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Implications of circular strategies on energy, water, and GHG emissions in housing of the Global North and Global South

Naomi Keena[a,*], Daniel R. Rondinel-Oviedo[a], Alejandra Acevedo De-los-Ríos[b,c], Jaime Sarmiento-Pastor[b], Andrés Lira-Chirif[b], Marco Raugei[d,e], Anna Dyson[f]

[a] Peter Guo-hua Fu School of Architecture, Faculty of Engineering, McGill University, Montreal, Quebec, Canada
[b] Instituto de Investigación Científica, Universidad de Lima, Peru
[c] Louvain Research Institute for Landscape, Architecture and Built Environment (LAB), University of Louvain, UCLouvain, Louvain-la-Neuve, Belgium
[d] School of Engineering, Computing and Mathematics, Oxford Brookes University, UK
[e] Center for Life Cycle Analysis, Columbia University, New York, USA
[f] Yale Center for Ecosystems in Architecture (Yale CEA), Yale School of Architecture, New Haven, CT, 06511, USA

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ABSTRACT

As urbanization continues to surge, building materials are poised to become a dominant contributor to global emissions. Traditionally, the building sector has focused on mitigating "operational carbon" linked to a building's day-to-day energy needs, such as heating, cooling, lighting, and equipment usage. However, there has been a paucity of studies on the environmental impacts associated with building materials across a building life cycle. This paper addresses this gap by conducting a life cycle assessment of housing stocks in two diverse case studies: Montreal (Canada) and Lima (Peru). These cities offer a North/South perspective, highlighting the challenges, opportunities, and potential solutions for decarbonizing the housing sector. The study investigates the potential of circular strategies and investigates three scenarios: selective deconstruction (allowing for reuse and recycling), recycling, and landfilling. The results underscore the potential of selective deconstruction in significantly reducing the overall environmental footprint of residential buildings. In Lima, for instance, selective deconstruction, when compared to landfilling, can cut greenhouse gas emissions, water consumption, and fossil resource usage by a substantial 70%, 67%, and 69%, respectively. These findings offer valuable insights for decision-makers in construction materials and waste management, encouraging the adoption of circular economy practices through informed guidelines and recommendations.

1. Introduction

The construction industry generates 39% of energy-related greenhouse gas (GHG) emissions globally (International Energy Agency & United Nations Environment Programme, 2018), nearly 60% of which come from emerging economies (Huang et al., 2018). Furthermore, the same industry is responsible for almost one quarter of the solid waste generated worldwide (Yeheyis et al., 2013), and this proportion can be even more significant in developing countries (Benachio et al., 2020). Therefore, addressing carbon emissions within the building sector is key to achieving the carbon neutrality goal in the 2015 Paris Agreement (World Green Building Council, 2019).

The GHG emissions (also commonly referred to as "carbon emissions") associated with a building can be divided into two groups: "embodied" carbon emissions, which includes the emissions from extraction, transformation, transportation, installation, maintenance and disposal of materials in the construction processes, and "operational" carbon emissions, which includes the indirect emissions from electricity and other fuels used during the service life of the building (Cao, 2017; Moussavi Nadoushani and Akbarnezhad, 2015). Embodied carbon is sometimes referred to as the "hidden" carbon emissions of buildings, as it is often not accounted for during the design phase (Monahan and Powell, 2011). Environmental policies have typically focused on in-use energy efficiency rather than material efficiency (Allwood et al., 2011). Globally, as energy grid mixes for electricity production becomes cleaner and buildings become more energy efficient, emissions associated with the operational phase will become less of a concern. Studies indicate "operational carbon" emissions are

* Corresponding author.
E-mail address: naomi.keena@mcgill.ca (N. Keena).

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projected to decrease from 75 per cent to 50 per cent of the sector in the next few decades (UNEP, 2023). However, the “embodied carbon” emissions associated with building materials throughout a building life cycle will continue to contribute to life cycle impacts if business as usual material practices continue. (UNEP, 2023). This is especially true in the face of rapid global urbanization, which is placing greater demand on construction materials with raw material consumption set to double by 2060 (OECD, 2019). Hence, this paper will focus solely on construction material use throughout a building cycle and its life cycle impacts.

The embodied carbon associated with the construction materials of a building increases gradually as it goes through different stages of its life cycle (Akbarnezhad and Xiao, 2017); in this sense, the accumulated embodied carbon may be regarded as a carbon “investment” which then may or may not be “recouped” if the materials themselves are re-used in subsequent buildings. Thus, at the end-of-life (EoL) phase of a building, strategies should be evaluated to determine whether such “invested” carbon is “lost” or partially recovered (Akbarnezhad et al., 2012).

From the built environment perspective, the conventional linear economy model of demolishing and landfilling has proven to be inefficient, and the accumulation of construction, renovation, and demolition (CRD) waste represents a global challenge (Purchase et al., 2021). While other industry sectors have successfully applied waste reduction strategies and policies, in the building sector the proportion of CRD waste that ends in landfills is still alarmingly high (Ajayi et al., 2015). As the US Environmental Protection Agency (EPA) reported, only 20% of worldwide CRD waste is recycled or reused: 53% thereof from demolition and 38% from construction and renovation (Pacheco-Torgal and Ding, 2013). There is a challenge to incorporate more integrated and sustainable waste management systems in developed countries, which are responsible for half of the world’s waste generation (Laurent et al., 2014). In emerging nations, increasing CRD waste is a critical problem due to rapid population growth and poor waste management systems (Esa et al., 2017). Likewise, there is a constant struggle to move from open to controlled landfill disposal (Guerrero et al., 2013). On the other hand, data on CRD waste materials in Latin America is not frequently monitored or reported, and classifications vary between countries (United Nations Environment Programme, 2023). There are key opportunities in the potential recycling of CRD waste in the region (United Nations Environment Programme & International Waste Management Association, 2015), as it accounts for a large share of the total waste - as high as 34% - though this largely varies among cities (Chen et al., 2022).

The transition to a Circular Economy (CE) has emerged as a viable alternative, with reduced environmental impacts and increased economic growth within the construction sector (Lieder and Rashid, 2016; Oluleye et al., 2022). Integrating a CE approach in the waste management of the construction industry has important implications for societies and governmental practices, promoting the diversion of material flows from landfills and reducing the demand for new raw materials (Akanbi et al., 2019; Mangialardo and Micelli, 2018; Papastamoulis et al., 2021) and, consequently, reducing overall carbon emissions. Within this context, the recovery of construction materials at the EoL of a building for recycling and/or reuse is a crucial step toward CE implementation (Hossain et al., 2020; Pan et al., 2015).

Bovea and Powell (2016) carried out a meta-analysis of 71 case studies that applied Life Cycle Assessment (LCA) to assess EoL materials management in buildings. Out of these, around 42% used the Ecoinvent life cycle inventory database, while 28% relied on other literature sources; moreover, 66% of the reviewed studies were developed in European countries. Also, little attention has been given to the life cycle environmental benefits of reuse rather than recycling, versus the use of new virgin materials (Densley Tingley and Davison, 2012). Where such studies have been conducted, the evidence demonstrates the significant energy, carbon dioxide, and resource savings of reuse (Hopkinson et al., 2019).

This study applies a (LCA) approach to two case studies, located in Montreal (Canada) and Lima (Peru), as two cities that offer a global North/South view of the challenges, opportunities, and potential levers faced in achieving the decarbonization of the housing sector.

In 2021, Statistics Canada reported that 62% of Canadian homeowners planned a home renovation, and that figure was 59% in Montreal (Government of Canada, S. C, 2021). Canada and the US governments have pledged to fund large-scale retrofit solutions (Magwood et al., 2021) for building stock with a focus on energy efficiency but failed to address embodied carbon emissions. Meanwhile, a recent report projected that by 2030 there would be a significant rise in the number of concrete structures becoming overburdened and in need of building system reparations such as structure and finishing (Vilches et al., 2017). The compound annual growth rate (CAGR) for global concrete restoration market size is set to increase to 6.7% by 2030 to reach a value of almost 26.4 million USD (Acumen Research and Consulting, 2022). The growth is projected to be most significant in North America, where many mid-century structures are experiencing premature deterioration primarily caused by poor building quality, improper design, and a failure to make timely repairs (Ürge-Vorsatz et al., 2015). Within this push for renovation, the ability to assess the quality of materials at EoL for application in a specific material recovery strategy is vital, considering that currently, 20–30% of Canada’s solid waste is CRD (Yeheyis et al., 2013).

