Towards circular and energy-efficient management of building stock:

an analysis of the residential sector of the Netherlands

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Table of contents

Chapter 1

General introduction

1.1 Importance of transforming the built environment towards a circular economy

1.1.1 From a linear economy to a circular economy

As the world moves toward a future where 70% of the population lives in urban areas, the linear "take, make, and dispose" economic model is still the dominant mode of production. This linear economic model results in serious resource supply risks and pressures of emissions and waste generation (Ellen MacArthur Foundation 2015a). The global resource extraction in 2010 is 10-fold higher compared to 1900 (Krausmann et al. 2017). As it is depicted in Figure 1.1 the global extraction of natural resources in 2011 was around 74 Gt (Aguilar-Hernandez et al. 2019). About 40% of the global material use was added to in-use stocks (30 Gt), and 54% were directly dissipated as emissions of or other combustion and biomass residues to the environment (40 Gt). Total waste generation amounted to 9 Gt, of which 25% was from stock depletion and 75% from direct waste generation. Global solid waste generation rate rose from fewer than 0.3 million tons per day in 1900 to more than 3.5 million tons per day in 2010, and it would double in 2025 and triple by 2100 (Hoornweg and Bhada-Tata 2012).

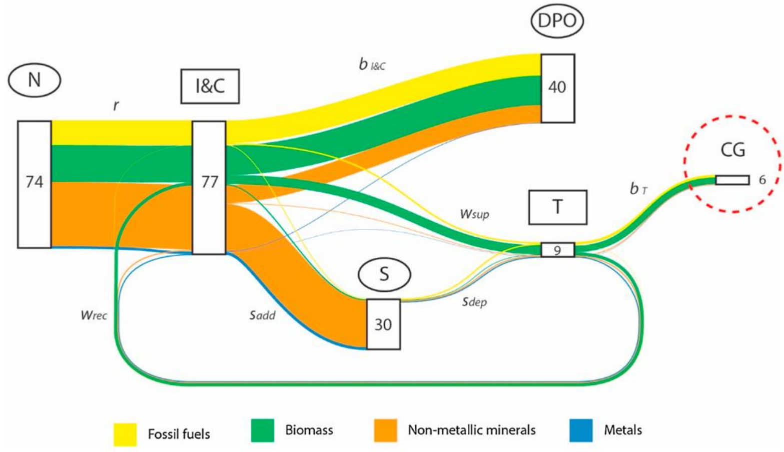

Figure 1.1 Sankey diagram of global material extraction, waste, and emission flows in 2011 (Aguilar-Hernandez et al. 2019). Numbers indicate the size of flows in Gt. Solid blocks indicate economic activities of: I&C=Intermediate sectors and final demand; T = waste treatment activities. Solid circles

indicate resource stocks of: N = Natural resources; S = Material in-use stocks; DPO = Domestic processed output. Colored lines indicate flows of: r = material extraction; W_{rec} = waste recovery; S_{add} = stock additions; S_{dep} = stock depletion, Wsup = waste generation; $b_{I\&C}$ = dissipative emissions, others combustion and biomass residues from intermediate activities and final demand; and b_T = dissipative emissions and others combustion and biomass residues from waste treatment. Dashed circle denotes circularity gap (CG).

An alternative “circular economy” would close loops in industrial ecosystems through lifetime extension of products, reuse of products, refurbishing and recycling, so that the generation of waste is prevented and resources are kept in use for much longer periods. This circular development model seeks to ultimately decouple global economic development from finite resource consumption. Korhonen et al. (2018) illustrated the win-win-win potential of circular economy to all the three dimensions of sustainability: economic, environmental, and social. The Ellen MacArthur Foundation quantified benefits of the circular economy development path that it could reduce primary material consumption in Europe by 32% by 2030 and 53% by 2050 while increasing European GDP as much as 11% by 2030 and 27% by 2050, compared with 4% and 15% in the current development scenario (Ellen MacArthur Foundation 2015b).

The concept of circular economy is complex, and different interpretations of the concept have been provided. Blomsma and Brennan (2017) presented interpretations of what a circular economy means according to different actors, such as seminal thinkers, think tanks, advisory and legislative institutions, academics, and businesses. Despite the diverse interpretations, a circular economy is most frequently depicted as a combination of reducing, reuse, and recycling activities, often summarized as “3R” (Kirchherr et al. 2017). For example, the Ellen MacArthur Foundation outlined a conceptual framework of the circular economy for technical and biological cycles, as shown in Figure 1.2. In a biological cycle, biologically-based materials are designed to flow back to the system via processes such as composting and anaerobic digestion. Inert materials such as concrete are classified as technical nutrients used in the technical cycle. Lifetime extension (e.g. by designing buildings for long life, maintenance and refurbishing) is usually the preferred circular strategy for technical nutrients. Next to this, technical cycles can be closed by reuse, repair and recycling of end-of-life (EoL) goods such as waste concrete at product-level (building), component-level (concrete elements), and material-level (constituents like gravel, sand, and cement).

In recent years, by promoting the adoption of closing-the-loop production patterns, circular economy is receiving increasing attention worldwide (Ghisellini et al. 2016). Major efforts have been done to develop theories and practical implementation tools in support of a circular economy (Kalmykova et al. 2018). The European Union (EU), USA, Korea, Japan, China, and Vietnam have adopted circular economy in material circularity policies at country and regional levels (Sakai et al. 2011).

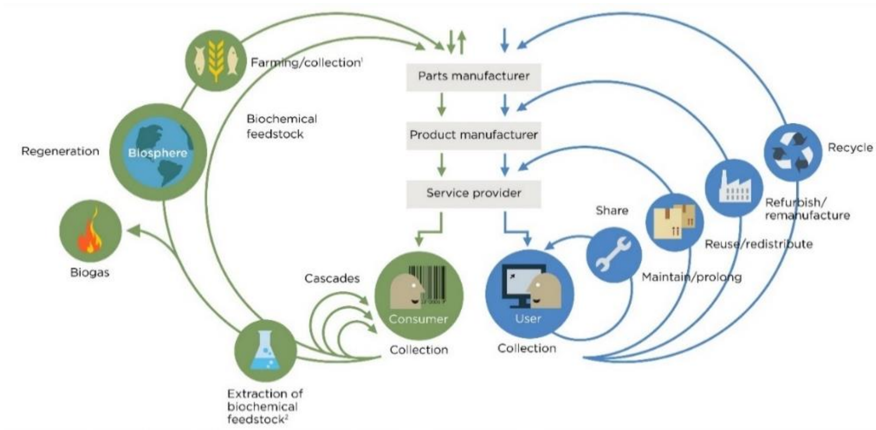

Figure 1.2 Outline of a circular economy framework by the Ellen MacArthur Foundation (2013)

1.1.2 Circularity gap within the built environment

Despite the increasing policy attention on circular economy strategies, the cycles of materials in the built environment are still far from closed. Global consumption of building materials is soaring. As indicated in Figure 1.1, approximately 40% of global material outputs in 2011 ended up as an addition to in-use stock (i.e. material for buildings, infrastructure, and products with a long lifespan). The built environment dominates here: it accounted for roughly half of the raw material use globally in 2015 (IRP 2019). In 2015, the total material input to satisfy the need for dwellings was 41 billion tonnes, and the accumulated building stock in the last two centuries until 2015 totals 832 Gt (de Wit et al. 2019).

Waste generated by the construction sector is usually called construction and demolition waste (CDW). CDW is one of the heaviest and most voluminous waste streams generated in the European Union (EU), and it accounts for approximately 25%–30% of all waste generated in the EU (EC 2020a). Large fractions of CDW are inert and recyclable. Over 90% of the value embedded in CDW comes from metals such as steel and copper (Zhang et al. 2020b). Metals are a high-value stream in CDW and are often already recycled to a high degree (Koutamanis et al. 2018). However, in terms of volume, most of the materials in in-use stocks in the built environment are non-metallic mineral materials (see Figure 1.1). Water use apart, concrete is the most used material in the global economy by mass (Monteiro et al. 2017), with a consumption of 3 tons per capita per annum (Gagg 2014). The production of concrete has exceeded that of other building materials over the past half-century (Miller et al. 2018). Globally, around 3.8 Gt of cement and 17.5 Gt of aggregate were used as constituents for concrete production in 2012 (Miller et al. 2016).

De Wit et al. (2019) quantified the global gap of material circularity and found that global material circularity was only 9% in 2015. About 4.3 Gt of materials flow into Europe's built environment every year with more than half of the resources used for maintenance and renovation, the circularity of Europe's built environment is estimated at just 12%; while in China less than 2% of the built environment can be considered circular (de Wit et al. 2019). Therefore, considering the essential role the built environment is playing in material use and stock formation, improving the material circularity in the built environment is crucial to reduce the circularity gap.

1.1.3 Material circularity interventions for the built environment

The objective of the circular economy is to maximize value at each point in a product's life, instead of minimizing the costs of collection and disposal as that has been doing in traditional waste management (Stahel 2016). Based on the 3R principle of circular economy, the Waste Framework Directive (WFD) (EC 2008a) recommended a five-level "waste hierarchy" illustrating a priority order from the most preferable option of "prevention" at the top to the least preferable option of "disposal" at the bottom. Aguilar-Hernandez et al. (2018) summarized four categories of circularity interventions, namely product lifetime extension, closing supply chains, resource efficiency, residual waste management. Eberhardt et al. (2019) proposed three main circularity strategies for the built environment: (i) reuse of entire building; (ii) reuse of building elements/components/modules; (iii) recycling of building materials. Building upon their achievements, this study proposes a "*Building circularity framework*" to categorize the material circularity interventions for the built environment (see Figure 1.3).

The framework comprises three categories of interventions, which are: (i) Reduce resource use at the building level, such as eco-design for new buildings and lifetime extension of existing buildings, (ii) Reuse at the element-level, such as reuse of building components, and (iii) Recovery at the material-level, such as upcycling/recycling/downcycling of CDW.

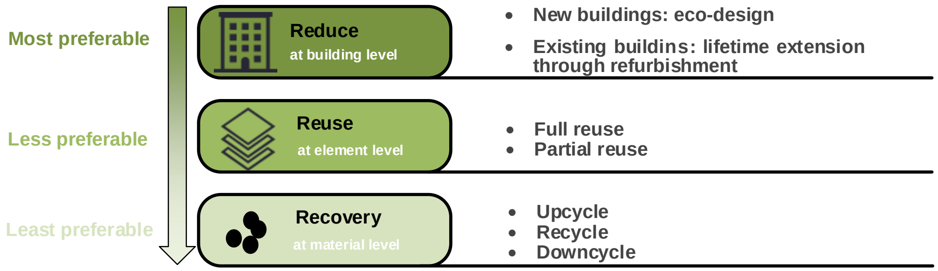

Figure 1.3 Building circularity framework based on 3R concept. 3R: reduce, reuse, and recycle. The circularity interventions are presumed based on (EC 2008a; Gharfalkar et al. 2015; Aguilar-Hernandez et al. 2018; Eberhardt et al. 2019).

Category I: Reduce resource use at the building level

Reducing resource use at the building level is the most preferable type of circularity intervention. For the built environment, it can be further classified into two: interventions for new houses and for existing houses.

For new buildings, the most important intervention is eco-design. Eco-designs at the concept stage can prevent CDW generation and improve material efficiency. Under this intervention, design for durability, lightweight design, and design for recyclability are the most considered strategies (Zhang et al. 2022). Prefabrication has been identified as a solution to reduce material use during concept and construction phases (Tam et al. 2006). An example of such an approach was developed in the EU-H2020 project - VEEP [1]. The project was launched in 2016 to develop the technologies that can produce clean aggregates from waste concrete and to design eco-friendly prefabricated concrete elements (PCEs) by using recycled aggregates. Using the technological possibilities offered by the VEEP project, this thesis will explore the potentials of massive retrofitting of the built environment on circularity and energy efficiency. In this thesis, as an illustrative technology the production and use of the PCE system for cladding walls in new houses (PCE-new) will be analyzed.

For existing buildings, lifetime extension is the most obvious circularity strategy saving the use of primary resources. However, particularly the older existing buildings often have a low energy performance which is problematics in view of the need to move to an economy with net-zero carbon emissions. Demolishing and rebuilding according to high energy standards to realize climate change targets is conflicting with the circularity pursuit of the building sector. Energy renovation seems a solution for existing buildings to achieve circularity and climate goals by prolonging the service life and improving

[1] The full name of VEEP project is “Cost-Effective Recycling of CDW in High Added Value Energy Efficient Prefabricated Concrete Components for Massive Retrofitting of our Built Environment” (http://www.veep-project.eu/).

energy performance simultaneously. There are many ways of how a building could be retrofitted to improve its energy efficiency, such as floor, roof, façade, ceiling, heating system, etc. (Akanbi et al. 2018). Renovating façades with PCEs is a common way to upgrade the energy performance of existing buildings. In this thesis, another PCE system (PCE-refurb) is used to refurbish walls of existing houses.

Category II: Reuse at the element level

Reuse at the element level is less preferable than waste prevention at the building level. It comprises full reuse that leads to reuse of an entire building element and partial reuse that only leads to reuse of some components of a building element (Zhang et al. 2021a).

With a growing trend towards prefabrication, concrete elements show an increasing potential for reuse The reuse of entire structural concrete components is extremely rare because structural components in building such as beams, columns, and floor slabs are often designed to resist very specific loading and there are limited opportunities to reuse them (Purnell and Dunster 2010). Besides, dismantling in-situ structures may cause structural damage to concrete components. Thus, partial reuse is considered more feasible than full reuse at the element level, especially for structural elements.

In the VEEP project, a novel multilayer PCE has been developed for full reuse of the element. Instead of permanent connections such as welded or grouted dowel connections to the wall, bolted connections and post-installed anchors are used in VEEP PCEs to allow the element to be quickly and easily disassembled. Thus, with the dismantlable connection design full reuse of the refurbishment PCEs, which are non-structural elements, becomes feasible. In this dissertation, the potential benefits of the improved reusability at the element level will be investigated.

Category III: Recovery at the material level

Recovery at the material level is the least preferable but unavoidable intervention, as the last resort to close the material loop. It comprises upcycling, recycling, and downcycling.

The EU “waste hierarchy” clearly defined relevant terminologies and priority order of waste managerial methods. Regarding waste concrete recovery, recycling waste concrete into secondary aggregate to substitute virgin aggregate for new concrete production is considered as a high-value-added option, which can be termed as Upcycling or Recycling. By contrast, processing waste concrete into filler for road base construction or site foundation elevation is a low-value-added way, also known as “downcycling” (Di Maria et al. 2018; Gharfalkar et al. 2015).

The downcycling of waste concrete as secondary materials after crushing is the most used way for end-of-life management of concrete. Waste concrete is fed into ordinary crushers (such as jaw crushers, cone crushers, and impact crushers) only to reduce particle size. The crushed concrete is usually applied as filler for building areas or road foundations. Any aggregates reclaimed by conventional crushers are not "clean": there is still a large amount of mortar attached to the surfaces of the aggregate (Ning 2012). Therefore, the quality of such reclaimed aggregates does not match the requirements for new concrete production.

For a high-value-added recovery, there are three main technologies to achieve the recycling or even upcycling of the waste concrete. They are: (i) wet process, (ii) dry recovery, and (iii) heating air classification system. In this dissertation, the environmental and economic performance of the different technologies will be compared in detail in Chapter 2.

1.2 Energy efficiency strategies for a low carbon built environment

1.2.1 Carbon emission and energy use of the built environment

Globally, the building sector (both residential and non-residential) accounts for 30% of the final energy use (including 55% of electricity consumption), which contributes approximately 37% of the carbon emissions if considering both construction and use phases (IEA 2020). The energy efficiency of the building sector is of high significance to realize a society with net-zero carbon emissions. For example, it is projected that the global energy use of buildings might double, or even triple by 2050 if no mitigating actions are taken (Chalmers 2014). The building sector can however reduce carbon emissions to near zero by 2050 through measures such as energy efficiency and isolation measures, the use of efficient electricity-based equipment, and decarbonization of heat and power supply (IEA 2020). The residential sector is the largest component in the built environment in terms of floor area (80%), final energy use (70%) and CO_2 emissions (60%, (IEA 2020)). Regarding energy use of buildings, operating energy accounts for 80%–90% of the life-cycle energy use of buildings, followed by embodied energy (10%–20%); whereas the energy use for demolition is negligible (Ramesh et al. 2010). In the operation phase, energy use is mainly driven by heating and cooling, lighting, ventilation, equipment, and appliances (O'Sullivan et al. 2004). Around 60% of the current building stock in use in the world was constructed when code requirements regarding energy performance did not exist (IEA 2020). Thus, the vintage of buildings tends to make a big difference to its heating demands. Also in the EU the existing housing stock, in general, remains poor insulation level, since the energy performance standards applied in the past 10–20 years (if available at all) are relatively weak when benchmarked against international best practices. Approximately, 35% of the EU buildings are over 50 years old and almost 75% of the building stock is energy-inefficient (EC 2010). The yearly renovation rate in Europe is still low, only 0.4–1.2%, depending on the country (EC 2010).

1.2.2 Energy efficiency interventions for the built environment

The EU reacted to the IPCC (Intergovernmental Panel on Climate Change)'s 1.5–2 ℃ target by formulating legislative goals of reducing energy use and greenhouse gas (GHG) emissions for the built environment in both the short- and long-term (Danish Energy Agency 2015). Representing its member states, the EU commits to reduce 32.5% gross energy use and 40% gross GHG emissions by 2030 and to reduce 80%–95% gross GHG emissions by 2050, compared to 1990 (EZK 2019). To meet the targets, various energy efficiency interventions are used in Europe, clearly focusing on operational energy savings using passive and active approaches, and addressing new and existing buildings (Sadineni et al. 2011), as shown in Figure 1.4.

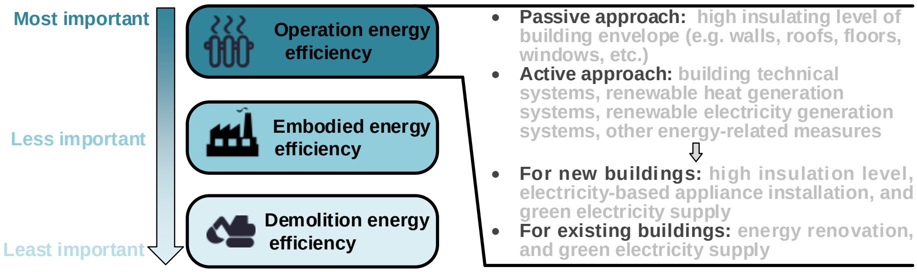

Figure 1.4 Building energy-efficiency framework from a life cycle perspective. Note: energy efficiency interventions are gathered from the European Energy Efficiency Platform (Economidou 2021).

As indicated before, operational energy use and related carbon emissions are the most dominant ones, following at a distance by embodied energy use and emissions, while energy use and emissions for demolition are relatively unimportant. This dissertation hence focused on operational energy used. The passive approach focuses on minimizing heat transfer through the building envelope by improving the insulation level of walls, roofs, floors, windows and so on. Due to the large pool of old buildings, for Europe, up-scaling building renovation by mitigating the energy dissipation from building heat loss is a priority (Staniaszek 2015). It is advocated that scaling up the use of novel technologies that can highly enhance the thermal insulation of a building's envelope (Morrissey and Horne 2011). In this dissertation, a novel multilayer PCE developed in the H2020 VEEP project will be used as an illustration to explore the GHG mitigation potential of the passive approach. The active approach reduces building energy use and GHG emissions by adopting efficient installations for building technical systems, renewable heat generation systems, renewable electricity generation systems, and other energy-related measures (Economidou 2021). As for the active heating system, an ongoing technological shift in Europe is from fossil-based heating equipment, such as the conventional central heating boiler, to natural gas-free alternatives such as heat pumps, residual heat or geothermal energy.

We further see that policy differentiates between new and existing buildings. For new buildings, the EU Directive 2010/31/EU (EC 2010) requires the member states to ensure that new buildings constructed after 2020 be nearly zero energy buildings (nZEBs) that have high insulation performance and are equipped with electricity-based appliances. For existing buildings, energy renovation [2] to achieve higher energy efficiency in combination with efficient, low carbon heating and cooling technologies is seen by the EU as an essential measure to realize a carbon-neutral built environment (Esser et al. 2019). This is because, due to the low construction rate, constructing new energy-efficient buildings alone is not sufficient to meet the short-term energy saving and GHG mitigation goals (Säynäjoki et al. 2012). Directive 2010/31/EU sets minimum energy use standards and cost-optimal levels when existing buildings are renovated (EC 2010). BPIE (Buildings Performance Institute Europe) investigated 16 EU regions (covering more than 60% of the gross EU floor area) and reported 97% of the building stock in Europe needs to be renovated to realize EU carbon neutral targets by 2050 (BPIE 2017). Moreover, the supply of green electricity is also crucial for both new and existing buildings.

1.3 Rationale for chosen cases: selected technologies in the context of the Netherlands

The situation about concrete recycling varies a lot in different countries. In China, it was reported that only 5% of CDW was recovered in 2014, with the remainder being landfilled or even illegally dumped (Zhang et al. 2018). In other countries, waste concrete is managed better. In the US, at least 83% of concrete waste was recovered in 1997 (Jin and Chen 2019). At the beginning of the 21st century, Japan recovered over 98% of waste concrete (Tam 2009). The WFD set up a target that requires member states to take any necessary measures to achieve a minimum target of 70% (by weight) of CDW by 2020 for reuse, recycling, and other recovery (EC 2008a). The Netherlands has already achieved the 70% goal decades ago. In Europe, the EU (28 countries) has had a 90% recovery rate of mineral CDW since 2016 (Eurostat 2021). The Netherlands is one of the best performers regarding CDW management in the world. Deloitte investigated the CDW management in 28 European countries via a maturity matrix of CDW Practice. The Netherlands scored best on every single indicator (Monier et al. 2017). In the Netherlands, the recovery rate of CDW has reached 95% since 2001 due to the

[2] Here, energy renovation is used as an umbrella concept that ranges from restoration, modernization, retrofit, refurbishment, and rehabilitation to simple maintenance, repairs and routine upgrades to deliver different levels of energy savings (Economidou 2021).

introduction of landfill tax in 1995 and landfill bans in 1997 (Scharff 2014), and waste concrete is almost 100% recovered (Monier et al. 2017).

In the Netherlands, the construction sector has been rendered a priority towards a circular and energy-neutral society because it is responsible for 50% of the depletion of the raw material, 40% of total energy use, 40% of waste generation, and 35% of GHG emission in the Netherlands (Dijksma and Kamp 2016). However, near 97% of waste concrete in the Netherlands is downcycled via backfilling. In 2015, only 3% of waste concrete was recycled for concrete manufacturing. The amount of waste concrete was expected to continue to rise from 11.1 Mt in 2014 to 16.3 Mt in 2025 (Zuidema et al. 2016), while the Netherlands has been already facing a saturation of road construction (Hu et al. 2013; Bio Intelligence Service 2011). It is hence important to explore upcycling options to channel recovered concrete aggregates from road base to the manufacturing of concrete. To improve the material efficiency within the construction sector, the Netherlands launched the "A circular economy in the Netherlands by 2050", aiming to achieve an interim target of 50% less use of primary raw materials (minerals, fossil, and metals) by 2030 and a long-term goal of a fully circular economy by 2050 (Dijksma and Kamp 2016). The built environment is one of the priority sectors.

Considering energy efficiency and climate neutrality, the residential building stock in the Netherlands is poorly insulated. Up to 2020, around 72% (Zhang et al. 2021c) of the building stock was constructed before 1995, when minimum energy performance requirements were introduced (Staniaszek 2015). The Netherlands reduced its GHG emissions by 17% in 2019 compared to 1990 (PBL et al. 2020). In the Climate Act (Government of the Netherlands 2019) the Netherlands set an ambitious interim target with a 49% reduction in national GHG emissions by 2030 and a long-term goal of a 95% reduction by 2050 for the country as a whole. To realize such overall targets, the Netherlands committed to an energy-neutrality target within the Dutch built environment by 2050. This means of 7.5 million dwellings, 80% are to be renovated to energy-neutral levels by 2050, which indicates 170,000 homes are to be renovated per annum (Staniaszek 2015). Prefabricated panels for buildings was used in over half (55%) of all construction projects in the Netherlands in 2016 (de Gruijl 2018).

We see hence that the Netherlands has ambitious circularity and GHG reduction targets, both in general and as for the built environment. The built environment is a sector of prime importance in realizing both circularity and GHG reduction targets. As a leading country on concrete recycling, the country still downcycles most of the material as filler. The Netherlands also has good data availabilities on material flows and energy use and related GHG emissions of buildings. This all makes the Netherlands a good geographical focus for this study, which aims to analyze how energy and carbon neutrality and circularity in the built environment can be fostered.

With regard to the technical solutions to be considered, this thesis obviously could not be comprehensive. We hence investigated a number of illustrative technologies that could make the Dutch built environment more circular and carbon-neutral. It concerns the technologies that support material recycling, element reuse, and lifetime extension of buildings, and low impact design of new buildings already presented in sections 1.1 and 1.2:

- Material recycling: for a high-value-added recovery, as discussed further in Chapter 2 there are three main technologies to achieve the recycling or even upcycling of the waste concrete. They are: (i) wet process, (ii) advanced dry recovery (ADR), and (iii) heating air classification system (HAS). In this dissertation, the environmental and economic performance of the different technologies will be compared in detail in Chapter 2.

- Refurbishing and lifetime extension of existing buildings, and element reuse: Renovating façades with PCEs is a common way to upgrade the energy performance of buildings. In this thesis, as an illustrative technology the so-called PCE-refurb system as developed in the EU H2020 project VEEP will be analyzed. This system is not only suitable for renovating existing buildings but also can be dismantled after their use allowing for elements reused.

- Eco-design of new buildings: In this thesis, as an illustrative technology the production and use of PCE-new for cladding walls to improve building energy efficiency in new constructions will be analyzed.

1.4 Methods to assess impacts of the circularity and energy-efficiency interventions for the built environment

To evaluate how new technological approaches can help to realize circularity and GHG reduction targets and if there are tradeoffs regarding other environmental problems and costs, life cycle assessment (LCA), life cycle costing (LCC), and material flow analysis (MFA) will be used in combination in this study. Below we discuss the state of the art and potential for methodological improvement in view of the intended application on the built environment.

1.4.1 Assessment of the environmental and financial impacts at the product level

The built environment provides constructed space to accommodate human activities. To transform the provision pattern of the built environment towards circularity and low carbon, technological interventions are the most direct measure to be considered.

Therefore, the first question is if the technological intervention – be it a novel panel or a new type of concrete/cement – does bring the expected benefits at the product level.

For this assessment, a life cycle perspective will be taken. This is because, due to problem-shifting, a life cycle perspective is essential for systems-wide environment management (Hunkeler et al. 2003). Using a life cycle perspective can avoid interventions that can mitigate impacts in one stage but may have more adverse impacts in other life stages (Huang et al. 2020).

To assess the environmental impacts, LCA will be used. Environmental LCA is an ISO standardized method, which is defined as "a product-oriented assessment of the inputs and outputs and the associated potential environmental impacts of a product system during its life cycle" (ISO 14040 (2006) and 14044 (2006)). Three quantification approaches have been developed (Guinée et al. 2011; Miah et al. 2017; Ibn-Mohammed et al. 2016): (i) process-based LCA (process-LCA) that is based on a product system model in which the product production is illustrated by input-output flows and unit processes, (ii) environmental input-output based LCA (EIO-LCA) that handles the environmental issue by using an input-output analysis; and (iii) hybrid LCA that combines the merits of both process-LCA and EIO-LCA. With specific technological inputs, this study will use process-LCA for the quantification.

As assessment will involve some novel technologies, whose process data are only available at lab or pilot scale, ex-ante (/perspective/anticipatory) LCA will be applied. Ex-ante LCA is specifically developed for new technologies that only function yet at the lab- or pilot-scale, and for which process data are only available at these scales. In an ex-ante LCA, lab- and pilot-scale data will be used to estimate its performance when the technology is mature and used at scale, by applying e.g. expert judgment, scenarios or learning curves for similar technologies (Cucurachi et al. 2018). In this study, process-based ex-ante LCA will be used to simulate the development of renewable electricity affecting housing energy use.

However, an optimal environmental-sound intervention may not be an economically viable option. Therefore, the EU addressed the importance of cost-effectiveness in its laws regarding the circular economy as well as energy renovation. For instance, the Action Plan for the Circular Economy (COM/2015/0614) proposed minimum conditions on cost-efficiency of waste collecting and recycling (EC 2015a). The Zero Waste Program (COM(2014) 398)) also aims to minimize the costs of recycling and reuse (EC 2014a). The Energy Performance of Buildings Directive (2010/31/EU) set cost-effectiveness as a critical requirement for building energy efficiency (EC 2010).

To assess the financial impacts, in parallel to the environmental assessment, LCC – life cycle costing will be used. LCC is a useful method to analyze the costs with the purpose to reduce the life cycle expense of a product. The LCC has a longer history than LCA. LCC has been developed by the Rand corporation after the Second World War (Novick 1959). The major difference between the traditional costing system and LCC is that the LCC approach comprises an expanded life cycle perspective, and thus includes not only investment costs, but also operational costs and sometimes even EoL costs. It also requires a more precise definition of the functional unit. LCC for a general purpose has not been standardized. LCC was vaguely defined by the building and construction assets standard ISO15686-5 (2008) as a technique that enables comparative cost assessments to be made over a special period. One obvious merit of LCC is the identification of factors that have the largest contribution to the total life cycle costs of a project (Korpi and Ala-Risku 2008). The application of LCC has expanded to many areas, such as waste management (Martinez-Sanchez et al. 2015), ship production (Utne 2009), packaging (Albuquerque et al. 2019), energy system (Zakeri and Syri 2015; Ristimäki et al. 2013), and building (Marszal and Heiselberg 2011; Sterner 2000; Kneifel 2010). Because LCC can provide a significantly better assessment of the long-term effectiveness of a project than alternative economic methods that focus only on first costs or operation-related costs in the short run, LCC is particularly suitable for evaluating the building design alternatives that satisfy a required level of building performance (Akhlaghi 1987).

Even so, using LCC alone is insufficient to capture the environmental soundness and the cost-effectiveness in one go. Besides the obvious costs of production activities, all processes involved in the life cycle may induce external costs related to environmental impact, as they consume resources, emit greenhouse gasses, and generate waste, which, however, usually are not considered in traditional LCC. It is hence best to view LCC as a tool for financial assessment, which has the potential to align with LCA for a combined evaluation of environmental impacts and traditional financial costs (Gluch and Baumann 2004).

The integration and alignment of LCA and LCC are however challenging since the methods have been developed for different purposes and have their roots in different scientific communities. *First, how can the perspectives on system boundaries of LCA and LCC be aligned?* An LCC needs to be conducted from the perspective of a cost bearer to clarify the system boundaries with regard to costs, while an LCA uses a functional unit as a starting point. Hunkeler et al. (2008) stated the perspective of a consumer is more suitable for an LCC as it seems able to cover a full life cycle of a product from production to disposal. But from the consumer's perspective, the costing processes of other actors are black boxes, an LCC as such is unable for cost optimization, because it cannot reflect on the cost details (e.g. production costs, EoL costs and so on) to capture the hot spots for cost-saving (Rebitzer and Hunkeler 2003). Swarr et al. (2011) presented a case study from a municipal perspective. Even though it is not a strictly financial perspective, as it contains externalities, it unveils the possibility of a perspective that can aggregate all actors in the life cycle of a product. However, this reasoning still cannot tackle the preceding questions “whose costs is the LCC accounting for?” and “how the LCC can be aligned with the parallel LCA?”.

Second, can differences in the calculation of overall results and their usual breakdowns be aligned between LCA and LCC? On the one hand, in LCC the life cycle cost can be broken down according to the life cycle phases (construction phase, operation phase, and disposal phase), cost bearers (manufacturer, consumer, and recycler), and expenditure categories (capital costs and operation costs) which are similar as in LCA. But often LCCs use more detailed, and random cost categories that do not align with LCA (e.g. costs incurred by actors, or costs for labor, depreciation of capital goods, consumables, and taxes). LCA adds up all impact over the full life cycle to total scores per impact category. Results of an LCC are expressed in various forms, including net present value (considering a discount rate or not, see below), payback period, and so on (Akhlaghi 1987). Such differences in how to express overall results and their breakdowns hamper an integrated presentation of LCA and LCC results.

Third, how can differences be dealt with in the practice to discount costs in LCC and not to discount environmental impacts in LCA? LCC usually applies a discount rate to costs that are incurred in the future. Hence, LCC results are sensitive to discount rates, especially when done for buildings since they have such long lifetimes. For instance, the benefits of reuse of building components in future due to applying design for reuse principles today will be hardly visible if the reuse takes only place in 50 years and a discount of 2–4% per year is applied. But in an LCA the discount rate for environmental impacts is usually set as 0% (Hunkeler et al. 2008). Applying discount rates on environmental impacts in the future, and taking this into account to calculate external costs is a long-standing controversial issue (Weitzman 2011; Arrow et al. 2014; Portney and Weyant 2013).

To sum up, LCA and LCC are increasingly used in combination to assess the environmental and financial impacts at the product level. However, the aforementioned methodological inconsistencies are hampering the application of their combined approach. This thesis presents various case studies (e.g. on recycling of concrete aggregates for prefabricated concrete panels) where we suggest some solutions to overcome such inconsistencies in the combined use of LCA and LCC, and at the same time to shed a light on the cost-effectiveness of several technological interventions for the circular and low-carbon transition in the built environment.

1.4.2 Assessment of up-scaled environmental and economic impacts at country level

The above assessment at the product level is not capable to analyze if circularity and carbon emission targets at a regional or national level are met. Therefore, in this study, to scale up the product-level LCA and LCC results to a national level, the tool of dynamic MFA will be used. With MFA we will compute a balanced set of material flows through the housing sector of the Netherlands over a large number of years into the future (until 2050). Integrating the product level life cycle studies in national scale MFA, the up-scaled benefits of technologies that support the transition of the Dutch built environment towards a circular and carbon-neutral system can be obtained.

MFA is an approach based on the law of conservation of mass to assess the metabolism of materials in a system (Brunner and Rechberger 2004). Regarding the object of analysis, Udo de Haes in 1997 (de Haes et al. 1997), and Graedel and Allenby in 2003 (Graedel and Allenby 2003) differentiated an MFA into substance flow analysis (SFA) and Bulk-material flow analysis (Bulk-MFA). Bulk-MFA and SFA share the same methodological framework. The difference between SFA and Bulk-MFA is that an SFA supports management strategy with regard to specific substances or compounds (Jeswani et al. 2010); while Bulk-MFA focuses on bulk materials (Huang et al. 2012a). SFA studies have been conducted for phosphorus in Denmark (Klinglmair et al. 2017), copper in China (Dong et al. 2019, 2020) and in Germany (Pfaff et al. 2018), Aluminum in the USA (Chen 2018; Chen and Graedel 2012) and China (Chen and Shi 2012). Bulk-MFA can investigate mass flows of a region at a macro-level and of a product at a micro-level. At a macro-level, a Bulk-MFA is also known as Economy-wide MFA (Eurostat 2018), which is based on national economy-wide material flow accounts that record all materials entering or leaving the boundary of the national economy (OECD 2008). Many other Bulk-MFA studies investigate only a selection of materials such as building materials as a whole (Condeixa et al. 2017; Mesta et al. 2017; Huang et al. 2018b), concrete in the Netherlands (Müller 2006) and in Taiwan, China (Hsiao et al. 2002), sand/gravel in Chongqing, China (Zhang et al. 2018), iron/steel in Beijing, China (Hu et al. 2010a), and timber in Switzerland (Mehr et al. 2018) and in Japan (Kayo et al. 2019). At a micro-level, a bulk-MFA is redeemed as a simplified LCA in which the mass is used as an indicator for assessing the environmental impact of the product (Kleijn 2000).

Regarding the quantitative methods to model a material flow system, van der Voet (1996) identified three possible methods, namely: (i) the accounting/bookkeeping model, which is usually applied to signaling, spotting trends, evaluation of ex-post; (ii) the static model, which is employed to analyze origins, compare regimes, and evaluate ex-ante; (iii) the dynamic model, which focuses on the evaluation of ex-ante, trends prediction, and scenario analysis. Of the three quantification methods defined by van der Voet (1996), only the dynamic approach is truly capable to analyze the long-term metabolism of the built environment. Augiseau and Barles (2017) proposed three ways to further distinguish MFA studies: (i) bottom-up or top-down: depending on whether there is a prior definition of processes in which materials circulate; (ii) retrospective or prospective: depending on whether exploring the past or the future; (iii) flow-driven (also known as demand-driven) or stock-driven: depending on whether focusing on input flows or stocks. The top-down method aims to determine the extent to which material flows and estimates of stock size can be derived from general economy-wide statistics, while the bottom-up model considers individual objects using a coefficient-based MFA (Schiller et al. 2017b). Regarding the distinction of stock-driven versus flow-driven approaches, the stock-driven approach is based on the hypothesis that stock is the driver for the material flows (Augiseau and Barles 2017); the flow-driven approach focuses on the physical inflows as exogenous model inputs, for example by extrapolating recent yearly average value of flows (Wiedenhofer et al. 2019). In this study, the estimation of the dynamics of the building stock was analyzed via a top-down approach, by gathering data from socioeconomic statistics. Next to this, a prospective approach was applied to generate what-if scenarios for stocks and flows in future years contrasting with a business as usual baseline.

Although it is a suitable instrument for the recognition of waste and resource problems and the development of solutions to the problems, MFA only looks at physical material flows in the economic system. Hence, impacts related to resource extraction and emissions, and economic impacts are not made clear. For a comprehensive assessment of the regional environmental and economic impacts, MFA is frequently connected to LCA and/or LCC. The following are illustrations for how such connections are made: (i) use LCA (or life cycle sustainability assessment) as an overarching framework, and use the MFA data to define a functional unit at a regional or national scale, and use this as a basis to perform the life cycle inventories (Nørup et al. 2019; Wäger et al. 2011; Goldstein et al. 2013; Rochat et al. 2013; Sevigné-Itoiz et al. 2015, 2014; Groleau et al. 2018; Zhang et al. 2018); (ii) conduct LCA and MFA separately, and use the results of the LCA as unit environmental-emission indicators and the results of MFA as up-scaling size factors (Lavers Westin et al. 2019; Millward-Hopkins et al. 2018; Lopes Silva et al. 2015; Mehr et al. 2018); (iii) use LCA and MFA together while independently express the results of MFA as mass-based indicators (such as total material requirements) and results of LCA as impact-based indicators (such as GHG emissions) (Rincón et al. 2013).

However, the previous studies did not explicitly consider the difference of temporality between LCA and MFA, in particular dynamic MFA. Generally, an MFA is conducted through the synchronic-temporality approach, while an LCA is modelled from the diachronic approach (Birat 2015). The temporal mismatch between the two methods may lead to biased results in life cycle impacts, especially for long-lasting goods like buildings. This study will explore how to harmonize the temporality aspect in the combined use of LCA/LCC and dynamic MFA, thus, to provide some insights on the potential benefits of large-scale implementation of the PCE technological system for the residential sector in the Netherlands.

1.5 Research questions

The overarching research question posed in this thesis is: *what are the potential impacts of the application of selected novel technological systems to enhance circular use of materials, energy efficiency and carbon neutrality in the residential sector of the Netherlands?*

Also in view of introduction on the choice of cases/technologies and methodologies above, five sub-questions were investigated in the thesis:

RQ1. Assessment of concrete recycling at the product (material) level

Is it possible to achieve environmental-economic win-win situation in high-grade concrete recycling? Would the innovations trigger any potential problem-shifts between different impact categories? **(Chapter 2)**

RQ2. Assessment of the PCE-new system for new building construction at a product (element) level

What are the environmental and economic implications of using the prefabricated element system (PCE-new) for cladding walls to improve building energy efficiency in new constructions in the Netherlands? **(Chapter 3)**

RQ3. Assessment of the PCE-refurb system for existing building renovation at a product (element) level

What are the environmental and economic implications of using the prefabricated element system (PCE-refurb) to over-clad existing buildings for energy refurbishment in the Netherlands? How is the applicability of the PCE-refurb under different climatic conditions? **(Chapter 4)**

RQ4. Assessment of turnover of the housing stock at the country level

How much CDW from the construction, demolition, and renovation of the Dutch housing stock will arise from 2015 to 2050? To which extent the CDW can be recycled as a feedstock in building energy renovation in the Netherlands? **(Chapter 5)**

RQ5. Assessment of implementing the recycling and prefabrication systems at the country level by combining dynamic MFA, LCA and LCC

What are the up-scaled environmental benefits and economic consequences of implementing the recycling and prefabrication systems in the Netherlands? To which extent can the proposed recycling and prefabrication system achieve the prospective circularity goal and decarbonization goal of the Netherlands? **(Chapter 6)**

The answers to these research questions will be provided in the conclusions (Chapter 7). That chapter also will reflect on lessons learned about the integration of MFA, LCA and LCC, in relation to the problems discussed in section 1.4.

1.6 Thesis outline

The thesis is outlined as seven chapters as shown in Figure 1.5.

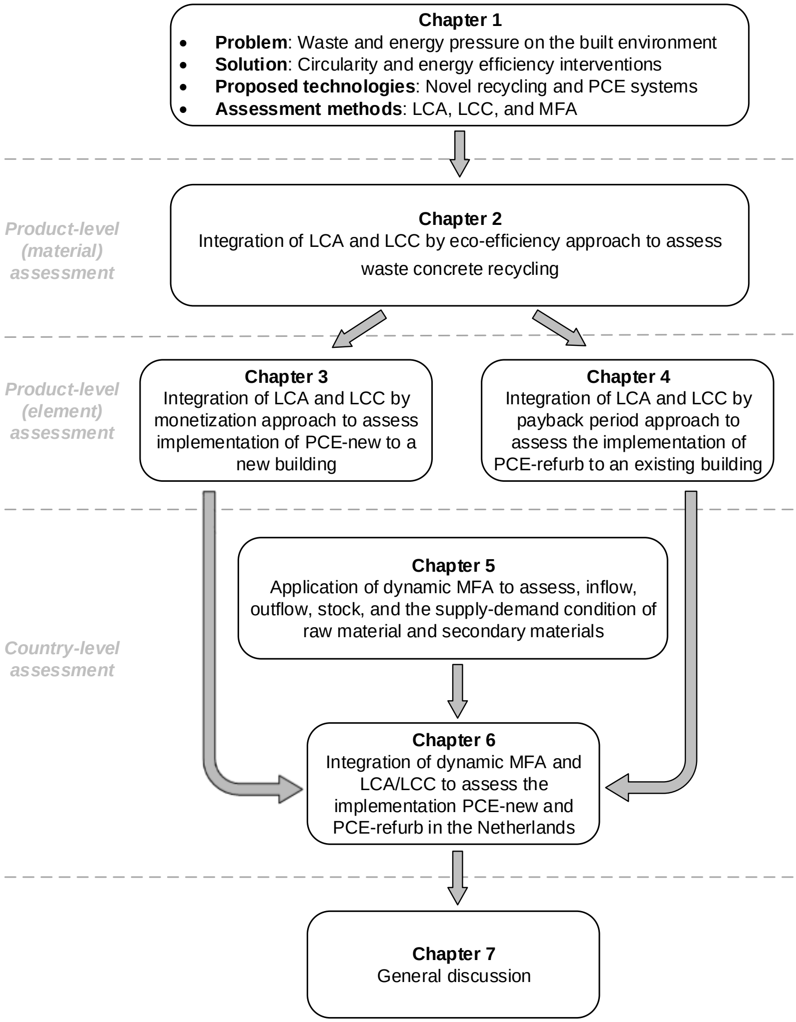

Figure 1.5 Conceptual scheme of the thesis and the topic of each chapter

The main content of each chapter is presented as follows:

Chapter 1: General introduction

This chapter provides a general introduction on the status quo and importance of the built environment, the potential material-circularity and energy-efficiency strategies, the major questions, and the thesis outline.

Chapter 2: Eco-efficiency assessment of technological innovations in high-grade concrete recycling

This Chapter investigates the environmental and economic impact of multiple technological systems for high value-added waste concrete recycling in a Dutch context. LCA and LCC were integrated by using an improved eco-efficiency analytical approach. In this study, four technological systems were analyzed for comparison: (i) BAU stationary wet processing; (ii) stationary ADR; (iii) mobile ADR; (iv) mobile ADR and HAS. This study proposed an overarching framework for LCA/LCC-type eco-efficiency assessment conforming to ISO standards, which is easy to reveal environmental-economic trade-offs between multiple scenarios. The study also aims to expose the future trend on cost-efficient technological routes for high-value concrete recycling.

Chapter 3: Life cycle greenhouse gas emission and cost analysis of prefabricated concrete elements for use as fa çade of new building

This chapter bases on the results of Chapter 3 to explore the environmental and economic impact of incorporating CDW into green PEC-new for constructing as the wall of new buildings, compared to a BAU PCE-new that is only fabricated by virgin materials. This study aims to determine whether the use of green PCE-new leads to lower carbon emissions and lower associated costs over the life cycle of a virtual four-story residential building under the Dutch climate condition of the Netherlands than a BAU PCE-new scenario by applying LCC and LCA. This chapter provides a case study on the alignment and/or integration of LCA and LCC in an independent and a combined manner via monetization. The simulation results will show how the internalization of carbon cost can strengthen the economic advantage of a product.

Chapter 4: Energy-carbon-investment payback analysis of the prefabricated envelope-cladding system for building energy renovation: cases in Spain, the Netherlands, and Sweden

This chapter aims to examine cross-state cases that investigate the performance of the PCE-refurb system for energy renovation of existing buildings on energy conservation, carbon abatements, and cost savings. Assessments are conducted in the Netherlands as well as other two European member states: Spain and Sweden. As the rest lifetime of existing buildings that were constructed in different vintages varies, the temporality of the LCA and LCC cannot be directly defined. Therefore, the results of the LCA and LCC are presented in payback periods. The study case in this chapter illustrates how material circularity strategies (recycling and reuse) and climatic conditions influence the energy/carbon/investment payback periods of refurbishment.

