbon steel		
	Quantity per beam	36		pcs.
	Weight	15,8		g/pcs

3.3.2. Downstream processes (as modelled according to EN15804)
The downstream processes (DP) cover the end of life (EoL) stages, referred to as modules C1–C4, along with module D in the BE LCA approach. There is no difference in whether the BAU or RU beam is assessed, except in the amount of material used in the design, as the handling and EoL processing must be presumed equal to the beams. In the case studies, only modules C3, C4 and D, are considered, as transport is presumed secondary to the hypothesis of the study. Additionally, crediting of secondary input materials at EoL is not allowed in the EN15804 system model according to the EoL formula in EN15804. Further details are discussed in the supplementary material in Appendix C.

Two modelling choices to determine the EoL waste treatment are applied, one following the conservative assumptions from the Norwegian cPCR, and the other considering the waste statistics of the national waste handling (Eurostata; Eurostatb). The calculations are described further in the supplementary material in Appendix C.

3.3.3. Modelling differences
Due to the different methodological choices for multifunctionality and allocation, the foreground system of the products changes. The adaptions from transferring the system model according to EN15804 to cut-off, by classification, APOS and consequential, are shown in the conceptual diagrams in Figs. 3 and 4. Additionally, by interchanging the background databases, these changes also occur in upstream and market data. When changing from attributional to consequential system models, the data are also modelled with marginals instead of averages. Where relevant, the background data used are 'market' processes, i.e., electricity, transport and avoided products. All applied datasets and processes are presented in the supplementary material in Appendix D.

3.3.3.1. Cut-off, by classification. The two cut-off system models, EN15804 and by classification, use the same methodological value choices to a great extent (see Fig. 3). The main difference is the modelling boundary to EoW, along with the prescribed option to model crediting (module D) for an EPD.

3.3.3.2. APOS. Due to the simplicity of the foreground system, no directly multifunctional product processes occur. However, in processing the wood some off-cuts and sawdust occur, which are presumed to be incinerated (due to the fraction size). Where the incineration process is attributed fully to the processing in the two cut-off models, APOS allows for the allocation of the upstream and treatment burdens to heat and electricity produced during incineration, under the assumption that this 'waste' is sold, for instance, as an allocatable product (as e.g. done in (Bergman and Alanya-Rosenbaum, 2017)). Similar options may or may not be allowed for the EPD system, i.e., internal production crediting, but due to ongoing discussions, this has been omitted from the two cut-off models.

Off-cuts from processing are modelled as waste output 'waste wood, untreated', with a treatment process for Municipal Solid Waste Incineration (MSWI). Heat and electricity are modelled as co-products from the processing process, using economic allocations (based on ecoinvent € price indexes).

- Web elements (€ = 160/m^3 wood)
- Electricity (€ = 0,107/kWh, energy produced in MSWI = 1,3 MJ/kg waste (ecoinventd))
- Heat (€ = 0,01060/MJ, energy produced in MSWI = 2,74 MJ/kg waste (ecoinventd))

Another key difference in the APOS model is the input and use of secondary material, i.e. reclaimed wood, which is no longer assumed to be burden-free, as the theory prescribes that the material should carry an allocated fraction of the upstream processing and potential treatment

Fig. 2. Illustration of case study beams. Left: BAU beam design, right: RU beam design.
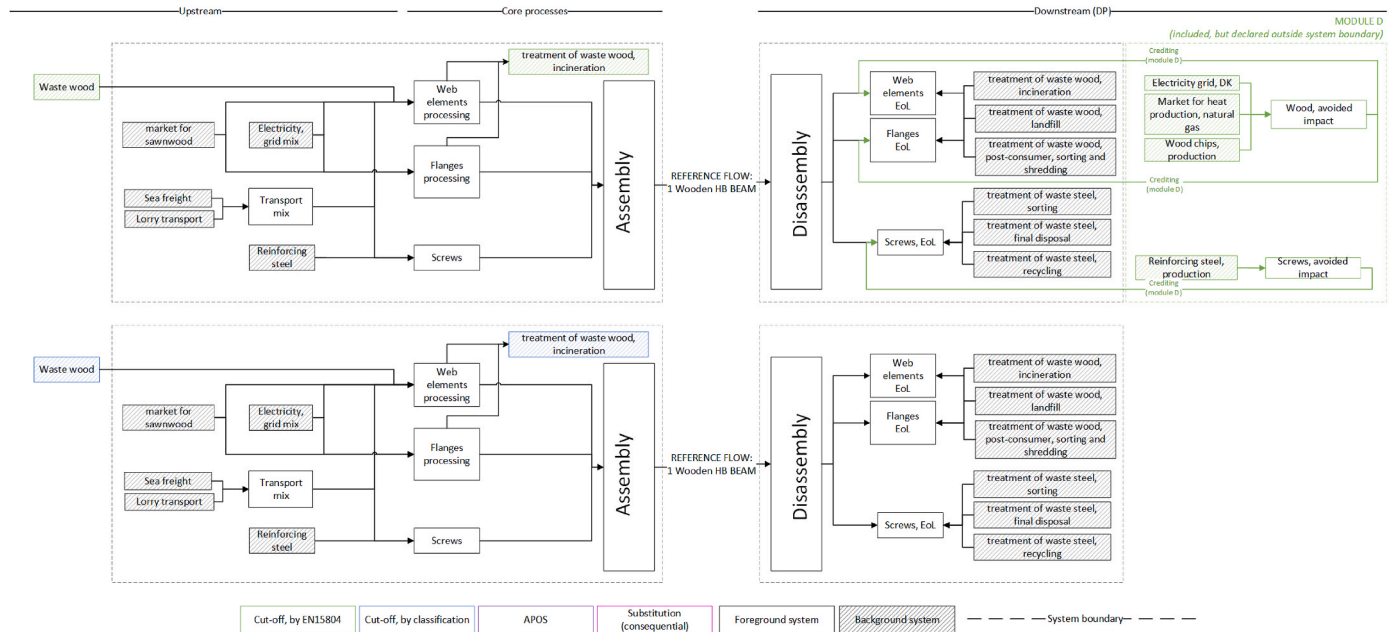