Shifting the focus to Peru, in the last 60 years, Lima has grown from 1.2 million to 10 million inhabitants (Espinoza and Fort, 2020; Matos Mar, 2012), and the city currently has a deficit of about 612,000 housing units (Gestión, 2017). The population is expected to grow at a 1.36% average rate for the next ten years (Instituto Nacional de Estadística e Informática, 2018), which, along with the growing demand for housing, increases the need for new building materials (Acevedo et al., 2018; Córdova, 1958). Given the growth in new construction, according to the latest report on global warming by the Peruvian Ministry of Environment (2021) the manufacturing and construction industry was responsible for 23% of GHG emissions of the energy sector, which at the same time represents 18% of total GHG emissions at a national level. If CE strategies are promoted, decarbonizing the manufacturing and construction sector holds great promise while reducing the need to import raw materials (Eberhardt et al., 2019a).

During recent decades, the use of reinforced concrete has become more widespread in order to create affordable mass housing (Acevedo and Llona, 2016). The extensive use of concrete is also due to its being perceived as a safe material in case of seismic events (Kiani et al., 2022). However, conventional use of concrete in construction results in high embodied carbon emissions. This is primarily due to the sheer volume of concrete needed. This situation presents important challenges to decrease the impact of concrete use, ranging from the reduction by design, through the implementation of design for disassembly (DfD) (Cai et al., 2019; Li et al., 2022; Salama, 2017; Xiao et al., 2017), to improvements in the associated extraction, manufacturing, and transport processes.

To investigate the potential benefits of a shift to a more CE approach, two different alternatives to landfilling are considered in this study for the EoL phase of the buildings, namely recycling and reuse.

Recycling has been the oldest strategy to deal with CRD waste sustainably. However, it is often a complex process that comprises preliminary steps for sorting large amounts of construction materials from demolition, and it ultimately generates materials of similar or often lower value than the originals (Maccarini and Avellaneda, 2013). For typical CRD waste, advanced recycling plants are based on the scheduling of different batches with segregated materials treated separately from mixed ones, which leads to different types and qualities of recycled aggregates (Galán et al., 2019). GHG emissions are mainly due to the electrical energy needed for the machines and the fuels required for transportation.

Reuse is an effective way to reduce the demand for virgin resources and the environmental impacts of construction and demolition, by not requiring in-plant processes, but just transport (Yuan and Shen, 2011). It has been argued that the focus should shift toward waste prevention and component reuse strategies (Joensuu et al., 2020). However, the reuse scenario is limited by social, economic, and legal factors (Da Rocha and Sattler, 2009). The development of assessment standards and certifications for secondary materials is key in assuring the safety and efficacy of re-use materials and to make sure they comply with building codes. Such standards can help overcome impediments to reuse such as legal limitations and a lack of social acceptance in using secondary materials (UNEP, 2023). Also, to achieve effective reuse practices, it is essential to start from the design phase, through DfD (Condotta and Zatta, 2021), which focuses on improving the ease of disassembly of the building components, where parts are selectively deconstructed to act as material banks for new buildings (Eberhardt et al., 2019b; Hopkinson et al., 2019; Bakker et al., 2014).

2. Materials and methods

This paper examines the potential for circular strategies to deal with construction, renovation, and demolition waste during the EoL phase of residential buildings. It examines two distinct housing typologies: that of a housing unit in a triplex building in Montreal, Canada and that of an apartment housing unit in Lima, Peru. Both chosen housing units are representative of typical "formal" housing in each location. The two housing typologies are illustrated in Fig. 1 and briefly described below. Section 2.1 describes the bills of materials (BOM) for the two housing units and Section 2.2 then discusses the LCA modelling, and the associated assumptions and limitations.

(a) Montreal Triplex Housing Unit

The traditional Montreal triplex housing has a timber frame construction and brick façade. The triplex's average gross floor area is 247 m^2, typically consisting of three apartments inhabited by three families of, on average, four members each (thus the triplex in total has 12 residents). For the purposes of this study one floor which represents one apartment unit is considered, with a floor area of approximately 80 m^2. The house is constructed by assemblies (floor, roof, foundation, walls, stairs, balcony, finishes, and windows), and each assembly is composed of different materials (i.e., concrete, insulation, bricks, wood, etc.) as highlighted in Fig. 1. The triplex was chosen for this study as it is representative of a typical Montreal housing unit given that 4 in 10 dwellings in the city are apartments with fewer than five stories, and apartments make up 58.4% of all Montreal dwellings (Government of Canada, S. C, 2021). For the purposes of this study, an entire triplex was modelled in a 3D software as described in Section 2.1 and the resulting materials and assemblies were divided by three. This approach was taken so that a portion of the roof and foundation materials could be attributed to each housing unit.

(b) Lima Formal Housing Apartment Unit

In the case of Lima, an average housing apartment was modelled. In 2019, 94% of the housing units in Lima were apartments. A typical apartment has an average floor area of 74 m^2, which hosts four residents (Cámara Peruana de la Construcción - CAPECO, 2019). The apartment is made up of assemblies (lightened slab roof, solid slab roof, lightened slab floor, solid slab floor, columns, beams, concrete walls, brick walls, windows, stairs, and foundation) and each assembly is composed of different materials. For this study the housing unit's construction materials are composed of those most used materials according to the Peruvian Chamber of Construction and central government statistics (Cámara Peruana de la Construcción - CAPECO, 2019; Instituto Nacional de Estadística e Informática, 2018), as illustrated in Fig. 1. Similar to the Montreal unit, for the foundation, the material for the whole building was calculated and then divided by the total number of apartments, allocating the corresponding amount for one apartment.

2.1. *Defining the BOM for the housing units in Montreal and Lima*

Two representative models (RM) of average dwelling units for, respectively, Montreal (Canada) and Lima (Peru) were created using the Building Information Modeling (BIM) software Autodesk Revit 2023 as illustrated in Fig. 1. Both are hypothetical models created according to average housing types as described above, including assemblies and

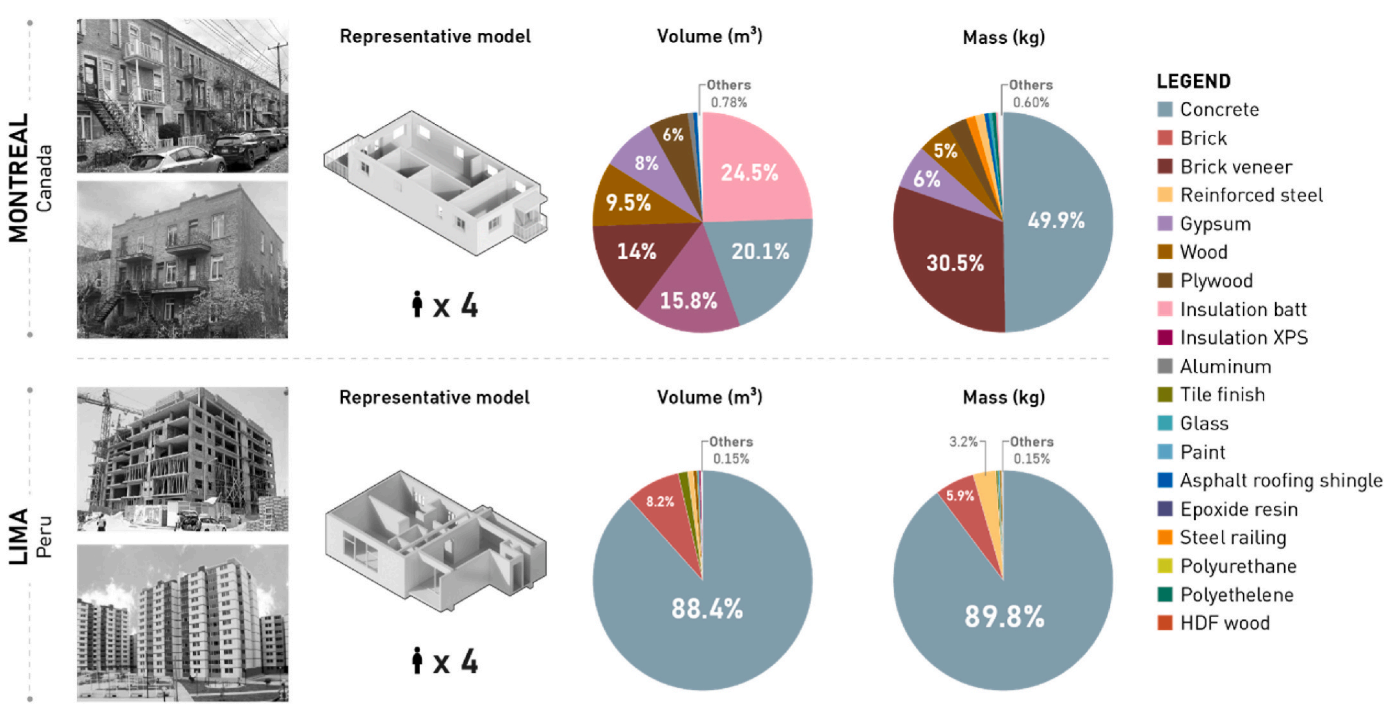

Fig. 1. Representative model of Montreal's triplex housing and Lima's formal housing apartment unit including a breakdown of the typical building materials used by mass (kg) and volume (m^3).