Chapter 5: **Recycling potential in building energy renovation: a prospective study of the Dutch residential building stock up to 2050**

This chapter uses a dynamic MFA model to explore the supply-demand balance of secondary raw materials made from CDW (including normal-weight and lightweight concrete, glass, insulation mineral wool, and steel) and the secondary raw materials demanded for manufacturing PCEs in building energy renovation in the Netherlands for the period 2015 to 2050. This study case will characterize the inflow, outflow, and stock of the Dutch housing sector and evaluate whether the CDW (construction waste, demolition waste, and renovation waste) is sufficient to support the extensive energy renovation in the Netherlands.

Chapter 6: Towards the 2050 circularity and decarbonization goals: Economic and environmental implication of material-energy efficiency renovation of housing stock in the Netherlands

Chapter 6 scales up the product-level results from Chapters 3 and 4 with the housing stock size from Chapter 5. This is realized by an integrated methodological framework that combines dynamic MFA with LCA and LCC. Except for the renovation scenarios (REN) that use the proposed PCE system to renovate the housing stock, additional two scenarios are established: the BAU scenario that does not apply any renovation strategy, and the Rebuild (REB) scenario in which old buildings are demolished and reconstructed instead of renovation. This study explores the product-level and country-level carbon mitigation, cost savings, and material footprint reductions of these three scenarios in the Netherlands from 2015 to 2050. It aims to reveal (i) the economic and environmental trade-offs of the rebuilding and renovation compared to the BAU scenario, and (ii) to what extent the circularity and decarbonization goals can be realized if up-scaling the PCE system to the residential building sector.

Chapter 7: General conclusion

This chapter is dedicated to a general discussion, conclusions, and recommendations of aspects related to employing LCC/LCA at a product level and MFA models at a regional level to support decision-making towards material circularity and energy efficiency of the built environment.

Chapter 2

Eco-efficiency assessment of technological innovations in high-grade concrete recycling

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Abstract

The increasing volume of construction and demolition waste (CDW) associated with economic growth is posing challenges to the sustainable management of the built environment. The largest fraction of all the CDW generated in the member states of the European Union (EU) is end-of-life (EoL) concrete. The most widely applied method for EoL concrete recovery in Europe is road base backfilling, which is considered a low-grade recovery. The common practice for high-grade recycling is a wet process that processes and washes EoL concrete into clean coarse aggregate for concrete manufacturing. It is costly. As a result, a series of EU projects have been launched to advance the technologies for high value-added concrete recycling. A critical environmental and economic evaluation of such technological innovations is important to inform decision making, while there has been a lack of studies in this field. Hence the present study aimed to assess the efficiency of the technical innovations in high-grade concrete recycling, using an improved eco-efficiency analytical approach by integrating life cycle assessment (LCA) and life cycle costing (LCC). Four systems of high-grade concrete recycling were analyzed for comparison: (i) business-as-usual (BAU) stationary wet processing; (ii) stationary advanced dry recovery (ADR); (iii) mobile ADR; (iv) mobile ADR and Heating Air Classification (A&H). An overarching framework was proposed for LCA/LCC-type eco-efficiency assessment conforming to ISO standards. The study found that technological routes that recycle on-site and produce high-value secondary products are most advantageous. Accordingly, policy recommendations are proposed to support the technological innovations of CDW management.

Keywords: Eco-efficiency assessment; Life cycle assessment; Life cycle costing; Concrete recycling; Construction and demolition waste; Technological innovation

Abbreviations

ADR	Advanced dry recovery
ADR-M	Mobile advanced dry recovery
ADR-S	Stationary advanced dry recovery
A&H	Advanced dry recovery and heating air classification system
BAU	Business-as-usual
CDW	Construction and demolition waste
C2CA	Project “Advanced Technologies for the Production of Cement and Clean Aggregates from Construction and Demolition Waste”
EC	European Commission
EC-JRC	Joint Research Centre of the European Commission
ESCAP	Economic and Social Commission for Asia and the Pacific
EU	European Union
EoL	End-of-life
HAS	Heating air classification system
HISER	Project “Holistic Innovative Solutions for an Efficient Recycling and Recovery of Valuable Raw Materials from Complex Construction and Demolition Waste”
ISO	International Organization for Standardization
LCA	Life cycle assessment
LCC	Life cycle costing
LCI	life cycle inventory
NCA	Natural coarse aggregate
RCA	Recycled coarse aggregate
RFA	Recycled fine aggregate
RUP	Recycled ultrafine particle
SETAC	Society of Environmental Toxicology and Chemistry
SS	Sieve sand
UNCED	United Nations Conference on Environment and Development
UNEP	United Nations Environment Program
VEEP	Project “Cost-Effective Recycling of CDW in High Added Value Energy Efficient Prefabricated Concrete Components for Massive Retrofitting of our Built Environment”
WBCSD	World Business Council for Sustainable Development
WP	Wet process for end-of-life concrete disposal

2.1 Introduction

Construction and demolition waste (CDW) is widely acknowledged as one of the most important sources of waste (Koutamanis et al. 2018). This is especially true for Europe, where the stock of buildings and infrastructure was built during World War II and renewal including demolition of such stocks is now a main activity for the building and construction sector. Eurostat estimated an annual CDW generation of 970 million tons in the European Union (EU)-27 (Vegas et al. 2015). The CDW has been identified by the European Commission (EC) (2001) as a priority stream because of the large amounts that are generated and the high potential for reuse and recycling embodied in these materials.

For this reason, the Waste Framework Directive (EC 2008a) requires member states to take any necessary measures to prepare for material recovery, by 2020, of at least 70% (by weight) of CDW. The current recycling percentages of CDW per European country vary between less than 5% in Montenegro and more than 90% in countries including Belgium, Portugal, and the Netherlands (Eurostat 2021). The vast majority of CDW is down-cycled, for instance in road foundation, or even landfilled in some European countries. For example, in 2003, the Spanish construction sector only recycled 10.3% of CDW, while 25.6% was deposited in inert waste landfills, and 64.1% was dumped in the absence of controls in debris sites, pits or watercourses (Rodr and Alegre 2007). In 2012, Switzerland recycled 51%, landfilled 26%, incinerated 8% (combustible fraction such as wood), and reused 15% on-site (Hincapié et al. 2015). In Europe, the average composition of CDW shows that up to 85% of the waste is stony waste (Gálvez-Martos et al. 2018) such as end-of-life (EoL) concrete. An alternative market of recycled aggregates derived from EoL concrete was already established in Europe, where the EoL concrete was reused for road base material (Anastasiou et al. 2014). Experts foresee that landfill of EoL concrete can be reduced to 0% and that the use of recycled concrete aggregates in road construction can significantly contribute to reaching the 70% target for CDW recycling in the EU (Bio Intelligence Service 2011).

The Netherlands has achieved 100% recycling of EoL concrete and has a more advanced concrete recycling and CDW management system than China (Zhang et al., 2018; Huang et al., 2018), Australia (Tam et al. 2010), Canada (Yeheyis et al. 2013) and other European member states (Eurostat 2021). The most common practice for concrete recycling in the Netherlands is simply crushing and subsequent use as a base in road construction, which is considered a low-grade or low value-added route. Currently, the most commonly applied method for high-grade recovery of concrete is the wet process, which produces clean aggregate for concrete by washing the coarse aggregate, leaving the fine fraction (sieve sands) for road base filling and generating sludge, which needs to be treated. A downside of the wet process is that it requires a large washing plant, which is expensive. Therefore, more than 90% of the waste concrete in the Netherlands is still processed low-grade for use in road base materials.

In the coming years, a continuous increase in the amount of CDW and EoL concrete is expected in Europe because of the large number of constructions built in the 1950s which are coming to the end of their life. At the same time, options for low-grade reuse will become more limited, since road construction will stabilize (Bio Intelligence Service 2011). So, higher value-added solutions are needed for the EoL concrete that cannot be absorbed in road construction.

In 2011, UNEP (2011) advocated "greening the waste sector", referring to a shift from less preferred waste treatment and disposal methods, such as landfilling, towards options that contribute to the highest reduction of the use of primary resources. The growth of the waste market, increasing resource scarcity and the feasibility of new technologies create opportunities for high value-added recovery options, also in the case of the EoL concrete. Technical progress and green technical innovation are necessary not only to improve the productivity of industries, but also to enhance the environmental benefits of reuse, recovery, and recycling (Song and Wang 2018). Governments are imposing more stringent regulations, while other parties, including suppliers, consumers, and banks, are formulating requirements for eco-products and green technology (Klostermann and Tukker 1998). Moreover, new products need to be prepared for upcoming challenges concerning lower carbon footprints, resource depletion and shortages and also concerning cost-effectiveness in a competitive marketplace (Zhang et al. 2019b). Over the last few years, novel technologies have been developed that aim to guarantee high-quality recycled raw materials for manufacturing new construction products, thereby closing the concrete loops.

In Europe, the EC funded an innovation project called C2CA (Concrete to Cement and Aggregate, www.c2ca.eu), which aims to develop a cost-effective approach for recycling high-volume EoL concrete streams into prime-grade aggregates and cementitious fines (Lotfi et al. 2014). The C2CA project proposes an innovative solution called Advanced drying recovery (ADR). It constitutes a dry alternative to the existing wet process, which significantly reduces the processing cost for high-grade recovery of the coarse fraction of EoL concrete. However, the initial plan to use the fine product of ADR as a feed-in kiln for cement production was not optimal due to the required long-distance transportation of fines.

In the C2CA project, the equipment for the ADR process was a semi-mobile facility that could not yet be used for in-situ EoL concrete processing. The challenge to make the ADR technology transportable for in-situ use was taken up by a follow-up project called HISER (Holistic Innovative Solutions for an Efficient Recycling and Recovery of Valuable Raw Materials from Complex Construction and Demolition Waste, www.hiserproject.eu). In this project, a mobile ADR set was developed that can be transported by one truck and assembled in one day.

Although the mobility of the ADR set has been improved, the fine fraction (0–4mm) materials generated during the high-grade concrete recycling are still not valorized, being left on-site or used as filling material for road base or land leveling. This issue was taken care of by the EC VEEP project (Cost-Effective Recycling of CDW in High Added Value Energy Efficient Prefabricated Concrete Components for Massive Retrofitting of our Built Environment, www.veep-project.eu). In the VEEP project, the ADR system was combined with a thermal treatment process called the Heating air classification system (HAS) to refine the fine fraction of the output of the ADR process for the production of high value-added products - clean secondary sand and cementitious fine materials.

The environmental benefits and economic consequences of different recycling routes are commonly assessed via eco-efficiency evaluation that combines Life cycle assessment (LCA) and Life cycle costing (LCC). Although the concept of eco-efficiency itself is not new or complex, a better specification is desirable to assess the co-benefits of technological innovations. A series of innovations in high-grade concrete recycling offers a good study case to investigate how technological development would influence the efficiency changes in CDW management. Using field data collected from the C2CA, HISER and VEEP projects, this study presents an eco-efficiency assessment, from a practical aspect, to understand whether each step of the innovation generates environmental benefits and if so, at what financial cost. Is it possible to achieve an environmental-economic win-win situation in high-grade concrete recycling? Would the innovations trigger any potential burden shifts (environmental and economic)? The findings of such an investigation are expected to shed light on the technological development of future concrete recycling and on the feasibility of a circular economy in the construction sector. Moreover, from a theoretical aspect, this case study on concrete recycling proposed a framework for LCA/LCC-type eco-efficiency assessment.

2.2 Literature review of eco-efficiency analysis

The concept of eco-efficiency was designed to guide the ecological and economic efficiency improvement in a production system within a company, by measuring the environmental impact caused per monetary unit earned. Eco-efficiency can be mathematically expressed as shown in Eq. (1) (Keffer et al. 1999).

$$Eco - efficiency = \frac{Value\ added}{Environmental\ impact} \tag{1}$$

ESCAP (2009) defines eco-efficiency as a key element for promoting fundamental changes in the way societies produce and consume resources, and thus for measuring progress in green growth. It is commonly accepted that eco-efficiency was first mentioned by Sturm and Schaltegger in 1989: "the aim of environmentally sound management is increased eco-efficiency by reducing the environmental impact while increasing the value of an enterprise” (Bohne et al. 2008). Later, it was popularized by the World Business Council for Sustainable Development (WBCSD) for the business sector in the course of the United Nations Conference on Environment and Development (UNCED) in 1992. Eco-efficiency was first developed academically in 1990 and prominently promoted by WBSCD in 2000 (Kicherer et al. 2007). Since then, eco-efficiency has been variously defined and analytically implemented, and in most cases, eco-efficiency is taken to mean the ecological optimization of overall systems while not disregarding economic factors (Saling et al. 2002). The “eco-efficiency assessment” is a concept rather than a specific appraisal tool. Eco-efficiency analysis can be deployed by using data envelopment analysis (DEA) as the efficiency measurement vehicle (Korhonen and Luptacik 2004). However, DEA is more likely to explore efficiency issues at the meso- and macro-level (Mardani et al., 2017; Chen and Jia, 2017; Tajbakhsh and Hassini, 2015; Atici and Podinovski, 2015; Gerdessen and Pascucci, 2013), whereas the environmental and economic impacts of technological innovations on concrete recycling are essentially product-level issues.

In 2012, eco-efficiency assessment was standardized in ISO 14045 (2012) as a quantitative management tool that enables the study of environmental impacts of a product system along with its product system value for a stakeholder from a life cycle perspective. In this manner, the eco-efficiency assessment which examines the life cycle of a certain product is more adaptable to product-oriented issues. The framework of eco-efficiency assessment, which is based on LCA standards, was outlined in 6 steps in ISO 14045 (2012), and in this framework, LCA is employed for “environmental assessment” conforming to ISO 14040 (2006) and 14044 (2006). ISO 14045 (2012) defines three ways to present a value system: functional value, monetary value, and other values (e.g. aesthetic, brand, cultural and historical). However, it does not specify the tool for the economic value assessment. Based on Equation 1, Bohne et al. (2008) argued that “value-added” cannot be used in a recycling-system context in the same way as at the firm level, because profits that stakeholders seek to make along the way do not necessarily increase the value of the material but arise from their performance of services, and “cost” is used to denote all economic transaction. As LCC is a methodology for the systematic economic evaluation of life cycle costs (ISO 2017a), we reckon that the financial analysis for this study via an LCC assessment would be an appropriate approach for making decisions on the cost-effectiveness of a product. We reviewed some typical LCA/LCC-type eco-efficiency studies and listed their methodological choices in Table 2.1.

Table 2.1 Literature related to LCA/LCC-type eco-efficiency analysis

Literature	Object	Impact category	Cost category	Form of eco-efficiency	Sensitivity analysis	Uncertainty analysis
(Zhao et al. 2011)	Municipal solid waste management	Global warming potential	Static cost	Two-dimensional diagram	×	×
(Bohne et al. 2008)	Construction and demolition waste management	EcoIndicator 99 method	Net present value	Two-dimensional diagram	√	√
(Woon and Lo 2016)	Municipal solid waste management	Human health-related impact	Future worth	Two-dimensional diagram	×	×
(Yang et al. 2008)	Municipal solid waste management	Climate change, Acidification, Eutrophication, Photochemical ozone synthesis	Static cost	Relative value	×	×
(Burchart-Korol et al. 2016)	Electricity production	ReCiPe 2008 H/A	Net present value	Absolute value	√	×
(Lee et al. 2011)	H_2 fuel cell bus	Global warming potential, Fossil fuels consumption, and regulated air pollutants.	Static cost	Two-dimensional diagram	×	×
(Ibbotson et al. 2013)	Scissors	Cumulative energy demand, World ReCiPe midpoint, and World ReCiPe endpoint.	Static cost	Two-dimensional diagram	×	×
(Lorenzo-Toja et al. 2016)	Wastewater treatment plants	CML-IA baseline, Global warming potential, and Eutrophication potential	Static cost	Two-dimensional diagram	×	×
(Vercalsteren et al. 2010)	Drinking cups	Carcinogenic, Respiratory effects caused by organics, Respiratory effects caused by inorganics, Climate change, Ozone layer, Ecotoxic emissions, Acidification/eutrophication, Extraction of minerals, Extraction of fossil fuels	Static cost	Two-dimensional diagram	√	×
(Fernández-García et al. 2016)	Interior partition walls	Acidification for soil and water, Eutrophication, Global warming (climate change), Ozone depletion, Photochemical ozone creation, Depletion of abiotic resources elements, Depletion of abiotic resources e fossil fuels	Static cost	Two-dimensional diagram	√	×
(Auer et al. 2017)	Glass container	Eutrophication potential (EP), Photochemical ozone creation potential (POCP), Global warming potential (GWP) and-Acidification potential (AP)	Static cost	Two-dimensional diagram	×	×

Table 2.1 shows that eco-efficiency assessment has been applied to multiple domains: waste management, energy, construction, and daily necessities. However, the assessment method is far from standardized yet. First, ISO 14045 (2012) did not specify the method for the product value system assessment, and a guideline on LCC and LCA under an overarching eco-efficiency framework is lacking. Second, in LCC cost structures were

broken down in different ways and life-cycle costs were randomly expressed in different cost forms. Third, even though sensitivity and uncertainty analysis are mandatory in ISO 14045 (2012), they are not common practice yet either on LCC and LCA separately or on the eco-efficiency index as a whole.

To fill the knowledge gap, the present study proposes a protocol 1) to embed LCA and LCC inside a joint eco-efficiency framework under ISO standards; 2) to add an additional "economic impact assessment" step to multi-dimensionally break down the cost structure and classify cost stressors; and 3) to present a solution for the quantification of sensitivity and uncertainty in an LCA/LCC-type eco-efficiency assessment.

2.3 Methods

2.3.1 Framework for integrating LCC and LCA for eco-efficiency

According to the ISO 14040 (2006) and 14044 (2006), an LCA is organized into four steps: (i) goal and scope definition; (ii) life cycle inventory (LCI) analysis; (iii) life cycle impact assessment; (iv) life cycle interpretation. We tried to apply the environmental LCC conforming to the guidebooks published by the Society of Environmental Toxicology and Chemistry (SETAC): *Environmental Life Cycle Cost* (Ciroth et al. 2008) and *Environmental Life Cycle Cost： a Code of Practice* (Swarr et al. 2011), in which LCC is classified into three types: conventional LCC, environmental LCC, and societal LCC. In this eco-efficiency study, the cost indicator is supposed to relate to the environmental indicator which is based on LCA; therefore the cost indicator was calculated according to the environmental LCC methodology (Swarr et al. 2011), in which the LCC is constructed in three steps: (i) goal and scope definition; (ii) LCI analysis; (iii) life cycle interpretation. According to the SETAC guide, LCC need not include the step of "impact assessment" as it is already clear that a lower cost is better. However, the types of cost and the time factor of the cost are also important when evaluating the economic impacts of technological innovations. We argue that not only the sum of the life cycle costs but also the breakdown of the cost structure needs to be investigated in the LCC analysis. Therefore, in this study, an "impact assessment" step is added to the LCC analysis, which consists of a definition of cost categories and cost impact category selection. In the first step, "cost category definition", cost breakdown structures were applied to present the cost distribution and to identify cost stressors. The second step, "cost impact category selection", introduced issues such as whether to employ a discount rate over time, and how the proposed life cycle cost will facilitate decision making. Figure 2.1 gives an overview of the updated integrated framework for the LCA/LCC type eco-efficiency analysis. The "economic impact assessment" step for the LCC analysis is depicted with a dashed rectangle in Figure 2.1, in analogy to the environmental impact assessment in LCA.

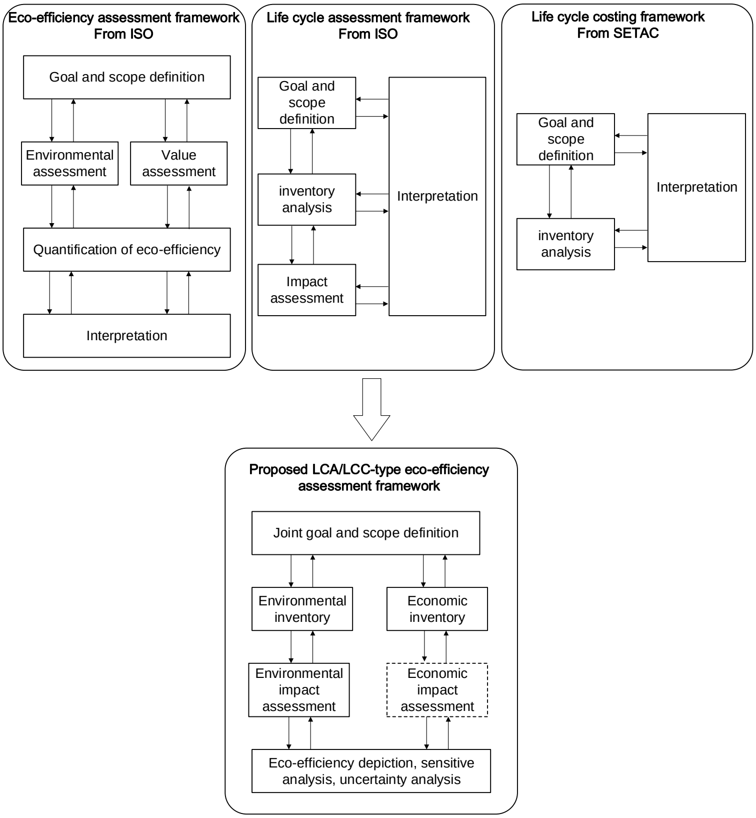

Figure 2.1 Proposed LCA/LCC-type eco-efficiency assessment framework, based on eco-efficiency assessment framework (ISO 2012), life cycle assessment framework (ISO 2006b), and SETAC environmental life cycle costing framework (Swarr et al. 2011; Ciroth et al. 2008)

2.3.2 Goal and scope definition

The goal of this study was to assess and compare the eco-efficiencies of four high-grade concrete recycling alternatives enabled by the technological innovations of ADR and HAS. The presently available high-grade recovery method — the wet process — serves as a reference to illustrate the potential changes led by the innovations. The geographic scope of the study is the Netherlands, where the field data of the case study were collected. The temporal scope of the study is recent years (2015 to 2019).

2.3.2.1 Description of the innovative technologies

Wet process

In 2010, when the C2CA project started, about 2% of the EoL concrete in the Netherlands was processed for high-grade applications, such as recovered clean aggregates for concrete. The commonly applied method is the wet process. Within the C2CA project, the wet process data were collected from a wet treatment plant located in Utrecht, which represents the BAU high-grade concrete recycling method. In the wet process, the pre-crushed concrete rubble (0–0.5mm) is transported by a truck to a stationary wet process treatment plant with a productivity of 150 ton/h. There the EoL concrete is broken down to 22 mm, and sieved into recycled coarse aggregate (RCA) above 4mm and sieve sand (SS) below 4 mm. Then the coarse fraction (4–22 mm) of the aggregates enters a long water bed for washing. After crushing and washing, the high-grade 4–22 mm RCA is sold for concrete manufacturing, substituting natural coarse aggregate (NCA). The washing residues are pumped to a thickener for sedimentation, and sludge is generated and sent to a landfilling site. The 0–4mm SS is a mixture of dirt, sand and hydrated cement, which prevents its high-grade application, e.g. clean sand for new concrete manufacturing. Consequently, SS is seen as a residue in the production of the 4–22 mm RCA. Due to its chemical inertness, SS is often piled up in-situ. However, if a nearby construction project needs to balance earthworks, SS could be given away free of charge or sold at a very low price. Since the application of SS is uncertain, in present study the environmental and economic impact of SS is cut-off. The mass balance of the investigated wet process is presented in Figure 2.2.

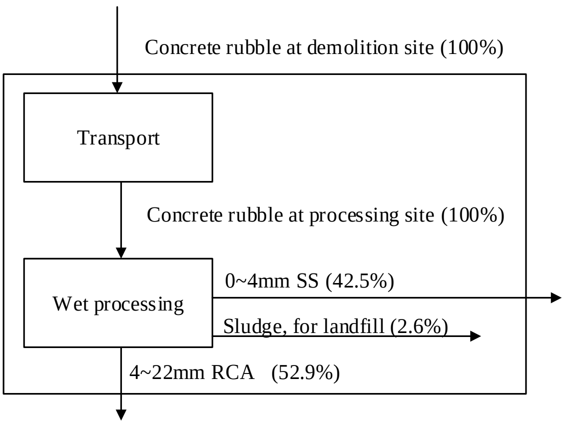

Figure 2.2 Mass balance of wet process

Advanced dry recovery (ADR)

When the C2CA project started, the ADR technology had already been successfully applied for the recovery of incineration bottom ash. In the C2CA project, the technology was used to recover the high-grade concrete aggregate. The original version of the ADR system was already much smaller than the wet processing plant; however, in the C2CA project, dismantling and assembling the ADR system took a week, which meant that in practice it was used as a stationary recycling plant. In the HISER project, the mobility of the ADR equipment was improved, and in the VEEP project, a truly mobile ADR set was developed, which can be transported by one truck and assembled and dismantled on-site within one day. In the case studies carried out in the C2CA, HISER and VEEP projects, the ADR process is combined with pre-crushing. In an ADR system, the EoL concrete of about 0.5 m is crushed to 22mm and sieved to a fraction above 12mm as a final product and below 12mm as ADR feed. The 12–22mm fraction is about 20% of the crusher output, which is quite clean and was used as clean coarse aggregate for concrete. About 80% of the crusher output is in the 0–12 mm fraction and is fed into the ADR set. The ADR breaks up the feed material and classifies it into 4–12 mm RCA, which is used as high-grade concrete aggregate, and 0–4 mm SS, which contains pollutants and for which no suitable high-value applications are found yet in the C2CA and HISER projects, hence it is usually stacked on site or left for land leveling or road foundation due to its inertness. As the mass balance of the ADR system (Figure 2.3) shows, the ADR set transforms 68% of its feed material into high-grade coarse aggregate and generates 32% of 0–4mm SS, for which suitable applications have to be found. Otherwise, the more concrete is recycled with the ADR system, the more 0–4mm fines will require disposal. Thus, the impact of 0–4mm SS is cut-off in the ADR process, as it is in the wet process.

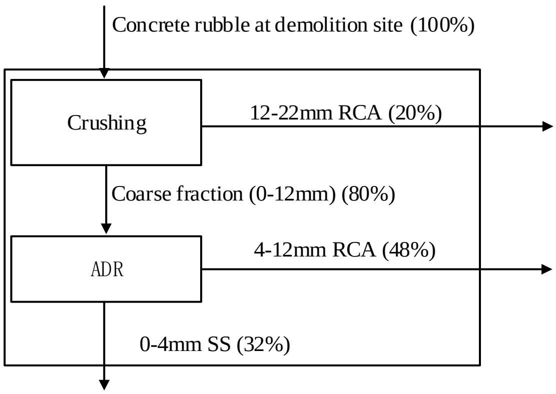

Figure 2.3 Mass balance of the Advance dry recovery (ADR)

Heating air classification system (HAS)

The VEEP project took up the challenge to valorize the fine products of ADR. In the exploration, the Heating Air Classification System (HAS) was proposed for treating the 0–4 mm SS residue of ADR. The HAS is capable of separating cementitious powder from the sandy part in the fine particle fraction. The HAS uses simultaneous heating, grinding and separation in a fluidized bed, which can remove most of the 0–0.125mm recycled ultrafine particles (RUP, 6.4%) from the 0.125–4mm recycled fine aggregate

(RFA, 25.6%). The RUP can be used to reduce cement consumption in concrete manufacturing, and the 0.125–4mm RFA can be used to substitute natural sand in concrete manufacturing. The mass balance of HAS is presented in Figure 2.4.

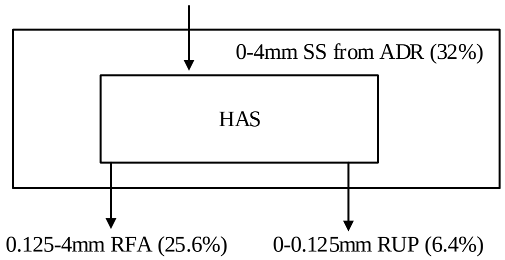

Figure 2.4 Mass balance of Heat air classification system (HAS)

Recycling systems used in the comparative evaluation

Offered by the series of innovations in high-grade concrete recycling, four systems representing the potential alternatives for the treatment of EoL concrete in the Netherlands are assessed to capture changes in eco-efficiency: (i) *BAU WP (wet process) system;* (ii) *ADR-S (stationary) system*; (iii) *ADR-M (mobile) system*; (iv) *A&H (ADR & HAS) system*. Details of each system are listed in Table 2.2.

Table 2.2 Four technological systems for concrete recycling in this study

Technological system	Description
S1. BAU WP (wet process) system	In the wet process system, EoL concrete is transported to the wet processing plant. Through the wet recycling process, 4–22 mm RCA for concrete manufacturing is produced together with the below 4mm SS.
S2. ADR-S (stationary) system	In the ADR-S system, crushed concrete rubble is transported by truck to the plant with a stationary ADR set. 4–12 mm and 12–22 mm RCA for concrete manufacturing is produced together with the below 4mm SS.
S3. ADR-M (mobile) system	In the ADR-M system, a mobile ADR set is transported and reassembled for in-situ treatment. 4–12 mm and 12–22 mm RCA for concrete manufacturing is produced together with the below 4mm SS.
S4. A&H (ADR & HAS) system	In the A&H system, mobile ADR and HAS sets are transported and reassembled at the demolition site for on-site production. 4–12 mm and 12–22 mm RCA, 0.125–4 mm RFA, and 0–0.125 mm RUP for concrete manufacturing are produced.

2.3.2.2 Functional unit

Comparability of assessment is particularly critical when different systems are being evaluated (ISO 2006a). Since the wet process, the ADR and HAS system deliver different products, each product system was expanded to ensure comparability. Since the residue 0-4mm SS is cut-off due to its uncertain position as a good or waste, the basket of functions for the comparison of the expanded product systems are: (i) EoL concrete treatment, (ii) coarse aggregate for concrete production, (iii) fine aggregate for concrete production, (iv) cementitious material for concrete production. Based on the mass balance of the combined ADR and HAS system the functional unit for the comparative study is defined as follows (see Figure 2.5):

a. *treatment of 100 tons of EoL concrete,*

b. *68 tons of 4–22mm coarse aggregate for concrete;*

c. *25.6 tons of 0.125–4mm fine aggregate for concrete;*

d. *6.4 tons of cementitious material for concrete.*

The reference flows of each system are presented in Table 2.3.

Table 2.3 Reference flows in each system under a coherent functional unit

	S1 BAU WP	***S2 ADR-S***	***S3 ADR-M***	***S4 A&H***
Functional unit-a: Treatment of 100 tons of EoL concrete	Transportation and treatment of 100 tons of EoL concrete to and in wet processing plant	Transportation and treatment of 100 tons of EoL concrete to and in ADR stationary plant	Transportation of ADR set and in-situ treatment of 100 tons of EoL concrete	Transportation of ADR and HAS sets and in-situ treatment of 100 tons of EoL concrete
Functional unit-b: Production of 68 tons of 4–22mm coarse aggregate	Production of 52.9 tons of [**4–22mm RCA]** and 15.1 tons of **<u>4–22mm NCA</u>**	Production of 20 tons of [**12–22mm RCA]** and 48 tons of [**4–12mm RCA]**		
Functional unit-c: Production of 25.6 tons of 0.125–4mm fine aggregate for concrete	Production of 25.6 tons of **<u>0.125–4mm sand</u>**			Production of 25.6 tons of **[0.125–4mm RFA]**
Functional unit-d: Production of 6.4 tons of cementitious material for concrete	Production of 6.4 tons of **<u>cement</u>**			Production of 6.4 tons of [**RUP]**

Note: Secondary products are marked in **[bracket]**; primary products are **underlined**. EoL denotes end-of-life, RCA means Recycled Coarse Aggregate, NCA means Natural coarse aggregate, RCA means Recycled coarse aggregate, RFA means Recycled fine aggregate and RUP represents Recycled ultrafine particle.

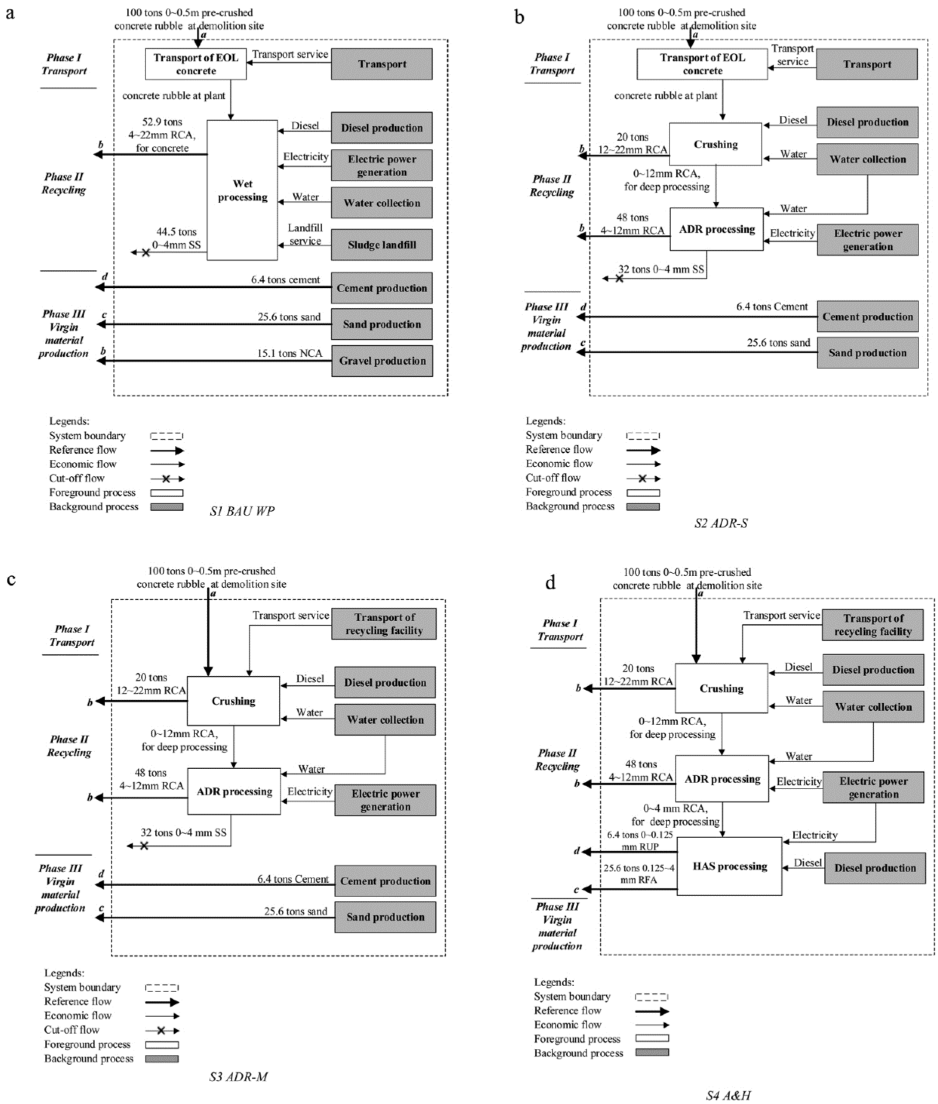

Figure 2.5 Flow diagrams for the four systems: (a). *S1 BAU WP* (wet process) system; (b). *S2 ADR-S* (stationary) system; (c). *S3 ADR-M* (mobile) system; and (d). *S4 A&H* (ADR & HAS) system, those flows which cross through the system boundary are functional flows, those lower-case letters at the end of the reference flow arrows refer to the sub-functional unit in Table 2.3.

2.3.3 Life cycle assessment (LCA)

2.3.3.1 Environment inventory analysis

System boundary and unit processes

Inventory analysis is the phase that defines the product system, including system boundaries, flow diagram with unit processes, data collection and allocation for multifunctionality (Guinée et al. 2001). Since the Netherlands is one of the major European countries involved in C2CA, HISER and VEEP projects for technological systems development. This study takes the Netherlands as the geographical reference area. Since selective demolition and sorting is a common practice in the Netherlands, very few contaminations are contained in the EoL concrete waste. To simplify modeling, unnecessary process like residue disposal is omitted in this study. It is assumed that the target EoL concrete for analysis does not contain any contamination. After selective demolition and sorting, EoL concrete generated at the construction site in the Netherlands will be crushed into 0–0.5m size and then sorted on-site, and the cost and impacts of this procedure will not be considered.

The life cycle considered in the study comprises three phases: *I) Transport*; *II) Recycling*; and *III) Virgin material production*. The first phase considers the transportation of the EoL concrete for treatment. It varies from different technology systems. For the off-site ones, it includes the transportation of the EoL concrete from the demolition site to the recycling plant. For the in-situ recycling pathways, it refers to the transportation of the processing equipment. The recycling phase is about processing EoL concrete into diverse secondary products, which can be used as raw materials for concrete manufacturing, so save virgin materials, accordingly. In order to guarantee compatibility across different technology systems, virgin material production processes are added in several systems, which are grouped in the phase of virgin material production. It is assumed that the transport costs for the secondary products are the same as for virgin materials to their next destination. Based on the defined 3 phases, the flows diagrams for 4 systems are depicted in Figure 2.5. As experiments have shown that the use of the secondary raw materials (0–0.125 mm RUP and 0.125–4 mm RFA) produced by HAS can reduce comparable amounts of virgin cement and virgin sand in concrete production (Technalia 2018). It is modeled as that the generation of HAS fine products 0.125–4mm RFA and 0–0.125mm RUP will lead to the avoided production of the virgin sand and cement.

Data collection

As indicated, process-based LCA was used for the environmental impact assessment. We used the software OpenLCA 1.7.4 to perform the LCA analysis with the Ecoinvent 3.4 database in combination with foreground processes, which are listed in Table A2.1 and Table A2.2 in the Appendix. The background processes that were linked to the foregrounds are listed in Table A2.3 in the Appendix. Data for the BAU WP system were obtained from an industrial wet treatment plant located in Utrecht, the Netherlands (within the C2CA project). Data for ADR was collected from the semi-mobile installation in the C2CA project and from the ADR demonstration in the HISER project. The mobile HAS data is gathered from the Recycling Lab at the Technology University of Delft, the Netherlands. Data of energy use and emissions were compared to those of relevant diesel-engine equipment for verification. For the technical systems which do not generate certain secondary products as specified in the functional unit in Table 2.3, the production of their natural counterpart materials (e.g. gravel, virgin cement, virgin sands) were modeled by using Ecoinvent datasets.

Multifunctionality

When a process delivers more than one function, we encounter a ‘multifunctionality’ problem. ISO 14040 (2006) recommends avoiding allocation by either dividing processes or expanding the system boundary. According to the data obtained, multifunctional processes cannot be divided into discrete sub-processes, thus the system boundary was expanded by using a basket-type functional unit. In S1 BAU WP especially, recycling of 100 tons of EoL concrete through WP will generate 52.9 tons of RCA but less than the amount of 68 tons in the functional unit. Thus, an additional 15.1 tons of NCA is produced in S1 BAU WP. Besides, in S1 BAU WP, S2 ADR-S, and S3 ADR-M, 25.6 tons of virgin sand and 6.4 tons of cement are produced.

2.3.3.2 Environment impact assessment

The impact assessment phase in an LCA includes the characterization of the result based on an impact category selected, followed by an optional normalization and weighting process (Guinée et al. 2001). ISO 14044 requires a deliberate assessment of all relevant impact categories for an LCA study; therefore, it is not allowed to leave out impact categories that have a significant impact. Besides, the evaluation of a range of novel technologies indicates the need for a broader environmental perspective. Joint Research Centre of the European Commission (EC-JRC) recommended a comprehensive ILCD life cycle impact assessment method. The impact categories in the ILCD method (ILCD 2011, midpoint, v1.0.10, August 2016) are shown in Table 2.4.

Table 2.4 ILCD impact assessment method

Impact category indicators	Units
Acidification	mole H+ eq
Climate change	kg CO_2 eq
Freshwater ecotoxicity	CTUe
Freshwater eutrophication	kg P eq
Human toxicity, cancer effects	CTUh
Human toxicity, non-cancer effects	CTUh
Ionizing radiation, human health	kBq U235 eq

Land use	kg C deficit
Marine eutrophication	kg N eq
Mineral, fossil & renewable resource depletion	kg Sb eq
Ozone depletion	kg CFC-11 eq
Particulate matter	kg PM2.5 eq
Photochemical ozone formation	kg NMVOC eq
Terrestrial eutrophication	mole N eq
Water resource depletion	m^3 water eq

Normalization and weighting are optional steps of LCA according to ISO 14040/14044 to rank the impacts of a system. However, decision-making becomes easier when the impacts are normalized, as this compares the contribution of a particular service with the overall environmental problems under eco-efficiency consideration (Kicherer et al. 2007). Normalization was based on "JRC EU 27, 2010, total [year]", which stands for impact in 2010 of the 27 European Union countries.

After normalization, the next step is to combine the normalized values via a weighting scheme. ISO 14045 (2012) regulates that weighting shall not be used in a comparative eco-efficiency analysis intended to be disclosed to the public. However, in order to present a solution to the sensitivity and uncertainty analysis of the final eco-efficiency results, this case study tried to weigh the environmental indicators in a relatively objective way. In fact, in an eco-efficiency context, it may be found that one recycling system is better than another for some impact categories but poorer for others. In that case, it is difficult to figure out whether the total environmental performance was improved or deteriorated. Thus, a weighting method is indispensable to aggregate all impact category indicators into one sole environmental score, making it possible to calculate an eco-efficiency ratio. There is no scientific basis for weighting LCA results as weighting requires value choices (ISO 2006a). However, the expert opinions about impact category weights are sensitive to either subjective biases in elicitation situations or in local characteristics (Seppälä et al. 2005), which may consequently result in a wide range of uncertainty. To render the results universally compatible and applicable for all EU member states, this study applied an equal weight (0.066) recommended by EC-JRC (2016).

2.3.4 Life cycle costing (LCC)

2.3.4.1 Economic inventory analysis

Data collection

LCC analysis shares the same system boundary as that of LCA. All costs are expressed in the currency of the Netherlands: euro (€). It is also a problem that some economic values keep fluctuating over time, such as the price of aggregate, which shifts with market supply and demand. We, therefore, used historically observed data from different sources and then adjusted those data according to confirmation with relevant actors. To perform the LCC study, Microsoft office 2016 Excel was used to investigate the main contributions of costs, connected with a parametric cost database. The cost data were validated by comparing them to the Ecoinvent 3.4 cost database to avoid noticeable deviation. Details and sources of the price data are presented in Table 2.5.

Table 2.5 Cost data in three life cycle phases and their sources

Life cycle phase	Explanation
Phase I Transport	**Transport cost (TC):** costs related to the transport of raw and ancillary materials, EoL concrete, products, and equipment. **Waste transport:** the transport cost is 0.1 €/t·km (including the cost of fuel and personnel costs of the staff)[1]. **Equipment transport:** The transport cost (including dismantling/reassemble) of ADR and HAS set is 2000 € (round trip). Transport cost of ADR and HAS for treatment per 100 tons of EoL concrete is 13.33 €; transport cost of ADR for treating per 100 tons of EoL concrete is 10.26 € [2].
Phase II Recycling	**Equipment cost (EC):** costs related to equipment and facility. In this study assuming the recycling company bought and owned the equipment, so equipment depreciation is selected standing for equipment cost. Hourly depreciation of each piece of equipment in this study is as follows: crushing set (including crusher: 1313, excavator: Case CX350D, Rubber-wheel loader: Case 921E): 147.67 €/h [3]; ADR with sensor: 83.73 €/h (ADR: 61.44 €/h, LIBS quality sensor: 22.29 EUR/h) [3]; HAS: 14.73 €/h [2]; wet processing plant: 3.23 €/t [4].
	Personnel cost (PC): costs related to wages and salaries. Wages and salaries in the construction sector are set as 35.9 €/man-hour [5]. Especially personnel cost for the wet processing plant is 0.65 €/t [4].
	Utility cost (UC): costs related to utilities (e.g. electricity, diesel, water). Diesel price is 0.73 €/L [6]; electricity price is 0.06 €/kWh [7]; water (for dust control) price is 0.16 €/L [1]; tap water (for wet process) is 0.003 €/L [1]. Lubricating oil for machines is omitted from this study.
	Waste treatment cost (WC): costs related to sludge treatment (wet process methods only). Sludge treatment is 25 €/ton (including transport) [4].
Phase III Production of virgin material	**Virgin material cost (VC):** costs related to the procurement of primary raw materials which cannot be produced through the wet process and ADR. NCA price is 10.2 €/ton [1]; sand price is 12 €/ton [1]; cement price is 75 €/ton [1].

Notes: [1] data from Strukton BV without documental support; [2] data from an investigation at Technology University of Delft; [3] data from HISER project unpublished report "Final Report of Integrated environmental and economic assessment for the HISER case studies" in 2018; [4] data from C2CA project unpublished report "Life cycle costing of concrete recycling: comparison between a conventional and the C2CA technology" in 2016; [5] EUROSTAT, Labor cost levels by NACE Rev. 2 activity (the Netherlands, 2018), via https://ec.europa.eu/eurostat/web/products-datasets/-/lc_lci_lev; [6] data from Ecoinvent 3.4 cost database; 7data from Eurostat "Electricity prices for non-household consumers - bi-annual data" via https://ec.europa.eu/eurostat/web/products-datasets/-/nrg_pc_205 ;

Multifunctionality

The solution for multifunctionality in LCC was the same as that of LCA, and system expansion was used.

2.3.4.2 Economic impact assessment

LCC quantifies costs to operate the same technological systems that were evaluated in LCA, while SETAC suggested not to have an impact assessment step for LCC (Swarr et al. 2011). Moreover, the life cycle costs of a product is a number expressed in monetary units; thus normalization and weighting are not performed either (Swarr et al. 2011). However, for different product systems, we were faced with different cost categories, while costs and benefits could also be incurred at different moments in time. An economic impact assessment was performed in this section to better align the economic information with the environmental ones generated by LCA. We propose two stages in the economic impact assessment: (i) cost category definition, which answers the question of how the cost will be structured in LCC analysis; and (ii) cost impact category, which answers questions on how the time factor will be considered and how the final cost value will be expressed.