Fig. 3. Simplified system boundaries on the two cut-off system models.

processes from the previous cycle. In the model, a proxy has been used as a provider, namely 'wood chips from waste wood'. The reasons for this are provided in the supplementary material in Appendix B.

For DP, the foreground system is changed to accommodate reasoning in the APOS system modelling. This entails the product system including all treatments of waste, as there are no products or co-product output/ reference products.

The adjustments for the APOS model are shown in Fig. 4.

3.3.3.3. Substitution. When changing to a consequential approach, several changes occur (see Fig. 4). However, it must be noted that the system boundaries of the foreground system were only expanded to a limited extent. This is further discussed in the discussion section. The beam is used as the reference flow, and only one service life-cycle is considered, whereas substitution of the beam into the construction market and cascading system could have also been included.

This modelling choice is based on the reasoning that, for the assessments to be comparable, a product approach must be chosen – i.e., must use the same assessment level across the used system models. That is why the CLCA is narrow and does not include all the butterfly effects of a decision.

Similar to APOS, the input data on reclaimed/secondary wood is changed. Due to the data available, a proxy is used (wood chips, from waste wood). This proxy use and the marginal assumption are discussed further in the discussion section.

Another change in the consequential model is the substitution system model, applied specifically in the DP. The substitution system model partially re-introduces the crediting from the EN15804 system, but it is

more inherent than the explicit declaration through module D (see Figs. 3 and 4).

3.4. Life cycle impact assessment (LCIA)

The reference LCIA method used is EF 3.0 (adapted method by openLCA). The method does not account for biogenic carbon sequestration, as opposed to the LCIA method for EN15804 (Sonderegger and Stoikou, 2023). To adhere to the BE LCA approach, an adapted EF 3.0 (called EN15804+A2 Method in the openLCA LCIA package) ought to be used. However, as the LCIA method is not applicable to the other system models in this study (Sonderegger and Stoikou, 2023), it has been omitted.

The sensitivity of the results is tested against robustness across other LCIA methods, IMPACT World+ (Bulle et al., 2019) (both damage and midpoints, default recommended) and ReCiPe 2016 (Huijbregts et al., 2017) (all three cultural perspectives: individualist, hierarchist and egalitarian), and both midpoint and endpoint. All methods applied are based on LCIA method pack 2.2.1 from openLCA (openLCA Nexus). The applied methods have been used due to their wide applicability and the option to consider both midpoint and endpoint, along with different cultural perspectives, in ReCiPe.