materials. The data for the models comes from census data, previously identified archetype studies, and local analysis and architectural surveys of existing housing (Kennedy, 2002; Legault, 1989). The building aspects considered in the RMs are structure, space plan, and skin, according to the service life of each assembly and material (Brand, 1995). The method involves calculating the type and mass of materials used in housing unit. As described above, the BOM considers the entire construction system to make the analysis more representative for each housing unit, assembly, and material.

Fig. 1 illustrates a breakdown of the housing units by percentage of building material by volume and mass. More extensive bills of materials (BOM) for the two housing units are reported in the Supplementary Materials. The BOMs were established based on the respective RMs. The RM for Montreal has a total mass of approximately 176,000 kg for the triplex, hence 58,000 kg for each floor. Two types of decomposition analysis were performed: by material and by assembly. As per Fig. 1, the analysis by material shows that concrete represents the highest percentage in mass (50%), followed by brick veneer (30%). In relation to the volume, the most representative material is insulation batt (25%) which is followed by concrete (20%). The analysis by assembly shows that the foundation and basement represent the largest mass share (50%), which is composed mainly of concrete; followed by walls (40%) composed mainly of brick veneer.

The RM for Lima weighs 126,000 kg, which leads to an average of 1700 kg/m^2. As illustrated in Fig. 1, the analysis by material shows that concrete once again represents the highest percentage by volume (88%) and mass (90%); followed by brick which represents 8.2% of volume and 6% of mass. In other words, for each square meter constructed in Lima, 1530 kg of concrete and 100 kg of brick are needed. Considering that a typical house is inhabited by four people, the amount of concrete per person per house is 28,000 kg, which is consistent with previous research (Rondinel-Oviedo and Schreier-Barreto, 2019). The analysis by assembly shows that the concrete wall is the element that represents the largest mass share (17%), followed by columns (14%) and beams (13%). The three assemblies are characterized by being composed mainly of reinforced concrete. In the case of walls, the mass of the concrete wall type is double that of the ceramic brick wall type.

The BOM for the RMs of both cities, highlights the different construction practices which are status quo in both locations. In Montreal, similar to most of North America, residential construction below five-storeys typically employs timber stick-frame construction using a framework of wooden vertical and horizontal members. This lightweight, cost-efficiency structure allows for the incorporation of insulation and involves the addition of a cladding material, water-proofing membranes, and an internal façade material, typically gypsum. Foundations and inclusion of a basement in such residential construction, involves the addition of concrete. This construction typology is highlighted in the BOM with a variety of materials, where no one material is dominant in terms of volume. In contrast, Lima's formal residential sector relies predominantly on reinforced concrete structures. Hence, per volume concrete is by far the most dominant material in Lima's residential buildings. As well as being cost effective, this construction typology meets seismic resistance standards which is essential in this region. Given the milder climate in Lima, additional insulation is not a requirement unlike the colder climate of Montreal.

2.2. LCA methodology

Life cycle assessment (LCA) is a method extensively used for assessing the environmental impacts of a wide range of human activities and processes, across all sectors. Over the course of the past two decades, it has become the de-facto standard approach in the building sector, to assess the carbon emissions of buildings materials, and specifically to compare alternative CRD waste management strategies (Di Maria et al., 2018; Junnila and Horvath, 2003; Nemry et al., 2010; Ortiz et al., 2010; Takano et al., 2015).

2.2.1. Goal and scope

As is outlined in Fig. 2, for each housing type, an LCA was carried out from cradle to grave, i.e., including the following life-cycle stages: materials sourcing and manufacturing, construction assembly, and EoL. The defined function was to provide shelter and comfortable housing to people. Consistently, the functional unit (FU) was set in both cases as "one average housing unit accommodating four persons".

The use phase was excluded from the scope of this study because the focus was on the materials and assemblies used to create the home, rather than the energy demand for operating it. Also, the GHG emissions arising from energy use during the use phase of a residential building are (i) a function of the primary energy source used to meet heating, cooling, lighting, and equipment needs; (ii) somewhat dependent on user behavior (which would have introduced an additional element of uncertainty); and – importantly – (iii) independent of the chosen EoL strategy, which is the main intended focus of this study.

It is also worthy of note that current research highlights the growing importance of considering embodied carbon emissions as buildings become more energy efficient and as energy supply becomes less carbon intensive (Berrill and Hertwich, 2021; Röck et al., 2020).

2.2.2. Inventory analysis and EoL scenarios

The LCA model was constructed using the GaBi LCA software version 9 (Sephera, 2023). All background processes were modelled using the Ecoinvent V3 life-cycle inventory (LCI) database (Wernet et al., 2016). For the best geographical representativeness, the closest local process for each material from Ecoinvent was chosen whenever possible; failing that, the global process was used instead.

Three EoL scenarios were considered and modelled, with the aim of highlighting the potential benefits of, respectively, reusing and/or recycling some of the building materials or complete assemblies: (i) selective deconstruction (allowing reuse and recycling where applicable), (ii) recycling, and (iii) landfilling.

In terms of EoL allocation, the avoided burden approach was used consistently for all scenarios, which is best suited to highlighting the benefits of recycling by assigning impact "credits" to the recycled materials, assuming that these will go on to displace equivalent quantities of their primary (virgin) counterparts (Obrecht et al., 2021). However, this assumption cannot be made in the case of all materials as some are downcycled, therefore, a quality ratio was added to the calculation of the environmental credits, as outlined in Table 1. An indirect proxy was relied upon to set a value for this quality ratio. Although quality factors are often determined by measures associated with lab tests from industry, in the case of building materials this information is not readily available. Hence, in order to develop an admittedly coarse quality factor, the Canadian Council of Ministers of the Environment (2019) report was used. The report categorizes CRD waste materials by their ease of diversion from landfills. The ease of diversion categories it uses are 'high value', 'simple to divert', 'complex to divert' and 'limited options'. 'High value' category materials are defined as having well-established recycling technologies and markets that are economically viable in most regions. 'Simple to divert' are materials which have proven diversion technologies and processes available but for which in most regions some level of support is required to make them economically viable. 'Complex to divert' refers to materials where technological options for diversion exist but they are complex, under development and/or not economically viable without significant support. Finally, the report defines 'limited options' materials as those having no technological options for diversion currently available. For the purposes of this study, the quality factor, as outlined in Table 1, includes all four diversion categories. We assigned a numeric factor to quantify or rank the quality of each of these categories where 'high value' is 1.0, 'simple to divert' is 0.75, 'complex to divert' is 0.5 and 'limited options' is 0. To elaborate further, in this study 'high value' assumes that the recycled product will have the same use as the original product. 'Simple to divert' assumes a level of downcycling to a product of lower quality but where the technological viability to do so is