Cost category definition

Given the diversity of LCC equations, the selection of LCC equations can play a central role in how LCC results are interpreted (Miah et al. 2017). The life cycle cost can always be broken down according to the life cycle phases, such as in the concrete recycling case, as shown in Eq. (2), where C_I, C_{II}, and C_{III} represent the costs of Phase I Transport, Phase II Recycling and Phase III Production of virgin material, respectively.

$$\textit{Life cycle cost} = C_I + C_{II} + C_{III} \tag{2}$$

On the other hand, the costs can also be categorized into different types of cost, such as transport cost (TC), equipment cost (EC), personnel cost (PC), utility cost (UC), waste treatment cost (WC), virgin material cost (VC). Thus, the life cycle cost can be estimated as in Eq. (3).

$$\textit{Life cycle cost} = TC + EC + PC + UC + WC + VC \tag{3}$$

If life cycle cost is estimated via Eq. (2), it will be clear how the cost is attributed to each phase; via Eq. (3), we would know the share for each category of cost. Thus, in this study, we could deploy cost structure breakdown using these two forms of cost category. In principle, further differentiation of costs and benefits is possible, i.e. which actors over the life cycle are confronted with costs and benefits. Since in this LCC analysis, there is only one stakeholder (the recycling company), adding actors as a third dimension was not considered here.

Cost impact category selection

In the cost impact category selection stage, two main issues were addressed: (i) will the incurring moment of the costs and benefits in time be considered? (ii) how will the final cost value be expressed? If costs and benefits are spread out over a long time span, a conscious decision is needed on whether one wants to discount costs and benefits that occur in the future, and which discount rate is applied, which leads to a dynamic-type

LCC model; on the other hand, if costs and benefits occur in a very short time span, discounting does not need to be considered, which results in a static-type LCC model (Ciroth et al. 2008). In this study, all unit processes take place in a short period; therefore, we add costs and benefits without considering any discounting over time. In fact, as mentioned in the SETAC LCC book (Ciroth et al. 2008), environmental LCC usually is a steady-state method. Discounting of the final result of an environmental LCC will be specified in detail in our further studies.

For the question of how to express the cost values, 9 approaches can be considered (see Table 2.6). Some other approaches were mentioned by Miah et al. (2017), such as “Net LCC” by Menikpura et al. (2016), “Total Annualized Equivalent Cost” by Pretel et al. (2016), and “Resale Value” by Minne and Crittenden (2015). These are conceptually overlapping with those in Table 2.6. Furthermore, the “Present Worth” method developed by Afrane and Ntiamoah (2012) includes the monetization of externalities, which does not fit in this eco-efficiency approach, where the environmental dimension is a separate one covered by LCA. For those systems that contain costs over longer time spans, discounting can play a role in considering the time value, and cost can be expressed in forms from C_1 to C_8. Since the costing system is defined as a static-state type, two possible LCC impact categories C_8 and C_9 were selected. Firstly, the exact cost of each technology was investigated without considering the time span; thus all costs in this LCC analysis are presented in static-state cost (C_9). Then, to make each technology more comparable in the form of an eco-efficiency indicator, the life cycle cost of each system was normalized based on the baseline reference in relative value (C_8).

Table 2.6 LCC impact categories

Label	LCC impact categories	Source	Costs over long time spans
C_1	Net Present-Value	(Akhlaghi 1987)	Yes
C_2	Net Annual-Value	(Akhlaghi 1987)	Yes
C_3	Net Savings	(Akhlaghi 1987)	Yes
C_4	Savings-to-Investment Ratio	(Akhlaghi 1987)	Yes
C_5	Adjusted Internal Rate of Return	(Akhlaghi 1987)	Yes
C_6	Payback Period	(Almutairi et al. 2015)	Yes
C_7	Global Cost	(EN 15459 2008)	It depends
C_8	Normalized Cost	(Zhao et al. 2011)	It depends
C_9	Static State Cost	(Luo et al. 2009)	No

2.3.5 Integration of LCA and LCC for eco-efficiency indicators

In this phase the environmental and economic results were elaborated by contribution analysis for identification of dominating factors, then the form of how the eco-efficiency indicator will be expressed was selected, and sensitivity and uncertainty analysis were conducted to evaluate the robustness. Firstly, should eco-efficiency be expressed graphically or numerically? It is clear from Figure 2.6 that there are two methods to express eco-efficiency: via numeric value and a two-dimensional diagram, and the last one is the most frequently used method. Providing a unified numeric value is convenient for decision making. However, it does not give easy insight into the relative scale and importance, and into the trade-offs between environmental and cost aspects. To overcome this drawback, the economic and environmental aspects can be plotted in a more visible and evident manner in a two-dimensional diagram. Therefore, a two-dimensional diagram was used to visualize the eco-efficiency results.

Secondly, there is an issue on whether the LCA and LCC results in the eco-efficiency graphs should be expressed in an absolute-value way (Huppes and Ishikawa 2005) or relative-value way (Woon and Lo 2016). In this study, evaluation of the eco-efficiency of technological innovations in high-grade concrete recycling would lead to different scores. The eco-efficiency of the existing recovery technology wet process was used as the reference basis. In this context, we believe LCA/LCC in the percentage form would better reflect the improvement of an innovative system compared to the BAU system, thus a modified eco-efficiency indicator was adopted, presenting LCA/LCA results in a relative-value method. The eco-efficiency was interpreted through a two-dimensional graph in Figure 2.6.

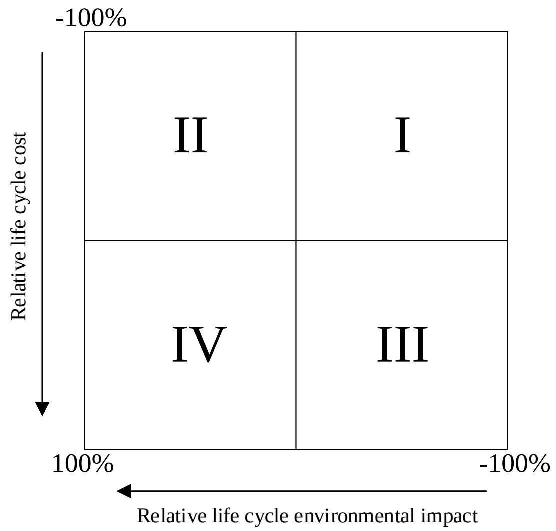

Figure 2.6 Eco-efficiency indicator graph

The cost saved is presented through a relative LCC index in Eq. (4) as the Y-axis of the graph; the relative LCA index is expressed through a relative LCA index in Eq. (5) as the X-axis of the graph. Zone I represents full eco-efficiency (lower environmental impact and cost); Zone II (higher environmental impact, lower cost) and III (lower environmental impact, higher cost) indicate half eco-efficiency; Zone IV depicts non-eco-efficiency (higher environmental impact and cost). Therefore, if the location of a recycling system is closer to the upper-right it represents a higher rate of eco-efficiency.

$$Relative\ LCC\ index = (\frac{LCC_{NOV} - LCC_{BAU}}{LCC_{BAU}}) \times 100\% \tag{4}$$

$$Relative\ LCA\ index = (\frac{LCA_{NOV} - LCA_{BAU}}{LCA_{BAU}}) \times 100\% \tag{5}$$

Where LCC_{NOV}, life cycle economic score of novel treatment; LCC_{BAU}, life cycle costs of BAU treatment; LCA_{NOV}, life cycle environmental score of novel treatment; LCA_{BAU}, life cycle environmental score of BAU treatment. The *S1 BAU WP* is set as the origin point.

2.4 Results

2.4.1 Results of LCA

Table 2.7 presents the indicator results calculated with OpenLCA, 15 impact categories for each system. The normalized results of 15 impact categories for each system are presented in Figure 2.7.

Table 2.7 Characterized life cycle environmental impact of four systems

Impact category	S1 BAU WP	S2 ADR-S	S3 ADR-M	S4 A&H	Unit
Acidification	1.77E+01	1.66E+01	1.23E+01	1.67E+01	mole H+ eq.
Climate change	5.15E+03	4.85E+03	4.24E+03	1.63E+03	kg CO_2 eq.
Freshwater ecotoxicity	1.41E+04	1.28E+04	8.86E+03	2.21E+03	CTUe
Freshwater eutrophication	7.18E-01	6.05E-01	5.65E-01	9.79E-02	kg P eq.
Human toxicity - carcinogenics	1.10E-04	9.51E-05	7.91E-05	4.56E-05	CTUh
Human toxicity - non-carcinogenics	6.20E-04	5.70E-04	4.20E-04	9.18E-05	CTUh
Ionizing radiation - human health	2.92E+02	2.58E+02	2.01E+02	1.13E+02	kg U235 eq.
Land use	1.10E+04	8.85E+03	5.66E+03	4.01E+03	kg SOC
Marine eutrophication	4.79E+00	4.48E+00	2.79E+00	7.20E+00	kg N eq.
Ozone depletion	2.80E-04	2.60E-04	1.40E-04	2.90E-04	kg CFC-11 eq.
Particulate matter	1.54E+00	1.43E+00	1.02E+00	2.05E+00	kg PM2.5 eq.
Photochemical ozone formation	1.42E+01	1.33E+01	8.14E+00	2.17E+01	kg C2H4 eq.
Resource depletion	9.26E-02	8.18E-02	5.05E-02	1.10E-02	kg Sb eq.
Resource depletion - water	1.20E+01	8.07E+00	7.73E+00	6.92E-01	m^3
Terrestrial eutrophication	5.47E+01	5.08E+01	3.22E+01	7.91E+01	mole N eq.

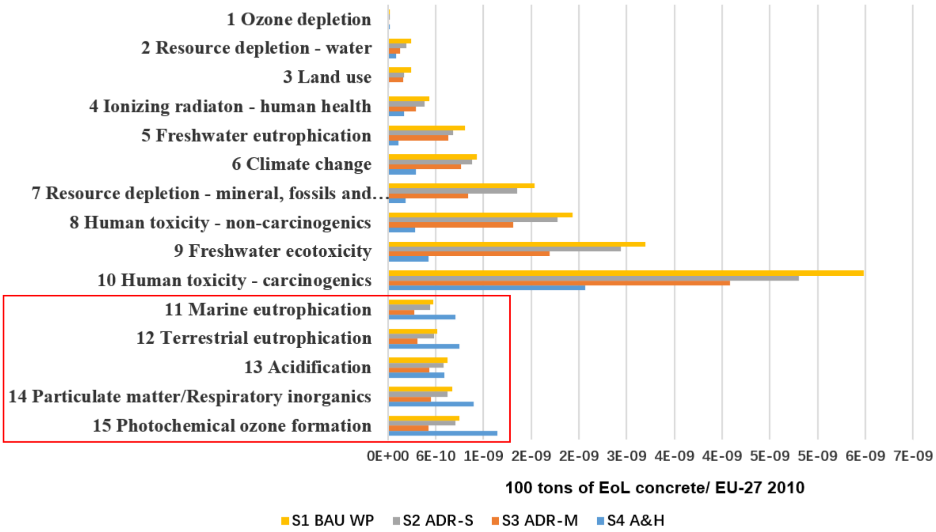

Figure 2.7 Normalized life cycle environmental impact of four systems

Figure 2.7 shows that from impact category indicator 1 to 10, the values of the environmental impact of systems from S1 to S4 presents an ascending trend; in contrast, from indicator 11 to15 (in the rectangle), S4 A&H has the highest environmental impact (resulting from diesel consumption). Thus, the selection of impact category method will probably affect the environmental performance superiority of S4 A&H. All 15 impact indicators are considered in this study, however, uncertainty on the choice of impact categories cannot be modeled due to the limitation of the software.

Figure 2.8 shows that generally technological development is associated with a clear descending trend in the weighted environmental impact. Firstly, transportability is essential for the comparative advantages of an EoL concrete waste recycling system. Transport accounts for around 25% of the life cycle environmental impact in stationary recycling methods (S1 BAU WP and S2 ADR-S). After optimization of the transportability of the recycling equipment (S3 ADR-S and S4 A&H), less than 1% of the life cycle environmental impact is contributed by transport. Another factor contributing to the comparative advantages of the HAS system is the high-value recovery of secondary raw materials. For the first three systems, S1 BAU, S2 DAR-S, and S3 ADR-M, the impact of virgin material production contributes 69%, 72% and 95% to their life cycle impact, respectively. Even though the HAS technology shows a surging increase of environmental impact in the recycling phase, from the calculated results we can see that its advantages can certainly be realized since the virgin cement and sand consumption can, in fact, be reduced by using the recovered RUP and RFA. Compared to the wet process (S1 BAU WP) and the ADR system (S2 and S3), HAS technology (S4 A&H) can reduce the total environmental impact by 31%–54%.

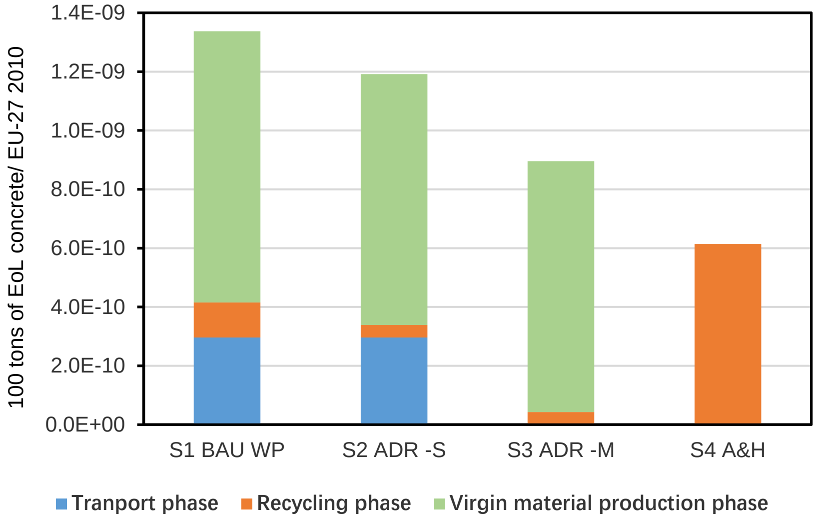

Figure 2.8 Distribution of the weighted environmental impact score in three phases

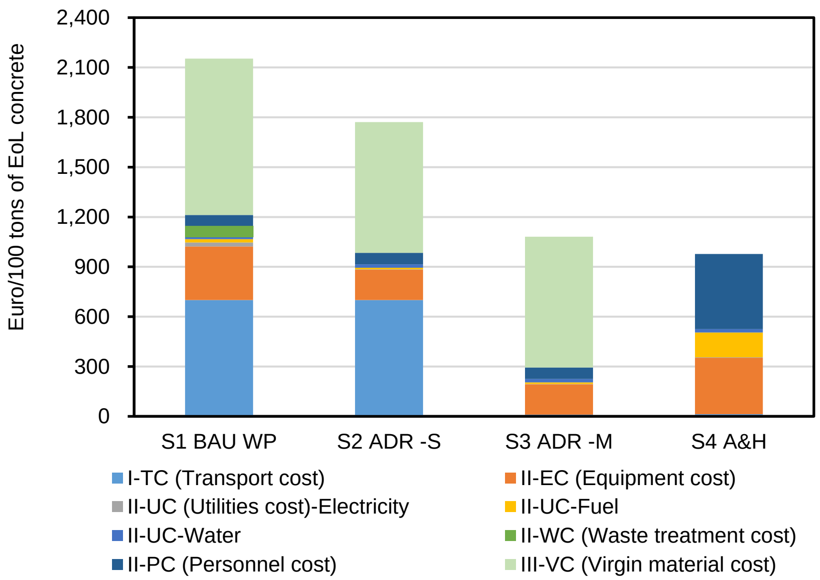

Figure 2.9 Life cycle costs of four systems

2.4.2 Results of LCC

The LCC results in Figure 2.9 show a similar trend as the LCA results. From an economic perspective, four systems show a descending cost trend, and S4 A&H is the most cost-efficient pathway. In general, cost savings are mainly realized by a reduction in transport and production of higher value-added materials. Compared to the stationary recycling methods (S1 BAU and S2 ADR-S), on-site recycling systems (S3 ADR-S and S4 A&H) can reduce the share of transport in life cycle costs from 33%–44% to 1%. Furthermore, the life cycle costs of S3 ADR-S are slightly higher (9%) than that of S4 A&H, although they both can be considered economically feasible methods for concrete recycling. However, there is a clear trade-off between virgin material cost (in S3 ADR-S) and personnel cost (in S4 A&H).

2.4.3 Eco-efficiency index

Based on the modified eco-efficiency Eqs. (5) and (6), the life cycle cost and life cycle environmental impact are translated into the relative life cycle cost and relative life cycle environment impact, respectively. Then those relative values are located in the eco-efficiency graph as shown in Figure 2.10. Graphically, all comparative systems are located in Zone I, and the S4 A&H is the best choice for concrete recycling from an eco-efficient perspective, as it can noticeably reduce both life cycle environmental and economic burdens by about 55%. The S1 BAU WP turns out to be the costliest and most environmentally unfriendly pathway, and S2 ADR-S only slightly improved the eco-efficiency by around 20%.

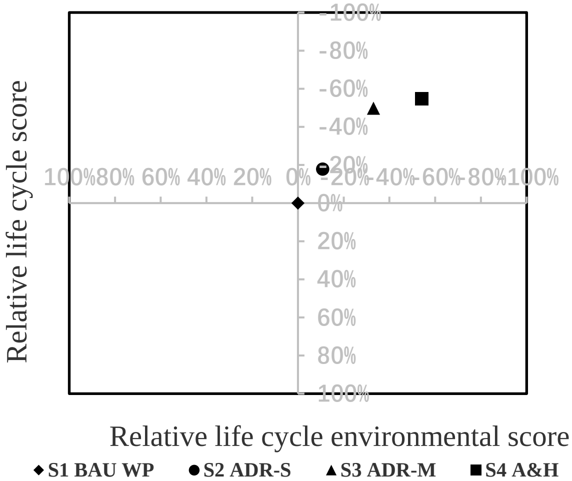

Figure 2.10 Eco-efficiency index diagram

2.4.4 Sensitivity analysis

For the assessments to be useful in the actual decision-making processes, knowledge of the uncertainty and sensitivity of the data is of great significance. In the assessment, LCA and LCC were used in an eco-efficiency assessment to estimate the environmental impact and economic value. The sensitivity and uncertainty in the calculation may result from inventory data, allocation options, characterization factors, and weighting factors. According to the Handbook on life cycle assessment (Guinée et al. 2001), sensitivity and uncertainty analysis cannot be made obligatory due to the limited functionality of LCA software, but it is recommended to implement such an analysis at least partially. Since the latest version OpenLCA 1.10.3 is unavailable for full sensitivity and uncertainty analysis, the present study only considered the sensitivity and uncertainty of critical cost data and unit process data from environmental and economic inventories. Based on the economic and environmental results highlighted, we list the most relevant 15 factors with respect to variations as shown in Table 2.8.

Table 2.8 Factors for sensitivity analysis

Factors category	Factor	Remark
Cost data	s_1	Diesel price
	s_2	Personnel cost
	s_3	Cement price
	s_4	Sand price
	s_5	NCA price
	s_6	Transport price
	s_7	WP plant depreciation cost
	s_8	ADR depreciation cost
	s_9	HAS depreciation cost
Unit process data	s_{10}	Distance of demolition site to wet processing plant
	s_{11}	Distance of demolition site to ADR Plant
	s_{12}	Distance of storage of ADR and HAS to a demolition site
	s_{13}	EoL concrete generation at a demolition site
	s_{14}	Unit diesel usage of HAS
	s_{15}	Productivity of HAS

The sensitivity analysis was conducted to identify the factors that are the most sensitive to economic and environmental performance. By decreasing 10% of each factor, their sensitivity is shown in Table 2.9. A positive value of sensitivity is presented in red and a negative value of sensitivity is presented in green. The darker its color, the more sensitive the factor will be. For stationary recycling systems, S1 BAU WP and S2 ADR-S are the most sensitive to transport-relative factors (price and travel distance), followed by cement price; in contrast, mobile recycling systems S3 ADR-M, S4 A&H are insensitive to transport. S3 ADR-S is the most sensitive to cement and sand price, while

S4 A&H is noticeably sensitive to HAS productivity, which, however, will not be improved currently. S4 A&H is also sensitive to personnel costs.

Table 2.9 Sensitivity analysis results (each factor decreased by 10%)

	LCC score				LCA score			
	S1 BAU	S2 ADR-S	S3 ADR-M	S4 A&H	S1 BAU	S2 ADR-S	S3 ADR-M	S4 A&H
s1	-0.09%	-0.05%	-0.09%	-1.53%	0.00%	0.00%	0.00%	0.00%
s2	-0.30%	-0.39%	-0.64%	-4.63%	0.00%	0.00%	0.00%	0.00%
s3	-2.23%	-2.71%	-4.44%	0.00%	0.00%	0.00%	0.00%	0.00%
s4	-1.43%	-1.73%	-2.84%	0.00%	0.00%	0.00%	0.00%	0.00%
s5	-0.72%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
s6	-3.25%	-3.95%	-0.09%	-0.14%	0.00%	0.00%	0.00%	0.00%
s7	-1.50%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
s8	0.00%	-0.76%	-1.24%	-1.37%	0.00%	0.00%	0.00%	0.00%
s9	0.00%	0.00%	0.00%	-1.61%	0.00%	0.00%	0.00%	0.00%
s10	-3.25%	0.00%	0.00%	0.00%	-2.22%	0.00%	0.00%	0.00%
s11	0.00%	-3.95%	0.00%	0.00%	0.00%	-2.49%	0.00%	0.00%
s12	0.00%	0.00%	-0.19%	-0.27%	0.00%	0.00%	-0.01%	-0.01%
s13	0.00%	0.00%	0.14%	0.15%	0.00%	0.00%	0.04%	0.07%
s14	0.00%	0.00%	0.00%	-1.43%	0.00%	0.00%	0.00%	-4.25%
s15	0.00%	0.00%	0.00%	6.96%	0.00%	0.00%	0.00%	4.25%

Note:	0.00%	0.00% ~ -2.00%	-2.00%~ -4.00%	-4.00%~-6.00%	-6.00% ~ -8.00%
		0.00% ~ 2.00%	2.00% ~ 4.00%	4.00% ~ 6.00%	6.00% ~ 8.00%

2.4.5 Uncertainty analysis

The factors which were evaluated in the sensitivity analysis were selected for the uncertainty analysis. Their values of the range were determined by consulting with relevant actors, as shown in Table 2.10. Since according to the HAS developer, the productivity and unit diesel usage of HAS will remain steady in the near future, therefore their uncertainty was not considered. A single standard error range of ±5% for the LCI data was chosen in this study, which is an accepted approach to the uncertainty of LCI data (Huijbregts et al. 2003). Thus, a market price fluctuation range of ±5% for LCC uncertainty factors (from u1 to u9) and environmental inventory data (u14) was selected. Apart from that, truck travel distance and the amount of EoL concrete generation at demolition site have a larger range of uncertainty, more than 50% of fluctuating rate was given to those factors u10 to u13 as shown in Table 2.10.

Table 2.10 Relevant factors for uncertainty analysis

Cost category	**Code**	**Value range of factor**
Cost data	u_1	Diesel price (€/L): 0.73 ±5%
	u_2	Personnel cost (€/man-hour): 34.8 ±5%
	u_3	Cement price (€/t): 75 ± 5%
	u_4	Sand price (€/t): 12 ± 5%
	u_5	NCA price (€/t): 10.2 ±5%
	u_6	Transport price (€/t·km): 0.1 ± 5%
	u_7	WP plant depreciation cost (€/t): 3.23 ± 5%
	u_8	ADR depreciation cost (€/h): 83.73 ±5%

	u_9	HAS depreciation cost (€/h): 14.73 ±5%
Unit process data	u_{10}	Demolition site to the wet processing plant (km): 70 ±50%
	u_{11}	Demolition site to ADR Plant (km): 70 ±50%
	u_{12}	Storage of ADR and HAS to demolition site (km): 20 ±50%
	u_{13}	EoL concrete generation at demolition site (t): -50% – +200%
	u_{14}	Other environmental inventory data in LCA: ±5%

Taking into account the uncertainty of those data, the final economic and environmental performance with uncertainty ranges of each scenario is shown in Figure 2.11. The stationary recycling systems S1 BAU and S2 ADR-S have a wider range of uncertainty mainly because of the fluctuation of truck travel distance.

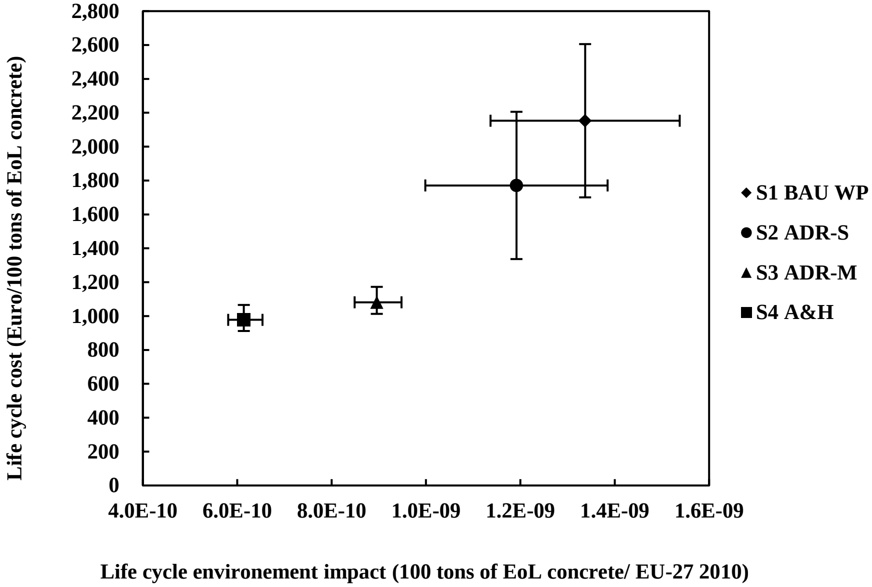

Figure 2.11 The uncertainty of eco-efficiency for four systems

2.5 Discussion

The results indicate that different technological innovations have different potentials to improve eco-efficiency. Technological innovations are responsible for improving product quality and reducing the recycling cost, while policies are responsible for fostering a functional market for the recycled concrete product to evolve. The results do not intend to be precise quantifications, but rather to demonstrate the potential contributions of those EoL concrete technological strategies toward sustainable growth.

Following the eco-efficiency assessment, if policymakers want to support the eco-efficient growth of concrete recycling networks and technologies, then they should impose relevant policies at least for the following perspectives. In this section, we discuss relevant policy implications in relation to currently existing policies at the EU, National (Dutch), and local levels.

2.5.1 Current policies related to CDW management

At the EU level, there are several policy frameworks related to recovery and recycling of CDW, for example, the *7th Environment Action Program, WFD (2008/98/EC)*; *Roadmap to a Resource Efficient Europe (COM(2011) 571 final)*, *Resource efficiency opportunities in the building sector (COM(2014) 445 final)*, *Towards a circular economy: A zero waste programme for Europe (COM(2014) 398 final)*, and *EU Construction and Demolition Waste Management Protocol*, *Landfill Directive (99/31/EC)*. The main policy drivers for CDW management and EoL concrete recycling are the WFD and the Landfill Directive (Bio Intelligence Service 2011). The WFD set the 70% goal for CWD recovering for EU member states, while the Landfill Directive covers the location and technical requirements for landfills and sets targets for landfilling reductions. According to the Landfill Directive, there are three classes of landfill: hazardous waste, non-hazardous waste, and inert waste. The European List of Waste (2000/532/EC) categorizes each class category of waste. However, according to Eurostat, only the data on mineral waste recycling rate for each member state is available, thus lacking a rule on verifying the compliance with the "70%" target. Additionally, the "70%" target did not mandatorily request the minimal "recycling" (as opposed to the downcycling) target. Therefore, it is no practical significance for countries such as the Netherlands which already achieved around 100% recovery rate by downcycling on CDW but with a negligible portion on recycling.

At the national level, the national regulation corresponding to the EU WFD is the *National Waste Management Plan*. With 95%, the recycling rate for CDW in the Netherlands is already far beyond 70%, the LAP2 sets the target for CDW as keeping the current recycling rate (despite the expected increase of CDW) while reducing the overall life-cycle environmental impacts of CDW management.

In the Netherlands, the process of implementation of sustainable construction regulations (including minimization of natural resource use) is a cooperative government and industry initiative. The predominantly responsible actor(s) for the implementation of sustainable construction regulation are local/municipal governments (PRC 2011). Additionally, to the aforementioned regulations, the non-legislative instrument *Green Deal* was launched by the Dutch government to support sustainable economic growth. A Green Deal is a mutual agreement or covenant under private law between a coalition of companies, civil society organizations and local and regional governments. Since

2011, more than 200 Green Deals have been signed. For the concrete sector, Green Deal 030 was completed in 2016, aiming to substantially reduce CO_2 emissions and achieve high-quality recycling of concrete by 2030.

At the local level, the main approach to stimulate concrete recycling is through Sustainable Public Procurement. The Dutch government has developed a set of sustainability criteria documents. These contain recommendations that public authorities can use to implement sustainable procurement practices for approximately 45 products, services and public works. Most relevant to the recycling of EoL concrete is the *Criteria for the Sustainable Public Procurement of Demolition of Buildings*, which set up minimum requirements on the demolition process and stony waste breaking-up process. The *Criteria for the Sustainable Procurement of Construction Works* addresses the use of secondary materials as a point for consideration at the preparatory stage of the procurement process. The core Sustainable Public Procurement criteria require the contractor to put appropriate measures in place to reduce and recover (reuse or recycle) waste that is produced during the demolition and construction process.

The Dutch governmental authorities have also set clear objectives to boost the market for Sustainable Public Products: the municipalities are aiming for 75% sustainable public procurement in 2010 and 100% in 2015. Provincial governments and water boards have set themselves the target of at least 50% in 2010, while the central government aspires towards 100% Sustainable Public Procurement in 2010. 100% Sustainable Public Procurement is understood to mean that all purchases meet the minimum requirements that have been set for the relevant product groups at the time of purchase. However, no mandatory requirement exists on the minimum use of recycled gravel, recycled sand, and recycled cementitious particle.

2.5.2 Potential policy options

At the EU level, the general high-level recycling goals are set. For countries such as the Netherlands, which are supposed to shift from downcycling to recycling, the EU should set more ambitious goals. For example, the goal could be set as "those member states who already achieved the goal of recovering 70% CDW, are encouraged to achieve a 20% recycling goal”. Setting more ambitious goals at the EU level is only possible if a clear definition of recycling (as opposed to downcycling, or energy recovery) is given, which is currently lacking. Waste registration systems of member states are not harmonized. For example, the 98% recycling rate of Dutch CDW includes energy recovery. Furthermore, the definition of “backfilling” should be strictly clarified in order to avoid "hiding" landfilling operations in this definition. Unfortunately, current waste registration systems and databases are not suitable for estimating EoL flows of CDW, and in particular concrete. It is, therefore, necessary to develop a more systematic waste registration system that includes quantities CDW is generated, and how it is treated.

Given more detailed information about CDW management, more precise decisions could be made by national governments.

At the Dutch level, concrete is mainly downcycled instead of recycled. Recycling of CDW has the potential to mitigate environmental impact compared to downcycling, but in current policy, there is no direct link between recycling targets and environmental and economic targets. Development of standardized LCA- and LCC-based tools for assessing the options can support environmental and financial performance-based policy-making for CDW treatment. In this study, technological routes that recycle concrete waste on-site and produce high-value-added secondary raw materials demonstrate an obvious advantage from an economic and environmental point of view. In addition, policies could also be enacted to set a minimum high-quality recycling share should be set regarding EoL concrete recovery in the upcoming National Concrete Agreement.

At the local level, Sustainable Public Procurement is a strong potential driver for CDW recycling, but it does not provide mandatory requirements on the minimum use of recycled materials. Standards for building materials are based on virgin materials and are not always useful for secondary materials. The VEEP project has demonstrated that with proper quality control of secondary material, the recycled aggregate concrete will not be noticeably different in terms of workability and strength, compared with concrete with natural aggregate. A minimum required share of recycled aggregates and cement should be introduced in local Sustainable Public Procurement criteria.

2.6 Conclusions

EoL concrete is the predominant constituent in CDW with a high potential for reuse and recycling. In EU countries, EoL concrete is usually downcycled for road bases or even used in landfills. It is important to shift from a less preferred EoL concrete treatment and disposal way towards methods maximizing resource efficiency. In Europe, novel technologies have been developed aiming to guarantee high-quality recycled secondary raw material from EoL concrete for use in the manufacturing of new concrete products, thereby closing the concrete loops. Eco-efficiency assessment provides a useful tool for steering decisions towards sustainable resource management, considering economic and environmental aspects at the same time. This paper presents a comparative eco-efficiency analysis methodology for assessing the environmental and economic performance of technological innovations ADR and HAS for EoL concrete recovery by comparing them to the BAU method wet process. This study proposes a framework protocol for LCA/LCC-type eco-efficiency assessment. Besides, an “economic impact assessment” step is proposed for LCC to specify cost breakdown structure, types of cost expressed, and cost stressors, in analogy with the “environmental impact assessment” step in LCA. Next, this case study presents a solution for conducting sensitivity and uncertainty analysis in an eco-efficiency assessment.

The study showed that the most advantageous technological routes are recycling on-site and producing high-value secondary products. The higher eco-efficiency performance system *S3 ADR-M* and *S4 A&H* reduced the life cycle environmental impact to a large extent and minimized the life cycle cost by ensuring the transportability of the recycling facility. However, for the fine fraction of HAS, the recovered product (0–0.125 mm RUP and 0.125–4 mm RFA) cannot replace cement and sand 100%, but it can reduce the use of cement and sand in the production of concrete. Calculation of the achievable reduction of cement and sand led to a modeling choice in favor of HAS. Besides, *S4 A&H* has the worst performance on some impact categories indicators such as photochemical ozone formation, acidification, etc., which, however, are compensated by other indicators under an eco-efficiency context, thus somehow concealing the energy-intensive personality of HAS. With respect to policy implications, relative policy recommendations are as follows: avoiding the transport of waste; enacting regulations and standards for secondary raw material; enhancing the publicity and promotions of technological innovations.

This study has several limitations. First, the cost data is largely based on a Dutch context, and higher availability and lower cost of primary material in some other EU member states will challenge the competitiveness and market share of secondary material. Second, this study used lab-scale data of HAS; the performance of HAS in a more developed stage (i.e. on a pilot-scale and industrial-scale) will be discussed in further research. Third, we excluded some factors, such as the exact distribution of the recycling plants, transportation cost of the products and virgin material to the next destination, the variation of some recycling technologies, and the uncertainty of impact category indicators selection, which may have influenced the results. Finally, this study demonstrates a preliminary concept of an “economic impact assessment” step for LCC with a case study on eco-efficiency assessment; a more comprehensive and systematic illustration will be presented in the near future.

Acknowledgements

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Appendix

Table A2.1 Unit process data for foreground processes. Notes: Based on investigations of RINA Consulting and Technology University of Delft, the Netherlands), the truck travel distance from the place where ADR is stored to the demolition site is assumed to be 20 km; distance from a demolition site to a recycling plant/facility is assumed to be 70 km. According to the C2CA project report "A quantified assessment of economics, potential environmental and social impacts of scenarios", a typical building demolished project contains around 15,000 tons of end-of-life concrete. The cost and environmental impacts of the transportation and dismantling/assembling of the mobile ADR set are calculated based on this amount of concrete waste to be treated per demolition site. The environmental impact from the transport of equipment was allocated based on the amount of concrete for disposal (100 tons) out of 15,000 tons.

	S1 BAU WP	***S2 ADR-S***	***S3 ADR-M***	***S4 A&H***
Transport	**Transport of EOL concrete** Products in: Transport 7,000 t.km (background process) Products out: Transport of EOL concrete: 7,000 t.km (Remark: Demolition site to wet processing plant: 70 km) Extensions in: N/A Extensions out: N/A	**Transport of EOL concrete** Products in: Transport 7,000 t.km (background process) Products out: Transport of EOL concrete: 7,000 t.km (Remark: demolition site to ADR Plant: 70 km) Extensions in: N/A Extensions out: N/A	**Transport of equipment** Products in: Transport 6.67 t.km (background process) Products out: Transport of equipment: 6.67 t.km (Remark: storage of ADR to demolition site: 20 km; Weight of ADR: 25 tons; coefficient: 100/15000=0.667%*) Extensions in: N/A Extensions out: N/A	**Transport of equipment** Products in: Transport 8.67 t.km (background process) Products out: Transport of equipment: 8.67 t.km (Remark: storage of ADR and HAS to demolition site: 20 km; Weight of ADR: 25 tons; weight of HAS: 7.5 tons; coefficient: 100/15000=0.867%) Extensions in: N/A Extensions out: N/A
Recycling	**Wet processing** Products in: Water: 670 L (background process) Electricity: 400 kWh (background process) Diesel: 27 L, (background process , the heat value of the	**Crushing** Products in: Water: 70 L (background process) Diesel: 1,300 L (background process) Crusher: Cut-off (Remark: depreciation of the equipment is negligible thus not considered) Products out: 12–22 mm RCA: 20 tons 0–12mm RCA: 80 tons Extensions in: N/A Extensions out: N/A (water is used for dust control, thus dust is not considered in this process)		

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<td rowspan="2"></td>
<td rowspan="2">diesel is set as 39 MJ/L, hereafter).
Transport 52 t.km (Remark: background process, for sludge disposal, Wet processing plant to landfill site: 20km)
Landfill 2.6 tons (Remark: background process, for sludge disposal)
Wet process plant: (Remark: Cut-off, depreciation of the equipment is negligible thus not considered).
Products out:
4–22 mm RCA: 52.9 tons
0–4 mm SS: 44.5 tons
Extensions in:
N/A
Extensions out:
N/A (water is used for dust control, thus dust is not considered in this process)</td>
<td colspan="3">ADR
Products in:
0–12mm RCA: 80 tons
Water: 70 L (background process)
Electricity: 36.8 kWh (background process)
Products out:
4–12 mm RCA: 48 tons
0–4 mm SS: 32 tons
Extensions in:
N/A
Extensions out:
N/A (water is used for dust control, thus dust is not considered in this process)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>HAS
Products in:
0–4 mm SS: 32 tons
Diesel: 192 L (background process)
Electricity: 0.32 kWh
Products out:
(background process)
0.125–4mm RFA: 25.6 tons
0–0.125mm RUP: 6.4 tons
Extensions in:
N/A
Extensions out:
N/A (water is used for dust control, thus dust is not considered in this process)</td>
</tr>
<tr>
<td>Virgin material production</td>
<td>Production of NCA
Products in:
Gravel: 15.1 tons (background process)
Products out:
4–22mm NCA: 15.1 tons
Extensions in:
N/A
Extensions out:
N/A</td>
<td>/</td>
<td>/</td>
<td>/</td>
</tr>
</table>

	Production of Sand Products in: Sand: 25.6 tons (background process) Products out: 0.125–4mm Sand: 25.6 tons Extensions in: N/A Extensions out: N/A	
	Production of cementitious material: Products in: Cement: 6.4 tons (background process) Products out: Cement: 6.4 tons Extensions in: N/A Extensions out: N/A	

Table A2.2 Operating weight of recycling facilities. Note: A truck scale is applied to measure the operating weight of each recycling facility. The crushing set, ADR, HAS were measured at an experimental trial site in Hoorn, the Netherlands, The DGR was measured in Helsinki, Finland.

Recycling facilities	Operating weight [Kg]
Crushing set:	
• Keestrack Destroyer 1313 Crusher	51,000
• CX350D Excavator,	35,900
• 921E Rubber-wheel loader	22.962
ADR	25,000
HAS	7,500
DGR	3,900

Table A2.3 Background processes linked to the foreground processes. Notes: Those processes are based on the LCA software OpenLCA 1.7.4 with the database Ecoinvent 3.4

Phase	Background process
Transport	**Transport of waste and facility:** market for transport, freight, lorry >32 metric ton, EURO3 | transport, freight, lorry >32 metric ton, EURO3 | Cutoff, U-GLO
Recycling	**Water:** market for tap water | tap water | Cutoff, U - Europe without Switzerland **Electricity:** market for electricity, high voltage | electricity, high voltage | Cutoff, U – NL **Diesel:** market for diesel, burned in building machine | diesel, burned in building machine | Cutoff, U – GLO **Waste transport:** transport, freight, lorry 16–32 metric ton, EURO3 | transport, freight, lorry 16–32 metric ton, EURO3 | Cutoff, U – RER **Waste landfill:** market for process-specific burdens, inert material landfill | process-specific burdens, inert material landfill | Cutoff, U - CH
Virgin material production	**NCA:** gravel and sand quarry operation | gravel, round | Cutoff, U - RoW **Sand:** gravel and sand quarry operation | sand | Cutoff, U - RoW **Cement:** cement production, blast furnace slag 36–65%, non-US | cement, blast furnace slag 36–65%, non-US | Cutoff, U - Europe without Switzerland

Chapter 3

Life cycle greenhouse gas emission and cost analysis of prefabricated concrete elements for use as façade of new building

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Abstract

Buildings are responsible for approximately 36% of carbon emissions in the European Union. Besides, gradual aging and a lack of adaptability and flexibility of buildings often lead to destructive interventions, resulting not only in higher costs but also in a large amount of construction and demolition waste (CDW). Recently, an innovative system (Ref. VEEP project) has been developed to recycle CDW for the manufacturing of energy-efficient prefabricated concrete elements for new building construction (PCE-new). By applying life cycle costing (LCC) and life cycle assessment (LCA), this study aimed to determine whether the use of VEEP green PCE-new leads to lower carbon emission and lower associated costs over the life cycle of an exemplary four-story residential building in the Netherlands than a business-as-usual (BAU) PCE-new scenario. This paper provides a case study on the alignment and/or integration of LCA and LCC in an independent and a combined manner (via monetization). This study examines how the internalization of carbon emission and discount rate will affect the final life cycle costs over a 120-year life span. The findings are that the key to the economic viability and environmental soundness of green PCE-new is to improve its thermal transmittance. Besides, internalization of external cost can monetarize the environmental advantage, thus, slightly expanding the cost advantage of low carbon options but also leading to larger uncertainty about the LCC result.

Keywords: life cycle costing, life cycle assessment, prefabricated concrete element, building façade, construction and demolition waste (CDW), industrial ecology.

Abbreviations

ADR	Advanced drying recovery technology
BAU	Business-as-usual
BAU-ex	BAU PCE-new scenario taking into account the external costs
CRSCA	Coarse recycled siliceous concrete aggregate
CDW	Construction and demolition waste
DGR	Dry Grinding & Refining system
EC	European Commission
EER	Energy efficiency ratio
EoL	End-of-life
EPS	Expanded polystyrene
EU	European Union
EU ETS	European Union Emissions Trading System
FRSCA	Fine recycled siliceous concrete aggregate
FWW	Mineral fiber wool waste
GW	Glass waste
HAS	Heating air classification system
LCA	Life cycle assessment
LCC	Life cycle costing
LCCs	Life cycle costs
PCE-new	Prefabricated concrete element for new building
RCA	Recycled concrete aggregate
URSCA	Ultrafine recycled siliceous concrete aggregate
RGUA	Recycled glass ultrafine admixture
RFUA	Recycled fiber wool ultrafine admixture
SEER	Seasonal energy efficiency ratio
SETAC	Society of Environmental Toxicology and Chemistry
VEEP	European Union Horizon 2020 project VEEP
VEEP-ex	Green PCE-new scenario taking into account the external costs

3.1 Introduction

There is a wide agreement that future economic growth must be driven by greater energy efficiency. The European Union (EU)'s current housing stock is thermally poor, and national energy performance standards are relatively weak when benchmarked against international best practices. Buildings are responsible for approximately 40% of energy use and 36% of CO_2 emissions in the EU (EC 2010). Currently, about 35% of the EU's buildings are over 50 years old and almost 75% of the building stock is energy-inefficient, while the yearly renovation rate is only 0.4–1.2%, depending on the country (EC 2010). The European building industry needs new technologies, products and materials to minimize that energy dependence. More renovation of buildings can lead to significant energy savings, potentially reducing the EU’s total energy use by 5 to 6% and lowering CO_2 emissions by about 5% (EC 2018). One of the key strategies for cutting the energy use of buildings through energy renovation is scaling up the use of novel technologies for highly efficient thermal insulation of a building’s envelope (Morrissey and Horne 2011).

At the same time, one of the largest solid waste streams is construction and demolition waste (CDW). The European Commission (EC) has identified CDW as a priority stream because of the large amounts that are generated and their high potential for reuse and recycling (EC 2011a). In 2005, the EU-27 member states generated approximately 461 Mt of CDW, and the generation volume is expected to reach 520 Mt in 2020 (excavated material excluded) (EC 2011a). Therefore, by 2020, the Waste Framework Directive (2008/98/EC) requires EU member states to take any necessary measures to prepare a minimum of 70% of CDW (by weight) for reuse, recycling and other material recovery, including the use of non-hazardous CDW for backfilling (EC 2008b).

The Netherlands has noticeable performance over CDW management. Nearly 98% of the CDW generated in the Netherlands can be recycled, which is more than in the other member states (EC 2012a). End-of-life (EoL) concrete represents 40% of CDW in the Netherlands, and 100% of this stream is recycled, with more than 97% of it being used in road construction as road base material (Hu et al. 2012). While road construction is expected to remain stable, there is a need for shifting from traditional recycling approaches to novel recycling and recovery solutions. In particular, the fine fraction (0-4 mm), which constitutes roughly 40% of the recycled concrete, is often down-cycled because its incorporation into new concrete still faces technical barriers (Lotfi et al. 2015). Also, some minor (e.g. glass) and emerging (e.g. mineral wool) CDW streams, currently accounting for about 0.7% of the total CDW generation, are expected to grow until 2030 as a consequence of the European regulations on building energy efficiency and building retrofitting (EC 2014b). In global terms, no technological and business solutions have yet been found for recycling those emerging CDW streams, which so far are mostly landfilled. Thus, more advanced and appropriate solutions should be developed to ensure the effective and efficient use of natural resources and to mitigate the associated environmental and economic impacts.