3.5. Sensitivity analyses and uncertainties

Data quality and representativeness are key reporting aspects in EN15804. However, in this study, the key element is to be consistent in the modelling representation across the system models whereas
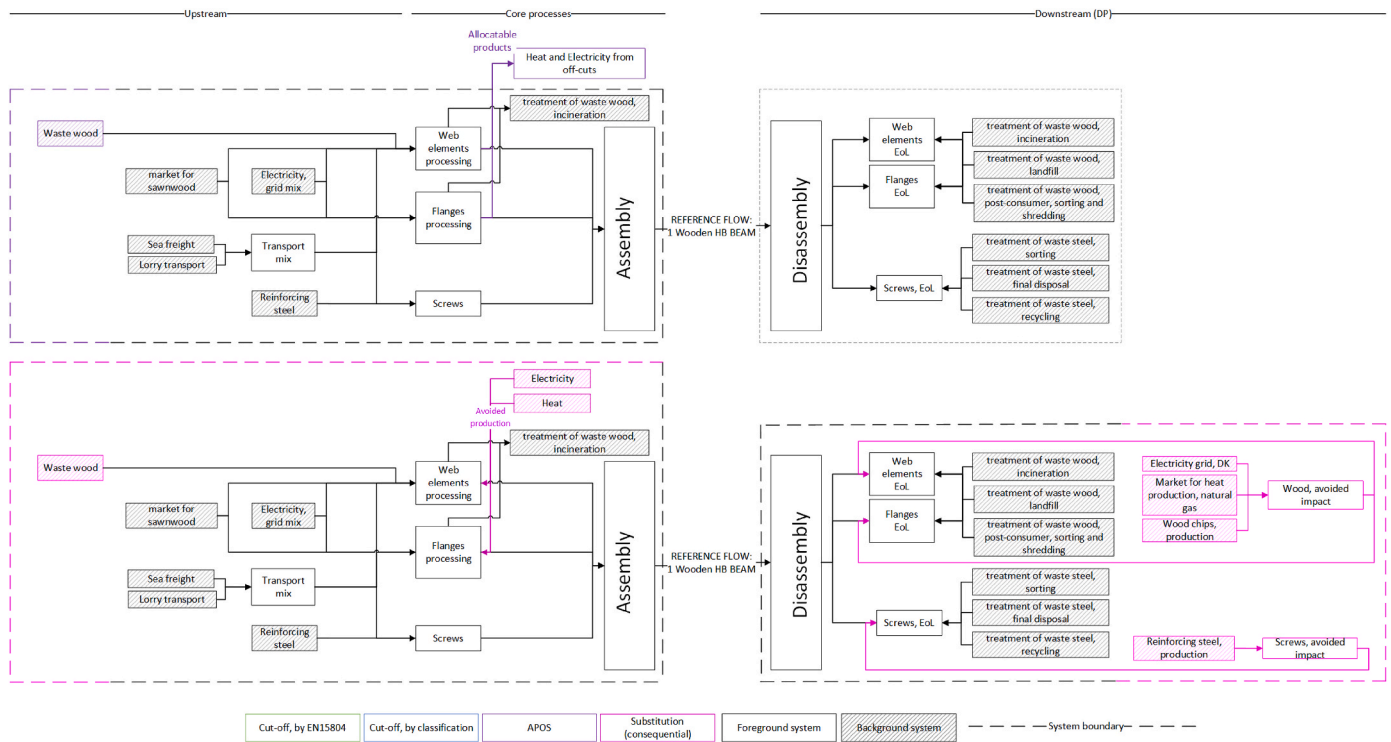
Fig. 4. Simplified system boundaries on the APOS and 'substitution (consequential)' system models.

Table 3
Aggregated results of the beam, upstream (UCP) and downstream (DP), applying **conservative EoL treatment**. EF 3.0 Method (adapted). RU values, normalized against BAU values. System models: cut-off by EN15804 (both incl. and excl. module D crediting), cut-off by classification, APOS, substitution (consequential).

		Cut-off, EN15804				Cut-off, By classification		APOS		Substitution, Consequential	
	System model, and boundary inclusion	Incl. Module D		Excl. Module D							
	Case study model	BAU	RU	BAU	RU	BAU	RU	BAU	RU	BAU	RU
	Acidification	1	0.51	1	0.48	1	0.48	1	0.55	1	4.48
	Climate change	1	0.85	1	0.50	1	0.50	1	0.60	-1	2.02
	Climate change - Biogenic	1	1.13	1	0.55	1	0.55	1	0.64	1	-1.10
	Climate change - Fossil	1	0.86	1	0.50	1	0.50	1	0.60	-1	2.01
	Climate change - Land use and LU change	1	0.37	1	0.37	1	0.37	1	0.43	1	0.46
	Ecotoxicity, freshwater	1	0.63	1	0.52	1	0.52	1	0.62	1	2.01
	Ecotoxicity, freshwater - inorganics	1	0.49	1	0.47	1	0.47	1	0.55	1	0.87
	Ecotoxicity, freshwater - metals	1	0.67	1	0.53	1	0.53	1	0.64	1	2.17
	Ecotoxicity, freshwater - organics	1	0.45	1	0.43	1	0.43	1	0.47	1	0.21
	Eutrophication, freshwater	1	0.73	1	0.53	1	0.53	1	0.61	1	18.36
	Eutrophication, marine	1	0.51	1	0.49	1	0.49	1	0.56	1	1.74
	Eutrophication, terrestrial	1	0.51	1	0.49	1	0.49	1	0.56	1	1.40
	Human toxicity, cancer	1	0.75	1	0.71	1	0.71	1	0.78	1	0.88
	Human toxicity, cancer - inorganics	-	-	-	-	-	-	-	-	-	-
Conservative EoL modelling	Human toxicity, cancer - metals	1	0.80	1	0.71	1	0.71	1	0.77	1	2.32
	Human toxicity, cancer - organics	1	0.74	1	0.72	1	0.72	1	0.79	1	0.69
	Human toxicity, non-cancer	1	0.61	1	0.53	1	0.53	1	0.60	1	1.48
	Human toxicity, non-cancer - inorganics	1	0.54	1	0.52	1	0.52	1	0.61	1	-0.68
	Human toxicity, non-cancer - metals	1	0.76	1	0.55	1	0.55	1	0.62	1	4.04
	Human toxicity, non-cancer - organics	1	0.63	1	0.49	1	0.49	1	0.58	1	-2.12
	Ionising radiation	1	0.87	1	0.48	1	0.48	1	0.55	1	0.01
	Land use	1	0.35	1	0.35	1	0.35	1	0.46	1	0.62
	Ozone depletion	-1	0.00	1	0.51	1	0.51	1	0.60	-1	-1.44
	Particulate matter	1	0.43	1	0.43	1	0.43	1	0.57	1	-2.64
	Photochemical ozone formation	1	0.49	1	0.47	1	0.47	1	0.55	1	1.06
	Resource use, fossils	1	0.94	1	0.48	1	0.48	1	0.57	-1	0.82
	Resource use, minerals and metals	-1	0.44	1	0.57	1	0.57	1	0.66	1	-3.30
	Water use	1	1.17	1	0.60	1	0.60	1	0.60	1	29.17