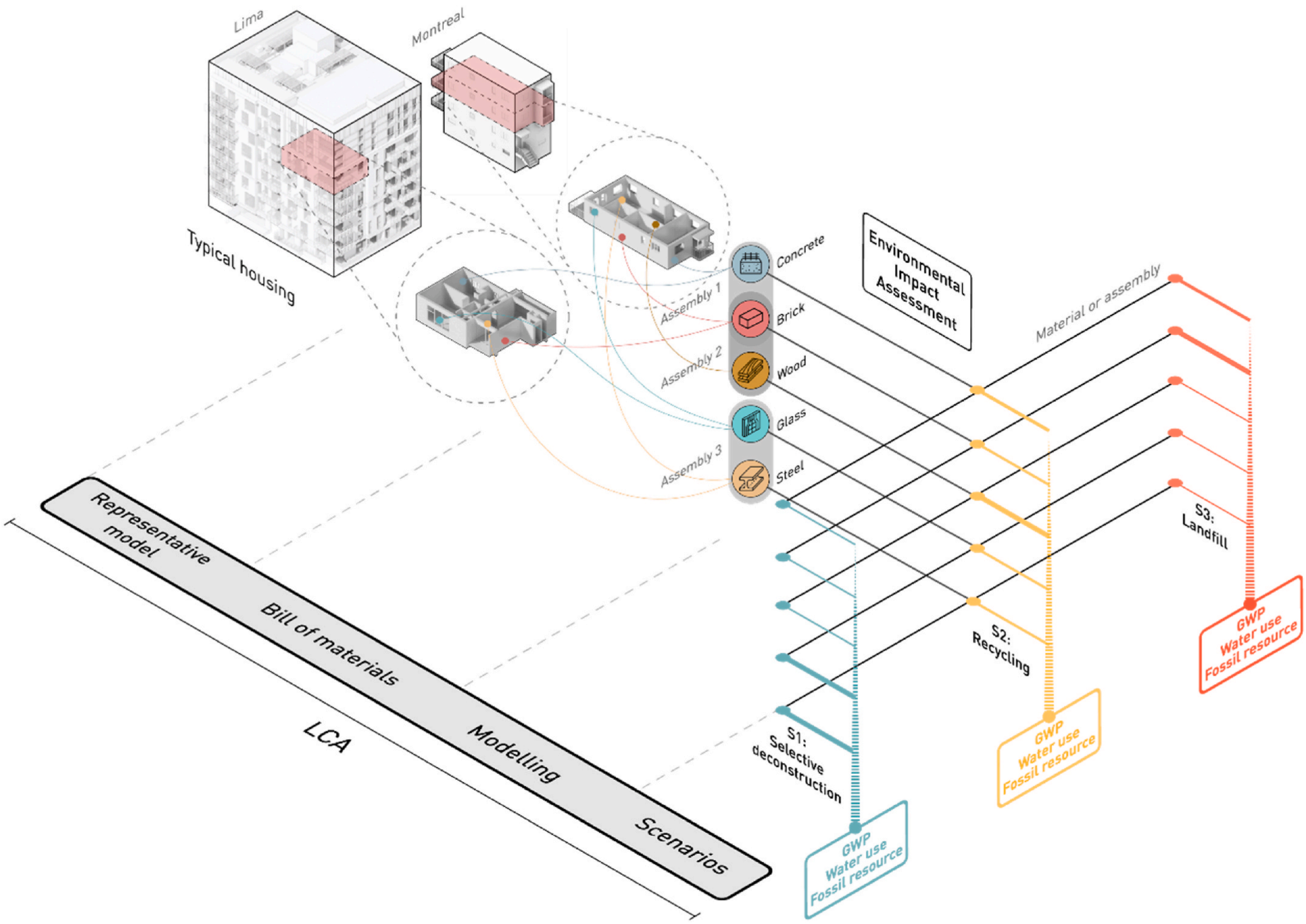

Fig. 2. Methodological Framework: illustrating 1) the typically housing in Montreal and Lima, 2) the development of a representative model (RM) for both housing types, 3) the bill of materials for those RMs, 4) using the LCA model an environmental impact assessment was conducted, 5) the three scenarios that were investigated via the LCA.

well established, e.g., concrete recycled to aggregates for roads and pavements. 'Complex to divert' assumes a level of downcycling where there is a significant reduction in the quality of the original product and the technological viability of recycling is complex and under development, e.g., architectural glass that is recycled to glass wool insulation or ceramic fritted and printed glass. The main issue with recycling architectural glass is overcoming the contamination issues due to laminates and coatings on glass. It is very difficult to completely remove these interlayer materials from the glass even with pulverizing techniques of delamination although recent studies highlight advancement and improvements in this process (DeBrincat and Babic, 2018). A detailed breakdown of the recycling assumptions made in this study which are based on a thorough literature review of the state-of-the-art recycling techniques for construction materials, is outlined in Table 1.

In the first scenario **(S1 – Selective Deconstruction),** the assumption was made that implementation of suitable DfD strategies would result in selected materials and assemblies being reused for a second life cycle, after their first use in the original building. A literature review was performed and the reusability shares for each material were established as outlined in Table 1. From a modelling perspective, when a material or assembly is destined for reuse, its contribution to the RM was amortized over two life cycles, and hence all its associated life-cycle impacts were considered to be halved. In some cases, however, the reuse percentage was lower than 100% because it was assumed (based on literature as indicated in Table 1) that there would be some losses in the recovery process for reuse. According to these literature-informed shares, in this scenario, the overall percentage of reuse in Montreal ended up being 77% by mass, while 21% of the total mass was recycled and 2% ended up being landfilled. In Lima, those percentages were, respectively, 84%, 15%, and 1%.

In the second scenario **(S2 – Recycling),** the assumption was made that no direct reuse would be possible, but that instead selected materials would be separately collected, and then closed-loop recycled to produce the same materials again or, if this is not possible, downcycled to materials of lesser value. The quality factor, as described above, indicates the level of downcycling, if any. According to these considerations, in S2 overall, in Montreal 94% of materials were recycled and in Lima 96% were recycled.

Finally, in the third and last scenario **(S3 – Landfill),** the assumption was made that all the materials would be 100% disposed of in landfills. Note that although some materials (like metals) are in fact currently partly recycled in both cities, the simplifying assumption of 100% landfilling was applied across the board in this scenario, to provide a 'worst-case' scenario.

2.2.3. Environmental impact assessment

The environmental impacts were assessed using the Environmental Footprint 3.0 (EF 3.0) Life Cycle Impact Assessment (LCIA) method.

This study is part of a global research initiative on "embodied carbon" in building materials (UNEP, 2023), and therefore the main focus was on GHG emissions; specifically, the EF 3.0 aggregated indicator for "climate change" comprises fossil, biogenic and land use and land use

Table 1
Recyclability and reusability index.

Material	Scenario	REUSE % of reuse	Source of % of reuse	RECYCLING % of recycling	Source of % or recycling	Recycling factor [a]
Technological viability and recycling output: high value						
Aluminum	S1-Selective Deconstruction	100%	Diyamandoglu and Fortuna (2015)	–		1.0
	S2-Recycling	0%		100%	Diyamandoglu and Fortuna (2015)	1.0
Reinforced steel	S1-Selective Deconstruction	85%	Eberhardt et al. (2019b)	98%	American Institute of Steel Construction (2023)	1.0
	S2-Recycling	0%		98%		1.0
Steel railing	S1-Selective Deconstruction	100%	Gorgolewski (2006)	–	–	1.0
	S2-Recycling	0%		98%	American Institute of Steel Construction (2023)	1.0
Wood (finished and structure)	S1-Selective Deconstruction	90%	Piccardo and Hughes (2022)	100%	Diyamandoglu and Fortuna (2015)	1.0
	S2-Recycling	0%	–	100%		1.0
Technological viability and recycling output: simple to divert						
Brick	S1-Selective Deconstruction	75%	Ergun and Gorgolewski (2015)	90%	Nordby et al. (2009)	0.75
	S2-Recycling	0%		90%		0.75
Concrete	S1-Selective Deconstruction	85%	Eberhardt et al. (2019b)	97%	Chen et al. (2022)	0.75
	S2-Recycling	0%		97%		0.75
Gypsum	S1-Selective Deconstruction	40%	Rasmussen et al. (2019)	94%	Jiménez-Rivero and García-Navarro (2016)	0.75
	S2-Recycling	0%		94%		0.75
HDF Wood	S1-Selective Deconstruction	90%	Diyamandoglu and Fortuna (2015)	95%	Mercante et al. (2012)	0.75
	S2-Recycling	0%		95%		0.75
Plywood	S1-Selective Deconstruction	60%	Piccardo and Hughes (2022).	100%	Diyamandoglu and Fortuna (2015)	0.75
	S2-Recycling	0%		100%		0.75
Technological viability and recycling output: complex to divert						
Asphalt roofing shingle	S1-Selective Deconstruction	25%		90%	Fagan (2021)	0.5
	S2-Recycling	0%		90%		0.5
Glass	S1-Selective Deconstruction	90%	Hartwell and Overend (2019)	90%	Hartwell and Overend (2019)	0.5
	S2-Recycling	0%		90%		0.5
Insulation Batt	S1-Selective Deconstruction	0%		0%		0.5
	S2-Recycling	0%		0%		0.5
Polyethylene	S1-Selective Deconstruction	90%	Lima's fieldwork	7%	Kulkarni (2018)	0.5
	S2-Recycling	0%		7%		0.5
Tiles and cladding panels	S1-Selective Deconstruction	100%	Rasmussen et al. (2019)	65%	García-Ten et al. (2015)	0.5
	S2-Recycling	0%		65%		0.5
Technological viability and recycling output: limited options						
Epoxide resin	S1-Selective Deconstruction	0,0%		0%	Kulkarni (2018)	0
	S2-Recycling	0%		0%		0
Insulation XPS	S1-Selective Deconstruction	90%	Fabian et al. (2004)	0%	Wiprächtiger et al. (2020)	0
	S2-Recycling	0%		0%		0
Paint	S1-Selective Deconstruction	0%		0%		0
	S2-Recycling	0%		0%		0
Polyurethane	S1-Selective Deconstruction	0%		0%	Vefago and Avellaneda (2013)	0
	S2-Recycling	0%		0%		0