More and more businesses in the construction sector, as well as governments and even consumers are seeking eco-products that are not only financially viable but also bring in environmental, and even social benefits (Zhang et al. 2019b). Also, new products need to meet upcoming challenges concerning climate change and lower carbon footprints, resource depletion and shortages increasing restrictions on the use of toxic substances, lower embodied energy, and best positioning in competitive markets. Over the last few years, novel technologies have been developed aiming to guarantee high-quality recycled raw material for use in new construction products, thereby closing the loops in the manufacturing of concrete and insulation material. In Europe, an innovative and integrated technological system was designed and developed in a VEEP project for the massive retrofitting of the built environment, aiming at cost-effectively recycling CDW and reducing building energy use. VEEP's core technologies include advanced drying recovery (ADR), heating air classification system (HAS) and Dry Grinding & Refining (DGR), which provide the scientific-technological basis for new green concrete recipes containing high levels of upgraded CDW recycled materials. With VEEP, CDW will contribute at least 75% of the total weight of the new concrete, and at least 10% of cement will be replaced by recycled supplementary cementitious CDW materials. VEEP also allows for higher resource efficiency in the novel multilayer precast concrete elements for new building envelopes (PCE-new), through the combination of concrete and superinsulation material manufactured by using recycled CDW materials as raw materials.

Given the need for eco-efficient thermal insulation materials for renovating a building's envelope, it is of great significance to explore the environmental impact and cost-effectiveness of green PCE-new as the building fa çade. Hence the main research question of the present study was "Is the use of concrete façade elements containing secondary materials more economic and environmentally advantageous than the use of elements that are only made of primary material?". This study aimed to answer this question by comparing the economic costs and GHG emissions of two types of PCE-news, one of which is made of both virgin and secondary raw material from the VEEP technological recycling system, the other being a conventional PCE-new with only virgin material. The comparison was based on an integrated life cycle assessment (LCA) conforming to ISO 14040 (2006) and ISO 14044 (2006) and life cycle costing (LCC) based on the SETAC's definition (Swarr et al. 2011; Hunkeler et al. 2008). This study built upon the LCA-LCC analysis framework proposed in Zhang et al. (2019) and explored the potential to harmonize LCA and LCC from stakeholders' perspectives by internalizing the foreseeable environmental costs – carbon emission costs, and on the factor of time by considering "discounting" effects. Through a comparative life cycle assessment of PCE-new panels, this study provides a Dutch case for global issues with respect to CDW generation, GHG emission and energy efficiency in the construction domain. As the evaluation includes a use phase of 120 years it provides an excellent opportunity to investigate the effects of "discounting" on the harmonization of LCA and LCC methods in a combined study.

3.2 Methods

3.2.1 Goal and scope definition

Goal

The goal of this study was to determine the financial effects and carbon mitigation of manufacturing and using the innovative green PCE-new as façade for a new building in comparison to those of manufacturing and using conventional PCE-new.

Scope and scenarios

Normally there are five processes in the life cycle of PCE-new: material preparation; manufacturing; installation; in use; disposal. The life cycle that is considered in this study comprises four phases: (I) material preparation, (II) PCE-new manufacturing, (III) PCE-new installation, (IV) PCE-new in use, and (V) PCE-new disposal, as shown in Figure 3.1. Two scenarios are assessed in the study: *VEEP* and *BAU*. In both scenarios, PCE-new will be manufactured to improve the energy efficiency of a building. The differences are that PCE-new from BAU uses virgin material and conventional insulation material. In the VEEP scenario, secondary raw material and the novel insulation material aerogel will be incorporated in PCE-new.

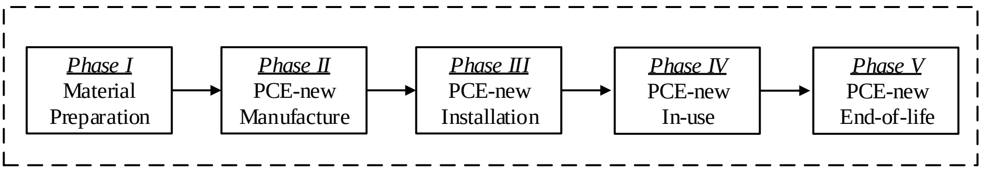

Figure 3.1 Fives phases of this study for the assessment of the PCE-new

3.2.1 Description of the VEEP system

VEEP ADR and HAS technologies

The combined innovative ADR technology and HAS was developed for the simultaneous cost-effective production of high-quality coarse and fine recycled concrete aggregates from concrete waste for green concrete production and green aerogel production. Siliceous concrete waste will be studied in this study. Given that selective demolition and sorting are common practices in the Netherlands, it is assumed that concrete waste fed into ADR and HAS does not contain residue. In this study, EoL siliceous concrete waste is considered as the target concrete waste for producing coarse recycled siliceous concrete aggregate (CRSCA), fine recycled siliceous concrete aggregate (FRSCA), and ultrafine recycled siliceous concrete aggregate (URSCA). The details of ADR and HAS were described in Chapter 2.

VEEP DGR technology

Evolved VEEP Dry Grinding & Refining Recovery (DGR) technology is currently a stationary recycling facility that can produce secondary raw material: recycled glass ultrafine admixture (RGUA) and recycled fiber ultrafine admixture (RFUA), with an average purity level higher than 90%, from emerging building glass waste (GW) and insulating mineral fiber wool waste (FWW). These waste materials will first be pre-crushed by a mobile hammermill when larger amounts of material are processed. Materials are fed three times through the hammermill to achieve a suitable particle size. The whole process is sealed by a small vacuum in the hammermill feeding opening so that no dust or particles can escape from the process. From the hammermill, the milled material is transported pneumatically through a cyclone separator to the collecting bag. Recycled mineral microfibers and ultrafine cementing particles (particle sizes lower than 200 microns) are obtained from this process in order to hopefully incorporate these silica-rich particles effectively into new concrete formulations and aerogel composites for the subsequent manufacture of panels.

VEEP aerogel production

While the BAU scenario involves the use of the conventional insulating material EPS, the VEEP project includes the development of a green cost-effective aerogel using secondary raw materials from CDW. The production of this aerogel relies on the integration of the following steps: (i) low-cost water-glass-based precursor production by using silica-containing CDW recycled materials such as FRSCA, RGUA or RFUA; (ii) gasification; (iii) higher efficient multi-solvent low-temperature supercritical drying. Aerogels can be manufactured in different forms: monolithic, powder, blankets, granules, etc. In the VEEP project, the chosen strategy for preparing aerogel composites is the employment of fibers during the sol-gel step. These fibers will contribute to the mechanical performance of silica-based aerogel materials, allowing the use of the aerogel in the novel precast concrete elements. However, since the VEEP green aerogel is still under development, the present assessment uses lab-scale data. Additionally, due to concerns about business confidentiality, the details of the data will not be disclosed in this study.

Green PCE-new production

The VEEP green recipe concrete contains secondary material, including CRSCA, FRSCA and URSCA from ADR and HAS, RGUA and RFUA from DGR. green PCE-new will be manufactured using the VEEP concrete and the aerogel EPS, as well as rebar cages and welded nets.

3.2.2 Functional unit

Principally, the same functional unit at the system level will be defined for the LCA and LCC. The average lifetime of residential buildings in the Netherlands is 120 years (Sandberg et al. 2016). The functional unit selected for this study was maintaining the thermal comfortableness of 1 m^2 flooring area of a building with the application of a PCE-new façade and active heating and cooling for 120 years based on reference scenarios. In both scenarios, it is assumed that the required building façade per 1 m^2 of flooring area amounts to 0.55 m^2 of PCE-new.

3.3 Life cycle inventory analysis

The LCA software OpenLCA 1.9 was selected to perform the LCA analysis as an assessment instrument with a database of Ecoinvent 3.4 (Allocation, cut-off by classification). For the LCC study, Microsoft office 2016 Excel was used to investigate the main contributions of costs. The system boundaries of the two scenarios are shown in Figure 3.2.

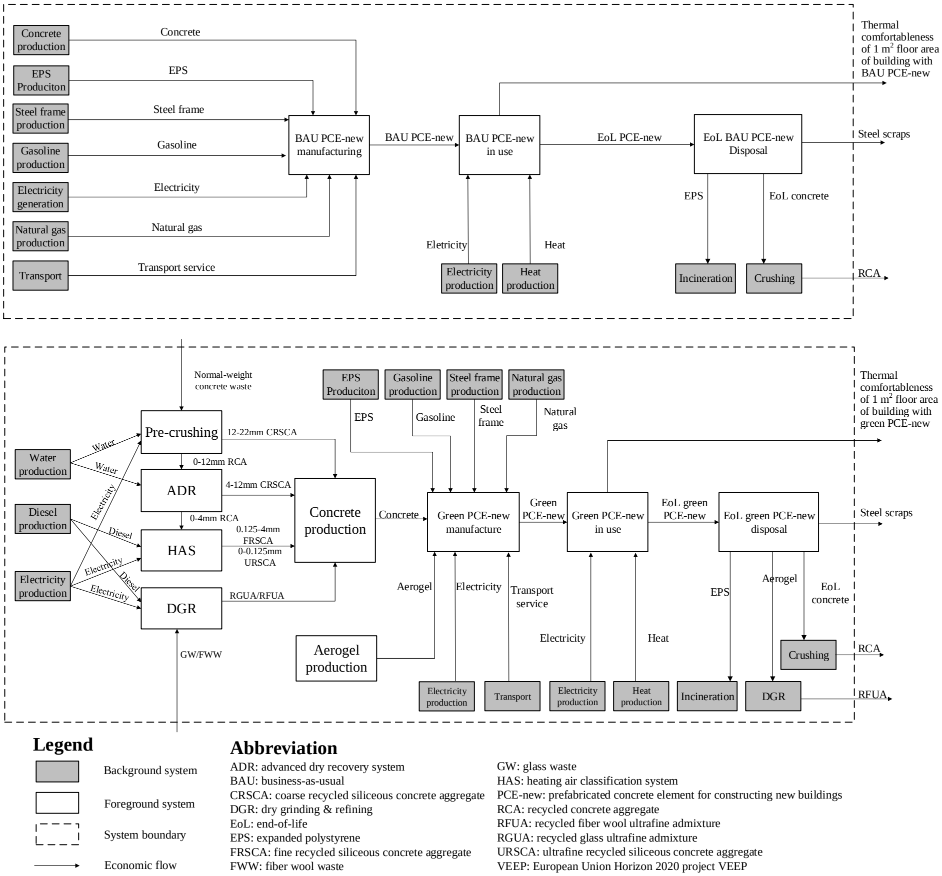

Figure 3.2 System boundaries for the BAU scenario (above) and the VEEP scenario (below)

3.3.1 Environmental inventory

The environmental inventory has been carried out for the BAU and VEEP scenarios and organized into five phases: (I) Material preparation, (II) PCE-new manufacturing, (III) installation, (IV) PCE-new in-use, (V) Disposal of PCE-new as defined in Goal and Scope Definition (Figure 3.1). The details of how the five phases are modeled in shown as follows.

3.3.1.1 Material preparation

In the Material preparation phase, the raw material for manufacturing the green PCE-new will be prepared, including concrete materials (aggregate, cement, additive, etc.), thermal insulation material, rebar cages, and welded nets. In the BAU scenario, only primary raw material was used, while the green PCE-new used both primary and secondary raw material. As the transport of raw material does matter in urban mining (Zhang et al. 2018), the transport costs of virgin material and recycled material are considered.

How the crushing, ADR, HAS are modeled are referred to in previous research (Zhang et al. 2019a). The mass flow diagram of the dry grinding and recovery (DGR) process is shown in Figure 3.3. For the sake of simplicity, it is assumed a 1:1 substitution rate between recycled and virgin materials for GW/FWW.

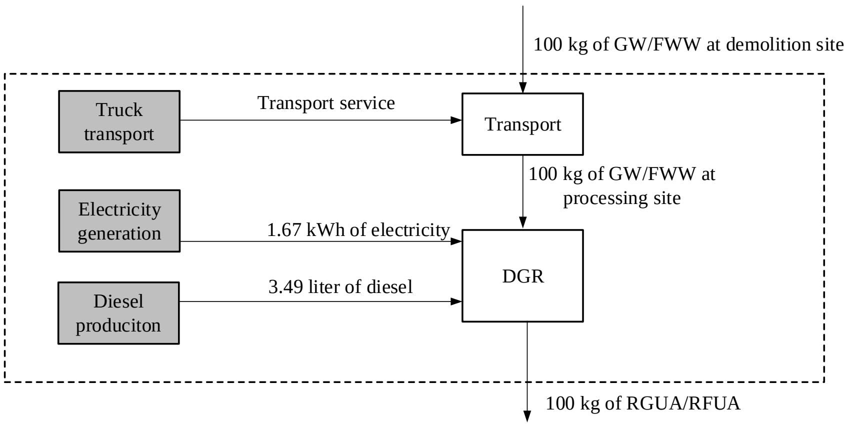

Figure 3.3 Flow diagram of dry grinding & refining (DGR) process. DGR: dry grinding and refining; FWW: fiber wool waste, GW: glass waste; RFUA: recycled fiber wool ultrafine admixture; FGUA: recycled glass ultrafine admixture.

Multifunctional processes in the VEEP scenario include crushing of concrete rubble, ADR, HAS and DGR, which are described in Table 3.1. Allocation is applied to distribute the environmental and economic impacts of functional flow from a multifunctional process. The allocation method for LCC and LCA is process-based allocation. The costs and environmental impact of multifunctional processes are both allocated via the mass of each product.

Table 3.1 Processes with multifunctionality in the VEEP scenario

Process name	Multifunctionality category	Functional flows	Allocation shares	Category
Pre-crushing	Recycling	EoL concrete treatment	50%	Non-target service
		Coarse fraction (0-12mm)	40%	Intermediate product
		Coarse fraction (12-22mm)	10%	Non-target product
ADR	Co-production	CRSCA (4-12mm)	71%	Target product
		Fine fraction (0-4mm)	29%	Intermediate product
HAS	Co-production	FRSCA (0.125-4mm)	80%	Target product
		URSCA (0-0.125mm)	20%	Target product
DGR	Recycling	GW/FWW treatment	50%	Non-target service
		RGUA/RFUA	50%	Target product

Material for the PCE-new and its source is shown in Table A3.1. Lubricating oil for machines is omitted from this study. For the commercially confidential concerns, the exact amounts of the concrete constituents were not shown.

Application of secondary raw material from urban mining in concrete production, especially for on-site recycling, can considerably reduce transportation impact compared to virgin material from conventional mine extraction (Zhang et al. 2018). Besides, Göswein et al. (2018) also proved that transport matters when recycled material is involved. Thus, to make certain if the transport is important when the life cycle of concrete expands, the difference between the transportation of recycled material and virgin material is considered in the material preparation phase. Background process "market for transport, freight, lorry >32 metric ton, EURO3 | transport, freight, lorry >32 metric ton, EURO3 | Cutoff, U - GLO" is selected for the transport simulation. According to the survey at the Delft University of Technology and RINA consulting, it is assumed that truck average travel distance of recycled material is 20 km; the truck travel distance of virgin material is 50 km.

3.3.1.2 PCE-new manufacture

In the PCE-new manufacturing phase, a family of ribbed panels is selected for both scenarios in order to reduce the consumption of concrete, reduce the weight of the panel, and improve the thermal performance of the panel. The cross-section perspective of green PCE-new and BAU PCE-new are shown in Figure 3.4.

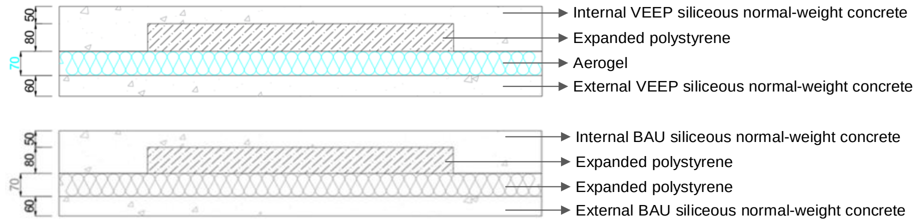

Figure 3.4 Cross-section perspective of green PCE-new (above) and BAU PCE-new (below)

Green PCE-new is a sandwich panel with higher insulation properties due to the material green aerogel and higher contents of secondary raw material from CDW. The BAU PCE-new taken as the reference is also in the form of a sandwich panel with the same stratigraphy but manufactured from traditional concrete (benchmark siliceous concrete produced without the use of secondary raw material) and expandable polystyrene (EPS) as the insulation layer. The sandwich panels will be manufactured with those materials in the material preparation phase in the plant and will then be transported to the construction site.

Both PCE-news have the same structure, thickness, but with different materials. The dimensions of both PCE-news are length 3.6 m; width 2.4 m; thickness 0.26 m. The thickness of concrete layers was designed according to the regulations Eurocode 2, EN 206-1, EN 14992, which requires a minimum of 130 mm for the ribs, 50 mm for the concrete layer between ribs (protected) and 60 mm for the external concrete layer possibly exposed to the environment aggressions. Thermal transmittance measures how effective a material is as an insulator. The thermal transmittance of the green PCE-new is 0.19 W/(m^2 K), then compliant with the project requirements. The thermal transmittance of the BAU PCE-new taken as the reference with EPS as insulation layer (same thickness) is 0.32 W/(m^2 K). It is assumed that all the materials are produced and proceeded in the PCE-new manufacturing plant in the Netherlands. The bill of materials related to the production of one unit of green PCE-new and BAU PCE-new are shown in Table 3.2.

Table 3.2 Component and structure of PCE-new

	Material	Thickness [mm]	Volume [m^3] per unit of PCE	Weight [kg] per unit of PCE
VEEP	VEEP Concrete	60	1.19	2485.46
	Green Aerogel	70	0.61	42.35
	EPS	80	0.45	13.5
	Rebar Cages & welded nets	50	/	117.10
BAU	Concrete	60	1.19	2485.46
	EPS	150	1.05	31.65

Rebar Cages & welded nets	50	/	117.10

Energy usage related to green PCE-new and BAU PCE-new manufacturing is presented in Table A3.2.

3.3.1.3 PCE-new installation

After fabrication, the PCE-new is transported to the construction site for installation. Compared to BAU PCE-new, a dismantlable connecting and anchoring system is applied to conveniently install green PCE-new. Since 90% of the expense on installation is the cost of labor, the dismantlable connecting and anchoring system can also enable a reduction of labor costs through quick installation. It is assumed 30% of the installation cost can be saved by the system. Transport of the PCE-new was assumed 50 km. The input for installation of the BAU and green PCE-new is shown in Table A3.3.

3.3.1.4 PCE-new in-use

In the PCE-new in-use phase, dynamic thermal simulation (DTS) was performed to compare the thermal performances of the two concrete façade elements. The selected case study building was a typical residential building in the capital of the Netherlands, Amsterdam (52°22'N, 4°54'E). The life span of the prefabricated building is assumed to be 120 years. For the climate zone of Amsterdam, the cooling need is rather low. However, to reflect the entire thermal performance of the application of the two PCE-news, this study did take the cooling need into account along with the heating need. DTS at building scale were carried out on a virtual residential multi-story building.

Software DesignBuilder and EnergyPlus were used for dynamic thermal simulations. Figure 3.5 presents the virtual building for the case study. DTS at building scale were carried out on a virtual residential multi-story building. This building is composed of 4 floors for a total floor area equal to 1767 m^2. The gross floor area of PCE1 used for the façade is equal to 967 m^2, of which 234 m^2 for the first floor, 253 m^2 for the second floor, 253 m^2 for the third floor, and 227 m^2 for the top floor. The gross area of PCE-new used for façade is equal to 967 m^2. Thus, per building floor area needs 0.55 m^2 of PCE-new.

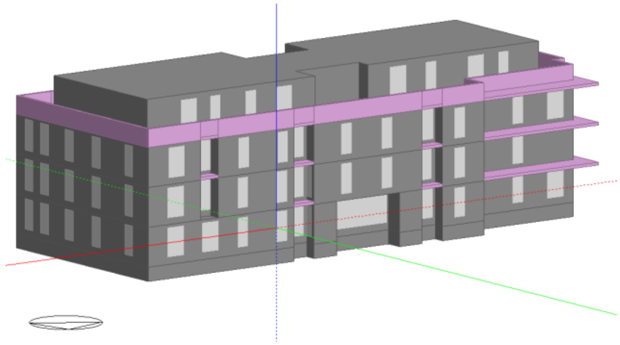

Figure 3.5 Reference virtual building for the case study

Designed thermal conductivity and Thickness of materials in PCE-new of properties used for all DTS are presented in the following Table A3.4 and Table A3.5. According to EN ISO 6946 (2017), no correction (on thermal transmittance) must be applied if the thermal conductivity of the connection, or a part of it, is lower than 1 W/(m·K). For the CHRYSO Flexo connector of PCE-new is made of material with low thermal conductivity (λ<0.231 W/(m·K)), no correction on thermal transmittance was taken into account. For the aerogel, a conservative thermal conductivity value is considered. This value is obtained for a temperature equal to 40 ℃ and for a relative humidity equal to 90%. For these calculations, superficial thermal resistances are not considered. The thermal transmittances of the envelope elements of the building are presented in Table A3.6. The thermal transmittance of selected PCE-new solutions is calculated according to the standard EN ISO 6946 (2017). The heating and cooling scenarios considered are presented in the following Table 3.3.

Table 3.3 Heating and cooling reference Scenarios

	Heating	Cooling	Application
Temperature in occupation	21 ℃	26 ℃	Monday, Tuesday, Thursday, Friday: 00:00 to 10:00 and 18:00 to 24:00 Wednesday: 00:00 to 10:00 and 13:00 to 24:00 Saturday, Sunday: 00:00 to 24:00
Temperature in vacancy	18 ℃	30 ℃	Monday, Tuesday, Thursday, Friday: 10:00 to 18:00 Wednesday: 10:00 to 13:00

According to the dynamic thermal simulation in terms of need heating and cooling referred to the climate of Amsterdam is presented in Table 3.4.

Table 3.4 Need heating and cooling with BAU and green PCE-new in façade

	BAU PCE-new kWh/(m²year)	Green PCE-new kWh/(m²year)
Heating need	29.25	25.90
Cooling need	3.35	3.84

It was assumed that the heating of the building was performed with a boiler of installed power that generates the thermal energy by combustion of natural gas. The background process in Ecoinvent 3.4 “heat production, natural gas, at boiler modulating >100kW | heat, district or industrial, natural gas | Europe without Switzerland” is used for heating need calculation. Concerning cooling, it was assumed that the cold air is produced by an air conditioner using electrical power. According to norm DIN V 18599 (DIN 2011) electricity demand of the air conditioning can be calculated using the following Eq. (1):

$$Electricity\ demand = E_{coldness} / (EER \times f) \quad (1)$$

where $E_{coldness}$ is usable energy for cooling needs (in kWh); *EER* is energy efficiency ratio; *f* is average part load factor. Multiplication of the *EER* with the average part load factor gives the annual seasonal energy efficiency ratio (SEER). According to norm DIN V 18599 (DIN 2011) for split air conditioning (>12kW), the SEER is around 4.7, this means that with 1 kWh of electricity 4.7 kWh of cooling is produced. According to the assumptions above, calculated the Nm^3 of natural gas for heating and the electrical kWh for cooling are shown in Table 3.5. Background process “market for electricity, low voltage | electricity, low voltage | Netherlands” is applied for cooling need calculation.

Table 3.5 electrical energy for the cooling need

	BAU PCE-new	green PCE-new
Electricity [kWh/(m^2·annum)]	0.71	0.82

3.3.1.5 Disposal of PCE-new

When the target building enters its EoL stage, the building will be demolished, and the PCE-news will be deconstructed from the building in structurally intact condition and will be further dismantled manually in situ. A novel anchoring and connection system was applied to the green PCE-new. Dismountable internal epoxy connectors were set between concrete layers, which enable the PCE-new itself as well as the constituents inside the PCE-new to be disassembled more easily. Due to a lack of data, the impact of dismantling the PCE-new from the building and disassembling the PCE-new is currently not considered. Steel, concrete and insulation materials were separated from each other.

Given the high prices for metals, they appear to be in an almost closed-loop (Koutamanis et al. 2018). Metals are collected from the CDW and further re-melted in furnaces to produce new iron and steel. In the disposal phase, steel treatment is processed by collecting and selling it directly on-site. The environmental impact of the follow-up re-melting process will not be considered in the study.

EoL concrete is recycled by crushing on-site with a crusher. The crushing process referred to the previous study (Zhang et al. 2019a). The Allocation method for crushing in the EoL phase of the VEEP scenario is presented in Table A3.7.

For insulations, disposal options for fibrous materials include landfill or incineration (Karatum et al. 2018). EPS and aerogel are both recyclable. The detailed data for EPS recycling is unavailable. It was assumed that EPS is disposed of through incineration. The incineration of the EPS referred to the process "market for waste expanded polystyrene | waste expanded polystyrene | Cutoff, U – GLO". The aerogel was assumed recycled via the DGR.

3.3.2 Economic inventory

To align with the environmental analysis, the economic inventory has been carried out for the BAU and VEEP scenarios and organized into the same five phases as defined in Goal and Scope Definition (Figure 3.1). While different from the environmental assessment, the economic assessment considers the stakeholders' perspective. It distinguishes the costs with clear cost-bears, being producers' costs or consumers' costs, from those without clear cost-bears, being society's costs. The former is termed internal costs, which can be inventoried by monitoring real transactions. However, internal private costs include transfer payments to governments, such as payment for emission allowances in the European Union Emissions Trading System (EU ETS). The transfer payments are currently not discussed in this study. And the latter is termed as external costs, following the scope defined in environmental LCC (Nakamura and Rebitzer 2008) only the "external costs expected to be internalized", which in this study only the carbon emission costs are inventoried.

3.3.2.1 Internal costs

The economic inventory of the internal costs relies on the physical flows associated with the product system, which are the same as defined in the environmental inventory. The cost structure was broken down in this section based on the four defined phases. The geographic scope of the study is the Netherlands, where the field data of the case study were collected. Relevant cost data were collected and expressed in € (euro). The analysis took the Netherlands as the geographical reference area for the price background, and all cost categories are expressed in €. The cost data and their sources for internal costs calculation are presented in Table A3.8.

3.3.2.2 External costs

The external costs was inventoried for the carbon emissions of the BAU and VEEP systems. In 2003, a scheme of greenhouse gas emission allowances was established under the EU ETS. Larger European firms must deliver carbon allowances equal to their emissions, and it can buy or sell carbon allowances that it needs or does not need. The carbon emission costs rose to its highest level in more than a decade in Europe, surpassing 20 euros a metric ton, and it has been predicted that prices will rise to 35 or 40 euro/metric ton on average from 2019 to 2023, with market rates possibly reaching 50 euros in the winters of 2021 and 2022 (Morison and Hodges 2018). The EU ETS does not affect all companies it covers in the same way because of the differences in their reliance on energy and in their production methods (Leadership 2015). Along with this trend, the external costs related to GHG emission might be directly internalized to relevant actors in the future.

In this study, the CO_2 costs was seen as the "external costs expected to be internalized " which was defined in the environmental LCC (Nakamura and Rebitzer 2008). The data for monetization and their sources for external costs calculation are presented in Table A3.9. Those monetary data derived from the VITO is for studies focusing on the comparison of impacts from different building materials or building lines in western Europe (De Nocker and Debacker 2017).

3.4 Life cycle impact assessment

3.4.1 Environmental impact assessment

Climate change poses a fundamental threat to habitats, species and people's livelihoods (Liu et al. 2017). Recent studies have identified a near-linear relationship between global mean temperature change and cumulative GHG emissions (Friedlingstein et al. 2014). For LCA, this study explores the potential of the green PCE-new for greenhouse gas emission mitigation. Global warming (kg CO_2 eq) from CML-IA version 4.4 issues in January 2015 was selected as the sole impact indicator, thus normalization and weighting scheme were not necessary.

3.4.2 Economic impact assessment

According to the environmental LCC guidebook, there is no need to make an impact assessment for LCC. The LCCs of a product is expressed in monetary units which are already comparable, thus there is no threshold and a lower cost is always better (Swarr et al. 2011). However, for a better interpretation of the economic results, (Zhang et al. 2019a) proposed adding an economic impact assessment step in the LCC analysis, which intends to answer three questions: (i) how will the life cycle cost be categorized? (ii) how will the moment of incurring costs and benefits in time be considered? (iii) how will the value of the final cost be expressed? In this study, the economic impact assessment has been implemented according to (Zhang et al. 2019a). While answering these questions, this study intended to explore the potential of sensibly using “external costs” to harmonize LCA and LCC from stakeholders’ perspectives, and the factor of time by investigating the effects of discounting.

Cost breakdown structure

Firstly, the costs are categorized according to the life cycle stages of the PCE-new, thus, the LCCs are estimated as in Eq. (2):

$$LCCs = C_I + C_{II} + C_{III} + C_{IV} + E \quad (2)$$

Where LCCs is life cycle costs; C_I is internal costs incurred in the material preparation phase; C_{II} is internal costs incurred in PCE-new the manufacturing phase; C_{III} is internal costs from PCE-new in the in-use phase; C_{IV} is internal costs from PCE-new in the EoL phase. *E* is the external costs related to GHG emission. The external costs of carbon emission was added to the LCCs to demonstrate to what extent it would affect the economic viability.

Discounting scheme

Whether a study should use a discount rate and if so, which rate, is highly dependent on the study’s defined goal and scope. However, according to the LCC guide book *Environmental Life Cycle Cost* published by SETAC (Hunkeler et al. 2003), environmental LCC usually is a steady-state method, as is the complementary LCA, and discounting the final result of environmental LCC specification is not consistent nor easily carried out and is therefore not recommended. However, in this study, the life span of the prefabricated building is assumed to be 120 years, and therefore the discount rate has to be considered even though it may not be consistent.

Internal costs discounting

Regarding the internal costs, the private discount rate is used for heating and cooling costs, and EoL costs. The costs in the material preparation phase and the PCE-new manufacturing phase will not be discounted, nor will the GHG emission costs. Islam et al. (2015) reviewed building-related LCC studies with consideration of the time value of money and found that discount rates ranged from 2% to 8% worldwide, and from 2.5% to 4% in Europe. Moore and Morrissey (2014) found that the discount rate was usually significantly lower in developed countries. With respect to the time factor on costs, the private discount rate was considered to modify costs incurred in different life cycle stages. The historical time series interest rate in the Netherlands and the Euro area are depicted in Figure 3.6. It can be seen that the interest rate in the Netherlands as well as in the Euro area presents a descending trend and arrives around 0% in 2020. The private discount rate of November 2020 was set as -0.52% for this Dutch case study, while a range of (-1%, 3%) is considered for uncertainty analysis.

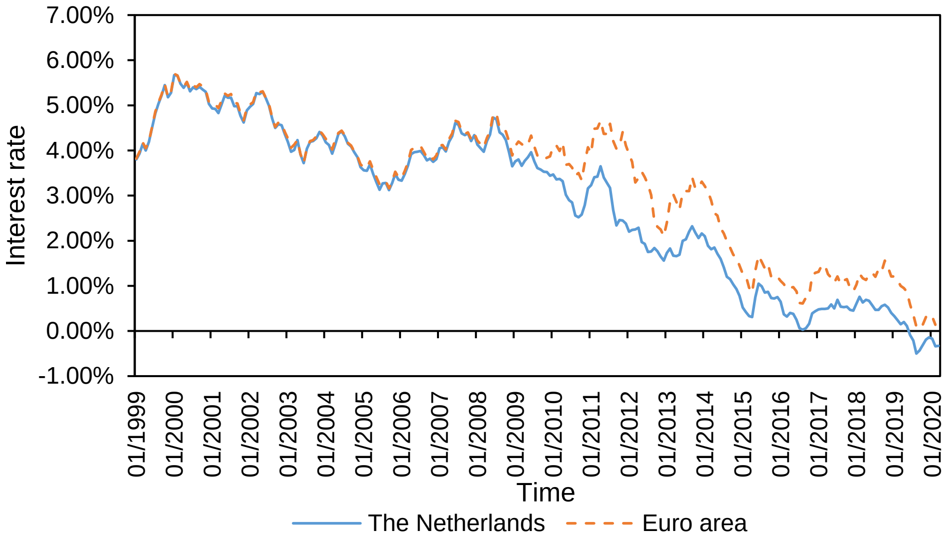

Figure 3.6 Time series interest rate in the Netherlands and Euro area, data from De Nederlandsche Bank (2020).

External costs discounting

As for external costs, a social discount rate was used. Applying discount rates to actualize external costs is a long-stand controversial issue (Weitzman 2011; Arrow et al. 2014; Portney and Weyant 2013). However, this study does not aim to solve the social discounting dilemma once for all but tries to present an example of integrating externality into private costs accounting.

Under the context of the EU ETS, the carbon costs are more likely to be regarded as “real cash flows” in the near future. For the consistency of externality calculation, the associated discount rate of 3% was suggested (De Nocker and Debacker 2017). Weitzman (1998) stated the ‘‘lowest possible’’ interest rate should be used for discounting the far-distant future part of any investment project, as Hunkeler et al. (2008) suggested to use the 0.001% rate for discounting of the externalities for an environmental LCC. Tol (2008) found the uncertainty on the social cost of carbon is incredibly large. Even though using unacceptably low discount rates, that the fat tail effect may dominate the conclusions (Weitzman 2011). Thus we used a relatively low discount rate of 0.01% for climate change impact as suggested by (Hunkeler et al. 2008).

The financial result will be expressed as a net present value (NPV) in Euro. The costs incurred at the material preparation stage, at PCE-new manufacturing stage, at the disposal stage was regarded as NPV directly, whereas the costs incurred at the in-use stage was regarded as annual values (A), and costs incurred at the disposal stage regarded as final values (F), which were transferred into NPV according to Eqs. (3) and (4).

$$NPV = A[\frac{1}{i} - \frac{1}{i(1+i)^n}] \quad (3)$$

$$NPV = F\frac{1}{(1+i)^n} \quad (4)$$

Where A is annual energy costs; F is final EoL costs; *i* is (private/social) discount rate; n is building Life span, 120 years. For LCA, discounting of environmental impacts is seldom performed.

3.5 Results

The primary results of the LCA and LCC analysis are presented separately along with the contribution analysis, followed by a sensitivity analysis and uncertainty analysis.

3.5.1 Contribution and comparison analysis

Life cycle greenhouse gas emission

The Life cycle environmental impacts of the 120-year life cycle of the green PCE-new and BAU PCE-new are summarized in Figure 3.7. Generally, the BAU scenario and the VEEP scenario have similar distributions of life cycle GHG emissions. In both scenarios, around 88% and 83% of the life cycle GHG emission is consequent from the operation of the building in the in-use phase. Due to the climate zone of Amsterdam, the cooling need is negligible in both scenarios. The Energy demand for heating accounts for over 90% of operation emissions.

The VEEP system does not show an obvious advantage over the BAU scenario on GHG mitigation. The life cycle GHG emission of the VEEP scenario is only 4.06% lower than that of the BAU scenario. Besides, even using secondary raw materials the VEEP scenario emits more GHG than the BAU scenario. This is because the production of aerogel is a carbon-intensive activity, over 68% of the emission in the material preparation phase is original from aerogel production.

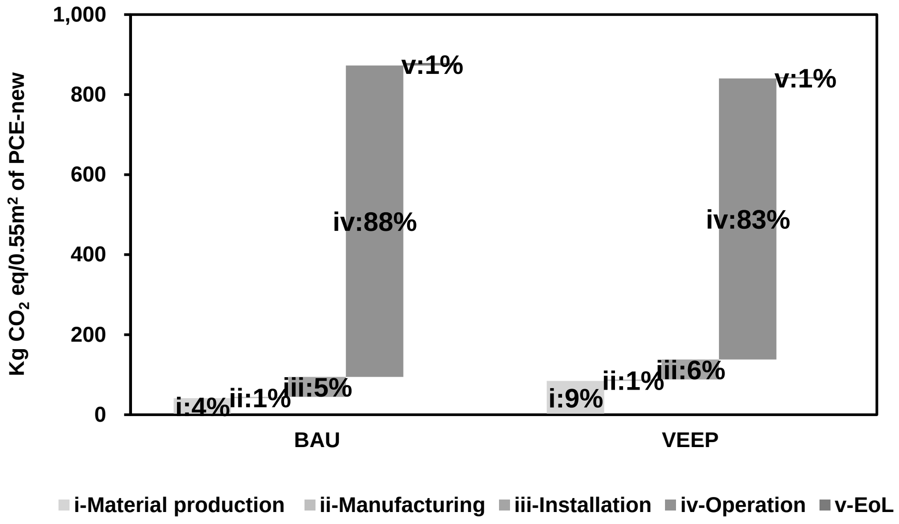

Figure 3.7 Life cycle GHG emission of BAU PCE-new (left) and green PCE-new (right). Note: EPS represents Expandable polystyrene; BAU represents Business-as-usual PCE-new technological scenario; VEEP represents VEEP technological PCE-new scenario; EoL represents end-of-life.

Life cycle costs

The LCC results on the 120-year life cycle of the green PCE-new and the BAU PCE-new are summarized in Figure 3.8. The LCCs of the two scenarios in the figure include internal costs incurred in five life cycle stages and external CO_2 costs. There is an obvious mismatch of the contribution of the installation phase in LCCs and life cycle emission. The emission of the installation phase only accounts for 5–6% in life cycle carbon emission, however, making up 45%–49% in cycle costs. This is because more than 90% of the installation costs are personnel costs, whilst using labor does not generate GHG.

The green PCE-new and BAU PCE-new have a similar distribution of LCCs. The LCCs of the BAU PCE-new is 501.63 €/0.55m^2. The costs incurred in the installation phase (49%) and the in-use phase (44%) together account for more than 90% of the LCCs. For the VEEP scenario, the LCCs of green PCE-new is about 483.66 €/0.55m^2. The costs of the installation phase and in-use phase account for 87% of the LCCs. The green PCE-new does not present a noticeably economic advantage over the BAU scenario. The LCCs of the green PCE-new is 3.58% lower than that of the BAU PCE-new. Even though considering carbon costs, the cost reduction just increases by 4.04%.

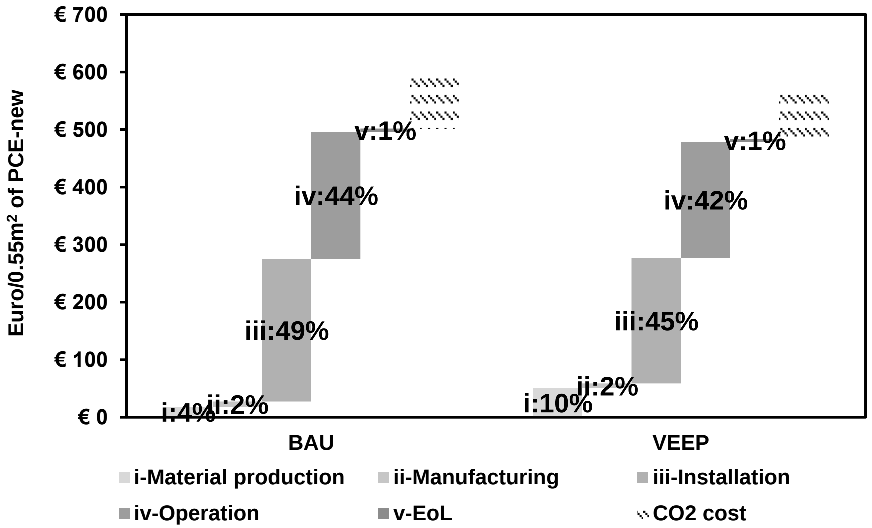

Figure 3.8 Life cycle costs of BAU PCE-new and green PCE-new. Note: Note: EPS represents Expandable polystyrene; BAU represents Business-as-usual PCE-new technological scenario; VEEP represents VEEP technological PCE-new scenario; EoL represents End-of-life.

3.5.2 Sensitivity analysis

To better understand how the CO_2 costs would affect the LCC results in reality, we assumed scenarios in which CO_2 costs are directly borne by relevant actors and are seen as internal costs in the LCCs. In this case, the discount rate is applied to the CO_2 costs that were allocated to each phase accordingly. We established two new scenarios, BAU-ex and VEEP-ex, which do consider the CO_2 costs. Cumulative cost curves of the four scenarios are projected in Figure 3.9. If CO_2 costs are considered, the cost reduction performance of VEEP-ex (compared to BAU-ex) is slightly better than VEEP (compared to BAU).

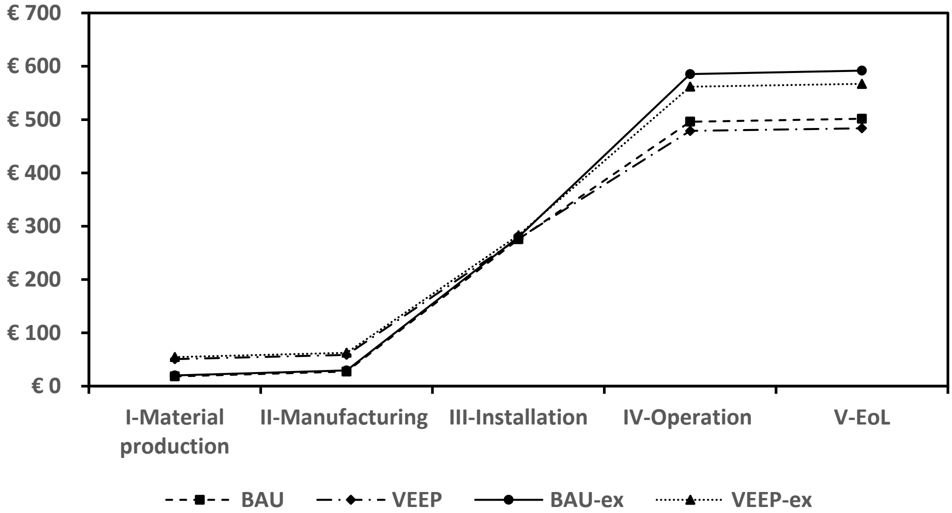

Figure 3.9 Cumulative life cycle costs of four scenarios: BAU, VEEP, BAU-ex, VEEP-ex. Note: Note: EPS represents Expandable polystyrene; BAU represents Business-as-usual PCE-new technological scenario; VEEP represents VEEP technological PCE-new scenario; BAU-ex represents BAU PCE-new scenario taking into account the external costs; VEEP-ex represents green PCE-new scenario taking into account the external costs; EoL represents end-of-life.

The robustness of these scenarios was first verified using a sensitivity analysis. As explained in the contribution analysis, 9 factors related to heating, installation, the aerogel production are considered in the sensitivity analysis as listed in Table 3.6 were considered.

Table 3.6 Factors for robustness analysis. Note: (a) as mentioned in Section 3.3.2.2; (b) for those LCI data that do not have a source of uncertainty range, a single standard error range of ±5% for the LCI data was selected in this study, which is seen as an accepted assumption regarding the uncertainty of LCI data (Huijbregts et al. 2003); (c) data from a survey to the Keey Aerogel in March 2020; (d) data from a survey to the Nobatek in March 2020; (e) from literature (De Nocker and Debacker 2017).

Code	Factors	Value	Range of uncertainty
f_1	Private discount rate	-0.52%	(-1%,3%) [a]
f_2	Social discount rate	0.01%	(-1%,3%) [a]
f_3	Heating price [€/kWh]	0.04	0.04 ±5% [b]
f_4	Aerogel cost [€/0.01m^3]	10.00	(8.00,12.00) [c]
f_5	Reduction of labor cost in installation phase	30%	(10%, 50%) [d]
f_6	CO_2 monetary indicator for construction phase [€/kg CO_2 eq]	0.045	(0.023,0.09) [e]
f_7	CO_2 monetary indicator for in-use phase [€/kg CO_2 eq]	0.11	(0.055,0.22)[e]

f_8	CO_2 monetary indicator for EoL phase [€/kg CO_2 eq]	0.14	(0.070,0.280) [e]
f_9	BAU/VEEP heating demand [kWh/(m²year)]	29.25/25.90	29.25 ± 5%/25.90 ± 5% [b]

The sensitivity analysis was conducted to identify the sensitivity of the 9 factors by decreasing 10% of each factor. The results are depicted in a radar plot in Figure 3.10. All scenarios are sensitive to heating related factors such as heating price and annual heating demand. Scenarios including external costs are more sensitive to heating demand while scenarios only containing internal costs are more sensitive to the heating price. Furthermore, four scenarios are relatively sensitive to the market interest rate but insensitive to the social discount rate. Besides, the VEEP and VEEP-ex scenarios are also sensitive to the reduction rate of labor cost in the installation phase.

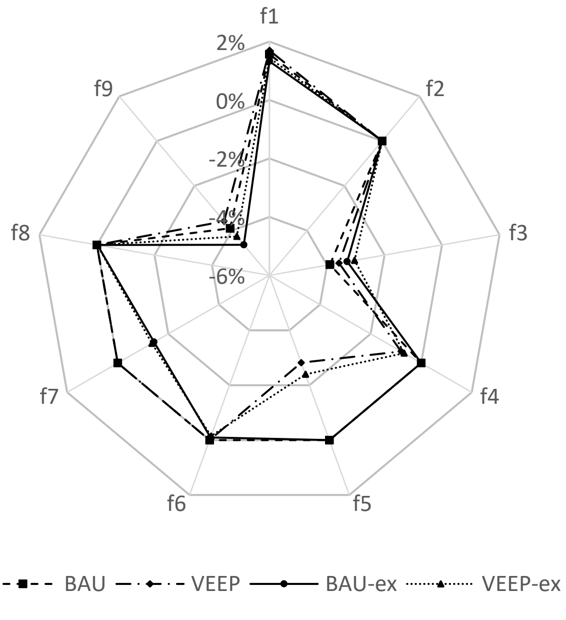

Figure 3.10 Sensitivity analysis of relevant factors in the BAU and VEEP scenarios (each factor decreased by 10%). Note: BAU represents Business-as-usual PCE-new scenario; VEEP represents green PCE-new scenario; BAU-ex represents BAU PCE-new scenario taking into account the external costs; VEEP-ex represents green PCE-new scenario taking into account the external costs.