temporal, geographical and technological representativeness is of less importance.

Sensitivity and uncertainty are addressed in the supplementary material to qualify the outcomes of the study. This is studied on several levels: 1) considering different LCIA methods for robustness, 2) performing input value variations (+10%), and 3) qualitative uncertainty evaluations, based on the pedigree matrix.

4. Results

With respect to the aim of the study, namely to illustrate how the choice of the system model and boundaries of the LCA study can affect the decision support, the results are shown as relative comparisons with normalized values. The normalization enables quick and easily interpretable graphics, showing whether the RU exceeds the impacts of the BAU across several system models and impact categories.

Absolute values are often used in practice and for decision support against benchmarks. While this might help the assessor to decide whether the calculated system complies with a set target or requirement, there might be an issue in the specificity and certainty of these values. The results show how values change based on system models, partially invalidating the certainty of benchmark conclusions. However, the quantitative values and benchmarking are not relevant to the main hypothesis. Thus the 'absolute' quantitative values are shown in the supplementary material in Appendix E.

The output from the primary modelling is aggregated into one life-cycle value and presented with LCIA method EF 3.0. Further LCIA methods addressed for sensitivity assessments are shown in the supplementary material in Appendix F. The aggregation of the results goes

against the prescribed granularity in EN15804, where only modules A1-A3 are allowed to be aggregated. For further transparency, the results are shown separately in the supplementary material in, Appendix E.

The impact category 'Human toxicity, cancer - inorganics' results in '0' for all results and system models using the EF3.0 LCIA method and is therefore omitted from the results. The lack of value is presumably from a linking error in the database, as other LCIA methods result in values for this category. For all LCIA methods, only hydrazine is characterized by this impact category.

4.1. Decision support recommendations, normalized results

The results shown are normalized against the BAU beam results within each system model – i.e., the reference value of '1' is the BAU beam in all system models. The results cannot be compared across system models, as the numerical values, and thus the normalization reference, differ.

The aggregated model results are shown in Table 3 and Table 4. The disaggregated results for UCP and DP are further discussed in Appendix E. Interpretation of the normalized values should focus on whether the normalized RU beam values are below '1' (black numbers) or more than '1' (red marking), and if above '1', then to what extent. It can also be seen whether the BAU beam reference or normalized RU beam, imposes environmental impacts (black numbers), or avoids them (green numbers).

The internal relation between the BAU beam and RU beam impact, within each of the attributional system models, appears similar across the different models. The reason might be that the foreground system is rather simple, while the product designs are quite similar. Thus, the

Table 4

Aggregated results of the beam, upstream (UCP) and downstream (DP), applying **statistics EoL treatment**. EF 3.0 Method (adapted). RU values, normalized against BAU values. System models: cut-off by EN15804 (both incl. and excl. module D crediting), cut-off by classification, APOS, substitution (consequential).