[a] Values indicated are interpretations derived from qualitative values provided in a report from the Canadian Council of Ministers of the Environment (2019). Guide for identifying, evaluating and selecting policies for influencing construction, renovation and demolition waste management (p. 36).

change (LULUC) GHG emissions (kg CO_2-eq). It is also worth noting that the EF 3.0 method differs from other climate change impact assessment methods, such as e.g, the method adopted in EN15804+A2, in one significant way. The latter methods explicitly account for biogenic CO2 sequestration during biomass growth (resulting in large initial "negative emissions" for bio-materials such as wood), and then report correspondingly large CO2 emissions when these materials are incinerated. Conversely, EF 3.0 excludes these two specific flows from the accounting, thereby implicitly assuming that any CO2 sequestered during plant growth will later be released at EoL; therefore, when using EF 3.0, at no point during its life cycle does any material have a large "net negative" carbon budget (other biogenic emissions such as methane are instead explicitly accounted for because those have different global warming potential than CO2, and so they still actively contribute to climate change, beyond the "net zero" budget of straightforward CO2 uptake-and-release). In so doing, the EF 3.0 method prevents potentially misleading interpretations of "negative emission" materials in reduced-boundary cradle-to-gate analyses (i.e., when EoL is not included in the assessment). Another way of looking at this is that EF 3.0 adopts a "cautionary principle" approach that acknowledges that any amount of biogenic carbon that is initially sequestered will at some point end up being released, thereby resulting in a net zero contribution to climate change. It is however acknowledged that one draw-back of the EF 3.0 approach to climate change impacts is that it fails to highlight the possible short-term benefit of temporary carbon sequestration by bio-materials, resulting from the time-delayed release (at EoL) of the biogenic CO2 initially sequestered during biomass growth.

Furthermore, as water is a critical element in Peru and the use of fossil resources is an essential topic in the climate debate, the study also assessed water consumption (in units of m^3 water, world eq.). Finally, the overarching importance of reducing the reliance on non-renewable fossil fuels is addressed by the inclusion in the assessment of the fossil primary energy resources indicator (in units of MJ oil-eq.).

2.2.4. Limitations and uncertainties

One limitation of this study is that certain housing related materials were not included as part of the study such as fixtures, fittings, furniture, as well as mechanical, electrical, and plumbing services. In general, it was expected that these materials would be reused or re-sold before demolition. Secondly, the current real scenario for the EoL (i.e., a true current "baseline") was not modelled due to a lack of consistent and uniform data regarding the final use by material, both in Montreal and Lima. Although several semi-structured interviews were performed (Keena, et al., 2022a; Keena and Rondinel-Oviedo, 2022; Rondinel-Oviedo, 2021), the collected data was not consistent enough and more data collection is needed. Hence, since a baseline of the 'true' EoL scenario for both cities is a moving target, we instead chose to evaluate consistent scenarios where the percent of landfill, recycling, and/or reuse was the same across both cities. Given that a goal of this study is to offer a generally applicable method to other cities worldwide, we chose to omit a baseline that will constantly shift depending on location and avoid a scenario where the results are skewed and potentially misleading. Hence, for consistency we demonstrated the worst-case scenario (all goes to landfill), and the best-case scenario (selective deconstruction). Finally, average transport distances were simply estimated from the city center (due to higher construction density) to landfills and recycling facilities.

3. Results and discussion

3.1. Environmental impacts of housing in Montreal, and Lima

The environmental impacts of a typical housing unit in Montreal and Lima based on the defined FU are presented in Table 2.

As outlined in Table 2 and illustrated in Fig. 3, the total embodied carbon emissions in Montreal were reduced by 63% by using selective deconstruction (S1) instead of landfill. The reduction in embodied carbon emission was 48% when recycling (S2) was used as opposed to landfill. In the case of Lima, the reduction in GHG emissions was even greater, with a 70% reduction in embodied carbon emissions in S1 and a 50% reduction in S2.

In terms of "embodied" water use (i.e., once again consistently calculated over the full life cycle of the building, but excluding the use phase), the total volume used in the "landfilling" scenario (S3) in Montreal was 14,600 m^3. This is equivalent to 5.8 full Olympic swimming pools. The corresponding results for the S3 in Lima was 16,500 m^3 of water use across one housing unit life cycle, equivalent to 6.6 full Olympic swimming pools. Fig. 4 illustrates that in Montreal, selective deconstruction (S1) results in 53% less water use than landfilling. In Lima, the equivalent comparison results in 67% less water use. Recycling (S2) as opposed to landfilling, results in a 29% savings in water use in the case of Montreal, and a 45% saving in water use in the case of Lima. These large reductions in water use in S1 and S2 in Lima are significant, particularly in the face of growing climate change where water scarcity is becoming a growing issue throughout Peru (UNOPS, 2023). Also, a comparison between recycling and reuse processes in the two locations, found that re-use via selective deconstruction (S1) in Montreal consumes 26% more water than in Lima and recycling (S2) consumes 15% more.

Lastly, in terms of "embodied" fossil resource use, the S3 scenario results point to 274,000 MJ in Montreal, and 325,000 MJ in Lima. As illustrated in Fig. 5, the potential reduction in fossil resource use by using selective deconstruction (S1) as opposed to landfill (S3) is 48% in the case of Montreal and 69% in the case of Lima. Similarly, recycling (S2) as opposed to landfill (S3) results in a 28% reduction in fossil resource use in the case of Montreal and a 48% reduction in the case of Lima.

When interpreting of all these life cycle impact results, it is important to underline that, in all cases, the largest shares of the impacts always occur during the early life-cycle phases of material sourcing and manufacturing. The key reasons for the improvements (reduction in impacts) shown for scenarios S1 and S2, relative to S3, are, respectively: (i) because these material sourcing and manufacturing impacts are "amortized" and "shared" over two building lifetimes (S1); and (ii) because impact "credits" are assigned to the materials that can be recycled (S3), to account for the displacement of their virgin counterparts.

3.2. Decomposition analysis considering construction assemblies and materials

Figs. 3–5 illustrate the results of a decomposition analysis whereby the total "embodied" GHG emissions, water use, and fossil resource use for all three scenarios are broken down with respect to the main building assemblies (walls, columns, floors, etc.), and then to the building materials (concrete, brick, steel, etc.).

3.2.1. "Embodied" GHG emissions

When considering the construction assemblies, Fig. 3 (upper panel) shows that in Montreal, the three construction assemblies with the highest GHG emission impacts are the walls, the floors, and the foundation (the latter more so in the S3 scenario, and far less in S1 and S2). These impacts are related to the low reusability and recyclability indices of some component materials such as gypsum or insulation materials (see Table 1). For Lima, the assemblies with the highest impact are the concrete walls, the foundation, the concrete beams, and the concrete columns (the latter two more so in the S3 scenario, and far less in S1 and S2).

Table 2
Environmental impacts of typical housing units in Montreal, and Lima. The FU is one dwelling for 4 people. All results are rounded to three significant digits to avoid misrepresenting the level of precision attainable in LCA calculations.