3.5.3 Uncertainty analysis

The factors which were evaluated in the sensitivity analysis were also selected for the uncertainty analysis. The value ranges of those factors are determined by a variety of sources, as shown in Table 3.6. Values of those factors were assumed a uniform distribution varying from minimum to maximum value. To what extent the volatility of the factors affecting the LCCs is shown in Figure 3.11. Generally, scenarios that accounted for externality, BAU-ex and VEEP-ex, have a wider range of uncertainty which is mainly originated from the external discount rate and the monetary indicator for the in-use phase. For all four scenarios, the largest uncertainty stems from the fluctuation of the internal discount rate. Since the EoL costs of the four scenarios are negligible, the uncertainty of monetary indicators for the EoL phase barely affects the LCCs.

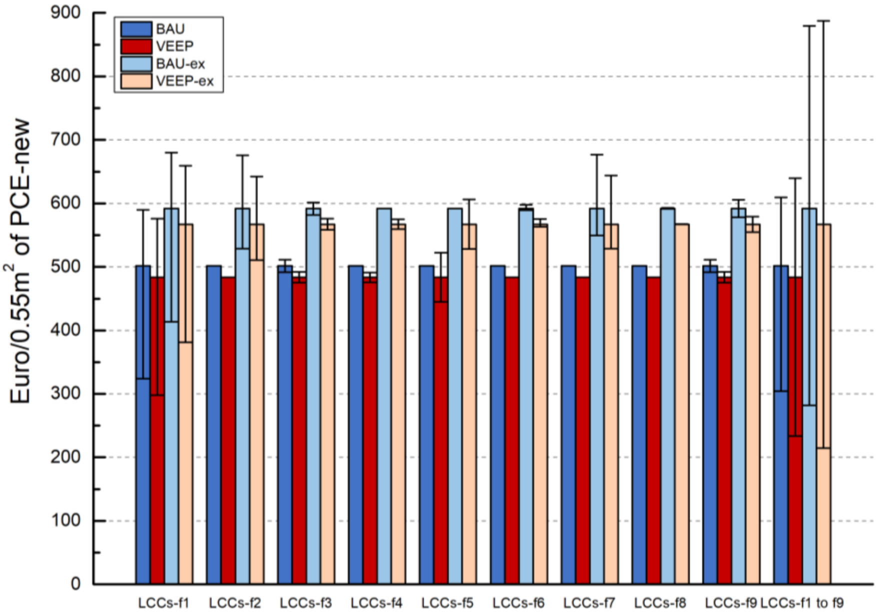

Figure 3.11 Uncertainty analysis of the LCCs: cohort LCCs-f1 to LCCs-f9 present the extent to which the fluctuation of the factors affects the uncertainty of the LCCs respectively; cohort LCCs-f1 to f9 show the summarized uncertainty of LCCs when considering all factors. Note: BAU represents Business-as-usual PCE-new technological scenario; VEEP represents VEEP technological PCE-new scenario; BAU-ex represents BAU PCE-new scenario taking into account the external costs; VEEP-ex represents green PCE-new scenario taking into account the external costs; LCCs represents Life cycle costs.

3.6 Discussion

Previous studies (Islam et al. 2014; Dong et al. 2005; Wan Omar et al. 2014; Dissanayake et al. 2017; Ottelé et al. 2011) assessed the economic and environmental performance of wall assemblage in buildings. However, it is impossible to validate the outcomes of this particular case study by comparing it to other studies, because LCA and LCC studies of residential building facades vary considerably in terms of functional units, assumptions, database, wholesale price index, and system boundaries. Additionally, they vary in building typology and life span, local climate, and inclusion or exclusion of maintenance. Results from the analysis of contribution, sensitivity and uncertainty are further discussed in this section.

At a certain degree of uncertainty, the green PCE-new system only shows slightly better economic and environmental performance in comparison with the BAU PCE-new. A few points that were not considered in the robustness analysis need to be mentioned. **First**, in the VEEP system, the production costs and environmental impacts of the secondary raw materials recovered from CDW was estimated using a mass-based allocation method. The prices used in the case study reflect only the Dutch market situation under current environmental regulations and resource conservation policies. Applying VEEP technologies in other regions with different market and policy situations may amplify or worsen the potential economic advantages of VEEP scenarios. **Second**, since green aerogel is under development. It is not financially viable yet, nor did it apparently improve the thermal performance of the green PCE-new compared to EPS. However, as a promising insulating material aerogel is crucial to the ongoing building energy renovation. This study assumed a conservative thermal conductivity value of 0.0157 W/(m·K) for the green aerogel. However, the target thermal conductivity of the green aerogel is equal to 0.012 W/(m·K). If this requirement is satisfied, the U value of green PCE-new will be below 0.17 W/(m^2·K). Besides, the production of green aerogel is currently carbon-intensive. In the EoL phase, this study assumed that aerogel was recycled by the DGR. Whereas the aerogel blankets have the potential to be fully reused, which can avoid a large amount of GHG emission from the new aerogel production. **Third**, including external CO_2 costs in LCC increases the financial advantages of low-carbon options, even though the promotion of economic viability by including externality is not significant. This indicates a government could use policy tools to propagate the use of low-carbon products by raising environmental taxes or emission fees. **Finally**, while previous research found that the LCC approach is sensitive to changes in discount rates (Islam et al. 2015), which is in accordance with the findings of this study. A high discount rate can take the edge off the economic advantages of the VEEP system in the in-use phase, flipping the evaluating results from country to country.

3.7 Conclusions

This paper presents an integrated environmental LCC and LCA study exploring to which extent the LCCs and environmental impact of building envelopes can be reduced by applying a green PCE-new system containing secondary material as opposed to a BAU PCE-new. LCA was used to estimate the GHG emission during the main life cycle phases of both PCE-news. While LCC was used to examine the systems' financial performance in parallel with LCA. To explore how externality will affect the PCE-new's economic performance, the GHG emission was internalized via monetary indicators, leading to two additional scenarios, BAU-ex and VEEP-ex.

The final results show that the life cycle GHG emission of the VEEP scenario is only 4.06% lower than that of the BAU scenario, and the majority of the carbon mitigation results from the heating energy saving. From the economic perspective, the LCCs of the green PCE-new is also 3.58% lower than that of the BAU scenario. If externality is considered, the difference in LCCs is slightly larger, amounting to 4.04% in favor of the green PCE-new, but this also leads to greater uncertainty. In the VEEP scenario, about 68% and 76% of the life cycle GHG emission and LCCs result from green aerogel production. However, the aerogel does not present an obvious advantage in energy saving for green PCE-new in the in-use phase if assumed a conservative thermal conductivity.

Sensitivity and uncertainty analysis were carried out to understand the robustness of the results. It was found that both BAU and green PCE-new are noticeably sensitive to heating demand. Thus, further work should focus on optimizing the thermal transmittance of the PCE-new. For green PCE-new, the green aerogel was shown to be the main cost stressor and GHG emitter in the material production phase. As it is under development, currently it does not show a noticeable economical advantage over EPS, nor does it show a better thermal performance than EPS. Moreover, the biggest uncertainty of the results from the discount rate is due to its wide range of possibilities.

It is necessary to note some limitations of the study. **First**, to reduce the uncertainty from monetization to some point, GHG emission was selected as the sole environmental impact indicator. However, other impact categories, such as resource depletion can be also significant in CDW management. **Second**, the dynamic thermal simulation was conducted using two different insulating materials in the PCE-news on the condition that both PCE-news maintain the same thickness. However, other potential scenarios could be established to compare the BAU and VEEP scenarios from multiple angles, such as choosing two PCE-news that use the same insulation or PCE-news with the same U value. **Third**, due to the limitations of the OpenLCA software, partial sensitivity and uncertainty analysis were performed. **Last** but not least, the controversial issues of monetization and discounting in an integrated LCC-LCA study have not been elaborated. On the one hand, the discount rate was inconsistently applied to LCC and LCA, and each cost component of LCC. On the other hand, in BAU-ex and VEEP-ex scenarios, the external cost was internalized thus market-related discount rates were applied. But the issue of discount rate for social cost including real externalities is much more complex, which is not discussed in the study.

Nevertheless, this combined LCA and LCC study of the PCE-new case explored the potential to resolve the inconsistency between the two analytical methods, from stakeholders' perspective and the factor of time, by including external costs and discounting. The study shows that to support sensible decision making, a systematic method to standardize the treatment on to be internalized external costs specify the discounting scheme should be developed for the combined use of LCA and LCC. These factors will be examined in our future studies.

Acknowledgements

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Appendix

Table A3.1 Bill of Material related to green PCE-new and BAU PCE-new

The material of the PCE-new		BAU	VEEP
Concrete	Cement (CEM III/A)	Unit process referred: "market for cement, blast furnace slag 36-65% | cement, blast furnace slag 36-65%, non-US | Cutoff, U - Europe without Switzerland"	Unit process referred: "market for cement, blast furnace slag 36-65% | cement, blast furnace slag 36-65%, non-US | Cutoff, U - Europe without Switzerland"
	URSCA	/	From HAS
	RGUA	/	From DGR
	Limestone	Unit process referred: "market for lime, packed | lime, packed | Cutoff, U-RoW"	Unit process referred: "market for lime, packed | lime, packed | Cutoff, U-RoW"
	RFUA	/	From DGR
	Siliceous sand	Unit process referred: "market for silica sand | silica sand | Cutoff, U - GLO"	/
	FRSCA	/	From HAS

The material of the PCE-new		BAU	VEEP
	Siliceous gravel	unit process referred: “market for silica sand | silica sand | Cutoff, U - GLO”	/
	CRSCA	/	From ADR
	Superplasticizer	Unit process referred: “market for plasticiser, for concrete, based on sulfonated melamine formaldehyde | plasticiser, for concrete, based on sulfonated melamine formaldehyde | Cutoff, U - GLO”	Unit process referred: “market for plasticiser, for concrete, based on sulfonated melamine formaldehyde | plasticiser, for concrete, based on sulfonated melamine formaldehyde | Cutoff, U - GLO”
	Water	Unit process referred: “market for tap water | tap water | Cutoff, U - Europe without Switzerland”	Unit process referred: “market for tap water | tap water | Cutoff, U - Europe without Switzerland”
Insulation material	EPS	Unit process referred: “polystyrene foam slab production, 100% recycled | polystyrene foam slab | Cutoff, U - RoW”	Unit process referred: “polystyrene foam slab production, 100% recycled | polystyrene foam slab | Cutoff, U - RoW”
	Aerogel	/	From VEEP aerogel production
Steel frame	Rebar Cages & welded nets	Unit process referred: “reinforcing steel production | reinforcing steel | Europe”	Unit process referred: “reinforcing steel production | reinforcing steel | Europe”

Table A3.2 Energy usage related to green PCE-new and BAU PCE-new manufacturing

Energy carrier	Energy usage per unit of PCE-new	Unit processes referred
Electricity	39.70 kWh	“market for electricity, high voltage | electricity, high voltage | Cutoff, U - NL”
Natural gas	292.41 MJ	“natural gas, burned in gas motor, for storage | natural gas, burned in gas motor, for storage | Cutoff, U - NL”
Diesel	107.06 MJ	“market for diesel, burned in building machine | diesel, burned in building machine | Cutoff, U - GLO”

Table A3.3 Input for installation of the BAU and green PCE-new

Input for per m2 of PCE-new	Amount	Unit processes referred in Ecoinvent 3.4
Transport	307.69 t km	“market for transport, freight, lorry 16-32 metric ton, EURO3 | transport, freight, lorry 16-32 metric ton, EURO3 | Cutoff, U – GLO”
Foundation slabs	0.08 m^3	“concrete production 30-32MPa, RNA only | concrete, 30-32MPa | Cutoff, U – RoW”

Steel structure	15.63 kg	"market for sheet rolling, steel | sheet rolling, steel | Cutoff, U – GLO"
Floor slabs	0.03 m^3	"concrete production 30-32MPa, RNA only | concrete, 30-32MPa | Cutoff, U – RoW"
Electricity	7.50 MJ	"market for electricity, high voltage | electricity, high voltage | Cutoff, U – NL"

Table A3.4 Designed thermal conductivity and Thickness of materials in green PCE-new

Material (external layer to internal layer)	Designed thermal conductivity-λ [W/(m. K)]	Thickness [mm]
External concrete	2.080	60
Aerogel	0.0157	70
EPS	0.031	80
Internal concrete	1.900	50

Table A3.5 Designed thermal conductivity and Thickness of materials in BAU PCE-new

Material (external layer to internal layer)	Designed thermal conductivity-λ [W/(m.K)]	Thickness [mm]
External concrete	2.080	60
EPS	0.031	70
EPS	0.031	80
Internal concrete	1.900	50

Table A3.6 Thermal transmittance of the building envelope for dynamic thermal simulation with PCE-news used as façade

Element	Thermal transmittance [$W/(m^2 K)$]
Floor	0.181
Roof	0.118
Glazing	1.300
BAU PCE-new	0.320
Green PCE-new	0.190

Table A3.7 Allocation method for crushing in EoL phase of VEEP scenario

Process name	Multifunctionality category	Functional flows	Allocation shares	Category
Crushing	Recycling	EoL concrete treatment	50%	Target service
		RCA	50%	Non-target product

Table A3.8 Internal cost data and sources

Life cycle phase		Cost detail and its sources
Material preparation phase	Siliceous concrete	**Raw material:** Virgin siliceous sand/gravel price is 31.20 €/metric ton, data referred to process "market for silica sand | silica sand | Cutoff, U - GLO" [a]; Virgin cement price is 61.50 €/metric ton, data referred to process "market for cement, blast furnace slag 36-65% | cement, blast furnace slag 36-65%, non-US | Cutoff, U - Europe without Switzerland" [a]; Limestone powder price is 122.00 €/metric ton, data referred to process "market for lime, packed | lime, packed | Cutoff, U - RoW" [a]; Superplasticizer price is 1280 €/metric ton, data referred to process "chemical production, organic | chemical, organic | Cutoff, U – GLO" [a]. **Utilities:** Diesel price is 0.73 €/L, data referred to the process "diesel, burned in building machine | diesel, burned in building machine | Cutoff, U – GLO" [a]; Water (for dust control) price is 0.16 €/L [b]; Non-household electricity price is 0.06 €/kWh [c]. **Personnel:** Wages and salaries in the construction sector are set as 35.9 €/man-hour [d]. **Equipment:** Hourly depreciation of each piece of equipment in this study is as follows: HAS is 14.73 €/h [e]; crushing set is 147.67 €/h [e]; ADR is 83.73 €/h [e]; DGR set 3.18 €/h [f]. **Transport:** transport cost of raw material and waste is 0.1 €/km·t [g] (hereafter).
	Aerogel	The comprehensive unit cost for aerogel is 10.00 €/m^2 (thickness 1cm) [h].
	EPS	EPS price is 1240.00 €/metric ton [a], data referred to process "polystyrene foam slab production, 100% recycled | polystyrene foam slab | Cutoff, U - RoW".
	Steel frame	Steel frame price is 537.00 €/metric ton [a], data referred to process "reinforcing steel production | reinforcing steel | Cutoff, U-RER".
PCE-new manufacturing phase	Equipment	Equipment depreciation is for per m^2 of EPS is as follows [i]: 0.80 €/m^2 for BAU PCE-new; 0.70 €/m^2 for green PCE-new.
	Personnel	Personnel cost for per m^2 of PCE-new is as follows [i]: 13.40 €/m^2 for BAU PCE-new; 11.7 €/m^2 for green PCE-new.
	Utilities	Energy cost per m^2 of EPS is as follows [i]:

		Electricity cost is 0.30 €/m^2 [i]. Natural gas cost is 0.30 €/m^2 [i]. Gasoline cost is 0.40 €/m^2 [i].
	Transport	Transport cost per panel is 35.75 €/per panel, 50km of transport distance is assumed [i].
PCE-new installation phase	Foundation slabs	3.99 € per m^2 of VEEP/BAU PCE-new [i].
	Steel	3.99 € per m^2 of VEEP/BAU PCE-new [i].
	Floor slabs	1.49 € per m^2 of VEEP/BAU PCE-new [i].
	Personnel cost	354.17 € per m^2 of BAU PCE-new; 247.92 per m^2 of green PCE-new [i].
	Electricity	0.17€ per m^2 of VEEP/BAU PCE-new [i].
	Transport	4.17 € per m^2 of VEEP/BAU PCE-new [i].
PCE-new in-use phase	Heating energy	Cost related to air water heat pump. Heating cost is 0.04 €/kWh, referred to the process “market for floor heating from air-water heat pump | heat, air-water heat pump 10kW | Cutoff, U - Europe without Switzerland” [a].
	Cooling energy	Cost related to electricity for household cooling. The electricity price is 0.21 €/kWh [c].
EoL PCE-new disposal	EoL concrete crushing	“Crushing” in the PCE-new disposal phase was modeled as same as the “crushing” process in the material preparation phase which referred to literature (Zhang et al. 2019a).
	EPS incineration	Reception of insulation at waste processor 80.00 €/metric ton excludes transport [b].
	Aerogel recycling	The aerogel is recycled by DGR.
	Steel recycling	Sell of other ferrous metals at demolition site 133.12 €/metric ton [b].

Notes:
[a] Data from database Ecoinvent 3.4 for OpenLCA 1.7.4;
[b] Data from HISER project report D5.4 “Final Report of Integrated environmental and economic assessment for the HISER case studies” via www.hiserproject.eu.
[c] Data from Eurostat “Electricity prices by type of user” (the Netherlands, 2017), via https://ec.europa.eu/eurostat/tgm/table.do?tab=table&init=1&language=en&pcode=ten00117&plugin=1;
[d] Data from Eurostat, Labour cost levels by NACE Rev. 2 activity (the Netherlands, 2018), via http://ec.europa.eu/eurostat/web/products-datasets/-/lc_lci_lev ;
[e] Data from an interview with Dr. Abraham Gebremariam and Dr. Francesco Di Maio from the Technology University of Delft on July 2018;
[f] Data obtained from interviewing with Mr. Ismo Tiihonen on July 2018;
[g] Data investigated via interviewing with Mr. Frank Rens from Strukton BV in November 2018;
[h] Data obtained from VEEP inner report No. D3.3 on May 2019, authorized by Dr. Francisco Ruiz and Dr. Kanda Philippe from Keey Aerogel;
[i] Data from VEEP inner report No. D6.2 on June 2019 from RINA Consulting;

Table A3.9 Monetary values for monetization of global warming

Indicator per phase	Applied to phases in this study	Descriptions
Construction	Material preparation phase, manufacturing phase	Monetary indicator (low/central/high) for global warming of building materials in the construction phase is 0.023/0.045/0.09 €/kg CO_2 eq. The central value 0.045 €/kg CO_2 eq is selected for assessment.
Use phase	In-use phase	Monetary indicator (low/central/high) for global warming of building materials in the use phase is 0.055/0.11/0.22 €/kg CO_2 eq. The central value 0.11 €/kg CO_2 eq is selected for assessment.
End of life	EoL disposal	Monetary indicator (low/central/high) for global warming of building materials in the EoL phase is 0.070/0.140/0.280 €/kg CO_2 eq. The central value 0.140 €/kg CO_2 eq is selected for assessment.

Chapter 4

Energy-carbon-investment payback analysis of prefabricated envelope-cladding system for building energy renovation: cases in Spain, the Netherlands, and Sweden

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Abstract

Buildings have become a major concern because of their high energy use and carbon emissions. Thus, a material-efficient prefabricated concrete element (PCE) system was developed to incorporate construction and demolition waste as feedstock for residential building energy renovation by over-cladding the walls of old buildings. By conducting a life cycle assessment and life cycle costing using the payback approach, this study aims to explore the life cycle performance of energy conservation, carbon mitigation, and cost reduction of the PCE system in three European member states: Spain, the Netherlands, and Sweden. The results show that the energy payback periods for Spain, the Netherlands, and Sweden were 20.45 years, 17.60 years, 19.95 years, respectively, and the carbon payback periods were 23.33 years, 16.78 years, and 8.58 years, respectively. However, the financial payback periods were less likely to be achieved within the building lifetime, revealing that only the Swedish case achieved a payback period within 100 years (83.59 years). Thus, circularity solutions were considered to shorten the PCE payback periods. Using secondary materials in PCE fabrication only slightly reduced the payback period. However, reusing the PCE considerably reduced the energy and carbon payback periods to less than 6 years and 11 years, respectively in all three cases. Regarding cost, reusing the PCE shortened the Swedish payback period to 29.30 years, while the Dutch and Spanish cases achieved investment payback at 42.97 years and 85.68 years, respectively. The results can be extrapolated to support the design of sustainable building elements for energy renovation in Europe.

Keywords: life cycle assessment, life cycle costing, building energy renovation, payback, construction and demolition waste, prefabricated concrete element

Abbreviations

ADR	Advanced drying recovery
BAU	Business-as-usual
CDW	Construction and demolition waste
CED	Cumulative energy demand
CRLWCA	Coarse recycled lightweight concrete aggregate
DGR	Dry Grinding & Refining system
EC	European Commission
EER	Energy efficiency ratio
EoL	End-of-life
EPS	Expanded polystyrene
EU	European Union
FRLWCA	Fine lightweight recycled concrete aggregate
GHG	Greenhouse gas
HAS	Heating air classification system
LCA	Life cycle assessment
LCC	Life cycle costing
LCEA	Life cycle energy analysis
$LCCO_2A$	Life cycle carbon emission analysis
IPCC	Intergovernmental Panel on Climate Change
PCE	Prefabricated concrete element
PCE-new	Prefabricated concrete element for new building construction
PCE-refurb	Prefabricated concrete element for existing building retrofit
RCA	Recycled concrete aggregate
RFUA	Recycled fiber wool ultrafine admixture
RGUA	Recycled glass ultrafine admixture
URLWCA	Ultrafine recycled lightweight concrete aggregate
VEEP	European Union Horizon 2020 project VEEP

4.1 Introduction

As of late, the building sector has become a primary contributor to global warming and resource depletion, in which buildings account for approximately 40% and 33% of global energy use and greenhouse gas (GHG) emissions (Atmaca 2016b). By 2050, it is projected that the global energy use of buildings might double, or even triple (Chalmers 2014). The European Union (EU) reacted to the IPCC (Intergovernmental Panel on Climate Change)'s 2 °C target by formulating legislative goals of reducing energy use and GHG emissions for the built environment in both the short- and long-term (EZK 2019).

In the EU, building sector legislature has been prioritized as it has the potential to meet certain GHG mitigation and energy-saving targets. Currently, more than 30% of buildings in the EU are more than 50 years old, and over 70% of the building stock is energy-inefficient (EC 2010). Thus, improving the overall energy performance of both old and new buildings is necessary. However, the construction of new energy-efficient buildings does not meet the short-term GHG mitigation goals (Säynäjoki et al. 2012). Therefore, renovating existing buildings would enable the EU to meet its 2030 goals of 32.5% energy savings and a 40% GHG emissions reduction, as compared with 1990 (EZK 2019).

EU-level legislative initiatives have been introduced for building renovations. In particular, directive 2012/27/EU requires member states to establish national strategies for cost-effectively renovating more than 3% of the central government's gross building stock each year (EC 2012b). Directive 2010/31/EU set minimum energy use standards and cost-optimal levels for old building renovations (EC 2010). Amendment 2018/844 to Directive 2010/31/EU introduced a clear target for the full decarburization of the EU's building stock by 2050. However, an investigation of 16 EU regions (covering more than 60% of the gross EU floor area) indicated that over 97% of buildings must be renovated to accomplish the EU's 2050 decarburization goal (BPIE 2017). Hence, cost-efficient energy-saving solutions are necessary to support the current energy goals of the EU.

Up-scaling building renovation by mitigating the energy dissipation from building heat loss is a priority for the EU building stock (Staniaszek 2015). Recently, an EU project, the European Union Horizon 2020 project VEEP (VEEP), developed a technological system to use recycled construction and demolition waste (CDW) to manufacture prefabricated concrete elements (PCE). These PCEs have been used to improve the thermal performance of buildings by either being constructed as an envelope of new buildings (PCE-new) or by over-cladding the envelope of existing buildings (PCE-refurb). As the life cycle performance of PCE-new has been previously evaluated (Zhang et al. 2020a), this study examines the performance of PCE-refurb.

Life cycle costing (LCC) and life cycle assessment (LCA) are typical appraisal tools of life cycle management methodology (Hunkeler et al. 2003). An integrated implementation of LCC and LCA can provide broader insights into sustainable products and technologies (Miah et al. 2017). Therefore, this study aims to employ LCC and LCA simultaneously to evaluate the potential of implementing VEEP PCE-refurb in residential buildings to save costs, conserve energy, and mitigate GHG emissions under different EU member states' climatic contexts. The results of the LCC and LCA are expressed for investment, energy, and carbon payback periods, and the applicability of VEEP PCE-refurb to EU residential buildings is examined. The results of this study will support policymakers in selecting cost-effective and material-efficient paths for building energy renovation in the EU.

This study is outlined as follows: Section 4.2 illustrates the details of the technological system and main methods; Section 4.2.4 states the results; Section 4.4 discusses the application potential of the PCE-refurb system at an EU-wide scale and the reusability of PCE-refurb, and Section 4.5 presents the conclusions of the study.

4.2 Materials and methods

This section presents the basic materials and methods used in this study. Section 4.2.1 details overviews of the literature related to LCC- and LCA-based payback period methods in the field of building energy renovation, proposing a conceptual framework for an energy-carbon-investment payback period analysis. Based on this conceptual framework, Section 4.2.2 defines the goal and scope of the assessment system. Section 4.2.3 presents the life cycle environmental and economic inventory LCC and LCA, and Section 4.2.4 details the life cycle environmental and economic impact analysis.

4.2.1 Life cycle management of building energy renovation

4.2.1.1 Overview of life cycle energy, carbon emission, and cost analysis

As one of the main techniques for life cycle management (Hunkeler et al. 2003), LCAs are commonly used to explore opportunities in GHG emissions mitigation and energy efficiency in the building sector (Sharma et al. 2011). Based on an LCA, the life cycle carbon emission analysis (LCCO$_2$A) and life cycle energy analysis (LCEA) specifically focus on the life cycle CO_2 equivalent emissions and the energy use of buildings, respectively.

The Building Assessment Information System (CEN 2012) defines four life cycle stages for building performance assessment: production, construction process, use, and end-of-life (EoL) stages. An LCEA is usually employed to calculate the overall energy-related inputs to buildings from a life cycle perspective (Ramesh et al. 2010). Analogously, an LCCO$_2$A accounts for the total CO_2 equivalent emission outputs from a building over different phases of its life cycle (Chau et al. 2015). Energy (Ramesh et al. 2010) and GHG emissions (Lu and Wang 2019) in the operation stage normally account for 80%–90% of a building's life cycle energy use and GHG emissions, followed by embodied energy and emissions, which accounts for 10%–20%. Meanwhile, the demolition energy (Ramesh et al. 2010) and emissions (Lu and Wang 2019) are almost negligible, contributing approximately 1%.

In an LCEA and $LCCO_2A$, building materials production and building construction are often grouped into one stage. For example, studies on the LCEA (Ramesh et al. 2010; Cabeza et al. 2014; Chau et al. 2015; Atmaca 2016b) and $LCCO_2A$ (Atmaca 2016b; Chau et al. 2015; Lu and Wang 2019) modeled the life cycle energy and emission of three stages: (i) embodiment (manufacturing and construction), (ii) operation (operation and use), and (iii) demolition (/EoL). Therefore, estimating the life cycle energy use and life cycle GHG emissions of buildings can be determined by summing all the energies and emissions incurred during their life cycle, as expressed in Eqs. (1) and (2):

$$Life\ cycle\ energy\ use = E_E + E_O + E_D, \tag{1}$$

$$Life\ cycle\ carbon\ emission\ = C_E + C_O + C_D, \tag{2}$$

where E_E denotes the energy use incurred in the embodiment phase, E_O represents the energy use incurred in the operation phase, E_D denotes the energy use incurred in the demolition phase, C_E denotes the GHG emissions incurred in the embodiment phase, C_O represents the GHG emissions incurred in the operation phase, and C_D denotes the GHG emissions incurred in the demolition phase.

Despite their popularity, it is debated whether LCEAs and $LCCO_2As$ are stand-alone methodologies, a step, or indicators to be included in the life cycle inventory analysis or the life cycle impact assessment in an LCA. Chau et al. (2015) reviewed the literature regarding LCAs, LCEAs, and $LCCO_2As$ and found that an LCEA focuses on energy input and an $LCCO_2A$ on outputs, while an LCA considers both environmental inputs and outputs. In this manner, the LCA is an overarching environmental assessment that includes both LCEAs and $LCCO_2As$. Conversely, the cumulative energy demand (CED) is a key index for both LCAs and LCEAs. Klöpffer (1997) stated that the CED is an inventory indicator that does not rely on any assumptions. However, Frischknecht (1997) explained that some assumptions are necessary to develop CED factors (Frischknecht et al. 2015). Instead of employing the LCEA or $LCCO_2A$ as independent methods, this study used a standard LCA, which conforms to ISO 14040 (ISO 2006a) and ISO 14044 (ISO 2006b), as an appraisal tool to explore the life cycle energy and carbon emissions of the PCE-refurb system.

Regarding economic assessment, LCC is a financial assessment tool that explores the costs incurred during the life cycle of a product system (ISO 2017a). There are multiple cost breakdown structures for an LCC, such as lifecycle-based, stockholder-based, and expenditure-based (Zhang et al. 2019a). The selection of the cost breakdown structure depends on the user's goal and scope. Owing to the characteristics of a life cycle perspective, the life cycle cost of a building is usually estimated based on the building's life cycle. According to ISO 15686–5 (ISO 2017a), the life cycle cost of a building consists of construction costs, operation and maintenance costs, and EoL costs, as shown in Eq. (3).

$$Life\ cycle\ cost\ = I_E + I_O + I_D, \tag{3}$$

where I_E represents the construction costs incurred in the embodiment stage, I_O denotes the operation and maintenance costs incurred in the operation phase, and I_D represents the EoL cost incurred in the demolition phase. External costs, such as environmental or social costs, are not considered in this study.

For the consistent application of LCAs and LCCs, the Society of Environmental Toxicology and Chemistry Europe working group defined an environmental LCC (Ciroth et al. 2008; Swarr et al. 2011), which is not meant to consider environmental externalities but has a methodological framework similar to a standard LCA. This study employed both an LCA and environmental LCC (hereinafter referred to as LCC) to investigate the energy and carbon reductions and economic viability of the PCE-refurb system.

4.2.1.2 Payback period method

Several systematic reviews have been conducted on LCCs (Sesana and Salvalai 2013; Islam et al. 2015; Atmaca 2016b; Lu et al. 2021), LCAs (Chau et al. 2015; Ramesh et al. 2010; Vilches et al. 2017; Singh et al. 2011; Sesana and Salvalai 2013; Buyle et al. 2013; Anand and Amor 2017; Islam et al. 2015; Atmaca 2016b; Lu et al. 2021), LCEAs (D'Oca et al. 2018; Sadineni et al. 2011; Chau et al. 2015; Ramesh et al. 2010; Sesana and Salvalai 2013; Deng et al. 2014; Sartori and Hestnes 2007), and LCCO_2As (Chastas et al. 2018; Chau et al. 2015) for buildings and the building sector. These reviews demonstrated that estimating the life cycle ecological and economic performance by summing all the impacts incurred during each life cycle stage over a lifetime is the most straightforward and commonly employed method for comparing building performances.

However, in some studies, the temporal scope of the LCA was not directly defined, or the goal of a study was to explore a breakeven time, making comparison impossible. For instance, in this study, the lifespan of a PCE-refurb is dependent on the remaining lifetime of the building. Because the remaining building lifetime varies, due to different construction times, the temporal span of a PCE-refurb cannot be directly set. In this case, it would be more straightforward to evaluate the life cycle results using the payback period approach.

The payback period method is used to appraise the economic attractiveness of capital investments (Yard 2000). Despite its methodological deficiencies, a payback period is employed as a primary sieve or constraint for investment appraisal (Weingartner 1969), representing the amount of time it takes to recover the cost of an investment, as expressed in Eq. (4) (Yard 2000).

$$Investment\ payback\ period = \frac{I}{CF} = \frac{1-(1+IRR)^{-L}}{IRR}, \quad (4)$$

where I is the investment outlay, CF denotes the annual cash flow, L represents the economic life, and IRR denotes the discount rate that makes the net present value equal to 0.

Regarding energy efficiency issues, the payback method is commonly used in energy efficiency and low-carbon projects, such as photovoltaics (Peng et al. 2013) and building energy renovation (Vilches et al. 2017). Table 4.1 summarizes studies related to the payback method, wherein the estimation of the energy and carbon payback periods are expressed by Eqs. (5) (Ardente et al., 2011; Asdrubali et al., 2019; Berggren et al., 2013; Comodi et al., 2016; Huang et al., 2012; Lu and Yang, 2010; Papaefthimiou et al., 2006), and (6) (Lu and Yang 2010; Huang et al. 2012b; Asdrubali et al. 2019; Ardente et al. 2011):

$$Energy\ payback\ period = \frac{E_E}{E_O}, \quad (5)$$

$$Carbon\ payback\ period = \frac{C_E}{C_O}, \quad (6)$$

where E_E is the initial embodied energy, E_O is the annual operational energy saving, C_E is the initial embodied GHG emission, and C_O denotes the annual operational GHG savings.

Table 4.1 Payback period literature of building energy renovation

Source	Topic	Area	Main findings
(Leckner and Zmeureanu 2011)	Net Zero Energy building with solar power	Quebec, Canada	The energy payback time is 8 – 11 years in the cold climate of Quebec, suggesting, with the high investment of the solar system, the financial payback may never be achieved (6 – 39 years).

(Säynäjoki et al. 2012)	Residential building Renovation	Finland	The carbon payback period of rebuilding new dwellings is several decades longer than that of renovating existing buildings, but; the period of renovation is 25 years less than rebuilding.
(Papaefthimiou et al. 2006)	Electrochromic window	Greece	The energy payback period is 8.9 years when considering aluminum frames.
(Berggren et al. 2013)	Heating energy sources	Sweden	The energy payback period of renewable heating alternatives (photovoltaic, solar thermal, and heat pump). Heat pump is the most promising option, with an energy payback period of less than 1 year.
(Dylewski and Adamczyk 2014)	Insulation material for exterior wall of a building	Poland	The economic payback periods of these materials (up to 24 years) are much longer than the ecological payback periods (up to 4 years).
(Huang et al. 2012b)	Overhang shading for campus buildings	Hong Kong, China	The energy payback period of the shading system is approximately 46 years; the carbon payback period is approximately 64 years.
(Bull et al. 2014)	School buildings refurbishment	Hong Kong, China	Mean discounted financial payback (32.1 years) is longer than carbon payback (3.9 years).
(Asdrubali et al. 2019)	Nearly-Zero Energy-level retrofit for school building	Turin, Italy	The carbon and energy payback periods show the same trend (3 – 7 years); the economic payback periods (5 – 30 years) are higher than the environmental periods. Retrofitting related to generators presents the biggest energy-saving potential.
(Faludi and Lepech 2012)	Rebuild of commercial building	San Francisco, California, USA	The carbon payback of a new building with no solar versus that with an existing one is approximately 7 years; a net-zero-energy building with rooftop solar is approximately 6.5 years. A full EcoIndicator99 impact payback for a new building with no solar is 20 years; a solar net-zero

			building is 7 years as compared with a building with existing operation.
(Hossain and Marsik 2019)	Highly energy-efficient house	Rural Alaska, USA	The carbon payback period of a house with a high insulation level is 3 years compared with a typical house.
(Schwartz et al. 2016)	House complex refurbishment	Sheffield, UK	Advanced refurbishment can reduce the carbon payback from over 160 years to less than 60 years, as compared with ordinary refurbishment. Updating from heating by waste combustion to natural gas can reduce the carbon payback from 56 – 58 years to 16 years.
(Mohammadpourkarbasi and Sharples 2013)	Eco-Refurbishment of dwellings	Liverpool and London, UK	The carbon payback time of refurbishment is less than 7 years.
(Dodoo et al. 2010)	Wood-framed apartment retrofit	Växjö, Sweden	The energy payback period is less than 4 years.
(Ardente et al. 2011)	Public buildings under different retrofit strategies	Brno, Czech; Gol, Norway; Plymouth, UK; Copenhagen; Denmark; Stuttgart, Germany; Vilnius, Lithuania	Regarding different retrofitting actions, the carbon payback period ranges from 0.4 years to 1.9 years; the energy payback periods are in the range of 0.4 – 2.1 years.
(Vilches et al. 2017)	-	-	When considering all environmental impact categories, the payback period of an energy retrofit building is less than 7.5 years
(Comodi et al. 2016)	Domestic hot water systems with unglazed and glazed solar thermal panels	Rome, Italy; Madrid, Spain; Munich, Germany	The energy payback of an unglazed panel system is 2 – 5 months and that of a glazed panel is 5 – 12 months. The carbon payback of an unglazed panel system is 1 – 2 months, while that of a glazed panel is 12 – 30 months. The economic payback is 9 – 11 years/8 – 13 years for systems with unglazed/glazed panels when compared with a natural gas boiler, and 3 – 4 years/4 years

			for those compared with an electric boiler.
(Lu and Yang 2010)	Roof-mounted building-integrated photovoltaic (PV) system	Hong Kong, China	The energy payback time of a PV system ranges from 7.1 to 20 years; the carbon payback time is 5.2 years.

These studies demonstrate that the payback period method is a suitable approach to handle issues related to building energy renovation as it can be used for different purposes, such as the environment, energy, and economic payback period, or as an integrated ecological payback period that includes multiple environmental impact categories. The payback method can also be modified to assess various topics, including building materials, building elements, buildings, and the area of buildings. These payback studies also manifest in energy renovation projects in which economic investment has a longer return period to return than embodied carbon emissions and energy use.

However, research gaps exist in the literature as studies do not consider the influence of material circularity in the EoL phase. Although the EoL impact accounts for approximately 1% of the life cycle energy and GHG emissions, utilizing secondary materials and reusing EoL products has the potential to significantly reduce the impact of the embodiment phase. Therefore, this study aims to examine cross-state cases to investigate the energy-carbon-investment payback period of the PCE-refurb system for building energy renovation and evaluate how material circularity influences the payback periods.

4.2.1.3 Methodological framework

The energy/carbon/investment payback periods herein indicate the length of time required for the cumulative cost/energy/GHG reduction from the implementation of PCE-refurbs to equal the cost/energy/GHG incurred in the embodiment and demolition phases. Based on Eqs. (4), (5), and (6), the energy, carbon, and investment payback periods are calculated with Eqs. (7), (8), and (9), respectively. This study applies process-based LCA and LCC to quantify PCE-refurb performance in different European cities, namely, Madrid, Amsterdam, and Stockholm.

$$T_E = \frac{E_E^{PCE-refurb} + E_D^{PCE-refurb}}{E_O^{BAU} - E_O^{PCE-refurb}}, \tag{7}$$

$$T_C = \frac{C_E^{PCE-refurb} + C_D^{PCE-refurb}}{C_O^{BAU} - C_O^{PCE-refurb}}, \tag{8}$$

$$T_I = \frac{I_E^{PCE-refurb} + I_D^{PCE-refurb}}{I_O^{BAU} - I_O^{PCE-refurb}}, \tag{9}$$

where T_E represents the energy payback period, $E_E^{PCE-refurb}$ denotes the embodied energy use for manufacturing the PCE-refurb, $E_O^{PCE-refurb}$ denotes the energy demand for heating and cooling in the building operation phase after PCE-refurb refurbishment, $E_D^{PCE-refurb}$ is the energy use for the treatment of EoL PCE-refurb in the demolition phase, and E_O^{BAU} is the energy demand in the operation phase of a building with a business-as-usual (BAU) wall as a façade. Similarly, T_C represents the carbon payback period, $C_E^{PCE-refurb}$ represents the embodied GHG emission for PCE-refurb manufacturing, $PCE-refurb$ denotes the GHG emissions incurred in the operation phase of a building after refurbishment with PCE-refurb, $C_D^{PCE-refurb}$ demonstrates the GHG emissions for treating EoL PCE-refurb in the demolition phase, and C_O^{BAU} denotes the GHG emissions in the operation phase of a building with a BAU wall as a façade. Finally, T_I represents the carbon payback period, $I_E^{PCE-refurb}$ denotes the investment for PCE-refurb manufacturing incurred in the embodiment phase, $I_O^{PCE-refurb}$ denotes the operation costs incurred in the operation phase of a building after refurbishment with PCE-refurb, $I_D^{PCE-refurb}$ denotes the cost for PCE-refurb EoL treatment incurred in the demolition phase, and I_O^{BAU} represents the GHG cost incurred in the operation phase of a building with the BAU wall as the facade.

The LCA in this study was outlined using the four steps determined by the ISO standards: (i) goal and scope definition, (ii) life cycle inventory analysis, (iii) life cycle impact assessment, and (iv) results interpretation. The CED and global warming potential were considered impact category indicators that belong to the life cycle impact assessment step. An LCC was performed using the same four steps, as proposed by Zhang et al. (2019). The conceptual framework of this study is shown in Figure 4.1. Note that the LCA focuses on energy inputs and GHG emission outputs from the system, whereas the LCC focuses on the investment inputs released from the system, thereby representing the three life cycle phases in the assessment. In the embodiment phase, both virgin and recycled raw materials were incorporated into the fabrication of PCE-refurb. In the operation phase, individual air conditioning was assumed to model the demand for household cooling. For heating energy demand, residential buildings in different member states were assumed to be equipped with different household heating systems based on the TABULA database (TABULA 2017). During the demolition phase, the impact of recycling and reusing PCE-refurb on the payback period was evaluated. Thus, this study used the payback method to investigate the energy-carbon-investment payback period of the proposed PCE-refurb system with the main research objective of determining what quantity of GHG mitigation, energy saving, and economic earnings from the operation phase offsets the additional inputs required in the embodiment and demolition phases.

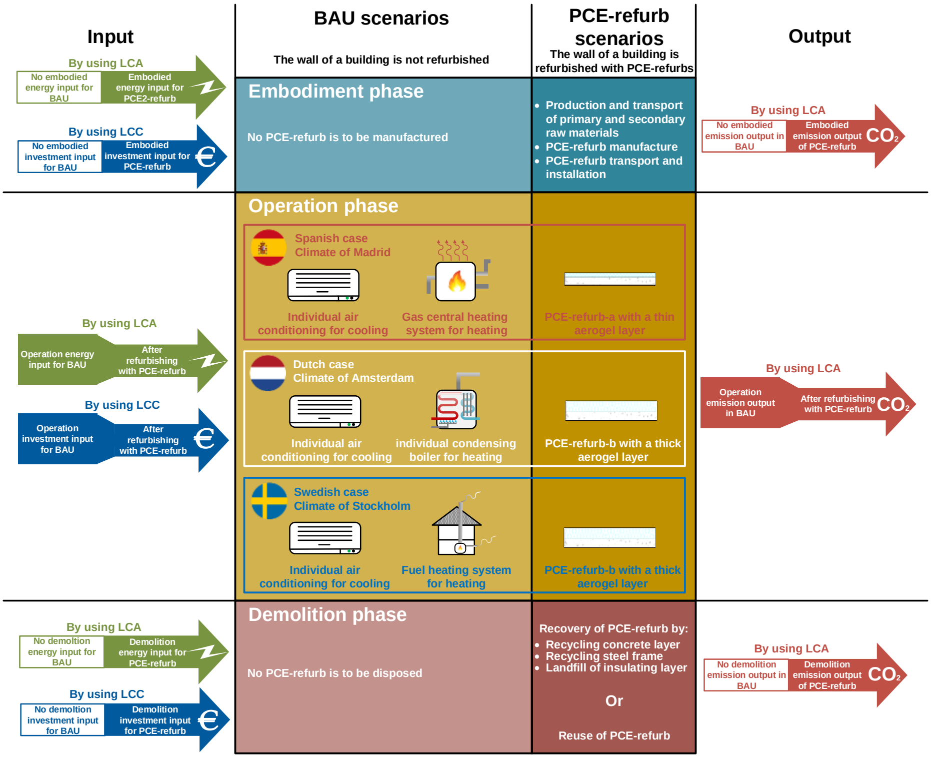

Figure 4.1 Conceptual framework of the study. LCA: life cycle assessment; LCC: life cycle costing; BAU: business-as-usual; PCE-refurb: prefabricated concrete element for old building refurbishment.

4.2.2 Goal and scope definition

4.2.2.1 Goal and scope

The goal of this study is to compare the energy and carbon payback periods for fabricating and operating the proposed PCE-refurb system as an energy retrofitting strategy for existing buildings with a conventional wall as a fa çade compared with those with conventional walls without any retrofitting in different EU member states Spain, Sweden, and the Netherlands. Herein, the LCEA and $LCCO_2A$ building analyses included three phases: embodiment, operation, and demolition. The embodiment phase includes the manufacturing and transportation of raw materials for the fabrication of PCEs. The operation phase includes the cooling and heating needs related to the use of buildings with or without the application of PCEs. Finally, the demolition phase includes the PCE dismantling and the transport of EoL materials for either disposal or treatment. Note that the object of interest is the PCE, not the building.

The system boundary for this assessment was the geographical boundaries of each studied city. Therefore, all the productive activities during the three life cycle phases are assumed to be conducted within each state. The capital cities Madrid, Amsterdam, and Stockholm were selected as the study areas. The climates and locations of these cities are listed in Table 4.2.