System model, and boundary inclusion / Case study model	Cut-off, EN15804 Incl. Module D BAU	RU	Excl. Module D BAU	RU	Cut-off, By classification BAU	RU	APOS BAU	RU	Substitution, Consequential BAU	RU
Acidification	1	0.46	1	0.45	1	0.45	1	0.53	-1	-0.14
Climate change	1	0.53	1	0.49	1	0.49	1	0.59	-1	-0.19
Climate change - Biogenic	1	0.59	1	0.54	1	0.54	1	0.64	1	0.26
Climate change - Fossil	1	0.53	1	0.49	1	0.49	1	0.60	-1	-0.19
Climate change - Land use and LU change	1	0.37	1	0.37	1	0.37	1	0.43	1	0.41
Ecotoxicity, freshwater	1	0.54	1	0.51	1	0.51	1	0.62	-1	0.02
Ecotoxicity, freshwater - inorganics	1	0.47	1	0.45	1	0.45	1	0.54	1	1.25
Ecotoxicity, freshwater - metals	1	0.55	1	0.53	1	0.53	1	0.64	-1	-0.00
Ecotoxicity, freshwater - organics	1	0.44	1	0.43	1	0.43	1	0.47	1	0.43
Eutrophication, freshwater	1	0.56	1	0.52	1	0.52	1	0.61	-1	-0.17
Eutrophication, marine	1	0.45	1	0.44	1	0.44	1	0.53	-1	0.00
Eutrophication, terrestrial	1	0.45	1	0.44	1	0.44	1	0.53	-1	0.16
Human toxicity, cancer	1	0.74	1	0.71	1	0.71	1	0.80	1	1.06
Human toxicity, cancer - inorganics	-	-	-	-	-	-	-	-	-	-
Human toxicity, cancer - metals	1	0.74	1	0.71	1	0.71	1	0.77	-1	0.01
Human toxicity, cancer - organics	1	0.73	1	0.70	1	0.70	1	0.82	1	0.68
Human toxicity, non-cancer	1	0.54	1	0.51	1	0.51	1	0.59	-1	0.16
Human toxicity, non-cancer - inorganics	1	0.53	1	0.51	1	0.51	1	0.60	1	0.28
Human toxicity, non-cancer - metals	1	0.56	1	0.53	1	0.53	1	0.61	-1	-0.13
Human toxicity, non-cancer - organics	1	0.52	1	0.48	1	0.48	1	0.58	1	0.26
Ionising radiation	1	0.51	1	0.48	1	0.48	1	0.57	1	0.29
Land use	1	0.35	1	0.35	1	0.35	1	0.46	1	0.52
Ozone depletion	1	0.58	1	0.50	1	0.50	1	0.60	1	0.07
Particulate matter	1	0.42	1	0.42	1	0.42	1	0.57	1	0.22
Photochemical ozone formation	1	0.45	1	0.44	1	0.44	1	0.52	-1	7.88
Resource use, fossils	1	0.52	1	0.48	1	0.47	1	0.57	-1	-0.20
Resource use, minerals and metals	1	0.65	1	0.57	1	0.57	1	0.66	1	0.11
Water use	1	0.64	1	0.58	1	0.58	1	0.59	-1	-0.17

Row group label (vertical): Statistics EoL modelling

adaptions applied follow the same trends for BAU and RU beam models. An exception is when including module D crediting in the EN15804 model and for the consequential system models, which apply substitution and avoided impacts. These crediting options introduce both numerically positive and negative values (i.e., imposed and avoided impacts respectively).

Different mechanisms appear across the system models and their normalized results, as seen numerically in Tables 3 and 4. Some of these are due to the introduction of the option or possibility of crediting and substitution (EN15804 module D and consequential, through system expansion, respectively). The 'mechanisms' can be divided into four groups: 1) both the BAU and RU design impose impacts on the impact category; 2) both the BAU and RU designs indicated negative impacts on the impact category, due to the crediting of avoided impacts or substitution; 3) the BAU design indicated negative impacts on the impact category, due to crediting or substitution, but the RU design imposes impacts on the impact category; and 4) the BAU design imposed impacts on the impact category, but the RU indicated negative impacts on the impact category.

For the system models cut-off by EN15804 (excl. Module D crediting), cut-off by classification and APOS (only group 1) appears. For the EN15804 system model, including Module D crediting, groups 1) and 3) appeared. For the results of the consequential model, all four groups appear.

Looking at the normalized results for the EN15804 system model in Tables 3 and 4 would lead to the interpretation that the RU is more environmentally beneficial than the BAU. This is the case because the aggregated results show fewer impacts across all assessed impact categories (i.e., they all represent results on 'group 1'). An exception is the scenario where a conservative EoL modelling is applied and module D crediting is included, as shown in Table 3. Here, the normalized results deviate in some impact categories, also showing mechanisms from 'group 3'. This is caused by the relation between production impacts (UCP) and EoL crediting in the DP, where crediting is only *allowed* for primary material. This means that all web and flange material in the BAU beam can be credited in module D, whereas only the flange material in the RU beam is credited. The reduced impacts due to the burden-free secondary material input do not counterbalance the benefits from crediting primary material.

The dynamics of the aggregated results are shown Fig. 5, using the system model cut-off EN15804. It shows how 'burden-free' input and crediting at EoL influences the results. It becomes evident how the cut-off modelling is beneficial for low impacts by the RU design, e.g., 50% of BAU impacts for climate change. However, it also shows how the EoL modelling favours the BAU design with a potential of much higher crediting. Looking at climate change, for example, including module D

crediting, leads to 65% less avoided impact at the DP of RU than the BAU beam (see Fig. 5). Conversely, when excluding module D crediting, the emissions from the RU beam are only 13% less than from the BAU beam. The disaggregated results are detailed in Appendix E.