	Montreal			Lima		
	S1	S2	S3	S1	S2	S3
GHG emissions ($kgCO_2$-eq)	7150	10,100	19,300	9790	16,200	32,300
Water Use (m^3)	6840	10,400	14,600	5400	9000	16,500
Fossil Resource Use (MJ)	143,000	196,000	274,000	102,000	169,000	325,000

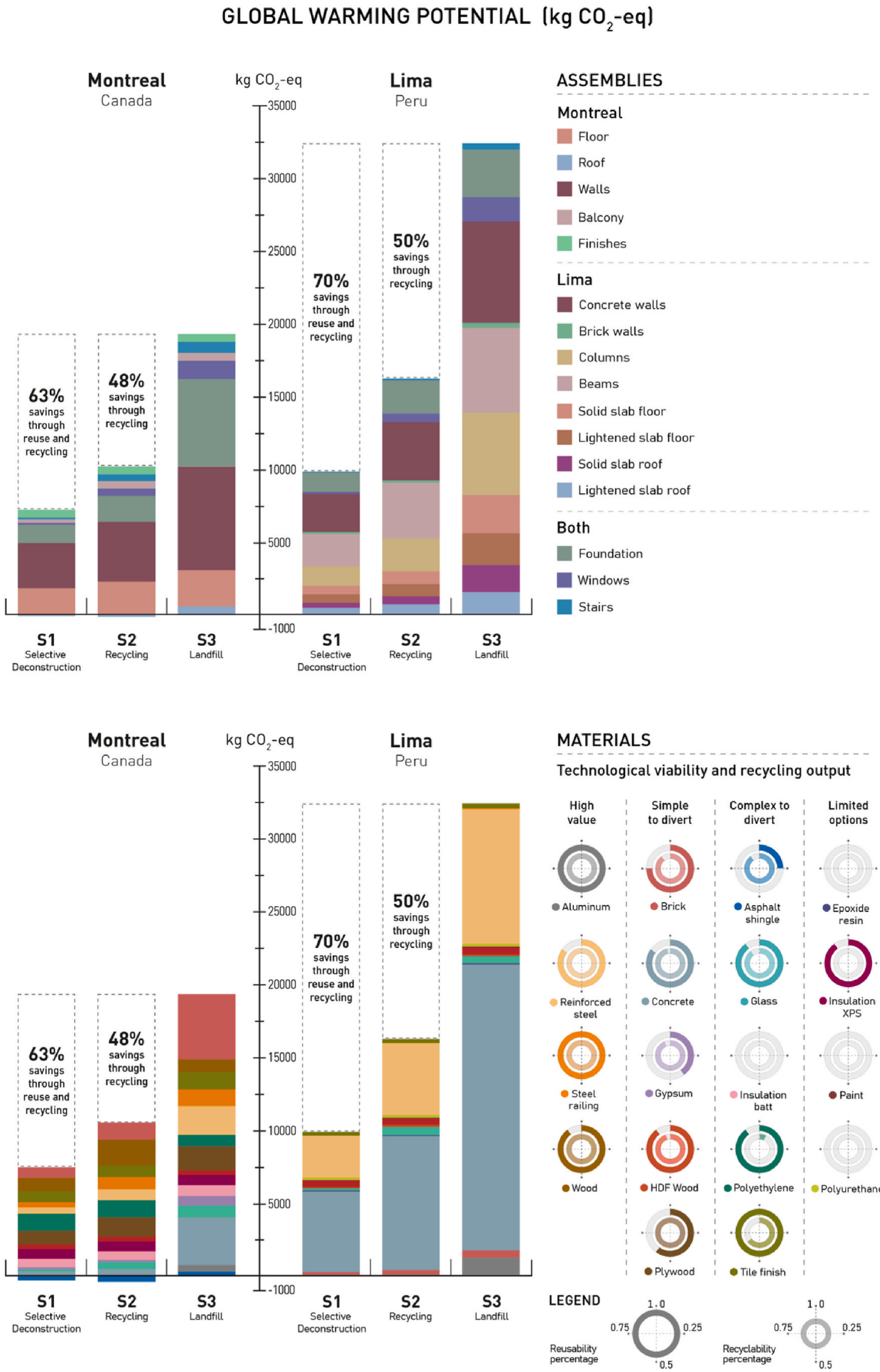

Fig. 3. "Embodied" GHG emissions (calculated over the full life cycle of the building, but excluding the use phase), by construction assemblies and materials for each city; three scenarios (S1 – S3).

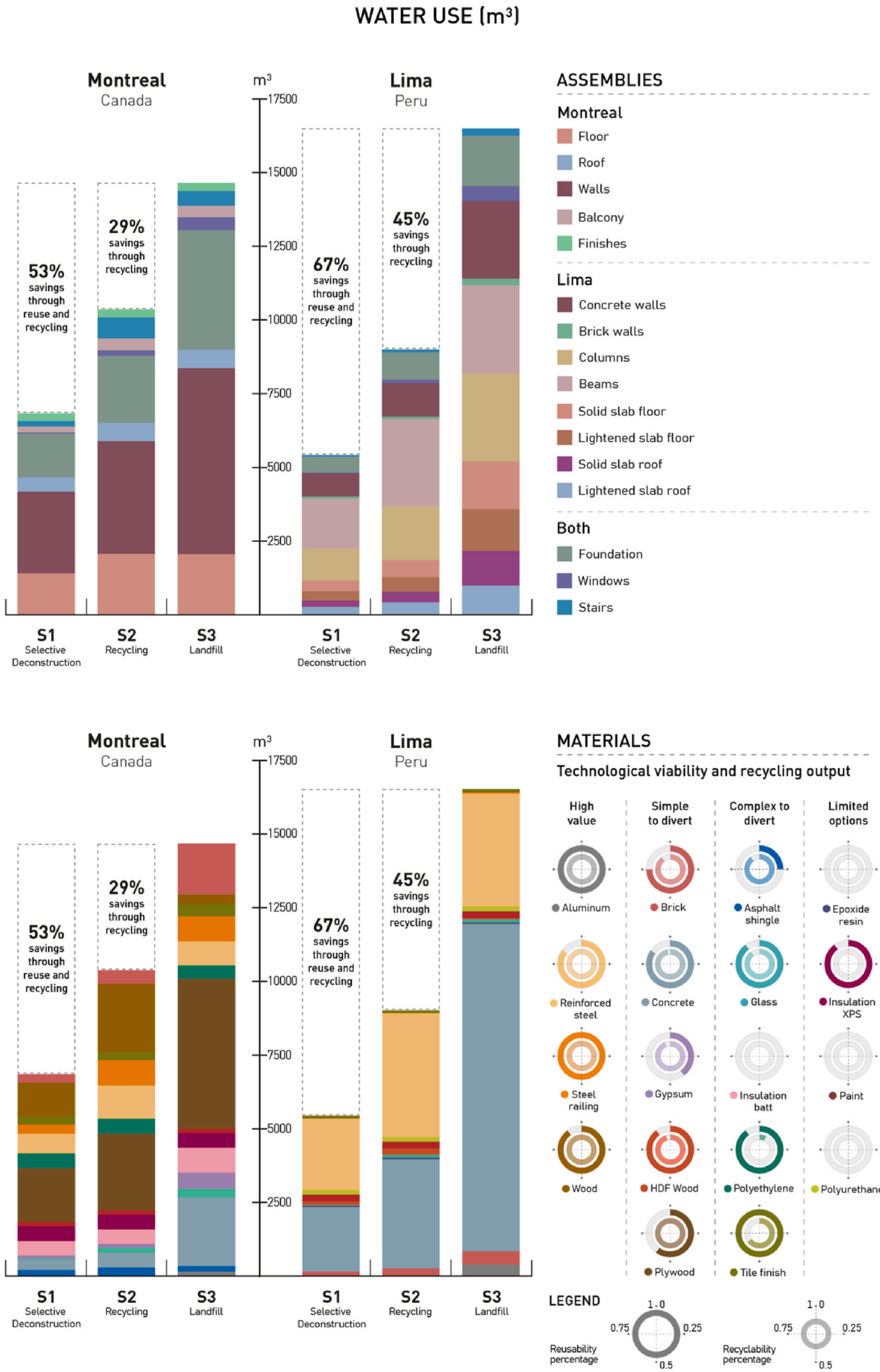

Fig. 4. "Embodied" Water Consumption (calculated over the full life cycle of the building, but excluding the use phase), by construction assemblies and materials for each city; three scenarios (S1 – S3).