Table 4.2 Climates and locations of three case cities. The data source for information about Madrid (Wikipedia 2020a), Amsterdam (Wikipedia 2020b), and Stockholm (Wikipedia 2020c)

	Madrid, Spain	Amsterdam, the Netherlands	Stockholm, Sweden
Location in Europe	Southern Europe	Western Europe	Northern Europe
Coordinates	40°25'N, 3°43'W	52°22'N, 4°54'E	59°19'46"N, 18°4'7"E
Climate	Mediterranean climate which transitions to a cold semi-arid climate	Oceanic climate	Oceanic climate with humid continental

4.2.2.2 Technological systems for building energy renovation

The technological system in the VEEP project involves advanced drying recovery (ADR) integrated with a heating-air classification system (HAS) to completely recycle the EoL lightweight concrete. The produced secondary coarse and fine concrete aggregate and cementitious particles were used for the production of green lightweight concrete and green aerogel in the PCE-refurb. Furthermore, a dry grinding and refining (DGR) system was developed to reprocess glass and insulating fiber wool waste to produce secondary ultrafine admixtures to substitute cementitious materials in the concrete, such as cement and lime.

Two technological scenarios were considered herein: a BAU traditional wall and a BAU traditional wall retrofitted with different types of PCE-refurbs. The cross-sections of the traditional wall in the BAU scenario and VEEP PCE-refurbs for over-cladding the traditional wall are illustrated in Figure 4.2. A typical façade for a residential building presented on the left of Figure 4.2 was selected as a benchmark reference. Regarding the climate difference between Madrid, Amsterdam, and Stockholm, alternative structures were applied to the PCE-refurb designs. In particular, PCE-refurb-a, which has a thinner aerogel layer, was employed for the Madrid case, while PCE-refurb-b, which has a thicker aerogel layer, was implemented for the Amsterdam and Stockholm cases. The PCE-refurb-a is 2 m long, 2 m wide and 0.08 m thick, and the PCE-refurb-b is 2 m long, 2 m wide, and 0.12 m thick.

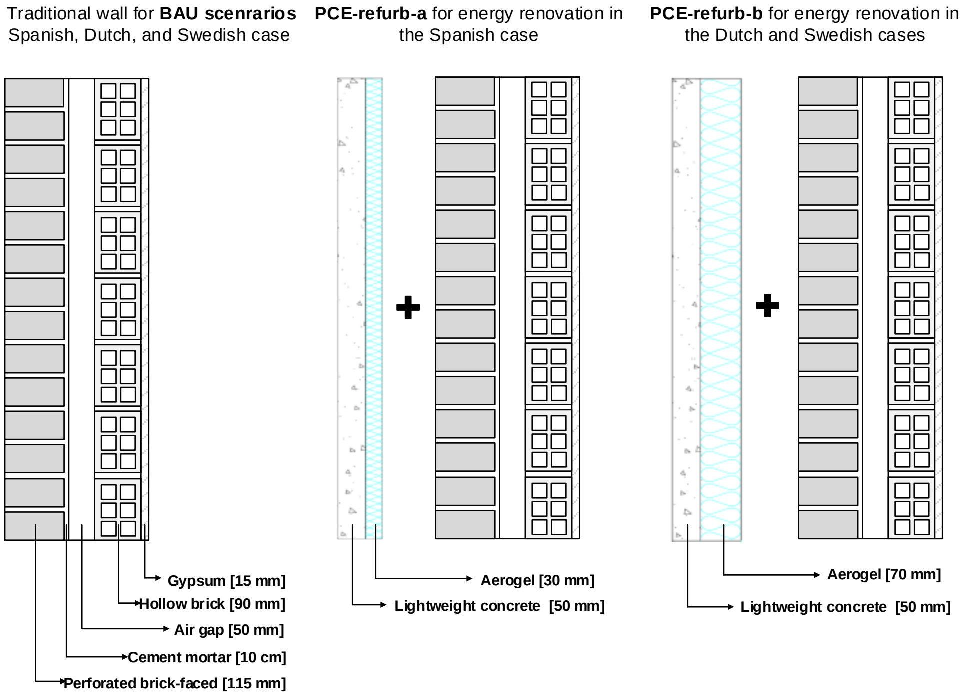

Figure 4.2 Cross-section diagrams of BAU traditional wall (left), and VEEP PCE-refurb-a (middle) to be implemented in Spain, and PCE-refurb-b (right) to be implemented in the Netherlands and Sweden. BAU: business-as-usual; PCE-refurb: prefabricated concrete element for old building refurbishment.

In the BAU scenarios, a typical traditional wall was selected as the benchmark reference for comparison with the PCE-refurb energy retrofitting scenario. Because no precast concrete elements are applied in the BAU scenario, the associated GHG emissions and energy use only occur in the operation phase.

Conversely, in the PCE-refurb scenarios, environmental impacts are incurred throughout the entire life cycle. In the embodiment phase, secondary raw materials are incorporated into a PCE-refurb. Integrated ADR and HAS technologies recycle EoL concrete, and DGR technology recovers glass waste. In the operation phase, dynamic thermal simulations were performed to compare the thermal performances of each scenario. A typical virtual residential apartment building was selected as a case study building for the thermal simulations. Finally, in the demolition phase, PCE-refurbs are dismantled and recycled. The specific features of the BAU and PCE-refurb scenarios are summarized in Table 4.3.

Table 4.3 Six scenarios developed based on technological and climate conditions. BAU: business-as-usual; PCE-refurb: prefabricated concrete element for old building refurbishment; ES: Spanish case; NL: Dutch case; SE: Swedish case; GHG: greenhouse gas.

	Spanish case	Dutch case	Swedish case
BAU	**BAU-ES:** traditional wall of the existing building under the climatic conditions of Madrid, Spain; associated investment, GHG emissions, and energy use are only incurred in the operation phase	**BAU-NL:** traditional wall of the existing building under the climatic conditions of Amsterdam, Netherlands; associated investment, GHG emissions, and energy use are only incurred in the operation phase	**BAU-SE:** traditional wall of the existing building under the climatic conditions of Stockholm, Sweden; associated investment, GHG emissions, and energy use are only incurred in the operation phase
PCE-refurb	**PCE-ES:** traditional wall of the existing building refurbished with PCE-refurb-a under the climatic conditions of Madrid, Spain; associated investment, GHG emissions, and energy use are incurred in the embodiment, operation, and demolition phases	**PCE-NL:** traditional wall of the existing building refurbished with PCE-refurb-b under the climatic conditions of Amsterdam, Netherlands; associated investment, GHG emissions, and energy use are incurred in the embodiment, operation, and demolition phases	**PCE-SE:** traditional wall of the existing building refurbished with PCE-refurb-b under the climatic conditions of Stockholm, Sweden; associated investment, GHG emissions, and energy use are incurred in the embodiment, operation, and demolition phases

The functional unit for the assessment was retaining the heating and cooling comfort for 1 m^2 floor area through (i) passive building façades (with or without the application of VEEP PCE-refurbs) and (ii) active heating by different heating systems and cooling by individual air-conditioning for 1 year based on the climate conditions in the Madrid, Amsterdam, and Stockholm. Based on the structure of the case study building, 1 m^2 of floor area requires 0.55 m^2 of PCE-refurb to over-clad the building façade.

4.2.3 Life cycle inventory analysis

The goal and scope definition step is followed by the life cycle inventory analysis, which further identifies the boundaries, background and foreground processes, and allocation scheme for a production system (Guinée et al. 2001). The system boundaries of the BAU and VEEP PCE-refurb scenarios are shown in Figure 4.3. The life cycle inventory is established according to the three phases of energy use and GHG emissions. The LCA software OpenLCA 1.9, with the Ecoinvent 3.4 Cutoff database, was used for the assessment.

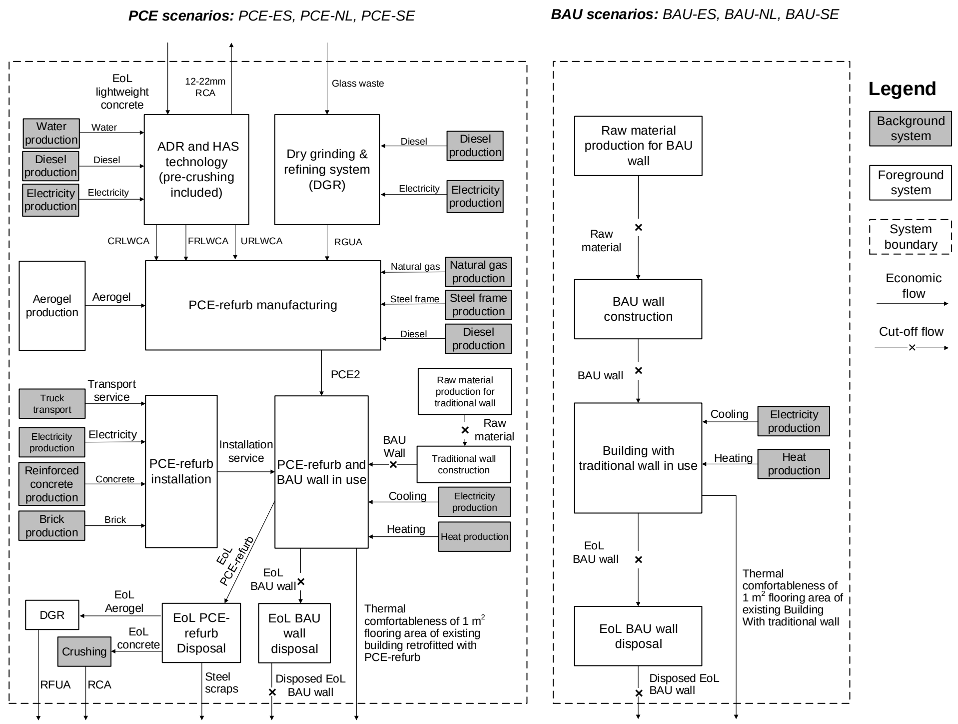

Figure 4.3 Assessment boundaries of the PCE-refurb (left) and BAU scenarios (right). ADR: Advanced dry recovery system; BAU: Business-as-usual; CRLWCA: coarse recycled lightweight concrete aggregate; DGR: Dry Grinding & Refining system; EoL: end-of-life; ES: Spanish case; FRSCA: Fine recycled siliceous concrete aggregate; FRLWCA: fine lightweight recycled concrete aggregate; HAS: Heating Air Classification System; NL: Dutch case; PCE-refurb: prefabricated concrete element for building refurbishment; RCA: recycled concrete aggregate; RGUA: recycled glass ultrafine aggregate; SE: Swedish case; URLWCA: ultrafine recycled lightweight concrete aggregate; URSCA: ultrafine recycled siliceous concrete aggregate.

4.2.3.1 Embodiment phase

The carbon emissions and energy use in the embodiment phase are only incurred in the VEEP scenarios. In this phase, virgin and secondary raw material transportation and preparation, PCE-refurb manufacturing, and PCE-refurb transport and installation were determined.

Secondary raw materials were extracted from the waste stream via ADR, HAS, and DGR to fabricate the PCE-refurb. A previous study explored the mass balance of integrated ADR and HAS technological systems for recycling both normal-weight siliceous (Zhang et al. 2019a) and lightweight concrete wastes (Zhang et al. 2021a), revealing that a larger

0–4 mm fraction is produced by processing lightweight concrete (48%) than normal-weight concrete (32%). The flow chart of ADR and HAS is shown in Figure A4.1.

DGR extracts secondary raw materials from glass, mineral wool, and fiber wool waste. In this study, DGR was used to recycle glass waste to produce recycled glass ultrafine admixture as a substitute virgin cement in lightweight concrete. The mass balance of the DGR system is shown in Figure A4.2 in the SI. Because the amount of residue from DGR is negligible, the recycling coefficient is assumed to be 100%.

As transport has proven to be of considerable importance in CDW recycling, especially when on-site recycling occurs (Zhang et al. 2019a, 2018), the impact of transportation of recycling facilities, raw materials, PCE-refurbs, and waste residue were considered in this study. The crusher (Keestrack Destroyer–1313), ADR, and HAS can be transported for on-site recycling. While DGR was once a stationary recycling facility, it has been optimized to process the CDW on-site. Therefore, all the recycling facilities in this study (crushing set, ADR, HAS, and DGR) were modeled as mobile. The truck travel distance from where recycling facilities are stored at the demolition site is assumed to be 20 km, and a typical building demolition project contains approximately 15 Kt of EoL concrete (Zhang et al. 2019a). According to the share of EoL concrete and glass waste in the CDW by weight (Zhang et al. 2020b), approximately 80 tons of glass waste is generated from a typical demolition site. The impact of recycling facility transport is allocated based on the waste recycled for PCE-refurb manufacturing and the gross waste generated from the demolition site. The operating weights of each facility are listed in Table A2.2 in Chapter 2.

In the PCE-refurb system, pre-crushing concrete rubble, recycling lightweight concrete waste by ADR and HAS, and recycling glass waste by DGR are multifunctional processes. Thus, allocation is applied to distribute the environmental impact of functional flow from these multifunctional processes. The allocation method for an LCA is based on process-based allocation. The energy use and GHG emissions of multifunctional processes are both allocated via the mass-based allocation scheme, as summarized in Table 4.4. Further, the detailed costs of virgin and secondary raw materials for the fabrication of PCE-refurb are listed in Table A4.1. Pre-crushing of concrete rubble, recycling lightweight concrete waste by ADR and HAS, and recycling glass waste by DGR are multifunctional processes. Allocation is applied to distribute the environmental impacts of functional flow from a multifunctional process. The allocation method for LCA is process-based allocation. The energy use and GHG emission of multifunctional processes are both allocated via the mass-based allocation scheme as presented in Table 4.4. The detailed bill of virgin and secondary raw materials for fabrication of a PCE-refurb is presented in Table A4.2. After extraction and refining, raw materials are transported to the factory to manufacture the PCE-refurb. It is assumed that the average truck travel distance of the recycled material is 20 km while that of virgin materials is 50 km (Zhang et al. 2020a). The energy utilities related to VEEP PCE-refurb manufacturing are listed in Table A4.3.

Table 4.4 Processes with multifunctionality in the PCE-refurb scenarios

Process name	Multifunctionality category	Functional flows	Allocation coefficient
Pre-crushing process	Recycling	EoL lightweight concrete treatment	50%
		Coarse RCA (0-12mm) production	40%
		Coarse RCA (12-22mm) production	10%
ADR process	Co-production	CRLWCA production	40%
		Fine RCA (0-4mm) production	60%
HAS process	Co-production	FRLWCA production	80%
		URLWCA production	20%
DGR process	Recycling	Glass waste treatment	50%
		RGUA production	50%

After fabrication, PCE-refurb is transported to the construction site for installation. It is assumed that the average truck travel distance of PCE-refurb is 50 km (Zhang et al. 2020a). The utilities and material inputs for PCE-refurb installation are listed in Table A4.4.

4.2.3.2 Operation phase

Dynamic thermal simulations were conducted to quantify the energy required to maintain heating and cooling under different climate conditions. Thermal assessments at the building scale were conducted on a typical residential multi-story building in Europe, as shown in Figure 3.5 in Chapter 3.

The thermal transmittance of the building walls varied from less than 0.2 W/(m ²·K) to more than 2.0 W/(m ²·K) depending on the construction age (Economidou 2011). Thus, a typical wall (as depicted in Figure 4.4) with an average level of thermal performance was selected for this case study. The thermal conductivities of the materials and components in the wall and PCE-refurb are listed in Table A4.5. The thermal transmittance of the traditional wall before and after PCE-refurb refurbishment were determined in accordance with ISO 6946 (ISO 2017b). The calculated thermal transmittance of each building element is listed in Table A4.6. The heating and cooling conditions considered herein are listed in Table 3.3 in Chapter 3.

The annual heating and cooling distribution requirements for Madrid, Amsterdam, and Stockholm based on the dynamic thermal simulations are shown in Figure 4.4. It is clear that with increasing latitude, more heating energy is required, while near the equator, more cooling energy is required. Overall, buildings (retrofitted or not) in the Netherlands and Sweden consume significantly more heating energy than those in Spain, while their cooling energy is negligible.

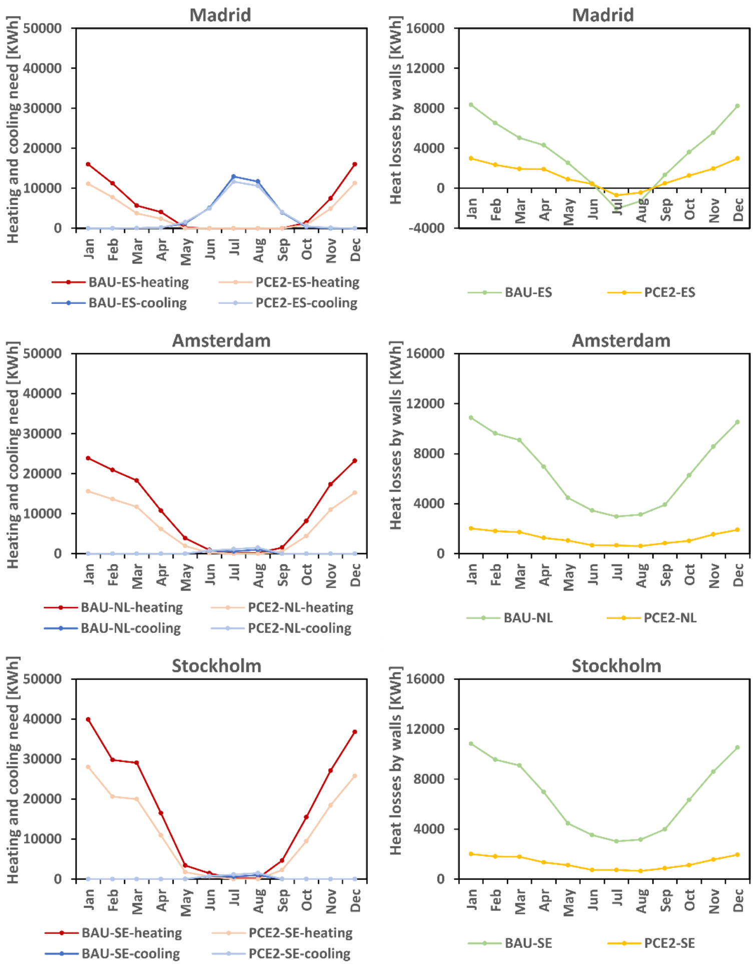

Figure 4.4 Distribution of the annual heating and cooling requirements and heat loss of virtual buildings in Madrid, Amsterdam, and Stockholm. BAU, business-as-usual; PCE-refurb, prefabricated concrete element for building refurbishment; ES, Madrid, Spain; NL, Amsterdam, Netherlands; SE, Stockholm, Sweden.

Based on the thermal dynamic simulations shown in Figure 4.4, the annual heating and cooling demand/floor area for both the BAU and VEEP scenarios in each region are listed in Table 4.5. Detailed information for modeling the heating and cooling demand is provided in Table A4.7.

Table 4.5 Annual heating and cooling demand/floor area for BAU and PCE-refurb scenarios. BAU: Business-as-usual; PCE-refurb: prefabricated concrete element for building refurbishment; ES: Spanish case; NL: Dutch case; SE: Swedish case.

	BAU-ES	PCE-ES	BAU-NL	PCE-NL	BAU-SE	PCE-SE
Heating need [kWh/(m ^{2}year)]	34.95	23.73	73.36	45.99	115.69	78.19
Cooling need [kWh/(m ^{2}year)]	19.89	18.81	1.31	1.98	1.55	1.99

4.2.3.3 Demolition phase

In the demolition phase, demolishing the VEEP PCE-refurbs and BAU traditional walls and disposing of the BAU traditional wall were not considered in the assessment. Further, the constituents of the PCEs (steel frame, concrete, and aerogel) were assumed to be recycled at this stage.

Specifically, steel is treated by collecting and then selling it directly on-site, and the environmental impact of the follow-up re-melting process was not considered in the study. The EoL lightweight concrete is recycled by crushing on-site with a crusher. Disposal options for fibrous materials include landfilling or incineration (Karatum et al. 2018). The aerogel is recyclable and reusable if it remains intact. Herein, it was assumed that the aerogel was recycled by DGR on-site. The reusability of PCE-refurb is examined in Section 4.1. Further recycling information is provided in the SI.

4.2.4 Life cycle impact assessment

The impact assessment step in an LCA characterizes the inventory results according to the target impact categories (Guinée et al. 2001). This study uses an LCA to quantify the GHG mitigation and energy saving potential of the PCE-refurb system. The "Global Warming (kg CO_2 eq)" from the "CML-IA, 4.4 issues, January 2015" database, and "OpenLCA LCIA methods 1.5.7" and Cumulative Energy Demand (MJ) (Frischknecht et al. 2003) from the "OpenLCA LCIA methods 2.0.3" database were selected as impact indicators. As an individual impact indicator is sufficient to estimate each type of payback period, the weighting scheme and normalization step were not considered in the LCA.

The cost category, time value of an investment, and cost results expression are discussed in the economic impact assessment (Zhang et al. 2019a). Herein, the LCC was performed from the homeowner's perspective. Therefore, the costing system only considers the real cash flows incurred by the owner, and environmental costs are excluded. Since 2020, the euro area has had zero interest (De Nederlandsche Bank 2020), which even reached a negative rate in developed areas, such as the Netherlands (Zhang et al. 2021a), thus, the interest rate was not considered for the payback estimation. The LCC result is expressed as the investment payback period.

4.3 Results

Section 4.3.1 presents the results of the embodiment, operation, and demolition phases of the LCA and LCC, which are converted into payback periods in Section 4.3.2.

4.3.1 Environmental and economic impacts of each life cycle phase

The results of energy use, GHG emissions, and the cost in each phase of all scenarios are presented in Figure 4.5. Figure 4.5 (a), (b), and (c) show similar trends, revealing that energy, emissions, and costs incurred in the demolition phase are nearly negligible. This is consistent with the conclusions of previous building life cycle assessment studies. Regarding the renewability of mixed energy, non-renewable sources, especially fossil energy, are the main energy sources in every phase. In the embodiment phase, the impacts of the Spanish case are less than those of the Dutch and Swedish cases because PCE-refurb-a uses less aerogel. Meanwhile, in the operation phase, all three cases show that PCE-refurb refurbishment reduces energy use, GHG emissions, and costs. As the operation impacts are expressed in annual values, they are not directly comparable to embodiment impacts. Thus, the results are aggregated into payback periods in Section 4.3.2.

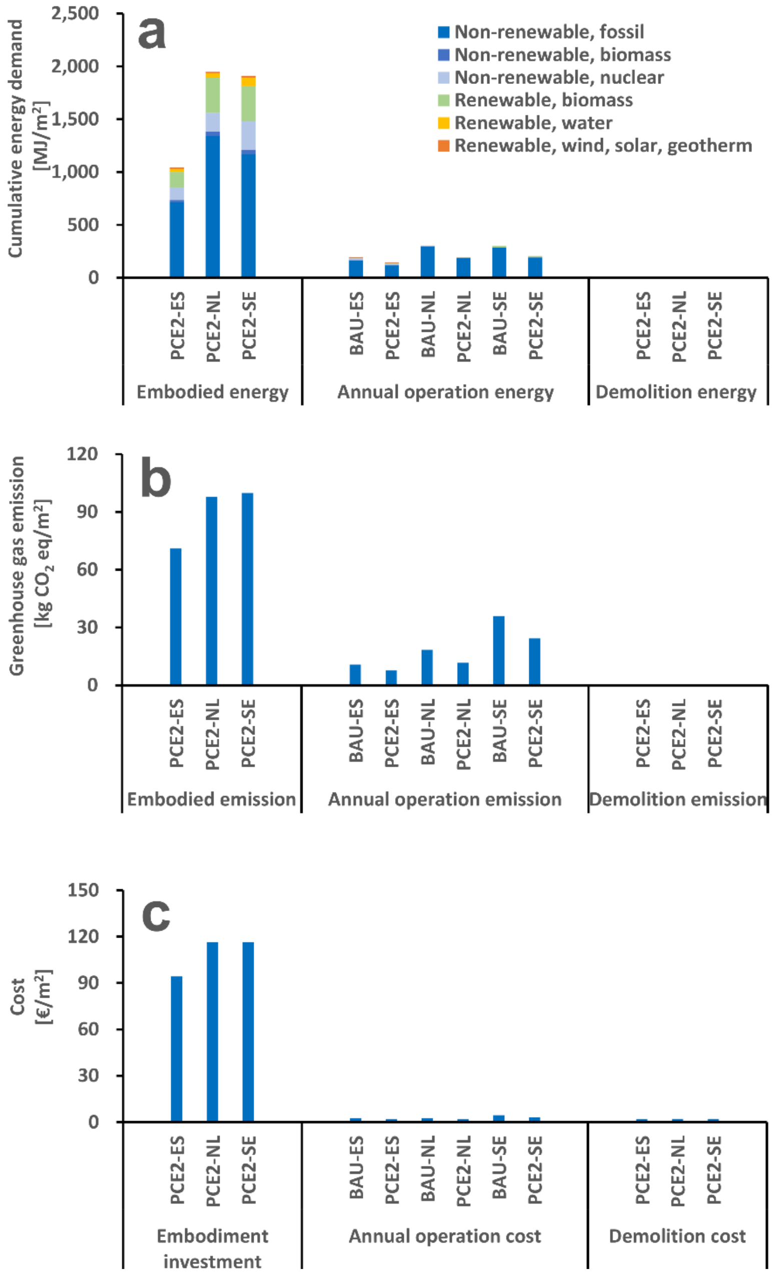

Figure 4.5 (a) Cumulative energy demand, (b) GHG emissions, and (c) cost in each phase of six scenarios. BAU: Business-as-usual; PCE-refurb: prefabricated concrete element for building refurbishment; ES: Spanish case; NL: Dutch case; SE: Swedish case.

4.3.2 Energy, carbon, and investment payback period

Using Eqs. (7), (8), and (9), the impact results were converted into payback periods. The energy, carbon, and investment payback periods of the PCE-refurb-ES, PCE-refurb-NL, and PCE-refurb-SE scenarios are shown in Figure 4.6. The differences between each energy payback period are subtle, ranging from 17.60 years for the Dutch case to 20.45 years for the Spanish case. However, disparities between the carbon payback periods are significant. The Spanish case has the longest carbon payback time of 23.33 years, the Dutch case has a middle-range payback of 16.78 years, while that in Sweden is considerably short, requiring only 8.58 years. These results indicate that the implementation of the PCE-refurb system in colder areas achieves shorter energy, carbon, and investment payback periods.

However, within a time span of 100 years, all the investment costs were not recouped. The Swedish scenario, which exhibited the best response, requires approximately 84 years to return the initial investment. Meanwhile, the investment payback periods of the Spanish and Dutch cases are more than 100 years, exceeding the average lifetime (120 years) of residential buildings in Europe (Sandberg et al. 2016). Therefore, financial payback will probably never be achieved if the PCE-refurb system is implemented in the Netherlands and Sweden.

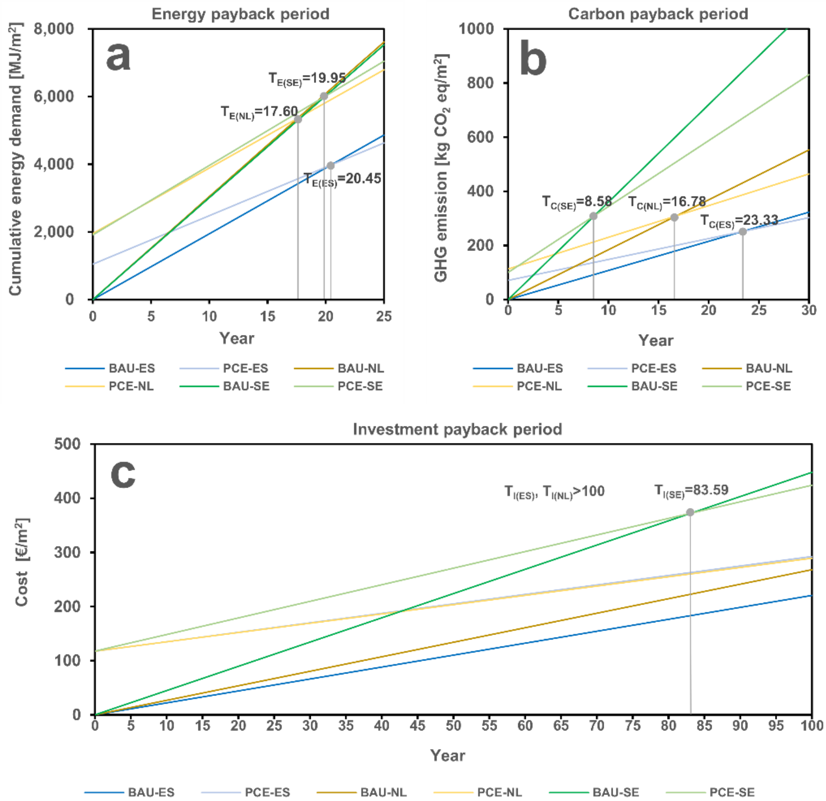

Figure 4.6 (a) Energy, (b) carbon, and (c) investment payback periods for Spanish, Dutch, and Swedish cases. BAU: Business-as-usual; PCE-refurb: prefabricated concrete element for building refurbishment; ES: Spanish case; NL: Dutch case; SE: Swedish case; T_E: energy payback period; T_C: Carbon payback period; T_I: investment payback period.

4.4 Discussion

This section discusses the impact of reusing PCE-refurbs, the application potential of the PCE-refurb system in an EU context, and the limitations of this study.

4.4.1 Influence of material circularity solutions on payback periods

Although the PCE-refurb system has relatively short energy and carbon payback periods, its economic payback is not achievable within the building's lifetime. To make the PCE-refurb system more cost-effective (EC 2012b, 2010), this section assesses how material circularity solutions, such as recycling and reuse, influence the payback periods, especially the economic payback period.

In the EoL stage of PCE-refurb, its components (concrete layer, aerogel layer, and steel frame) are assumed to be recycled. Recycling waste provides two functions: treating waste and producing secondary material. It can be seen from the LCC and LCA results that the demolition phase barely influences the payback period estimate as compared with the embodiment phase. However, the benefits of incorporating recycled materials into the production of green concrete and aerogels are not clear. Therefore, the influence of recycling was quantified using secondary raw materials in the embodiment phase. Consequently, the payback period of implementing a PCE-refurb that only contains primary raw materials was calculated.

Furthermore, as a non-structural element, one prominent merit of the PCE-refurb system is its reusability. Reusing PCE-refurb is realized by applying a dismantlable connecting and anchoring system that makes it possible to disassemble an intact PCE-refurb for reuse. As 90% of the cost of installation is the cost of labor (Zhang et al. 2021a), a dismantlable connecting and anchoring system can reduce labor costs through quick installation. Successful reuse can not only prevent the generation of waste but also avoid raw material consumption in the future production of PCE-refurbs. Therefore, the additional assessment in this section focuses on the extent to which reuse can avoid the additional PCE-refurb production in the embodiment phase. In this study, PCE-refurb reuse is modeled as (i) avoidance of 90% of the material and energy input in the embodiment phase for PCE-refurb manufacturing, and (ii) a reduction of the installation cost by 50% (Zhang et al. 2021a).

As shown in Figure 4.7, using secondary materials can slightly reduce all three payback periods. However, it does not shorten the investment payback periods in the Dutch and Swedish cases to less than 100 years. Nevertheless, reusing PCE-refurb decreases the payback period more than recycling. With reuse, the energy payback period of the three cases can decrease from approximately 20 years to 4.11–5.99 years, and the carbon payback period can be reduced by 3–11 years for all three cases. Regarding economic impacts, when reusing PCE-refurb, the Dutch and Spanish cases can achieve the investment payback at 42.97 years and 85.68 years, respectively. Meanwhile, the investment payback of the Swedish case can reach as low as 29.30 years.

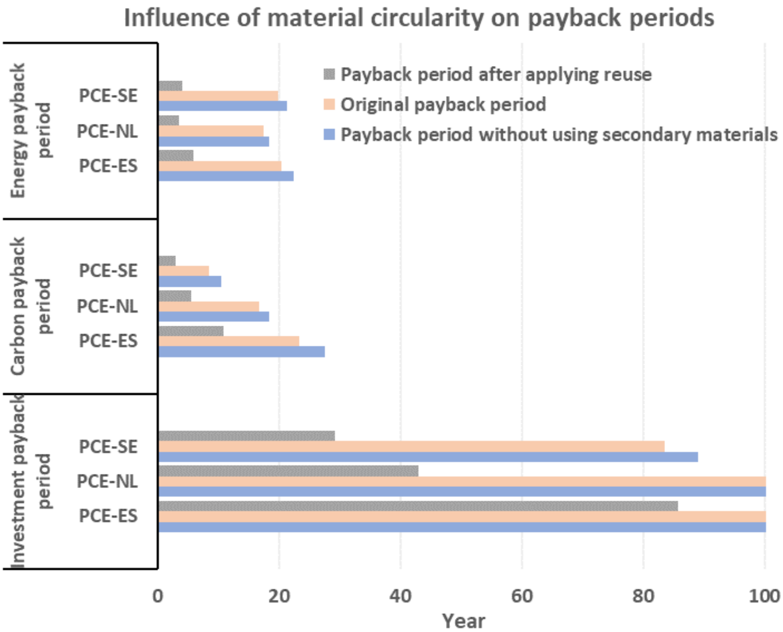

Figure 4.7 Influence of material circularity on energy, carbon, and investment payback periods. BAU: Business-as-usual; PCE-refurb: prefabricated concrete element for building refurbishment; ES: Spanish case; NL: Dutch case; SE: Swedish case.

4.4.2 Applicability of PCE-refurb system under EU context

This section evaluates the applicability of the PCE-refurb system in multiple EU member states. The system's energy use and associated costs and GHG emissions were modeled to directly relate to the thermal transmittance of building envelopes. In particular, the thermal transmittance considered were: the BAU wall (1.25 W/(m^2 K)), BAU wall retrofitted with VEEP PCE-refurb-a, and the BAU wall retrofitted with VEEP PCE-refurb-b. Each thermal transmittance was compared to the average-level building envelopes of EU building stock constructed at different times, as shown in Figure 4.8.

Sandberg et al. investigated 11 European countries and found that the average lifetime of European residential buildings was approximately 120 years (Sandberg et al. 2016). Thus, the potential building stock for refurbishment was considered to be constructed from 1900 to 2020. As shown in Figure 4.8, building stock constructed from 1900 to 1989 accounts for 75% of the total EU building stock. This stock has a higher thermal transmittance than that of the BAU traditional wall (illustrated by grey bar) used in this study. The thermal transmittance of the PCE-refurbs was even lower than the average thermal transmittance of the envelope of the buildings constructed after 2010, implying that the EU has a large potential market for the implementation of the PCE-refurb system for building energy renovation.

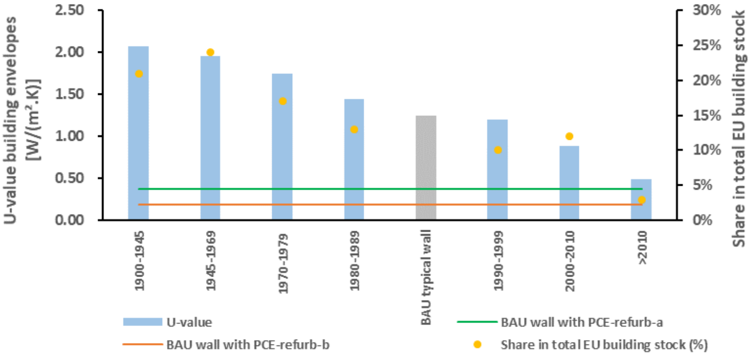

Figure 4.8 EU buildings' age and average thermal transmittance for building envelopes. Data source: (BPIE 2017); BAU: Business-as-usual; EU: European Union; PCE-refurb: prefabricated concrete element for building refurbishment; PCE-refurb-a: PCE-refurb containing a thin aerogel layer; PCE-refurb-b: PCE-refurb containing a thick aerogel layer.

However, the energy required for heating accounts for the largest share (approximately 70%) of building energy use (Economidou 2011). Considering the high importance of heating, heating demands in the BAU and PCE-refurb scenarios were compared with those of buildings in other EU member states that were constructed at different times, as shown in Figure 4.9. In general, the energy required for heating in each member state declined over time. With PCE-refurb implementation, the largest energy reduction potential was associated with the refurbishment of older buildings. In accordance with the EU's requirement for building energy efficiency, buildings constructed after 2000 require significantly less heating energy. For instance, the heating required in the PCE-refurb-ES scenario is higher than that of a building constructed after 1980 in a continental and Atlantic climate.

Southern EU member states, such as Spain and Italy, have lower heating demands because of their milder winters. Meanwhile, heating demands in northern European countries, such as Sweden, and Norway, remain relatively stable but generally require more heating than those of southern and western European countries. Note that because the heating energy demand of households depends on many factors, such as climatic characteristics, modeling methods, the efficiency of the heating system, and insulating levels of building facades, the results are shown in Figure 4.9 are not directly comparable. However, these results, to some degree, can demonstrate insights into the transitional trend of heating energy use in some EU member states and the application potential of PCE-refurb.

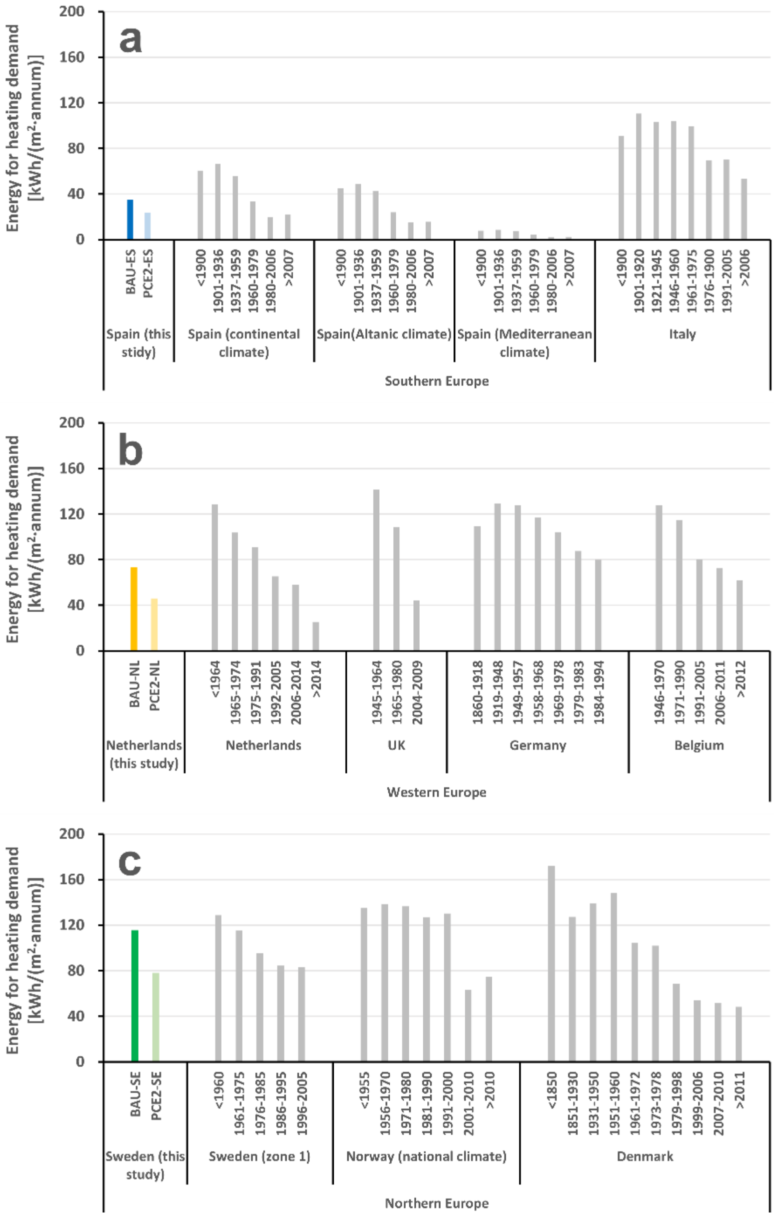

Figure 4.9 Annual/floor area energy required to meet heating demand of apartment blocks by construction year in (a) Southern Europe, (b) Western Europe, and (c) Northern Europe. Data collected from the TABULA database (TABULA 2017); BAU: Business-as-usual; ES: Spanish case; PCE-refurb: prefabricated concrete element for building refurbishment; NL: Dutch case; SE: Swedish case.

4.4.3 Limitations and outlooks

LCA and LCC building analyses are based on multiple simplifications and assumptions. Atmaca (2016) compiled the basic assumptions in building LCA and LCC analyses. Other than these assumptions, the specific limitations of this study are as follows.

First, the PCE-refurb lifetime was determined by considering the remaining lifetime of the building to be retrofitted. Because of the variance in building lifetimes, the payback method was applied. The lifetime prolongation of a product is a common method for reducing the life cycle environmental impacts (Aguilar-Hernandez et al. 2018). After refurbishment, the lifetime of a building is extended. However, because of the natural characteristics of the payback method (Yard 2000), it failed to consider the benefits of this prolonged lifetime.

Second, PCE-refurb implementation in buildings constructed in different periods will result in different payback periods. For example, renovating older buildings will lead to shorter payback periods than renovating newer buildings. Herein, only one BAU traditional wall with a thermal transmittance of 1.25 $W/(m^2 \cdot K)$ was selected as the benchmark for refurbishment, which does not provide comprehensive insights into building energy renovation.

Further, this study did not consider the time value of money. As the interest rates in the European area decreased to 0% in 2020, a steady-state costing system that did not consider interest rates was employed. Nevertheless, interest rates can considerably influence the results of an LCC (Swarr et al. 2011).

Finally, the payback assessment was conducted at a building element level in order to explore the environmental and economic performance of the PCE-refurb system. However, it is not clear if the PCE-refurb system can be scaled up to a regional level. For example, will EoL lightweight concrete generation be sufficient for massive building retrofitting? Thus, the dynamic building stock model should be combined with life cycle management to investigate the up-scaled benefits of PCE-refurb implementation at a regional level.

4.5 Conclusions

This study combined LCA and LCC analyses to determine the energy, carbon, investment payback periods for buildings renovated with the PCE-refurb system in the climatic context of three EU member states: Spain, the Netherlands, and Sweden. Two technological systems were considered: the BAU traditional wall and the BAU traditional wall retrofitted with PCE-refurb-a and PCE-refurb-b. In addition, a dynamic thermal simulation of the energy required to heat and cool a virtual residential apartment building was conducted.

The results show that the energy payback periods of the Spanish, Dutch, and Swedish cases were 20.45 years, 17.60 years, and 19.95 years, respectively. Meanwhile, the carbon payback periods for the three cases were 23.33 years, 16.78 years, and 8.58 years, respectively. However, the financial payback periods revealed that payback was unlikely to be achieved within the lifetime of a building, and only the Swedish case reached a payback period within 100 years (83.59 years). The impacts of material circularity on the payback period of PCE-refurb were also evaluated. The influence of recycling was quantified using secondary raw materials in the embodiment phase. However, the results show that using secondary materials in the PCE-refurb system only slightly reduces the payback periods. However, reusing the PCE-refurb can noticeably shorten the energy and carbon payback periods to 4.11–5.99 years and 3.03–10.82 years, respectively, for all three cases. Regarding cost, reusing the PCE-refurb reduced the payback period of the Swedish case to 29.30 years, and those of the Dutch and Spanish cases to 42.97 years and 85.68, respectively.

The applicability of VEEP PCE-refurb was evaluated by comparing the thermal transmittance and annual heating energy of EU buildings constructed at different times. The thermal transmittance of PCE-refurb-a and PCE-refurb-b were significantly lower than that of the average building envelope in the EU. Considering the lifetime, construction age, and energy performance of the EU building stock, the potential building stock for refurbishment was constructed from 1900 to 2020.

The integrated energy-carbon-investment payback analysis herein explored the life cycle stage of the PCE-refurb for building refurbishment. The results can be extrapolated to support the design and manufacture of sustainable building elements for building energy renovation in Europe. Further investigations will be conducted to integrate the life cycle management with the dynamic building stock model to address the question of region-level applicability and up-scaled ecological/financial benefits.

Acknowledgements

This work was supported by the European Union (grant number: 723582). Chunbo Zhang was supported by the China Scholarship Council (grant number: 201706050090). The authors also thank Arianna Amati from Rina Consulting, Ismo Tiihonen, Dr. Jaime Moreno Juez from Tecnalia, Dr. Abraham Teklay Gebremariam and Dr. Francesco Di Maio from the Delft University of Technology, Dr. Francisco Ruiz from Keey Aerogel, Frank Rens, and Eric van Roekel from Strukton BV for providing the data.