As the EoL statistics introduce both landfill (though minuscule amounts) and recycling as treatment options along with incineration, some of the inherent value choices in the system models appear. One is the handling of the cut-off system models, where recyclables are modelled without treatment (however, slightly extended until EoW according to EN15804). Additionally, for EN15804, crediting in module D is for the avoided production of wood chips instead of energy or electricity.

Further, the two cut-off system models (EN15804 and 'by classification') appear to give the same results. However, it is important to note that the results table only provides the internal relations between the impact from the BAU beam and the RU beam designs. I.e. the values that appear similar show only the RU results normalized against the corresponding BAU design. In the tables, only two decimals are used, but when expanding to the third decimal, they differ. In absolute values, the resulting impacts differ, though only very little. The modelling of these two systems is, however, also very similar, the only difference being the cut-off point applied in the background systems (the 'by classification' model cuts off right after the producing process, whereas the EN15804 data should be cut off at end-of-waste, i.e. potentially a little more processing occurs).

Looking at the aggregated results for the attributional system models cut-off by classification and APOS in Tables 3 and 4, they indicate that the RU design has less impact than the BAU. Across all impact categories, it can be concluded that the relative savings appear larger in the cut-off models than in the APOS model, an exception being water use. This might be due to the applied cut-off and the 'polluter pays'-principles, whereas the APOS system model, expands the system boundary and shares some of the burdens across product cycles.

Looking at the consequential system model, substitution, in Tables 3 and 4, the conclusion turns out different. The RU design no longer presumes 'decoupled material provided for free' but includes market mechanisms and substitutions. The magnitude of difference in the consequential model varies from 1 to ~30 times across impact categories and is mainly caused by the marginal assumption in the background data. The use of waste wood in the RU system is assumed to be 'taking' the resource 'waste wood' from heat production, which is then produced through other sources, e.g., hard coal, lignite and partially virgin wood. Looking into the upstream contributions, the main contributor to the RU exceeding the BAU is the use of hard coal for heat production, and in some categories lignite. Other marginal assumptions or national market specifics might have resulted in different results.
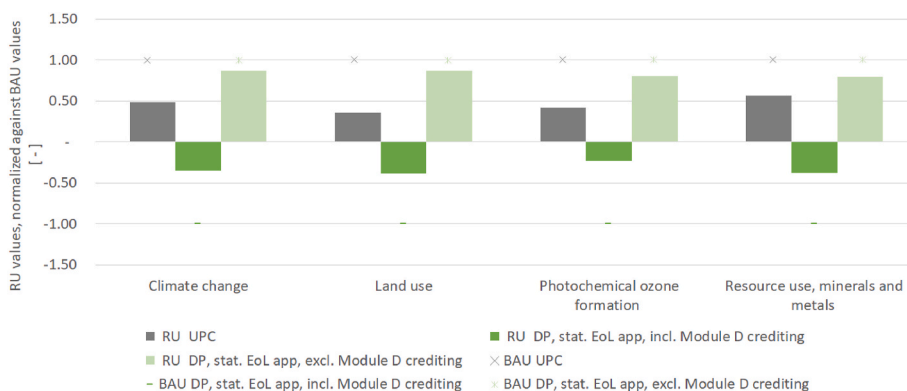

Fig. 5. Normalized values of the RU beam, against the BAU beam. Upstream (Production, in grey) and Downstream (DP, in greens) of the beam. EoL is modelled as statistical modelling. Results both incl. and excl. module D crediting. LCIA method: EF 3.0 Method (adapted). RU values, normalized against BAU values. System models: EN15804. Normalized values, considering the numerical reference point in BAU, i.e., whether avoided impacts (from substitution) result in negative indicator results.

4.2. Sensitivities assessment and uncertainty

All results are shown in the supplementary material in Appendix F. Only the central conclusions are listed here.

4.2.1. LCIA method variations

The results show that the change in LCIA method does not change the overall findings. This means that the RU beam indicates lower environmental impacts than the BAU beam in all three attributional system models, regardless of the LCIA method, cultural perspective or endpoint/midpoint. In the consequential system model, the RU beam indicates higher environmental impacts than the BAU beam, regardless of the LCIA method. An exception is when the single score from ReCiPe (2016) is used, where the results also indicate a negative impact (i.e., avoided impacts) from the RU beam design in the consequential system model.

4.2.2. Input value variations

The results show that smaller sensitivities appeared across all system models and LCIA methods. In particular, the consequential system model presented variations in every sensitivity scenario and impact method, where the parameters caused more than a ten-fold deviation in the results compared to the reference scenario. While many of these sensitivities may not be adjusted in this case due to the simplicity of the study, future assessments might qualify the data's representativeness. One highly influential parameter was the material quantity input, which was tested for further sensitivity regarding decision deviations.