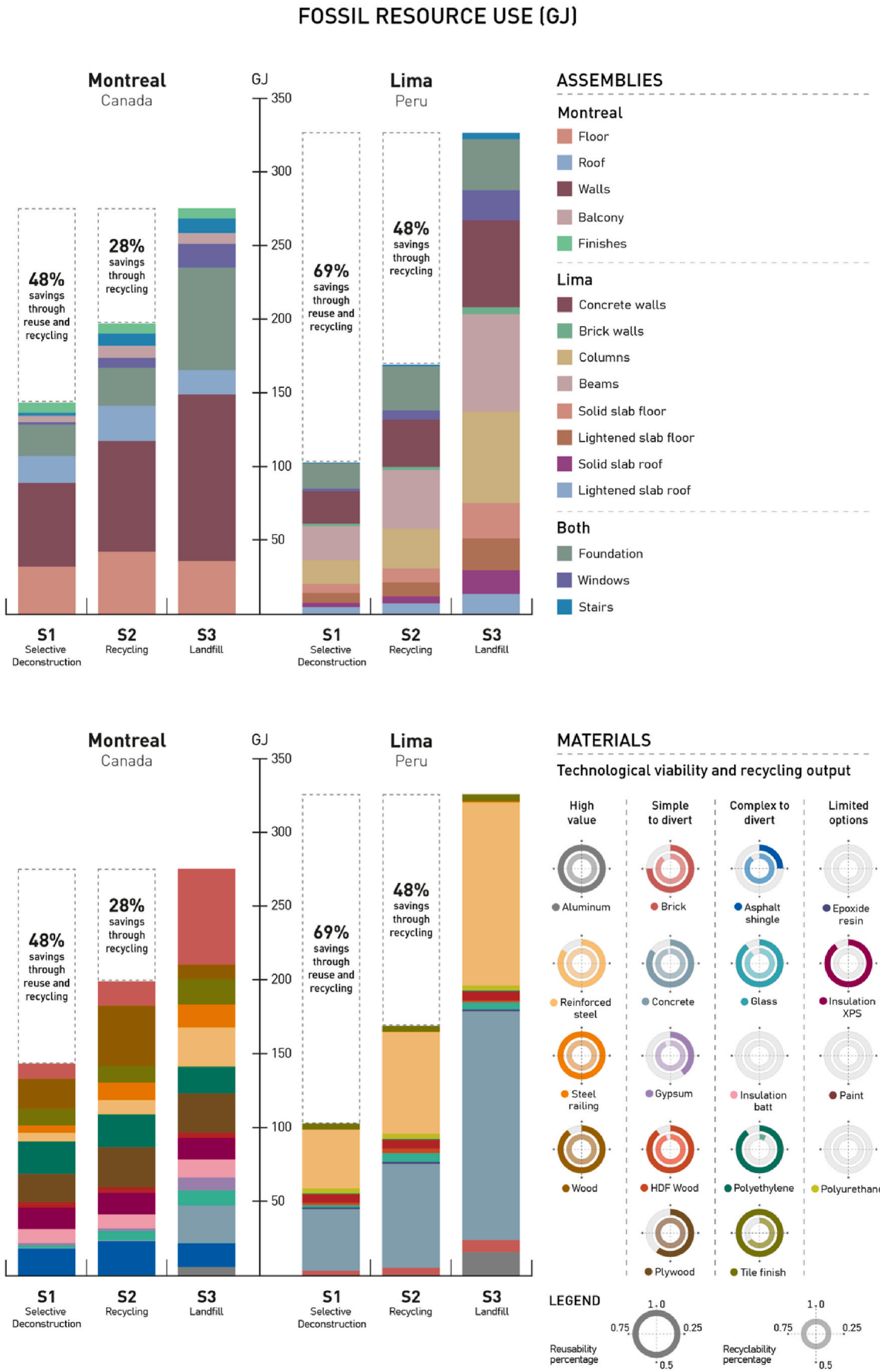

Fig. 5. "Embodied" fossil resource use (calculated over the full life cycle of the building, but excluding the use phase), by construction assemblies and materials for each city; three scenarios (S1 – S3).

When instead looking at the decomposition analysis in terms of materials (Fig. 3 - lower panel), some clear indications emerge, across both locations. In particular, it becomes clear that those materials which can be more readily recycled or reused are responsible for the largest share of the reductions in GHG emissions achieved under the two improved scenarios (respectively, S2 and S1), vs. the "worst case" scenario (S3). The three foremost such materials are brick, concrete and steel. More specifically, given that the GHG emission impacts under S3 in Lima are overwhelmingly dominated by concrete, and to a lesser extent by steel, moving respectively to S2 and S1 easily achieves the largest reductions in impact in this location. The situation for Montreal is a bit more nuanced, principally because the decomposition analysis for the "baseline" S3 scenario shows a more complex break-down, where no single material dominates clearly. As a result, when moving from S3 to S2 and S1, the relative percentages of impact by material shift, with structural wood becoming the largest contributor, in relative terms.

In the case of structural (sawn) wood, the environmental impact associated with waste was greatest in S2. This is due to the fact that for S2, as outlined in Table 1, it was assumed that 100% of the primary-cycle wood would be recycled. The recycling factor is 'high value' meaning that in the case of structural wood it is technologically feasible to recycle wood to be used in a secondary cycle as a structural component. However, when recycled, the wood typically ends up as a composite panel rather than a sawn wood component. In this study, one such typical recycling process was modelled, which converts sawn wood into an engineered board which can form a structural panel. However, this involves a process of firstly shredding and sorting the wood chips and then re-manufacturing those wood chips into an engineered board. This processing is both energy and water intensive, and that is why scenario S2 ends up being significantly more impactful than S3 in terms of GHG emissions, water use, and fossil resource use. In other words, in S2 the recycling impacts to produce the engineered composite panels are higher than the avoided impacts or 'credits' due to displacement of virgin sawn wood. As a result, in the case of structural wood materials, the S3 scenario, which assumes the single use of virgin sawn wood as part of a linear life cycle (ending with landfilling), is actually characterized by lower GHG emissions, water use, and fossil resource use. Finally, in S1 (the re-use scenario), similarly to S2, when wood is finally recycled, the same credit is also given for displacing an equal amount of virgin sawn wood. However, in S1 such calculation only applies to half of the virgin sawn wood, since the impacts (and credits) of the re-used materials are amortized over two life cycles, as explained in Section 2.2.2. This results in S1 having higher GHG emissions, water use, and fossil resource use than S3, but lower than S2.

3.2.2. "Embodied" water use

In terms of assembly decomposition, the results for water use largely mirror those previously discussed for GHG emissions. When moving to consider the decomposition by materials, in Lima, the impacts are once again largely dominated by concrete and steel, with concomitant strong reduction potentials when shifting from S3 to S2 and S1, respectively. In Montreal, instead, the material that ranks first in terms of water use in the S3 scenario is plywood, again with significant margins for reduction in S2 and S1, respectively.

3.2.3. "Embodied" fossil resource use

In terms of assembly decomposition, the results for fossil resource use largely also fall in line with those for GHG emissions and water use, for both locations.

Then, in terms of decomposition by materials, the results for Lima are once again, unsurprisingly, absolutely dominated by concrete and steel, with large margins for reduction in S2 and S1 (vs. S3), especially in the case of steel.

The results for Montreal are again more nuanced, since no single material clearly dominates in terms of relative importance in S3, and the reductions in S2 and S1 are proportional to the individual recyclability and re-usability of the individual materials. It is also worth noting that the impact of wood in S1 and S2 is respectively 350% and 700% higher than in S3, with wood being the only material that behaves in this way. The reason for this mirrors the "embodied" GHG emissions case, and is a result of the recycling process, as explained above in Section 3.2.1.

3.3. Discussion and comparison of results for Montreal vs. Lima

In this section, a comparison of the results for a housing unit in Montreal vs. Lima is outlined. In the case of S1, recycling and re-use in terms of selective demolition led to 63% and 70% decrease in environmental impacts compared to landfill, in Montreal and Lima respectively. There is high potential for mitigating environmental impact in both cities, despite the differences in materials and construction methods used. Such mitigation potential is primarily due to avoiding landfills and recovering material from selective demolition for reuse, through the application of DfD methods (Rios et al., 2015). Hence, it is important to note that such mitigation relies on the implementation of DfD strategies and the practice of deconstruction which are currently not ubiquitous in the construction industry. In addition, the implementation of such processes may face further challenges in cities where construction is less industrialized, as in the case of Lima.