Appendix

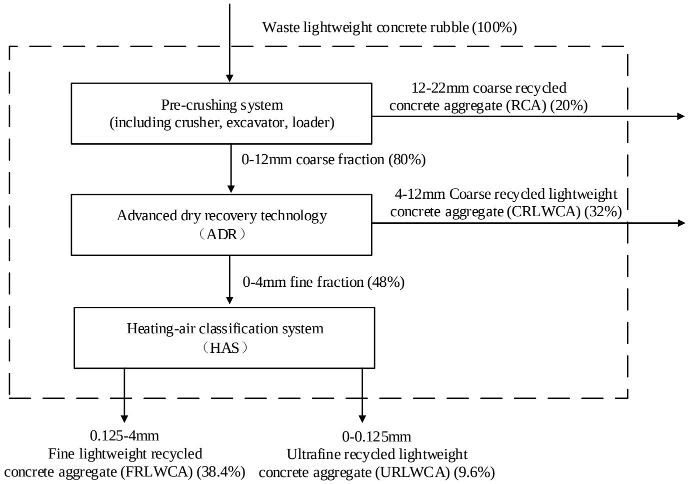

Figure A4.1 Mass balance of integrated ADR and HAS technology for recycling lightweight concrete waste

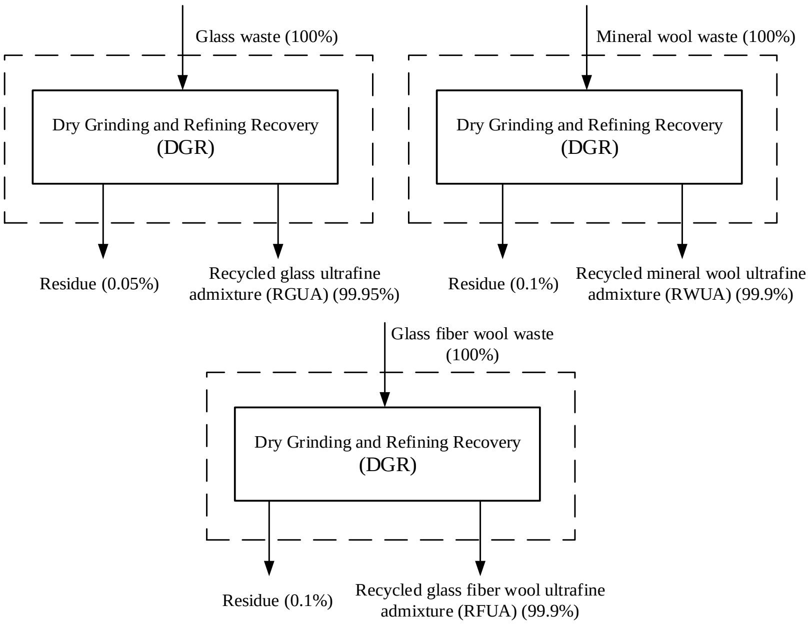

Figure A4.2 Mass balance of integrated ADR and HAS technology for recycling glass waste, mineral wool waste, and glass fiber wool waste

Table A4.1 Life cycle economic inventory

Items	Expenditure	Cost	Unit	Remark
Embodiment phase	Green lightweight concrete	93.82	€/t	The cost of green lightweight concrete for the PCE-refurb is expressed in an integrated unit cost. Data from (Zhang et al. 2021a)
	Green aerogel	10.00	€/0.01 m^2	The cost of green aerogel for the PCE-refurb is expressed in an integrated unit cost. Data from (Zhang et al. 2021a)
	Steel frame	537.00	€/t	Data from (Zhang et al. 2021a)
	Manufacturing of PCE-refurb	6.10	€/m^2	Data from (Zhang et al. 2021a)
	Installation of PCE-refurb	69.38	€/0.55 m^2	The cost incurred in the installation of 0.55 m^2 PCE-refurb for refurbishing per 1 m^2 floor area. Data from (Zhang et al. 2021a)
	Transport	0.1	€/km·t	Data from (Zhang et al. 2020a)
Operation phase	Electricity price in Spain	0.2239	€/kWh	Medium size household electricity prices in 2020 in Spain, data from the Eurostat (Eurostat 2020)
	Electricity price in the Netherlands	0.1427	€/kWh	Medium size household electricity prices in 2020 in the Netherlands, data from the Eurostat (Eurostat 2020)
	Electricity price in Sweden	0.1826	€/kWh	Medium size household electricity prices in 2020 in Sweden, data from the Eurostat (Eurostat 2020)
	Heating price in Spain	0.0360	€/kWh	Data from dataset "heat production, natural gas, at boiler modulating <100kW | heat, central or small-scale, natural gas | Cutoff, U - Europe without Switzerland" in Ecoinvent 3.4.
	Heating price in the Netherlands	0.0360	€/kWh	Data from dataset "heat production, natural gas, at boiler modulating <100kW | heat, central or small-scale, natural gas | Cutoff, U - Europe without Switzerland" in Ecoinvent 3.4.
	Heating price in Sweden	0.0382	€/kWh	Data from dataset "heat, at cogen, with supporting oil furnace 60%, 160kWe Jakobsberg, allocation exergy | heat, central or small-scale, Jakobsberg | Cutoff, U – RoW" in Ecoinvent 3.4.
Demolition phase	EoL concrete recycling	40.63	€/t	Data from (Zhang et al. 2021a)
	Aerogel recycling	75.18	€/t	Data from (Zhang et al. 2020a)
	Steel for sale	-133.12	€/t	Data from (Zhang et al. 2021a)

Table A4.2 Inventory of raw material for the fabrication per VEEP PCE-refurb. Note: For commercially confidential concerns, the exact amounts of constituents for lightweight concrete and aerogel are not given. Background process “market for transport, freight, lorry >32 metric ton, EURO3 | transport, freight, lorry >32 metric ton, EURO3 | Cutoff, U - GLO” in Ecoinvent 3.4 is selected for modeling transport of recycling facilities.

The material of the PCE	VEEP PCE-refurb-a	VEEP PCE-refurb-b	Sources of referred unit processes
Cement (CEM III/A)	/	/	Referred to “cement production, alternative constituents 6-20% | cement, alternative constituents 6-20% | Cutoff, U- Europe without Switzerland” in Ecoinvent 3.4
URLWCA	/	/	From the HAS
Lime sand	/	/	Referred to “market for sand | sand | Cutoff, U-GLO” in Ecoinvent 3.4
FRLWCA	/	/	From the HAS
CRLWCA	/	/	From the ADR
Expanded clay	/	/	Referred to “market for expanded clay | expanded clay | Cutoff, U-GLO” in Ecoinvent 3.4
Water	/	/	Referred to “market for tap water | tap water | Cutoff, U - Europe without Switzerland” in Ecoinvent 3.4
Masterglenium sky 526	/	/	Referred to “market for plasticizer, for concrete, based on sulfonated melamine formaldehyde | plasticizer, for concrete, based on sulfonated melamine formaldehyde | Cutoff, U-GLO” in Ecoinvent 3.4
Master X-Seed	/	/	Referred to “chemical production, organic | chemical, organic | global” in Ecoinvent 3.4
Sika AER 5	/	/	Referred to “chemical production, organic | chemical, organic | global” in Ecoinvent 3.4
Lime	/	/	Referred to “market for lime, packed | lime, packed | Cutoff, U - RoW” in Ecoinvent 3.4
RGUA	/	/	From the DGR
Aerogel	0.28 m^3	0.12 m^3	From VEEP aerogel production
Steel beams & welded nets	23.72 Kg	23.72 Kg	Referred to “reinforcing steel production | reinforcing steel | Europe” in Ecoinvent 3.4

Table A4.3 Energy usage related to VEEP PCE-refurb manufacturing. Note: Background process "transport, freight, lorry 3.5-7.5 metric ton, EURO3 | transport, freight, lorry 3.5-7.5 metric ton, EURO3 | Cutoff, U - RER" in Ecoinvent 3.4 is selected for modeling transport of secondary and virgin materials.

Energy carrier	Energy usage per unit of PCE-refurb	Sources of referred unit processes
Diesel	10.98 MJ/per PCE-refurb	"market for diesel, burned in building machine | diesel, burned in building machine | Cutoff, U - GLO" in Ecoinvent 3.4
Electricity	20.20 MJ/per PCE-refurb	"market for electricity, high voltage | electricity, high voltage | Cutoff, U – NL(/ES/SE)" in Ecoinvent 3.4
Natural gas	49.38 MJ/per PCE-refurb	"natural gas, burned in gas motor, for storage | natural gas, burned in gas motor, for storage | Cutoff, U - NL(/RoW)" in Ecoinvent 3.4

Table A4.4 Energy and material input for installation of 1 m^2 PCE-refurb. Note: Background process "transport, freight, lorry 3.5-7.5 metric ton, EURO3 | transport, freight, lorry 3.5-7.5 metric ton, EURO3 | Cutoff, U - RER" is selected for the transport of PCE-refurb.

Energy and material input	Amount	Sources of referred unit processes
Electricity [kWh]	4.21	"market for electricity, high voltage | electricity, high voltage | Cutoff, U – NL(/ES/SE)" in Ecoinvent 3.4
Brick [Kg]	62.85	"sand-lime brick production | sand-lime brick | Cutoff, U - RoW" in Ecoinvent 3.4
Reinforced concrete [Kg]	125.43	"concrete production 30-32MPa, RNA only | concrete, 30-32MPa | Cutoff, U - RoW" in Ecoinvent 3.4

Table A4.5 Thermal conductivity of materials for wall of building and PCE-refurb

Material	Thermal conductivity [W/(m K)]
Aerogel	0.0157
Green light weight concrete	0.5000
Perforated brick-faced	0.6560
Cement mortar	0.8000
Air gap (5cm)	0.2780
Hollow brick	0.2430
Gypsum	0.2500

Table A4.6 Thermal transmittance of the building envelope for dynamic thermal simulation with PCE-refurb in façade. Note: The thermal transmittance of the traditional wall before and after refurbishment with PCE-refurb systems were calculated based on ISO 6946 (ISO 2017b). BAU: Business-as-usual; PCE-refurb: prefabricated concrete element for building refurbishment.

Building elements	Thermal transmittance [W/(m²K)]
Floor	3.37
Roof	0.52
Glazing	1.30
BAU traditional wall	1.25
PCE-refurb-a (with thin aerogel layer)	0.52
PCE-refurb-b (with thick aerogel layer)	0.22
BAU traditional wall retrofitting with PCE-refurb-a	0.37
BAU traditional wall retrofitting with PCE-refurb-b	0.19

Table A4.7 Background processes in Ecoinvent for modeling household heating and cooling. Note: It is assumed individual air-conditioning is applied for cooling in these three areas. Energy demand for cooling is converted into household electricity use by a seasonal energy efficiency ratio. According to Norm DIN V 18599 (DIN 2011) for individual air conditioning (>12kW), the seasonal energy efficiency ratio is around 4.7. Therefore, 1 kWh of electricity can provide 4.7 kWh of cooling demand.

Heating and cooling input	Amount (BAU/PCE-refurb) [kWh/(m²annum)]	Sources of referred unit processes in Ecoinvent 3.4
Individual air-conditioning for household cooling in Spain	4.2300/4.0000	Data from dataset “market for electricity, low voltage | electricity, low voltage | Cutoff, U – ES”
Individual air-conditioning for household cooling in the Netherlands	0.2787/0.4213	Data from dataset “market for electricity, low voltage | electricity, low voltage | Cutoff, U – NL”
Individual air-conditioning for household cooling in Sweden	0.3298/0.4234	Data from dataset “market for electricity, low voltage | electricity, low voltage | Cutoff, U – SE”
Gas central heating system for household cooling in Spain	34.95/23.73	Data from dataset “heat production, natural gas, at boiler modulating <100kW | heat, central or small-scale, natural gas | Cutoff, U - Europe without Switzerland”
Individual condensing boiler for household heating in the Netherlands	73.36/45.99	Data from dataset “heat production, natural gas, at boiler condensing modulating <100kW | heat, central or small-scale, natural gas | Cutoff, U - Europe without Switzerland”
Fuel (oil) heating system for household heating in Sweden	115.69/78.19	Data from dataset “heat, at cogen, with supporting oil furnace 60%, 160kWe Jakobsberg, allocation exergy | heat, central or small-scale, Jakobsberg | Cutoff, U - RoW” in Ecoinvent 3.4

Table A4.8 Allocation scheme of processes with multifunctionality in the demolition phase in PCE-refurb scenarios. Note: The EoL lightweight concrete with be recycled on-site to produce RCA. The aerogel is assumed to be recycled by the DGR system. For recycling EoL lightweight concrete and aerogel, only the function waste treatment is considered. Based on the mass-based allocation method, 50% of the impact of recycling is supposed to be allocated to secondary material production.

Process name	Multifunctionality category	Functional flows	Allocation coefficient	Category
Pre-crushing process	Recycling	EoL lightweight concrete treatment	50%	Target service
		RCA production	50%	Non-target product
DGR process	Recycling	Aerogel treatment	50%	Target service
		RFUA/RWUA	50%	Non-target product

Chapter 5

Recycling potential in building energy renovation: a prospective study of the Dutch residential building stock up to 2050

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Abstract

Building energy and construction and demolition waste (CDW) are highly relevant but intertwined issues for the transition towards a carbon-neutral and circular built environment. Ongoing energy renovation uses an increasing number of emerging materials that pose a challenge for recycling. As a response, a novel technological system has been proposed to recycle CDW (including insulation mineral wool and lightweight concrete) for the manufacture of prefabricated concrete elements (PCEs) for use as fa çades for new (PCE-new) and retrofitting existing (PCE-refurbs) buildings. To explore how this novel system can improve recycling potential as part of building energy renovation efforts, the Dutch residential building stock was selected as a case study. Using a dynamic material flow analysis, we explore the supply-demand balance of secondary raw materials made from CDW (including normal-weight and lightweight concrete, glass, insulation mineral wool, and steel) and the secondary raw materials required for manufacturing PCEs in building energy renovation for the period 2015–2050. Our findings show that with advanced recycling technology, the secondary raw materials recovered from normal-weight concrete waste, glass waste, insulation mineral wool waste, and steel scrap will be more than sufficient to support the manufacturing of PCE-new walls, implying the possibility of closed-loop construction. However, for emerging materials such as lightweight concrete, the related waste will not be sufficient in the near future to meet the raw material demand for large-scale refurbishment with PCE-refurbs. Therefore, the Dutch case shows that the novel technology system offers a promising solution to CDW management problems in building energy renovation, but

primary raw materials will still be needed for the increased use of emerging materials such as lightweight concrete.

Keywords: material flow analysis (MFA), construction and demolition waste (CDW), prefabricated concrete element (PCE), recycling, energy renovation, building stock

Abbreviations

3R	Reduce, reuse, and recycle
ADR	Advanced dry recovery technology
CDW	Construction and demolition waste
CRLWCA	Coarse recycled lightweight concrete aggregate
CRSCA	Coarse recycled siliceous concrete aggregate
DGR	Dry grinding and refining system
EC	European Commission
EED	Energy Efficiency Directive
EoL	End-of-life
EPBD	Energy Performance of Buildings Directive
EU	European Union
FRSCA	Fine recycled siliceous concrete aggregate
FRLWCA	Fine recycled lightweight concrete aggregate
HAS	Heating-air classification system
MFA	Material flow analysis
ODYM	Open Dynamic Material Systems Model
PCE	Prefabricated concrete element
PCE-new	Prefabricated concrete element for new building construction
PCE-refurb	Prefabricated concrete element for existing building refurbishment
RFUA	Recycled fiber wool ultrafine admixture
RGUA	Recycled glass ultrafine admixture
SI	Supporting information
URSCA	Ultrafine recycled siliceous concrete aggregate
VEEP	European Union Horizon 2020 project “Cost-effective recycling of C&DW in high added-value, energy-efficient prefabricated concrete components for the massive retrofitting of our built environment”

5.1 Introduction

5.1.1 Potential of material circularity in building energy renovation

The building sector plays an essential role in resource depletion and waste management. The construction and operation of buildings in the European Union (EU) account for approximately half of all raw material consumption and generate approximately one-third of all waste (EC 2014c). It is generally recognized that a circular economy—with the principle of "Reduce, Reuse, and Recycle (3R)"—should become the basis of circular waste management and material cycles (Kirchherr et al. 2017). Legislative systems for waste management in the EU were established based on the 3R rule (Sakai et al. 2011). Following this, circular construction adopts the 3R rule for construction and demolition waste (CDW) management (Ghaffar et al. 2020). The essence of circular construction is to keep the components and materials of buildings in a closed loop and maximize their value as long as possible (Benachio et al. 2020). Closing the construction loop by recycling CDW is considered an effective means of improving material efficiency and reducing the adverse impacts of CDW.

A significant challenge, however, is that almost 75% of the overall European building stock is energy-inefficient (EC 2010). Considering the large amounts of greenhouse gasses emitted from the operation of buildings, improving energy efficiency is considered a critical strategy for achieving the EU's 2050 carbon-neutral goal (EZK 2019). The EU deems building energy renovation as a critical solution to shift to an energy-efficient and low-carbon built environment (Esser et al. 2019). Energy renovation is an umbrella concept that is acknowledged as a variety of interventions in buildings to deliver different degrees of energy savings (Economidou 2021). Moreover, employing advanced energy-efficient technologies in new construction also serves to establish a broader range of energy renovations (Esser et al. 2019). Accordingly, obsolete buildings in Europe are to be renovated or replaced to improve their energy performance, which increases the turnover of building materials as a result. Action 5 of Directive COM/2015/6317 (EC 2015b) calls for the "Development of new materials and technologies for the market uptake of energy efficiency solutions for buildings". In the context of extensive energy renovation in the EU, emerging high-performance materials such as insulating mineral wool, cellular and aerated glass, and lightweight concrete are increasingly used to reduce energy losses through building facades. Relative to 2015, the demand for such insulation materials is expected to increase in the EU by 3.5 % by 2027 (Pavel and Blagoeva 2018).

The demand for emerging materials to meet the demands of large-scale energy renovation not only increases the burden of resources but raises new problems surrounding their disposal. The main mineral-based insulating materials, such as stone wool and glass wool, are recyclable. One of the challenges for recycling is that insulation materials are lightweight, and the share of insulation also remains a small fraction of the total CDW. Therefore, the current EU weight-oriented CDW recovery targets and low disposal costs in some member states have no incentive to recycle insulating materials. In addition, the transport of insulation is costly because of its low weight-to-volume ratio. At the same time, concrete recycling is costly (Zhang et al. 2019a), hence the recycling of common (normal-weight) concrete waste has not been popularized in the EU, not to mention the recycling of emerging lightweight concrete. Therefore, establishing a cost-effective recycling solution is expected to greatly help close the loop of these emerging materials and support a more circular built environment.

5.1.2 The Netherlands as a case study

The Netherlands has the best practice of CDW treatment among EU member states and worldwide, with a recovery rate of 98% (CLO 2021). However, the Netherlands is also faced with the dilemma that the current destination for downcycled concrete—road base backfilling—is almost exhausted. Furthermore, extracting secondary raw materials from CDW via traditional wet-processing technologies for the building sector is costly (Zhang et al. 2019a, 2020b). For glass and insulation materials, it was reported that glass in CDW can be 100% recycled in the Netherlands; however, more than 60% of these insulation materials are landfilled and incinerated (Mulders 2013). Moreover, in the Netherlands, more than half of the raw materials (gravel, sand, and cement) used for concrete production are dependent on imports (Zhang et al. 2020b). Another crucial point is that a large portion of the dwellings in the Netherlands remain energy-inefficient (Staniaszek 2014). Therefore, the ongoing building energy renovation will likely further aggravate the demand for resources in the Netherlands.

One potential possibility for simultaneously moving towards a circular and low-carbon built environment could be considering CDW as feedstock for building energy renovation. In Europe, a novel technological system has been developed by the ‘VEEP’ EU project for recycling CDW in the manufacturing of green prefabricated concrete elements (PCEs), offering high insulation performance for the renovation of the residential building stock. An advanced dry recovery system (ADR) and heating air classification system (HAS) were developed to recycle normal-weight and lightweight concrete waste *in situ*; and a dry grinding and refining (DGR) system was designed to recover glass waste and insulating mineral wool on-site. Consequently, recycled materials are used to fabricate green PCEs. The green PCE solution is conceived both for new building envelope construction (PCE-new) and for existing building envelope refurbishment (PCE-refurbs). Details of the PCE system are presented in the Supporting Information (SI).

To investigate whether the integrated PCE system offers a promising solution for CDW recycling in building energy renovation in the Netherlands, and whilst considering the increased use of emerging materials, we sought to determine the extent to which CDW can be recycled as a feedstock in building energy renovation using the Dutch residential building stock as a case study. We apply material flow analysis (MFA) as a widely-used method for evaluating material metabolism by mass in the anthroposphere (Baccini and Btunner 2012). Among the three quantification approaches of MFA modeling defined by van der Voet (1996), dynamic MFA is usually applied to evaluate *ex-ante* and extrapolate trends. As we aim to unveil the recycling potential of emerging waste via an innovative recycling system, a dynamic MFA model was constructed for this study.

5.2 Literature review

To explore the recyclability of CDW, many studies have been conducted using the MFA approach. To position our study, a systematic literature review of MFA application in the field of CDW management was conducted. Relevant literature was searched for in the Web of Science Core Collection from 1945 to 2020 (search terms: TS = ("material flow analysis" OR "MFA") AND ("construction and demolition waste" OR "CDW")), yielding 32 results. After screening out five irrelevant studies, the remaining 27 studies are summarized in Table 5.1. It should be noted that this list is not exclusive.

Table 5.1 Literature related to material flow analysis of construction and demolition waste (CDW)

	Literature	Model	Region	Study aims/notes
1	(Lederer et al. 2020)	Static, 2014	Vienna	MFA was used to quantify how waste reduction, reuse, and recycling of mineral CDW from buildings and infrastructure can contribute to reducing the demand for raw material imports for construction minerals.
2	(Zhang et al. 2020b)	Static, 2015, 2025	The Netherlands	Quantifies how technological innovation could contribute to upgrading waste concrete treatment from downcycling to recycling.
3	(Marcellus-Zamora et al. 2020)	Static, 2007–2017	Philadelphia, USA	Characterizes the flow of recoverable CDW, quantify aggregated CDW diversion, and evaluate recycling patterns for a portion of the CDW.
4	(Gassner et al. 2020)	Dynamic, 1990–2015	Vienna	Estimation of material turnover of urban transport systems, including both infrastructure and vehicles.
5	(Wu et al. 2020)	Static, 2007–2017	Australia	Quantifies the compositions and generation of CDW and reveals its cross-regional mobility.
6	(Noll et al. 2019)	Dynamic, 1971–2016	Samothraki, Greece	Strategy design on reducing, reusing, and recycling CDW on islands where waste treatment options are limited.
7	(Tangtinthai et al. 2019)	Static, 2012	Great Britain, Thailand	Examines relevant policies on how to achieve more sustainable management of concrete and cement.
8	(Heeren and Hellweg 2019)	Dynamic, 2015–2055	Switzerland	Used a bottom-up probabilistic modeling approach to determine material stocks in Swiss residential buildings and associated carbon emissions.

9	(Jain et al. 2019)	Dynamic, 2012–2050	India	A bottom-up approach to explore how CDW generation rate varies across different classes of cities.
10	(Zhang et al. 2018)	Static, 2015	Chongqing, China	Explores the carbon mitigation and land-use reduction of different strategies for concrete waste management.
11	(Suzuki et al. 2018)	Dynamic, 1981–2015	Japan	Investigates the potential fate of engineered nanomaterials in the construction sector.
12	(Miatto et al. 2017c)	Dynamic, 1905–2015	USA	A bottom-up stock-driven model to evaluate long-term metabolism, and materials accumulated in the road network.
13	(Miatto et al. 2017a)	Dynamic, 1970–2010	Worldwide	Estimates the extraction of nonmetallic minerals and associated uncertainty about consumption by different sectors.
14	(Schiller et al. 2017a)	Dynamic, 1919–2010	Germany	Analyzes and quantifies the entire material cycle of bulk nonmetallic mineral building materials by considering the use of recycled aggregates in concrete building elements.
15	(Condeixa et al. 2017)	Dynamic, 2000–2010	Rio de Janeiro, Brazil	A bottom-up approach to assess the materials in use and further flows of CDW from the residential building stock.
16	(Lockrey et al. 2016)	Static, 2002–2025	Hanoi, Vietnam	Estimates construction and demolition concrete waste in Hanoi and Vietnam.
17	(Li et al. 2016)	Static, did not specify a time	A six-story building in Hebei, China	Proposes a model at a project level to quantify construction waste for building construction projects.
18	(Dahlbo et al. 2015)	Static, did not specify a time	Product-level, Finland	A combined method to holistically evaluate the environmental and economic performance of the CDW management system.
19	(Wiedenhofer et al. 2015)	Dynamic, 2004–2009	EU25	Quantifies stocks and flows for nonmetallic minerals in residential buildings, roads, and railways.
20	(Hu et al. 2013)	In general	In general	Examines concrete recycling as a case study to illustrate a framework of life-cycle sustainability analysis combining MFA with life-cycle analysis.
21	(Knoeri et al. 2013)	Static, did not specify a time	Product level	Provides a product- comparison of conventional concrete and concrete with recycled aggregates.
22	(Hoque et al. 2012)	Static, 2001	Catalonia, Spain	Analyzes resource consumption in the construction sector.
23	(Chong and Hermreck 2011)	Static, 2005, 2006	Las Vegas, Kansas, Portland, Seattle, USA	Quantifies energy demand for transporting and recycling construction steel.
24	(Hu et al. 2010b)	Dynamic, 1949–2050	Beijing, China	Quantifies the CDW in Beijing to support strategic waste management.
25	(Kapur et al. 2009)	Static, 2000–2004	USA	Develops a country-level stock and flow model to investigate the life-cycle of cement.
26	(Weil et al. 2006)	Static, did not specify a time	Product-level, Germany	A micro-level comparison of the environmental benefits of (per m^3) of concrete with or without recycled aggregates.
27	(Bertram et al. 2002)	Static, 1994	16 European countries	Copper mass balance assessment for waste management in multiple European countries.

Based on the literature review, MFA has been applied to investigate CDW at the product level (18, 21, and 26), building project level (17), regional level (1, 2, 3, etc.), and global level (13). The method has also been used in combination with life cycle assessment (10, 25, etc.) and life cycle costing (18) to evaluate the financial and environmental impact of CDW management. Most previous studies have focused on non-metallic mineral wastes such as concrete, whereas the recycling potential of emerging materials and renovation waste has not yet been examined.

Based on this review, regional-level dynamic MFA was selected for this study. Therefore, to fully consider the impact of the emerging waste (insulation mineral wool and lightweight concrete), we developed a dynamic MFA model to evaluate the supply-demand balance between the secondary raw materials made from CDW and the raw materials required for the manufacturing of PCEs for the period 2015–2050. Moreover, we explored how waste from energy renovation affects the mass accounting of CDW using dynamic MFA.

5.3 Methods and data sources

5.3.1 Conceptual framework

The estimation of the dynamics of the building stock was realized via a top-down modeling method based on gathered socioeconomic data. A prospective approach was applied because MFA aims to explore the “what-if scenario” of the future. As the waste flow was assumed to be determined by the change in stock, a stock-driven approach was used. Therefore, the MFA model applied to the Dutch case study presents a prospective, top-down, stock-driven model.

Müller (2006) developed a stock-driven model for estimating the diffusion of concrete in residential stock in the Netherlands from 1900 to 2100. Based on Müller’s modeling approach, we applied a three-layer stock dynamics model, as illustrated in Figure 5.1. The dwelling layer is the key layer for steering the turnover of the building stock. As part of the dwelling layer, data on population, floor area per capita, and building lifetime probability distribution were collected to calculate the construction, renovation, and demolition floor area for each year of study. Within the PCE layer, a geometry coefficient was used to determine the demand of PCE-new and PCE-refurbs per floor area of building construction and renovation. The outflow of the end-of-life (EoL) PCE was not considered because it is assumed to occur much later than the temporal scope of the accounting system. Finally, under the material layer, the waste intensity, material intensity, and recycling rate were investigated to understand the supply and demand conditions of secondary raw materials.

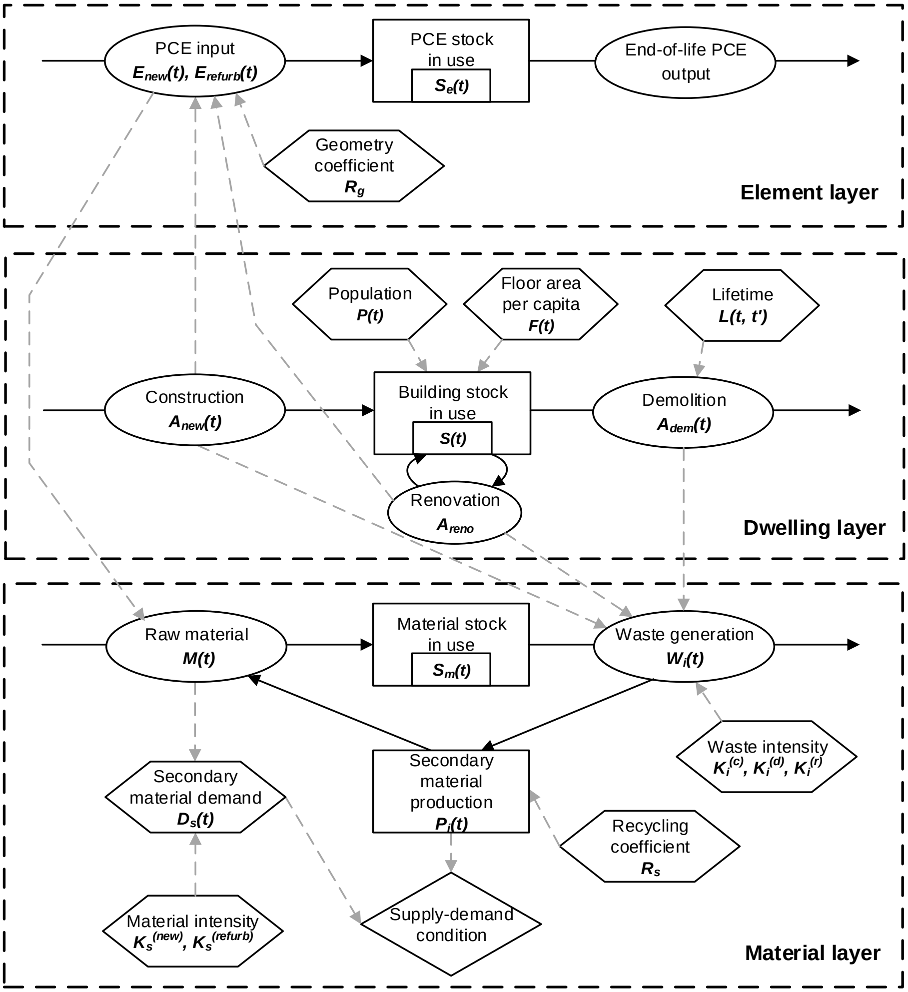

Figure 5.1 Conceptual framework of a three-layer dynamic material flow analysis model. Note: hexagons indicate drivers and determinants, rectangles represent processes, ovals with solid lines denote flows, and dashed lines with arrows denote influences between two variables.

5.3.2 Goal and scope definition

The goal of the MFA model was to estimate material inflows and outflows of the residential building stock of the Netherlands to support decision-making on the potential of material circularity in prefabrication-based building energy renovation. The geographical boundary of the assessment was the border of the Netherlands. The temporal scope of the assessment was from 2015 to 2050. Non-residential buildings, such as hospitals and schools, were excluded. The anthropogenic cycle of building materials is generally considered to consist of five life phases, as shown in Figure 5.2; we focused on the recycling phase only, in which CDW was assumed to be reprocessed to manufacture secondary raw materials. For waste and materials to be tracked, we focused on the target CDW and secondary raw materials, as shown in the dotted box in Figure 5.2.

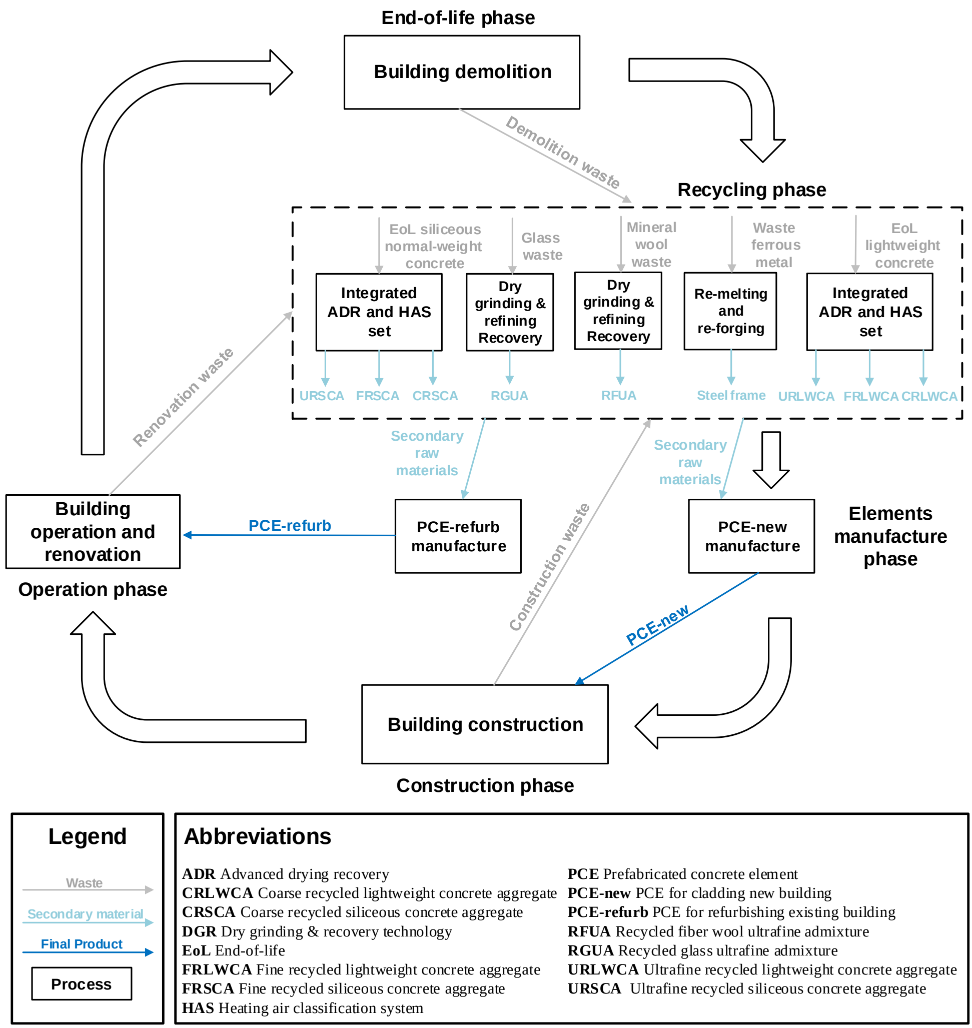

Figure 5.2 System boundary of the material flow analysis. Note: wastes and materials to be tracked in the system are shown in the dotted box

5.3.3 Characterization of parameters

Population

Historical population from 1900 to 2015 (CBS 2019a) and forecasted population from 2015 to 2050 (CBS 2019b) data were obtained for the Netherlands as shown in Figure 5.3(a).

Residential floor area per capita

To the authors' knowledge, there are no statistics available on the historical and forecasted residential floor area per capita in the Netherlands. Müller simulated the floor area per capita in the Netherlands from 1900 to 2100 based on the United Nations' average value (Müller 2006). Here, we used the standard flood area per capita scenario from 1900 to 2050, as shown in Figure 5.3(b).

Construction, demolition, and renovation

Computation of the construction and demolition floor area was based on the concept of building stock dynamics in Figure 5.1 and an operable Python-based framework called the "Open Dynamic Material Systems Model (ODYM)" developed by Pauliuk and Heeren (2020). We extended the ODYM using an additional renovation function, where the residential building stock was calculated using Eq. (1):

$$S(t) = P(t)F(t) \quad (1),$$

where *S(t)* is the gross residential floor area of year *t* (1900, 2050); *P(t)* is the population of year *t* (1900, 2050); and *F(t)* is the residential floor area per capita in year *t* (1900, 2050).

The newly constructed floor area for year *t* is given by Eq. (2):

$$A_{new}(t) = S(t) - S(t-1) + A_{dem}(t), \quad (2)$$

where $A_{new}(t)$ is the new construction floor area of year *t* (1900, 2050) and $A_{dem}(t)$ is the demolition floor area in year *t* (1900, 2050).

The annual demolition rate was modeled through Eqs. (3)–(6). *L(t, t')* in Eq. (4) is a probability distribution function that presents the probability that buildings built in year $t' < t$ will be demolished in year t. The lifetime distributions of buildings are commonly estimated with normal, log-normal, and Weibull distributions, although no evidence is available to indicate which probability distribution is best suited for dynamic stock modeling (Müller 2006; Miatto et al. 2017b). Therefore, we used a modified Weibull statistical distribution to approximate the lifetime of residential buildings in the Netherlands. The Weibull random variables t and t' are characterized by the shape parameter k and a scale parameter λ. The shape parameter k = 2.95 is specified according to the average level of buildings in Western Europe (Deetman et al. 2020). The scale parameter λ = 134.48 was determined as the average lifetime of Dutch residential buildings (E_{LF}), as shown in Eq. (5), in which $\Gamma(x)$ represents the gamma function as presented in Eq. (6). Müller (2006) compared different lifetimes for the Dutch building stock, specifically short (60 years), medium (90 years), and long (120 years). Deetman et al. (2020) found that estimations only match statistical data when a high average lifetime (130 years) of buildings in Western Europe is assumed. Thus, the average lifetime was assumed to be 120 years in our building stock modeling, as adopted by Sandberg et al. (2016). The resulting lifetime distribution of residential buildings in the Netherlands is shown in Figure 5.3(c).

$$A_{dem}(t) = \int_{t_0}^{t} A_{new}(t')L(t,t')\,dt', \tag{3}$$

$$L(t,t') = \begin{cases} k\lambda^{-k}(t-t')^{k-1}e^{-\frac{(t-t')^k}{\lambda^k}}, & t' < t \\ 0, & t' \geq t \end{cases}, \tag{4}$$

$$\lambda = E_{LF}/\Gamma(1+\tfrac{1}{k}), \tag{5}$$

$$\Gamma(x) = \int_0^\infty t^{x-1}e^{-t}dt, \tag{6}$$

The assumptions for the renovation of obsolete buildings were as follows: 1) Renovation started from t = 2015 to 2050; 2) buildings to be retrofitted were constructed from t'= 1900 to 2014; buildings constructed after 2014 were not retrofitted; 3) buildings to be renovated were separated from those buildings to be demolished, i.e., buildings that are supposed to be demolished by 2050 will not be renovated; 4) renovation floor area per annum was calculated based on Eq. (7). The gross floor area for renovation was equally allocated to each year between 2015 and 2050, amounting to an approximately 17 million m^2 floor area to be renovated per annum; and 5) for those buildings to be renovated, older buildings were preferentially renovated. The simulation results of the construction inflow, demolition outflow, and floor area for the renovation of each year are shown in Figure 5.3(d), and the dynamics of the building stock specified by construction cohorts are presented in Figure 5.3(e). The renovation of buildings in different construction periods (cohorts) is shown in Figure 5.3(f).

$$A_{reno} = \frac{S(2050) - \sum_{t'=2015}^{2050}\{A_{new}(t') - \sum_{t=t'}^{t=2050}[\int_{t'}^{t} A_{new}(t')\cdot L(t,t')dt']\}}{(2050-2015)+1}, \tag{7}$$

where A_{reno} is the renovation floor area in year t (2015, 2050).

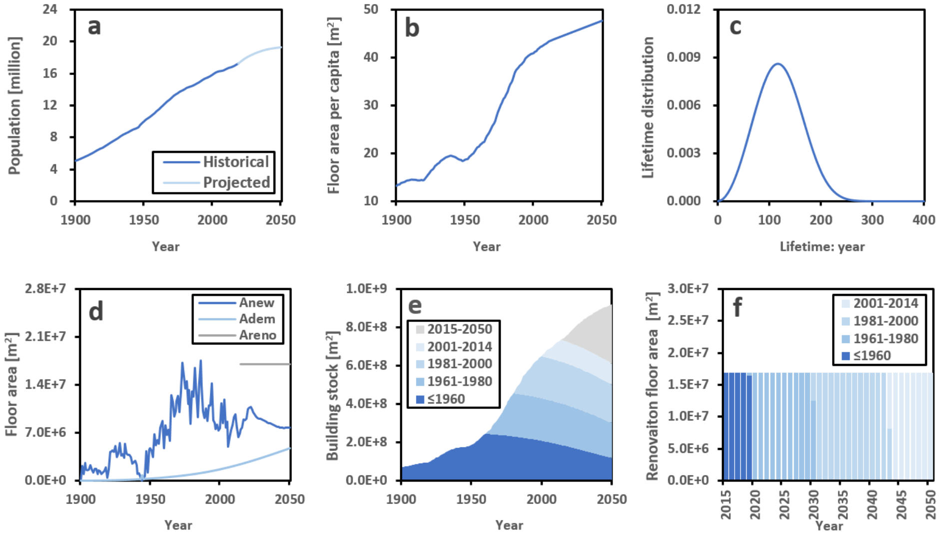

Figure 5.3 Estimation of parameter functions and simulation results for the Netherlands: (a) presents the historical and forecast population from 1900 to 2050; (b) demonstrates residential floor area per capita from 1900 to 2050; (c) shows the Weibull statistical distribution for modeling lifetime of dwellings; (d) presents construction, demolition, and renovation floor area of each year; (e) shows the dynamics of the building stock specified by construction cohorts; and (f) illustrates the vintage cohort of buildings to be renovated each year.

Demand of PCEs per floor area

The Agentschap NL of the Ministry of the Interior and Kingdom Relations in the Netherlands publishes data on the type and construction vintage of residential buildings. The Agentschap NL (2011) categorized the Dutch residential buildings into detached houses, semi-detached houses, terraced houses, maisonette houses, and apartments; and gave the number of homes and average floor area of each building type until 2005. The numbers of each type of house in different construction cohorts are presented in Table A5.1. The floor area of those examples is used to represent the average floor area of each type of house in the Netherlands as shown in Table A5.2. Based on the number (in Table A5.1) and average floor area (in Table A5.2) of each house, the stock share of each house (up to 2005) in gross residential stock share can be estimated, as shown in Table A5.3. The modified stock share of each building typology based on the Agentschap NL report is shown in Table 5.2. Details of the modifications are provided in the SI. We assumed that the share (m^2) of each housing category remains constant until 2050.

The required amount of PCEs (m^2) can be calculated based on the external wall surface and floor area of a building. To estimate the requirement of PCEs, we introduced a geometry coefficient (R_g) to denote the ratio of the gross external wall surface compared to the gross floor area of a building. The TABULA database contains comprehensive information about the typology of residential buildings for 21 European states. Yang et al. (2020) used this database to measure the geometric information of buildings in Leiden, the Netherlands. Here, R_g data for the different types of buildings were collected from the TABULA database (2017), as shown in Table 5.2.

Table 5.2 Ratio of external wall surface and floor area for different types of houses in the Netherlands

Building type	Stock share ($R_{stock}^{(bt)}$)	Building demonstrator	Reference code in the TABULA database	Construction vintage	External wall surface [m²] ($S_{wall}^{(bt)}$)	Floor area [m²] ($S_{floor}^{(bt)}$)	Geometry coefficient ($R_g^{(b)}$)
Detached house	15.98%		NL.N.SFH.03.Deta	1975–1991	144.00	169.00	0.85
Semi-detached house	11.39%		NL.N.SFH.01.Semi	Before 1964	97.80	121.00	0.81
Terraced house	33.60%		NL.N.TH.01.Mid1964	Before 1964	42.30	96.00	0.44
Maisonette	24.38%		NL.N.AB.02.Mai	1965–1974	598.40	1355.00	0.44
Apartment	14.65%		NL.N.AB.02.Por	1965–1974	951.40	1562.00	0.61

The weighted geometry coefficient of the Dutch building stock is R_g = 0.57, which was calculated using Eq. (8):

$$R_g = \sum[\left(\frac{S_{wall}^{(bt)}}{S_{floor}^{(bt)}}\right) R_{stock}^{(bt)}], \tag{8}$$

where R_g is the weighted geometry coefficient of the Dutch building stock, $S_{wall}^{(bt)}$ is the gross external wall surface of a certain type of reference building, $S_{floor}^{(bt)}$ is the gross floor area of a certain type of reference building, and $R_{stock}^{(bt)}$ is the gross stock of a certain building type.

Generation of CDW

According to statistics (Gálvez-Martos et al. 2018), construction of per m² floor area of a new building generates from 18 to 33 kg of waste concrete and demolition of per m² floor area of an obsolete building generates 840 kg of waste concrete. The median 26 kg is selected as the waste intensity coefficient for waste concrete for construction in the Netherlands ($K_{concrete}^{(c)}$). For demolition, from the global construction materials database developed by (Marinova et al. 2020), concrete intensity for different types of houses in Western Europe is in Table A5.4. Based on the stock share of each house type in the Netherlands in Table A5.3, the weighted concrete intensity is ($K_{concrete}^{(d)}$=) 902 kg per m^2 demolished floor area.

Zhang et al. (2020c) explored the composition of CDW generated in the Netherlands based on national waste flow statistic, share of concrete, ferrous metal, glass, and insulation in CDW by weight is 64.02%, 10.23%, 0.32%, and 0.07%. Thus the generation rate of glass and insulation per construction and demolition floor area can be calculated accordingly: $K_{ferrous}^{(c)}$ = 4.15 kg/m^2; $K_{glass}^{(c)}$ = 0.13 kg/m^2; $K_{insulation}^{(c)}$ = 0.03 kg/m^2; $K_{ferrous}^{(d)}$ = 144.08 kg/m^2; $K_{glass}^{(d)}$ = 4.51 kg/m^2; $K_{insulation}^{(d)}$ = 0.99 kg/m^2.

Waste from renovation is also considered in this study. Villoria Sáez (2018) estimated the ratio of renovation waste generation per renovation floor area for multiple retrofitting strategies. Amongst those strategies, the renovation by cladding system yields 30.62 kg of CDW per m^2 renovaiton floor area, and associated waste intensities are: $K_{ferrous}^{(r)}$ = 0/m^2; $K_{glass}^{(r)}$ = 0/m^2; $K_{insulation}^{(r)}$ = 0.04 kg/m^2; $K_{concrete}^{(r)}$= 28.50 kg/m^2.

The thermal insulation market in Europe in 2014 is shown in Table A5.5 (Pavel and Blagoeva 2018). Based on the market share, we assumed the insulating mineral wool waste (glass wool and stone wool) account for 58% of the gross insulation waste by weight.

CDW yielded from construction, demolition, and renovation activities were estimated using Eq. (9):

$$W_i(t) = A_{new}(t)K_i^{(c)} + A_{dem}(t)K_i^{(d)} + A_{reno}(t)K_i^{(r)}\ , \qquad (9)$$

where $W_i(t)$ is the waste i generated in year t; $A_{new}(t)$ is the new construction floor area of year t; $A_{dem}(t)$ is the demolition floor area in year t; $A_{reno}(t)$ is the renovation floor area of year t; $K_i^{(c)}/K_i^{(d)}/K_i^{(r)}$ is construction/demolition/renovation waste intensity coefficient: the amount of waste i generated per construction/demolition/renovation floor area.