4.2.3. Qualitative uncertainty evaluations

The quality is estimated from Fair to Very Good in terms of geographical and temporal representation. Technological representativeness was not addressed specifically, as the assessment did not make sense given that the specific production technology of the case was not prescribed. An exception was the denotation of the proxy processes applied.

5. Discussion

While this study has introduced many derived considerations and potential discussion points, the main and original hypothesis was that the BE LCA approach is not necessarily sufficient for evaluating the environmental impacts of product design strategies' decision support. To answer and address this, the discussion section has been organized into three topics: Decision support and conclusion pitfalls, System boundaries and influence, and Limitations of this study.

5.1. Decision support and conclusion pitfalls

When assessing the environmental benefits based on the BE LCA approach, this is often used for decision support and arguments for scaling a given solution as a basis for transition or legislation.

The results of the case study indicate that the RU beam design is environmentally beneficial overall compared to the BAU beam design in all three attributional models. Based on this, one might recommend this design be scaled up and applied to all waste wood. However, when looking at the consequential approach, it exemplifies how the reclaimed wood already has a use context. Thus, if the beam design begins to draw on this resource, the heat production value chain will have to use a different material, which in this case is modelled as coal in the background database.

The modelling and market mechanisms of marginal assumptions in the consequential model might be discussed, and adjustments would be interesting to assess – e.g., how would global markets react if the BE were to transition actively and move fast, and what would happen if the marginal energy carrier was not coal, but, e.g., gas. Although this was not applied further in the study, the case study highlights the risk of

overlooking mechanisms caused by product designs when only applying an attributional cut-off approach. While this may not have been of relevance to the same extent in a linear economy, under the assumption that 'everything is unlimited', the circular economy is a product of the realization that 'nothing is unlimited', thus challenging the approach of inconsequential consumption.

Another risk in decision support appears in the aim to continuously simplify the results, which are often aggregated at the building level for assessments to comply with a 'one-value' benchmark. However, this might disguise some partial conclusions, e.g., temporal impact differences, which also occurred in these results. Upfront production is mostly relevant here and now, whereas EoL is often a future 'potential', as in this respect are also crediting potentials.

Another angle on this is that, according to the current BE LCA approach and the standards prescribing calculation methods, one must use average or conservative approaches and assumptions when modelling EoL. However, the EoL of a given building, products etc. in the BE does not take place anytime near today's average treatment processes, as the BE operates with life spans of 20, 50, and 100 years. Thus, the modelling of both UCPand DP within the same timespan/technological framing cannot be said to produce valid and robust decision support. Both production and potential EoL treatment should still be considered in assessments and for decision support up front, but EoL especially ought to be supplemented with a study investigating the different paths and how modules C and D vary in the future, preferably considering the span of uncertainties in the future.

5.2. System boundaries and influence

Depending on the context of decision support, one should reflect upon the system model and boundaries applied, as they answer different questions. I.e. where an attributional model aims only to attribute the actual impacts related to a product or system, a consequential model considers the potential market consequences caused by the product or system. Such a reflection may be done by applying different value choices, world-to-model representations and incentives for action.

While LCA practitioners know this to some extent, and while it is heavily discussed in the LCA community beyond the BE, the rapid increase in LCA users in the BE might not know or understand the influence of applied system boundaries. Additionally, they may not be aware of the historical interpretations (and simplifications) applied, and thus the inherent bias in the BE LCA approach.

The case study showed that uncritically scaling up a solution based on an attributional approach might be misleading regarding the environmental benefits, at societal scale, as it overlooks consequences beyond the assessed system boundary. Instead, when aiming at societal-scale decision support, one recommendation is to supplement the assessments with either consequential LCA scenarios, or as a bare minimum a mapping of the potential societal consequences based on market mechanisms, existing value chains and changes to both.

Regardless of which system model or system boundary was applied, the case study also showed that the RU beam was never without any burden. Thus, the 'burden-free' assumptions of secondary materials at the building level imply an unsupported incentive to use secondary materials, which may in the best case provide underestimated results, and in the worst case be used for complying with benchmarks, such as legislative emissions limits.

5.3. Limitations

This study highlighted the complexity of interchanging the LCA approach while maintaining a reasonably comparable system assessed across system models as the boundaries change, but also the availability of data.

Due to the nature of the case study, a wooden product, biogenic carbon, could be a topic for discussion. While biogenic carbon inclusions

are widely discussed in the BE (Garcia et al., 2020; Rasmussen et al., 2021) and rigidly prescribed in the current standards and guidelines for the BE LCA approach (EN and 15804:2012+A2:2019, 2019), it was omitted from the final modelling and interpretation of results because the inherent material was the same in the designs, and thus beneficial properties were not to compete with other materials. Additionally, inclusion requires specific awareness of the carbon tracking and linking in the applied database.