S2 shows that recycling-only strategies (without reuse) led to a decrease in GHG emissions of 48% for Montreal and 50% for Lima. The most efficient materials in terms of recycling are metals such as steel and aluminum, which are commonly recycled due to their high economic value. The savings for S2 are proportionally lower for Montreal because of insulation and structural wood elements. As described above current structural wood recycling methods imply a change of material from lumber to composite panels which consume high amounts of water and energy and use additional materials such as resins in their transformation process to make the structural composite. The results also clearly outline the key materials which pose a challenge but also hold a high potential for recycling and reuse, in terms of reducing life-cycle GHG emissions: concrete, brick, gypsum, and glass. Technological advancements towards 1) making these materials more readily reusable and recyclable, and 2) more energy and material efficient recycling processes, could lead to significantly lower embodied carbon impacts during materials recovery.

For S3, the LCA shows that a new multifamily housing in Lima generates almost twice the environmental impact vs. an equivalent one in Montreal, if all the materials end up in a landfill (S3). This can be attributed to the housing structural frame in Lima (composed of reinforced concrete), which implies high emissions when it cannot be recycled. In fact, for both case studies, in S3, the lack of reuse or recycling in assemblies composed of high embodied carbon materials such as steel and concrete, results in a significant impact. The finishes (paint, sealants, etc.), which are not recyclable, have the same impact in the three scenarios, due to the low reusability and recyclability potential of those materials.

Finally, given that the construction industry often uses a metric of environmental impact per area of floorspace, e.g., $kgCO_2$-eq/m^2, as a means to benchmark the embodied carbon of buildings, we aimed to study the results using this metric, too. When considering the environmental impact per unit area, the results for scenario S3 show that a typical home in Lima has higher GHG emissions than in Montreal, by as much as 180 $kgCO_2$-eq/m^2. The results are aligned with benchmarked climate change impacts reported by Simonen et al. (2017) where the median is 462 $kgCO_2$-eq/m^2.

4. Conclusions

The rapid growth of urban environments in the Global South implies that the associated GHG emissions are likely to continue to grow. Therefore, material efficiency should be a key consideration in building design. Designing for extended service-life, disassembly, deconstruction, recycling, and reuse should all be encouraged. Recycling and reusing can promote the local value chain by reducing the demand for virgin imported materials.

The recommendations that are given to achieve Net Zero by 2050 (IEA, 2022; Masson-Delmotte et al., 2021) points towards the decarbonization of particularly the cement and steel industries. Fennell et al. (2022) outline how decarbonization of buildings can be achieved via four steps: using fewer materials to build, switching manufacturing processes, using low-carbon heat sources, and developing carbon capture and storage strategies (Fennell et al., 2022). Additionally, it is critical to facilitate the transition to local bio-based materials which means replacing some carbon-intensive technical components and materials with regenerative resources, which absorb and store carbon. Technologies and techniques are established that allow for the decarbonization of mineral-base materials as well as the increase of bio-based manufacturing and construction. However, the regulation, infrastructure, and financial frameworks needed to support them are lacking.

Although the substitution of conventional mineral-based building materials with biobased alternatives has been shown to offer one pathway towards reduced GHG emissions (Keena et al., 2022b), our results show that this is not a clear-cut solution when considering multiple life cycles. In other words, if a circular approach is followed, our findings, as highlighted in the case of structural lumber, indicate a scenario where the recycling process to convert that lumber into a secondary cycle engineered structural member, requires more energy and materials in re-manufacturing than sourcing the virgin sawn wood counterpart would. Additionally, the materials added in this recycling process typically include synthetic resins significantly limiting the recycling of any subsequent engineered boards of similar value. The limitation arises when biological and technical nutrients are mixed into a composite or hybrid with contamination of the wood making subsequent recycling very challenging. This mixing of nutrients would also make processes such as compositing difficult. As although the core material is biobased, the addition of the synthetic resins (i.e., wood is bonded with polyurethane reactive adhesives) would make composting very difficult as such hot-melt adhesives are rarely biodegradable. However, it is important to point out that the results are a snapshot in time, highlighting a current situation. They should not be interpreted as a (potentially counterproductive) message to rely solely on linear economies for natural resources, or to avoid recycling altogether in the case of wood. If demand for wood or other virgin biobased materials were to increase exponentially, there is a clear risk of deforestation, changing land use patterns, and loss of biodiversity if sustainable forestry is not practiced. Hence, more research is needed to refine the recycling processes of wood products and to avoid contamination of the biobased materials with synthetic additives.

In response to this limitation of biomaterials, a growing area of research is bio-resins which, if used, would prevent the mixing of biological with technical materials, and offer promise in the recycling and recovery process. However, whether synthetic or biobased, many resins used (e.g., polyurethane in the case of mass timber) have a shorter service life and per a circular economy logic, the structural components of a building should last and not be replaced for at least a hundred years to avoid the GHG emissions associated with EoL. The current service life of many laminated structural timbers is 60 years.

Contamination of materials is not limited to wood, but it is rather commonplace in many other building materials, too. For example, it is also seen in architectural glass. This is primarily due to the inclusion of adhesive products that complicate the recovery process.

Our results indicate that, for most materials, selective demolition with reuse offers the greatest reductions in GHG emissions. However, material reuse faces numerous operational challenges, as well as a lack of development and trade/business frameworks (Knoth et al., 2022). Operational challenges are associated with technical limitations (e.g., lack of deconstruction contractors, material degradation, seismic and fire-proof specifications), social limitations (e.g., stakeholder mindset and acceptance, perceived security, and absolute value), and legal limitations (e.g., the lack of a regulatory framework and a gap of technical studies). There is a necessity to promote a reuse market, involving incentives for the creation of CRD reuse centers that concentrate materials in a "one-stop shop" and the promotion of specialized contractors (Forrest, 2021). In both cases, elements with a higher value are resold before hitting sorting facilities, reaffirming the idea that economic drivers are as effective as legislation (King, 2021). Consequently, design for reuse should be addressed on multiple scales, including strategies for design for disassembly and stakeholders' participation (and training) from all building phases (Cruz Rios and Grau, 2020; Deplazes, 2012; Hossain et al., 2020; McClure et al., 2007; Rondinel-Oviedo and Schreier-Barreto, 2019).

The weight and dimension of the element or material are significant factors when it comes to reusability. Lighter and smaller elements that are designed with flexible joints will be more easily reused. In regions where reinforced concrete is commonly used, the reuse of structural elements is less viable, representing a challenge for engineers and architects since the design of reusable structural elements will need to address aspects such as the seismic resistance of materials in earthquake-prone regions. In contrast, in those regions where lighter materials are preferentially used, the potential for reusability is higher. However, today, due to challenges with recycling wood as highlighted above, secondary lightweight wood is mostly used for energy recovery. Many end-of-use strategies and pathways exist for different materials. Each strategy will offer different levels of carbon emission reductions as is illustrated in this study for the housing sector of Montreal and Lima.

It is vital to help the industry adapt and modernize, and it will be essential to periodically update building codes to consider technological advances, which will ideally encourage manufacturers to produce cement and concrete with the lowest "embodied" carbon. In the meantime, national and international standards must become more flexible to facilitate a shift to material reuse. For example, there is a need to develop standards to ensure the quality and efficacy of secondary materials. This will provide assurance to building sector actors who currently are faced with risk when specifying secondary materials in a building design. Additionally, existing regulations are a critical challenge, as legislation limit the reuse of recovered assemblies and materials and in addition prevent proper handling of CRD materials. This generates significant waste during construction and post-demolition, illegal dumping, and little or no recycling of non-metallic materials. Public-private partnerships for the use of recycled materials in construction, including social housing and infrastructure, should be also promoted.

Future studies could consider the housing stock at the urban or global scale to understand the potential GHG emissions savings in reusing and recycling the material stock in future constructions. This is particularly relevant in those countries that are currently undergoing and planning major retrofit programs. In addition, up-stream design choices of building with less, DfD, material substitution and light-weight design could be studied to understand how CE practices could result in lower GHG emissions in new construction. Finally, the results can be compared based on population and building growth according to projections (Global North and Global South) where the study allows the GHG emissions savings to be predicted if the reuse and recycling scenarios are carried out.

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

Data will be made available on request.

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List of abbreviations

BIM	Building Information Modeling
BOM	Bill of Materials
CDW	Construction Demolition Waste
CRD	Construction, Renovation and Demolition
DfD	Design for Disassembly
EoL	End of Life
FU	Functional Unit
GHG	Greenhouse gas
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
RM	Representative Model

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