The omission of biogenic carbon was only one of several prescribed requirements set out in modelling prescribed through EN15804. While the original modelling started from the BE LCA approach, several of the requirements were omitted, as they were either not relevant in comparing the system models nor applicable to the study. However, for the sake of usefulness in the industry, this might skew the understanding of EN15804 modelling requirements and possibilities. Even though it was intended to develop an EN15804 compliant study as a reference, this was not the case in the final study. One of the major deviations is the use of LCIA method EF 3.0, which was compatible across all system models.

Another limitation is the case-study beam design itself and its implications for the results. While the case study design was based on a specific design development of a beam using reclaimed timber in varying sizes, compared to a reference design with standardized pieces, this might have biased the benefits of RU. However, it might also show the current design approach of standardized units often being over-dimensioned. A sensitivity study was performed to check the influence of the case-study beam design's material consumption. The finding was that the material quantity inputs appeared sensitive: when interchanging the design material quantities, the RU beam design exceeded the BAU beam's impact in one impact category in the APOS system model. Several other impact categories showed close to no benefits.

This sensitivity of material input thus also affects replicability, as dynamics and markets might differ across materials, construction parts and buildings. Thus, for the results to be transferred and generalized, there is a need to extend the knowledge and understanding of other types of materials, value chains and circular design strategies.

Another inherent discussion lies in the system models' value choices and applied system boundaries. The system boundaries applied in the various system models were simplified for applicability, thus using existing background data without further manipulation to ensure compatibility. Several background processes might have been changed to represent the case study, thus making the use of proxy processes redundant (e.g., the use of wood chips as a reclaimed wood proxy in the APOS model) or, for the consequential model, adjusting the presumed market mechanisms for the specific waste fraction being considered, instead of using, e.g., wood chips.

Along with this comes the limitation that some of the system boundaries were limited. Especially with regard to the consequential modelling approach, the product focus of manufacturing might have been extended by considering the alternative use of virgin or reclaimed wood respectively in a parallel system to the assessed one, or the cascading market mechanisms and influence across industries. However, to ease comparability, align system boundaries and enable reporting a comprehensible study, this simplification was deemed acceptable.

6. Conclusions

This study has demonstrated the impact of changing the modelling approach and system boundaries on the conclusions drawn from an illustrative case study. To our knowledge, this is the first case of its kind in the built environment at this level of detail that considers decision support regarding circular economic design strategies. It shows that the conclusions might change when considering the dynamics and changes beyond the attributional system boundary, which might be crucial in decision support at a large scale, i.e., beyond the singular product or building value-chain.

For the case study particularly, the models indicate that the circular design strategy of using reclaimed material for the beam is environmentally beneficial compared to the reference design. However, when applying consequential LCA with system expansion and substitution, the reuse design exceeds all impacts. However, the findings only apply to this specific case study under the applied assumptions. To draw generalized conclusions and provide broader recommendations, a larger body of empirical evidence would be required.

It is crucial to further develop a comprehensive body of evidence to account for variations in the materials assessed, their position in the societal value chain and the presumed influence on different markets. The case study has highlighted the potentially misleading nature of uncritically scaling up a solution based on an attributional approach, as is currently often done when using LCA in the built environment. This is because it fails to consider the broader consequences beyond the assessed system boundary. To provide more robust decision support on a larger scale, it is advisable to supplement assessments with consequential LCA scenarios or, at the very least, a mapping of the potential societal consequences based on market mechanisms, existing value chains and associated changes. While the LCA method currently applied in the built environment may very well be used for declarations and accounting, much of the decision support is placed on the agenda of driving a transition. Thus, an assessment of a product or building cannot stand alone as decision support leading to firm conclusions.

6.1. Future investigations

This work presented a particular case study. Further development of a pool of evidence should be investigated because the results might fall out differently depending on the materials assessed, where in the societal value chain it is assessed and what markets it is presumed to influence.

Additionally, investigations into and the development of data representing industry-specific market mechanisms and how they might differ when industries approach a transition would be of great value. It could also tap into the societal aspects of how resources will be prioritized when unlimited availability is no longer presumed.

Further, the omission of biogenic carbon sequestration might be interesting to investigate in future work, together with how different methodological choices might impact conclusions. Another interesting aspect to consider is prospective LCA and how the dynamic changes in data and methodology might influence both the results and the conclusions.

CRediT authorship contribution statement

Sarah Cecilie Andersen: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Alexander Hollberg:** Writing – review & editing, Validation, Supervision, Conceptualization. **Xan Browne:** Writing – review & editing, Visualization, Funding acquisition, Data curation, Conceptualization. **Holger Wallbaum:** Writing – review & editing, Validation, Supervision, Conceptualization. **Harpa Birgisdóttir:** Writing – review & editing, Validation, Supervision, Conceptualization. **Olga Popovic Larsen:** Writing – review & editing, Supervision. **Morten Birkved:** Writing – review & editing,

Validation, Supervision, Methodology, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

We have shared modelling information in the Supplementary Material

Acknowledgement

The authors would like to thank Haitham Abu Ghaida for his feedback.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.dibe.2024.100398.

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