As an emerging material, lightweight concrete is not yet widely used in Europe (Thienel et al. 2020). The average lifespan of buildings in the Netherlands was assumed to be 120 years, and buildings to be demolished were mainly constructed around the 1900s. Thus, most concrete waste in the CDW is normal-weight concrete waste. Therefore, we conservatively assumed that the gross concrete waste contained 1% lightweight concrete (by weight). According to the insulation material market in Europe, insulating mineral wool accounts for 58% of the insulation material by weight (Pavel and Blagoeva 2018). Based on these assumptions, the estimated amounts of concrete waste, glass waste, ferrous waste, and insulation waste generated between 2015 and 2050 are presented in Figure 5.4.

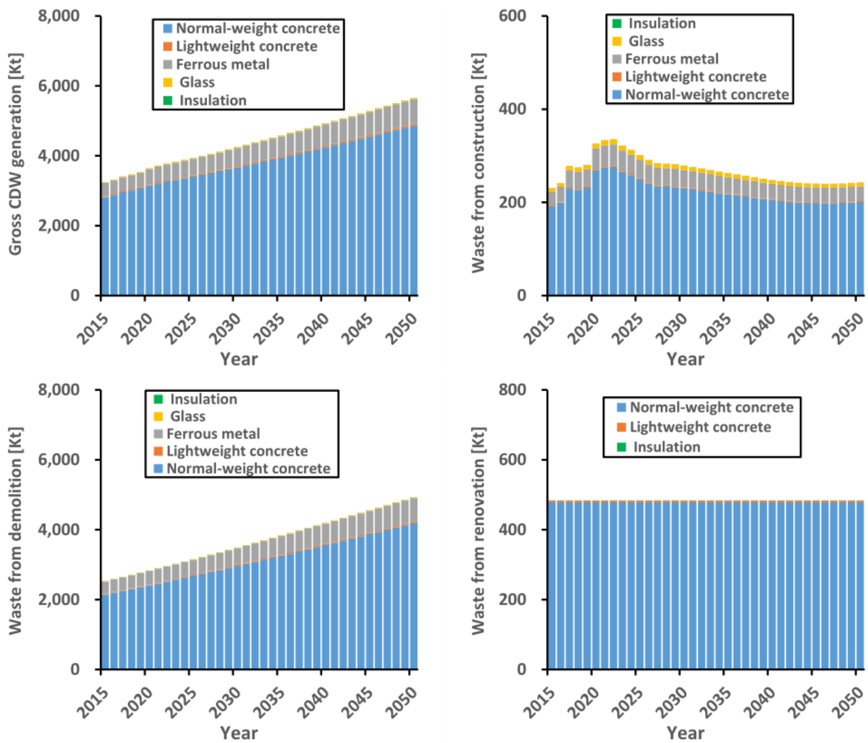

Figure 5.4 Estimated construction and demolition waste (CDW) generated from the construction, demolition, and renovation in the Netherlands for the period 2015–2050

Production of secondary raw materials

The recycling rate for steel is (R_{steel}=) 90% (Ruukki 2011). The data on the mass balance of ADR and HAS to process siliceous normal-weight concrete waste and lightweight concrete waste were collected from Strukton's CDW recycling site in Hoorn, the Netherlands. The siliceous concrete waste and lightweight concrete waste (around 14 tons for each type) were fed to the integrated ADR and HAS facility and the mass of the output streams were measured during experimental trials on site. The R_s for recycling siliceous normal-weight concrete waste are (Zhang et al. 2020a): R_{URSCA}=6.4%; R_{FRSCA}

= 25.6%; R_{CRSCA} = 68%; R_s related to lightweight concrete recycling are (Zhang et al. 2021a): R_{URLWCA}=9.6%; R_{FRLWCA} = 38.4%; R_{CRLWCA} = 52%.

The data on the mass balance of the DGR system was collected from experimental trials in Helsinki, Finland. The input mass of glass waste and insulating mineral wool waste (100 Kg for each) and the output mass of secondary raw materials were measured. The residue contained in the RGUA and RFUA is less than 1% by weight. Thus the recycling coefficient for RGUA and RFUA is assumed 100%, which means R_{RGUA} = 100% and R_{RFUA} = 100%.

The production of secondary raw materials was calculated according to Eq. (10):

$$P_s(t) = W_i(t)R_s \,, \tag{10}$$

where $P_s(t)$ represents the amount of secondary raw material made from waste i in year t, and R_s denotes the recycling coefficient of production of secondary raw material from waste. The potential productive capability of secondary raw materials via recycling waste is presented in Figure 5.5.

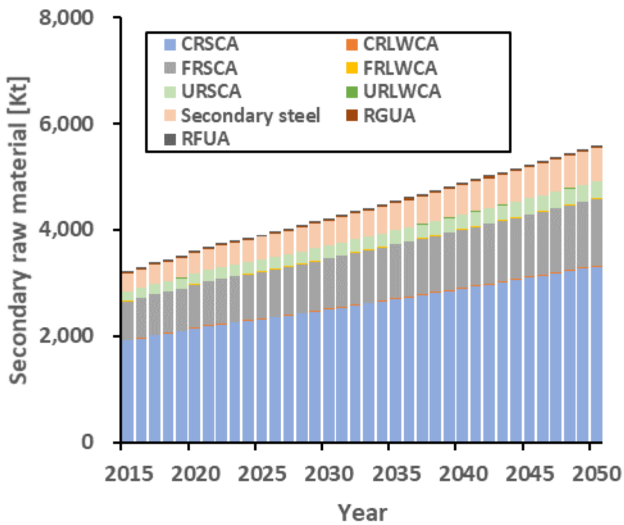

Figure 5.5 Potential productive capability of secondary raw materials in the Netherlands for the period 2015–2050

Demand for secondary raw materials

Based on the formulation of green concrete and aerogel, secondary raw materials used in green normal-weight and lightweight concrete, green aerogel, and steel frame for the

PCE-new and PCE-refurb are listed in Table A5.6. Due to commercially confidential concerns, the detailed recipes for concrete and aerogel is not disclosed.

Based on the formulation of VEEP PCE-new and PCE-refurb in Table A5.6 and the weighted geometry coefficient, the secondary raw materials demanded per m^2 construction and renovation floor area are presented in Table A5.7.

The secondary raw material demand of PCE-new and PCE-refurbs were computed using Eq. (11):

$$D_s(t) = A_{new}(t)K_s^{(new)} + A_{reno}K_s^{(refurb)}, \tag{11}$$

where $D_s(t)$ is the secondary raw material demand in year t, $A_{new}(t)$ is the construction floor area of year t, A_{reno} is the renovation floor area of each year, $K_s^{(new)}$ is the secondary raw material demand of PCE-new per construction floor area; and $K_s^{(refurb)}$ is the secondary raw material demand of PCE-refurbs per renovation floor area. The data sources for each parameter are presented in the SI. Based on these calculations, the total secondary raw materials required for the implementation of the PCE-new and PCE-refurbs are presented in Figure 5.6.

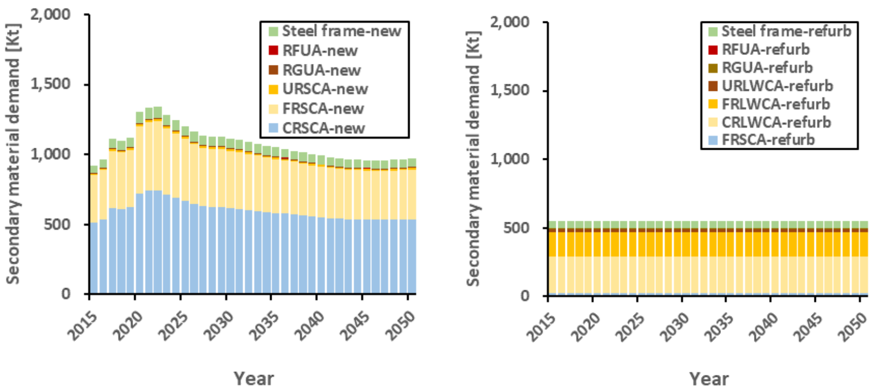

Figure 5.6 Secondary raw material demand for the manufacture of PCE-new (left) and PCE-refurb (right) in the Netherlands for the period 2015–2050

5.4 Results and discussion

5.4.1 Supply-demand analysis

Based on the potential supply of secondary raw materials (see Figure 5.5) and the demand for secondary raw materials for construction and renovation (see Figure 5.6), the supply and demand balance of each secondary raw material is presented in Figure

5.7. Based on this, the secondary raw materials (CRSCA, FRSCA, and URSCA) for PCE-new can be supplied in sufficient quantities, even with surplus quantities. The demand for steel frames, RGUA, and RFUA can also be fully met.

The CRLWCA, FRLWCA, and URLWCA for the production of PCE-refurbs are inadequate, however, to support significant refurbishment efforts. The deficit portion of these materials could be complemented by using virgin materials (e.g., expanded clay, sand, and cement) or by importing lightweight concrete waste from neighboring countries such as Germany or Belgium, although this is unlikely due to high transportation costs.

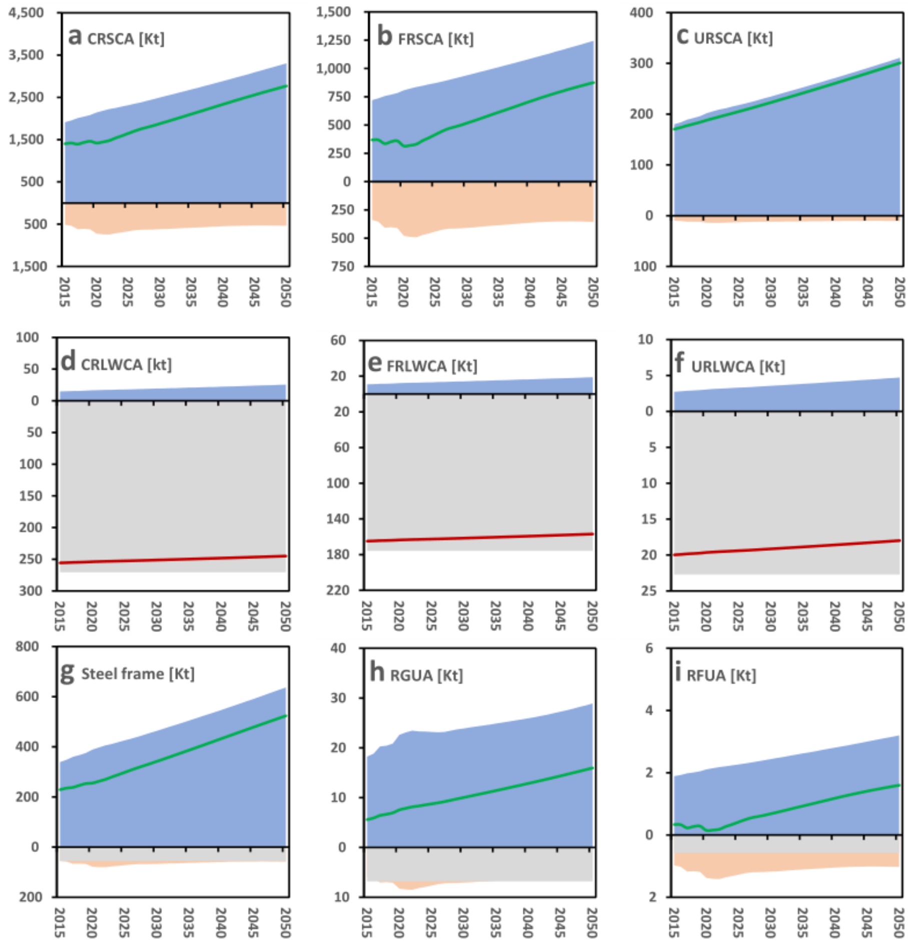

Figure 5.7 Supply-demand condition of secondary raw materials. Note: 1) zone (in blue) above 0 represents the supply of secondary raw materials, zone below 0 represents the demand of secondary

raw materials for building construction (in salmon) and building renovation (in grey); 2) curves in red indicate the deficient amount of secondary raw materials, curves in green indicate the surplus amount of secondary raw materials.

5.4.2 Comparison of secondary material surplus and primary material imports

The surplus or deficit of each secondary raw material was compared to the net import of the corresponding virgin raw material. The associated import and export data were collected from the UN Comtrade Database (2020). Because the data on iron and steel are presented as monetary values in the database, the comparison of these materials with reforged steel was excluded. For the comparison of gravel and CRSCA in Figure 5.8(a), the median trend of gravel net imports is approximately five times that of CRSCA since 2018; however, under conservative (lower confidence limit) conditions, the surplus of CRSCA can substitute all gravel imports from 2040 onwards. Concerning the net import of expanded clay in Figure 5.8(b), the overall volume is considerably smaller than that of gravel, fluctuating from 5 to 100 Kt between 1992 and 2018, and probably continuing to decrease to 2050. The deficit of CRLWCA stabilizes at approximately 180 Kt, which may cause the import of expanded clay to increase in the future.

For virgin sand imports in Figure 5.8(c), compared to the other raw materials, sand relies less on imports according to the trend of historical net imports, although with a large uncertainty range. The surplus of FRSCA and the deficit of FRLWCA are insignificant compared to the large uncertainty in net imports. In the case of the cement import in Figure 5.8(d), as with the net import trend of gravel, the Netherlands is and will be largely dependent on imports. The amounts of RGUA and URSCA surpluses and the URLWCA deficit are negligible compared to imports. Lastly, as shown in Figure 5.8(e), the net import of limestone follows an increasing trend. As insulation waste only accounts for less than 0.1% of the total CDW, the RFUA produced from insulation waste has an almost negligible effect on the import of limestone.

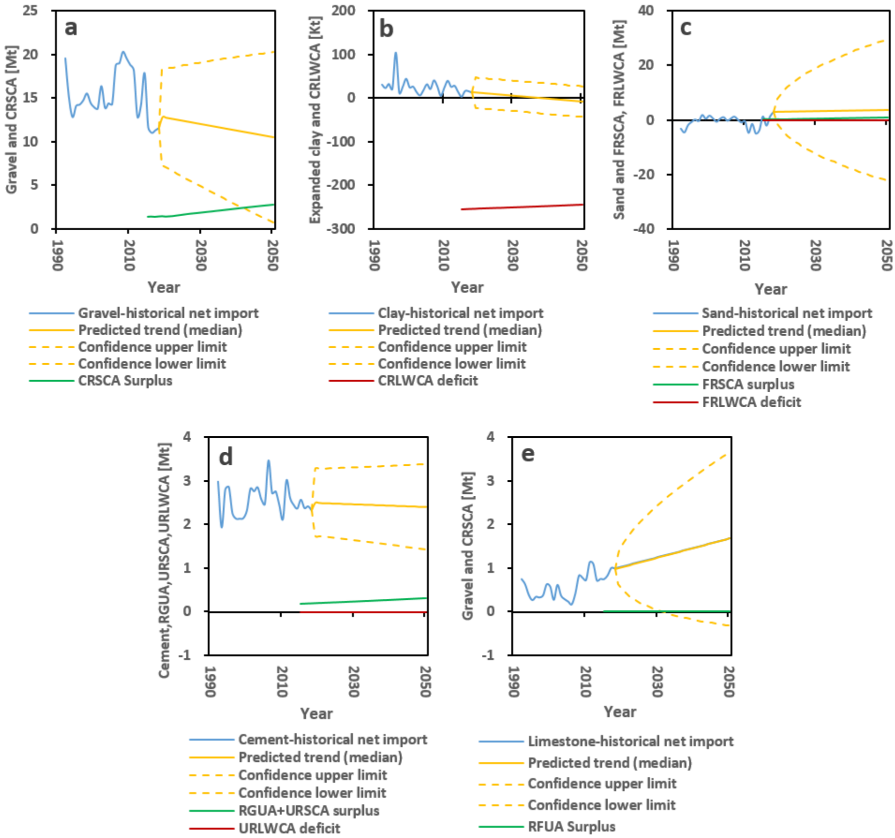

Figure 5.8 Comparison between virgin raw material net import (import subtracts export) and secondary raw material deficit and surplus in the Netherlands for the period 1990–2050: (a) represents the comparison of gravel net import and CRSCA surplus; (b) denotes comparison of expanded clays net import and CRLWCA deficit; (c) compares sand net import, and FRSCA surplus, and FRLWCA deficit; (d) compares cement net import, RFUA+URSCA surplus, and URLWCA deficit; and (e) compares limestone net import and RFUA surplus. Data were collected from the UN Comtrade database (2020). The predicted trends were obtained via linear regression with a 95% confidence interval.

5.4.3 Validation and uncertainty

The dynamic MFA model is based on multiple parameters, and the fluctuations of each parameter will, therefore, affect the final supply and demand balance. Owing to the lack of a valid reference for the fluctuation range of each parameter, it is impossible to conduct a full uncertainty analysis. Nevertheless, an examination of the uncertainty was performed based on those factors with a relatively strong influence on the results. Thus, we deem that the biggest uncertainties lie in the estimation of (i) annual construction, demolition, and renovation floor area; (ii) concrete waste intensity; and (iii) the share of lightweight concrete waste in gross concrete waste.

5.4.3.1 Annual construction, demolition, and renovation

The annual construction, demolition, and renovation floor area in this study were validated in reference to other data sources, the Environmental Assessment Agency (Staniaszek 2015), the ZEBRA2020 Data Tool (2020), Sandberg et al. (2016), and Statistics Netherlands (2020). Some of these sources measured the turnover of the building stock based on the number of dwellings instead of floor area, which makes their results incomparable. Therefore, we used relative indexes, namely construction rate, demolition rate, and renovation rate, to unify the comparison. Based on Figure 5.9(a), all of the construction rates present a decreasing trend from approximately 1.5% to 1%, while in Figure 5.9(b), demolition rates show a gradually increasing trend from approximately 0.3% to 0.5%. These renovation rates from the different sources demonstrate a notable disparity. Overall, the construction and demolition rates we applied in this study are in general accordance with these other sources.

As shown in Figure 5.9(c), the average historical renovation rate from Statistics Netherlands is approximately 0.5% while the renovation rates of other sources are much higher. To achieve the carbon-neutral goal by 2050, of the 7.5 million dwellings, 170,000 need to be renovated per annum in the Netherlands (Staniaszek 2015). Based on this, the equivalent renovation rate was set at 2.3% in 2015, amounting to approximately 17 million m^2 per annum.

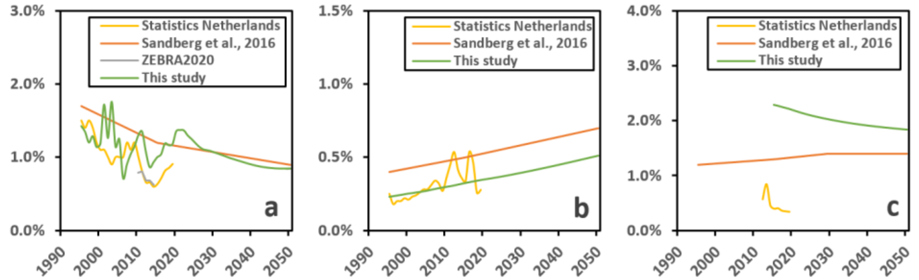

Figure 5.9 Comparison of (a) construction rate, (b) demolition rate, and (c) renovation rate for the Netherlands based on a range of sources

5.4.3.2 Concrete waste intensity

Concrete waste was the focal waste stream of our CDW estimates. The concrete waste intensity for demolition ($K_{concrete}^{(d)}$ = 902 kg/m^2) has a far greater contribution to gross concrete waste generation than construction ($K_{concrete}^{(c)}$ = 26 kg/m^2) and renovation

($K_{concrete}^{(r)}$ = 28.5 kg/m^2. Therefore, the uncertainty in waste concrete generation from building demolition ($K_{concrete}^{(c)}$) is discussed further in this section.

Concrete waste is commonly generated from four sectors: (i) the residential building sector, (ii) the non-residential building sector, (iii) civil engineering, and (iv) the building materials industry. Concrete waste produced from the residential sector accounts for approximately 30% of the gross concrete waste in the Netherlands (Zhang et al. 2020b), and the Environmental Data Compendium of the Netherlands (CLO) (2020) reported the generation of CDW between 1985 and 2016. Based on this, we estimated the concrete waste generated from residential buildings, as shown in Figure 5.10. These data show that the concrete waste released from residential buildings has stabilized at approximately 4,500 Kt per annum since 2000.

Notably, the concrete waste intensity varies for different types of buildings. For example, a timber-structured building generates up to 300 kg/m^2 of concrete waste (Gálvez-Martos et al. 2018). For concrete structure buildings, relevant data from a demolition project located on the de Kempkensberg in Groningen in the Netherlands (Hu et al. 2012) were collected to estimate the concrete waste intensity. This concrete high-rise building had 14 stories and a 6,174 m^2 of useful floor area, from which a total of 12,357 tons of concrete waste was generated during demolition, amounting to 2 tons of concrete per m^2 of floor area. This is in accordance with the medium-level concrete waste intensity of 2.1 t/m^2 in Müller's stock dynamics modeling (Müller 2006). The amounts of concrete waste based on different concrete intensities (300 kg/m^2, 902 kg/m^2, and 2,000 kg/m^2) were compared, as shown in Figure 5.10. If $K_{concrete}^{(d)}$ increases to 2,000 kg/m^2, gross concrete waste shows a sharply increasing trend. In contrast, at 300 kg/m^2, this trend is less than half of the historically probable trend. The selected median value (902 kg/m^2) was also lower than the actual trend. Therefore, the estimation of concrete waste in this study was relatively low compared to the reality. This may be because we assumed a high lifetime for residential buildings, leading to less generation of demolition waste. Moreover, we used static concrete waste intensity, whereas waste intensity is likely to increase over time.

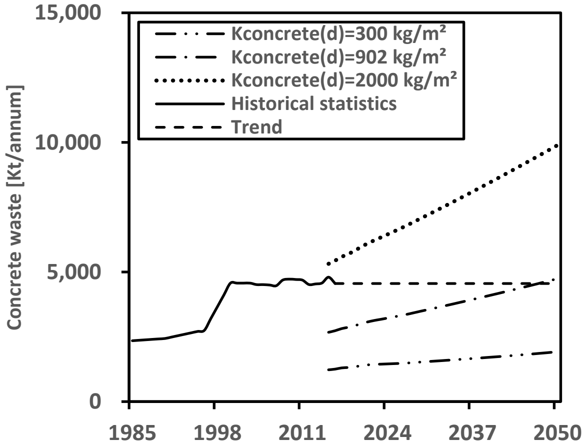

Figure 5.10 Concrete waste generation from the residential building sector in the Netherlands under different waste intensities. Note: $K_{concrete}$ denotes the waste concrete waste intensity for demolition.

5.4.3.3 Share of lightweight concrete waste

According to the Report and Data (2020), the global lightweight aggregate concrete market was valued at 37.2 billion USD in 2018 and is expected to reach 56.7 billion USD by 2026. In Europe, the lightweight aggregate concrete market is forecasted to increase from 23 million USD in 2018 to 40 million USD in 2026 (Reports and Data 2020). The share of lightweight concrete waste compared to gross concrete waste is assumed to remain stable at 1% until 2050. Quantification of the variations in this share can provide a more comprehensive assessment of the supply-demand connection. Therefore, we examined the level of uncertainty by modeling several scenarios in which the share of lightweight concrete waste would increase at different rates over time. The share was modeled starting with different initial values (1%, 3%, and 5%) and then increased linearly to 8%, 12%, and 20% between 2015 and 2050.

The results of the uncertainty simulation are shown in Figure 5.11. Under all conditions, the URLWCA is likely to be sufficiently supplied. For CRLWCA and FRLWCA, when the initial share is 1%, even though it increases to 8% by 2050, production barely meets the demand for widespread renovation until 2050. In this case, a large amount of virgin expanded clay and sand is produced or imported to replenish the CRLWCA and FRLWCA feedstock. If the initial share increases to 3%, the CRLWCA and FRLWCA supplies can sufficiently support building renovations with PCE-refurbs up to approximately 2045 with a high increase speed. If the share is started at 5% in 2015, the supply of FRLWCA and CRLWCA reach the break-even point by 2035. Finally, the production of URLWCA is barely able to sustain consumption under any of the assumptions. Primary sand and expandable clay are, therefore, needed to complement FRLWCA and CRLWCA by 2035 at the latest.

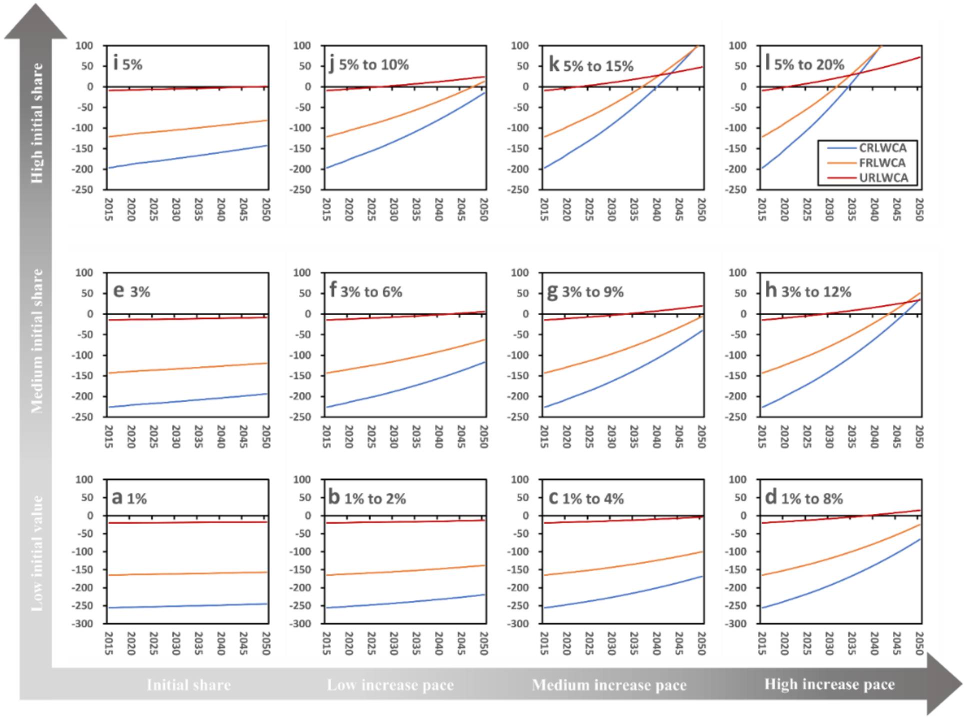

Figure 5.11 Supply-demand condition of CRLWCA, FRLWCA, and URLWCA in Kt. Note: (a) "initial share" means "initial value of the share of lightweight concrete waste to the total concrete waste remains at 1%, 3%, and 5% from 2015 to 2050; in Panel b, "1% to 2%" represents a linear share increase from 1% in 2015 to 2% in 2050. (b) zones above represent the amount of the surplus of secondary raw materials, zones below 0 represent the amount of the deficit of secondary raw materials.

5.5 Implications of this study

5.5.1 Constraints and opportunities of CDW management in the Netherlands

The EU has enacted a series of relevant directives on CDW management and energy efficiency. For example, the Waste Framework Directive (2008/98/EC) sets a 70% target for CWD recovery for EU member states (EC 2008a); the COM (2011) 571 aims to promote resource efficiency during the construction and renovation of buildings (EC 2011b); and the Energy Performance of Buildings Directive (EPBD, 2002/91/EC) (EC 2002) and Energy Efficiency Directive (EED, 2012/27/EU) (EC 2012b) request member states to employ cost-effective energy renovation measures to promote the energy performance of new and old buildings.

The residential building stock in the Netherlands has poor insulation level and is obsolete; approximately half the building stock was constructed between the 1950s and the 1970s—before minimum energy performance requirements were introduced in 1995 (Staniaszek 2015). In the Energy Agreement for Sustainable Growth (SER 2013), the Netherlands committed to achieving the ambitious goal of a carbon-neutral built environment by 2050. To support the EU's response to the Paris Climate Agreement, the Government of the Netherlands (2019) enacted a national climate agreement to achieve a 49% mitigation in national carbon emissions by 2030. Thus, an additional reduction of 3.4 Mt of greenhouse gas is required by 2030, and the Netherlands even called for increasing the European target to 55% by 2030. By 2050, the Netherlands is expected to achieve carbon-neutral status (EZK 2019), setting up a significantly limited carbon budget for the building sector.

For decades, the Netherlands has exceeded the EU target of 70% CDW management but upgrading the practice of road backfilling to high value-added recycling is urgently needed. Due to the topography of the Netherlands, domestic extraction of large quantities of stony mineral resources is not possible. Raw materials for the production of concrete, such as sand and gravel, are, therefore, must be imported and—in the future—recycled domestically. The Dutch government has outlined the goal for a circular economy in the Netherlands by 2050 (Dijksma and Kamp 2016), involving a 50% reduction in raw material use by 2030 and a fully circular economy by 2050. Therefore, to transition to a fully circular built environment, it is crucial to close the loop of the construction material supply chain, especially emerging materials used in energy renovation.

Prefabrication has been identified as a reliable solution for reducing CDW (Tam et al. 2006); waste concrete can be reduced by 52% to 60% as prefabricated products are cast off-site (Tam et al. 2005). Prefabrication also contributes to other on-site benefits, such as improved quality control, tidier and safer working environments, and improved environmental performance (Jaillon et al. 2009). According to the estimation of our model, approximately 8 million m^2 and 17 million m^2 of dwellings are to be constructed and renovated per annum. Therefore, the proposed PCE system presents a promising solution to upgrading the treatment of CDW, waste reduction, and energy renovation. The practice of prefabricating buildings is well established in the Netherlands, with prefabricated elements used in over half (55%) of all Dutch construction projects in 2016 (de Gruijl 2018). This lays a solid technical foundation for the implementation of the PCE system.

5.5.2 Influence of the density and typology of concrete on mass estimation

Concrete production is the main engine for rapid urbanization. The EU directive of resource efficiency opportunities in the building sector (COM/2014/0445) suggests that concrete waste should be a focal point for CDW management (EC 2014d). Indeed, the literature review in Table 5.1 shows that concrete waste management is a significant topic for MFA studies.

In general, the concrete in MFA studies is modeled as normal-weight concrete. In this study, normal-weight siliceous concrete and lightweight aggregate concrete were considered to represent normal-weight concrete and lightweight concrete, respectively. The normal-weight concrete includes other types of concrete, such as limestone concrete, which employs different formulations compared to siliceous concrete. Lightweight concrete can be categorized as lightweight aggregate concrete, foamed concrete, and autoclaved aerated concrete. Despite this diverse typology, the density of concrete is the key factor that could influence MFA because material flows are derived from physical mass data. A concrete waste intensity $K_{concrete}^{(d)} = 902$ kg/m^2 was applied to estimate the generation of normal-weight concrete. Because the waste intensity for lightweight concrete is unavailable, we simplified the estimation of lightweight concrete waste by assuming a share of 1%.

The densities of normal-weight concrete and lightweight concrete used in our analysis were 2,089 kg/m^3 and 1,963 kg/m^3, respectively. Assuming 1% of lightweight concrete waste by weight, the difference in the mass of gross concrete waste is approximately 2 Kt by 2030; if concrete waste comprises 1% of ultra-lightweight concrete (500 kg/m^3), the mass difference is 28 Kt over the same timeframe. With the gradual prevalence of lightweight concrete in building energy renovation practices, MFA studies should consider the effect of lightweight concrete on mass estimation.

5.5.3 Whether or not to consider renovation waste

The measurement of the composition and generation of CDW is a longstanding dilemma for MFA studies. The generation of renovation waste, in particular, is relatively difficult to estimate due to diverse retrofitting options, such as external insulation systems, cladding systems, and ventilated façade systems (Villoria Sáez et al. 2018) as well as different levels of renovation, i.e., minor, moderate, deep, and nearly zero-energy building levels (Economidou 2011). Thus, most of the MFA studies summarized in Table 5.1do not consider waste from building renovation. Table 5.3 provides some examples of waste intensity for the renovation of residential buildings in different regions. In more developed areas, the amount of renovation waste is growing rapidly (Cheng and Ma 2013). For instance, renovation waste accounts for 29% of the gross CDW by weight in

Norway (Bergsdal et al. 2008), and its intensity can reach up to 300 kg/m^2, which considerably exceeds the intensity of construction waste (41 kg/m^2 in this study). In developing countries such as China, renovation waste amounts to less than 1% of gross CDW (Ding et al. 2019b), and intensity could be lowered to 20 kg/m^2. The estimation of renovation waste based on the construction area, living area, and useful area can also yield differing results (Coelho and De Brito 2011).

Table 5.3 Examples of waste intensity for the renovation of residential buildings

Literature	Location	Amount [kg/m^2]	Remark
(Bergsdal et al. 2008)	Norway	60.13–89.47	Residential building
(Villoria Sáez et al. 2018; Thorpe 2008)	UK	147.84	Residential building, estimation based on volume (m^3) of waste generated per 100 m^2
(Villoria Sáez et al. 2018)	Spain	2.46–65.24	Residential building
(Coelho and De Brito 2011)	Portugal	347.3	Residential building, estimation based on a gross construction area
(Mália et al. 2013)	Portugal	28–397	Residential building
(Cochran et al. 2007)	USA	43.70–82.00	Residential building
(Ding et al. 2019a)	China	15.65–25.98	Residential building
(Ding et al. 2019b)	China	21.05	Residential building

We assumed that the Netherlands will undergo large-scale renovation, with more than 50% of the current stock (based on 2015 data) to be refurbished. Therefore, renovation waste was considered in the MFA model. The amounts of construction waste, demolition waste, and renovation are presented in Figure 5.12. Overall, the amount of renovation waste exceeds the amount of construction waste. It is noteworthy that we only estimated the renovation waste from the implementation of the proposed PCE cladding technology; deeper renovation is expected to yield more renovation waste. Moreover, we only estimated the wastes that can be incorporated into the PCEs, namely concrete, insulation, glass, and steel, which account for 77% of CDW by weight (Zhang et al. 2020b); minor waste streams, such as wood, plastic, and paper, are not included. Given the fact that building energy renovation has become a primary pathway towards a carbon-neutral built environment, considering renovation waste in MFA studies offers a more comprehensive means of CDW management.

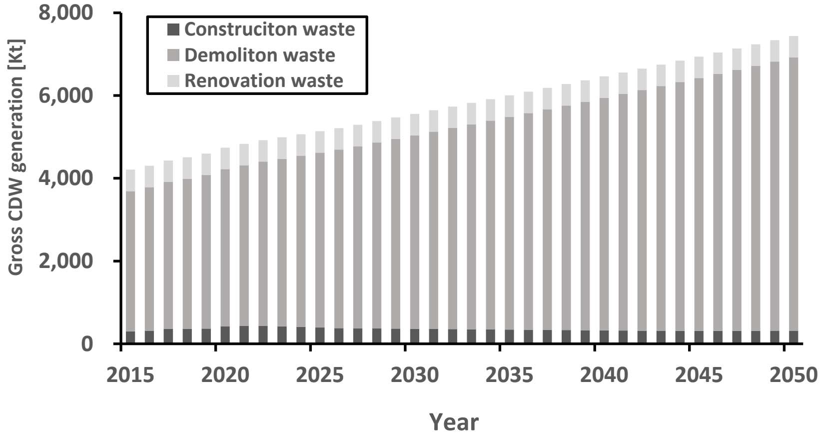

Figure 5.12 Generation of construction waste, demolition waste, and renovation waste estimated in the Netherlands for the period 2015-2050. Note: construction waste and demolition waste are estimated based on the share of concrete waste in CDW and the concrete waste intensity

5.6 Conclusions

The building sector is considered one of the main drivers of material depletion, waste generation, energy use, and greenhouse gas emissions. It is highly important and urgent, therefore, to accelerate the transition toward a carbon-neutral and circular built environment. Ongoing building energy renovation is accompanied by emerging materials such as mineral wool insulation and lightweight concrete, triggering new problems of disposal. This makes it harder to close supply chains in the building sector. The proposed PCE system delivers a potential solution by incorporating CDW into building energy renovations. Here, we constructed a prospective top-down stock-driven MFA model to explore the supply-demand condition of associated secondary raw materials for the PCE system for new building construction and existing building renovation in the Netherlands for the period 2015–2050. Compared to previous MFA studies, our model considers the recycling of glass, lightweight concrete, and insulation mineral wool in CDW through an on-site innovative recycling technological system in the context of building energy renovation in the Netherlands.

Our results show that secondary raw materials recycled from normal-weight concrete waste, namely CRSCA, FRSCA, and URSCA, can be sufficiently supplied, even with a large surplus. The reforged steel frames, RGUA, and RFUA required for building construction and renovation can also be sufficiently supplied. However, under the condition that lightweight concrete waste was assumed to account for only 1% of the gross concrete waste, the secondary raw materials CRLWCA, FRLWCA, and URLWCA for new lightweight concrete production are inadequate for supporting manufacturing of the PCE-refurb system. The deficit could be replenished using virgin materials or by importing lightweight concrete waste from neighboring countries. Based on a comparison of the surpluses/deficits of recycled materials to the net import of corresponding virgin materials, we found that the demand for main mineral resources in the Netherlands is highly dependent on imports. Only CRSCA shows potential for offsetting gravel imports assuming conservative imports. The other secondary raw materials do not appear to reduce the import of associated virgin materials.

Using uncertainty analysis, we quantified the influence of variations in (1) construction, demolition, and renovation floor area of each year; (2) concrete waste intensity; and (3) the share of lightweight concrete waste. We used construction, demolition, and renovation rates to compare the uncertainties of construction, demolition, and renovation activities each year from different sources. The results show that the construction and demolition rates are harmonized with historical statistics. The renovation rate is assumed to track the prospective energy renovation planning of the Netherlands and is, therefore, higher than the actual value. Regarding concrete waste intensity, owing to a conservative assumption of concrete waste intensity, the forecast waste concrete stream is relatively lower than the current statistics. Lightweight concrete was modeled with different initial shares in gross concrete waste with an increasing pace, starting from 1%, 3%, and 5% in 2015 and increasing linearly to 8%, 12%, and 20% by 2050, respectively. We found that the production of URLWCA can barely meet the demand under any of these cases, whereas primary sand and cement are still needed for the substitution of FRLWCA and CRLWCA until 2027.

This study has investigated the physical mass link between CDW recycling and secondary material demands in the context of building energy renovation in the Netherlands. However, the associated environmental and financial implications remain unknown. Our previous studies investigated the life-cycle carbon emissions and costs of the PCE system at the building level (Zhang et al. 2020a, 2021a). In the future, we aim to scale up the life cycle environmental and economic benefits of the proposed PCE system for energy renovation at a regional level.

Acknowledgements

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Appendix

Table A5.1 Numbers of each type of house in different construction cohorts

	Up to 1945	Up to 1964	1965-1974	1975-1991	1992-2005
Detached house	/	441,000	119,000	221,000	178,000
Semi-detached house	/	285,000	142,000	224,000	173,000
Row house	523,000	478,000	606,000	879,000	353,000
Maisonette house	/	2,266,000	22,000	94,000	40,000
Apartment-gallery	/	69,000	174,000	109,000	113,000
Apartment-porch	256,000	267,000	112,000	142,000	70,000
Apartment-other	/	99,000	125,000	125,000	136,000

Table A5.2 Average floor area of each type of house by building typology and construction cohort

	Up to 1945	Up to 1964	1965-1974	1975-1991	1992-2005
Detached house	/	130	144	154	172
Semi-detached house	/	110	123	123	132
Row house	102	87	106	106	114
Maisonette house	/	88	88	80	84
Apartment-gallery	/	72	82	68	79
Apartment-porch	59	66	71	70	74
Apartment-other	/	67	77	70	82

Table A5.3 Gross floor area of houses by building typology and construction cohort, and stock share

	Up to 1945	Up to 1964	1965-1974	1975-1991	1992-2005	Up to2005	Stock share
Detached house	/	57,330,000	17,136,000	34,034,000	30,616,000	139,116,000	15.98%
Semi-detached house	/	31,350,000	17,466,000	27,552,000	22,836,000	99,204,000	11.39%
Row house	53,346,000	41,586,000	64,236,000	93,174,000	40,242,000	292,584,000	33.60%
Maisonette house	/	199,408,000	1,936,000	7,520,000	3,360,000	212,224,000	24.38%
Apartment-gallery, porch, other	15,104,000	29,223,000	31,845,000	26,102,000	25,259,000	870,661,000	14.65%

Table A5.4 Concrete intensity for different types of houses in Western Europe

	Concrete intensity (kg /m^2)
Detached house	1507.04
Semi-detached house	1507.04
Row house	796.02

Maisonette house	567.99
Apartment	567.99

Table A5.5 Share of insulation material in Europe

Insulation material	Share
Glass wool	36%
Expanded polystyrene	27%
Stone wool	22%
Polyurethanes	8%
Extruded polystyrene	6%
Other	1%

Table A5.6 Formulation of secondary raw materials used for VEEP PCE-new and PCE-refurb

VEEP PCE	Component of PCE	Weight [Kg] per unit of PCE	Remarks
VEEP PCE-new (length 3.6 m, width 2.4 m)	Green normal-weight concrete	2658.41	URSCA used for per m^3 of concrete: 16.75 kg/ m^3 FRSCA used for per m^3 of concrete: 566 kg/m^3 CRSCA used for per m^3 of concrete: 880 kg/m^3 RGUA used for per m^3 of concrete: 10.05 kg/m^3 RFUA used for per m^3 of concrete: 1.675 kg/m^3
	Green Aerogel	42.35	FRSCA used for per 0.01m^3 of aerogel: 0.31 kg/m^3
	Expanded polystyrene	13.50	/
	Rebar cages & welded nets	117.10	Steel frame for per unit of PCE: 117.10 kg/PCE
VEEP PCE-refurb (length 2 m, width 2 m)	Green lightweight concrete	374.80	URLWCA used for per m^3 of concrete: 47 kg/ m^3 FRLWCA used for per m^3 of concrete: 364 kg/m^3 CRLWCA used for per m^3 of concrete: 560 kg/m^3 RGUA used for per m^3 of concrete: 14.1 kg/m^3 RFUA used for per m^3 of concrete: 1.2 kg/m^3
	Green Aerogel	19.60	FRSCA used for per 0.01m^3 of aerogel: 0.31 kg/m^3
	Steel beams & welded nets	23.72	Steel frame for per unit of PCE: 23.72 kg/PCE

Table A5.7 Secondary raw material demanded per 1 m^2 construction and renovation floor area

	Coefficient of secondary raw material demand	Value (kg/m^2)
$K_{URSCA(new)}$	Ultrafine recycled siliceous concrete aggregate (URSCA) for PCE-new	1.269
$K_{FRSCA(new)}$	Fine recycled siliceous concrete aggregate (FRSCA) for PCE-new	44.070
$K_{CRSCA(new)}$	Coarse recycled siliceous concrete aggregate (CRSCA) for PCE-new	66.662
$K_{RGUA(new)}$	Recycled glass ultrafine admixture (RGUA) for PCE-new	0.761
$K_{RFUA(new)}$	Recycled fiber wool ultrafine admixture (RFUA) for PCE-new	0.127
$K_{SF(new)}$	Steel frame for PCE-new	7.450
$K_{URSCA(refurb)}$	Ultrafine recycled lightweight concrete aggregate (ULWSCA) for PCE-refurb	1.293
$K_{FRSCA(refurb)}$	Fine recycled siliceous concrete aggregate (FRSCA) for PCE-refurb	1.194
$K_{FRLWCA(refurb)}$	Fine recycled lightweight concrete aggregate (FRLWCA) for PCE-refurb	10.010
$K_{CRLWCA(refurb)}$	Coarse recycled lightweight concrete aggregate (CRLWCA) for PCE-refurb	15.400
$K_{RGUA(refurb)}$	Recycled glass ultrafine admixture (RGUA) for PCE-refurb	0.388
$K_{RFUA(refurb)}$	Recycled fiber wool ultrafine admixture (RFUA)for PCE-refurb	0.033
$K_{SF(refurb)}$	Steel frame for PCE- refurb	3.260

Chapter 6

Integrated material-energy efficiency renovation of housing stock in the Netherlands: Economic and environmental implications

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Submitted and in review

Abstract

The Netherlands strives to achieve ambitious targets with regard to circularity and carbon neutrality in the built environment by 2050. Technical and social innovations are required to accelerate developmental progress to meet these ambitions. This paper presents a material-energy efficiency method of retrofitting building stocks with prefabricated concrete elements (PCEs) with high value-added recycled construction and demolition waste (CDW) as feedstock. The objective of this study is to investigate (i) the economic and environmental trade-offs of the PCE system compared to other options using both an actual-value approach and an annualized-value approach and (ii) to what extent circularity and decarbonization goals can be realized if this PCE system is up-scaled to the residential sector. By combining dynamic material flow analysis with life cycle assessment and life cycle costing, this study analyzes both the product-level and nationwide carbon mitigation, cost, and material footprint reduction potential of implementing this PCE system to the Dutch housing stock for the period 2015-2050. In addition to the proposed renovation scenario (REN), two additional scenarios were developed: the baseline business-as-usual (BAU) scenario, which does not apply any renovation strategy, or the rebuilding (REB) scenario in which old buildings are demolished and reconstructed instead of renovation. The key findings are: (i) the actual- and annualized-value approach may lead to different outcomes: this PCE system in the REN scenario achieves better comprehensive benefits in an annualized-value approach whereas shows a clear trade-off using an actual-value approach compared with the BAU scenario; (ii) the likelihood of achieving the circularity goal by 2030 is almost impossible; (iii) the REN and REB scenarios can achieve the goal of 49% carbon mitigation by 2030, however, only the REB scenario can achieve net-zero emissions by 2050. In the REN

scenario, carbon neutrality in the housing sector requires replacing traditional gas-based systems with electricity-based systems and achieving 100% renewable electricity supply nationwide.

Keywords: life cycle assessment, life cycle costing, construction and demolition waste, building energy renovation, material circularity, decarbonization

Abbreviations

3R	Reduce, reuse, and recycle
ADR	Advanced dry recovery technology
BAU	Business-as-usual
CDW	Construction and demolition waste
CRLWCA	Coarse recycled lightweight concrete aggregate
CRSCA	Coarse recycled siliceous concrete aggregate
DGR	Dry grinding and refining system
EC	European Commission
EoL	End-of-life
EU	European Union
FRSCA	Fine recycled siliceous concrete aggregate
FRLWCA	Fine recycled lightweight concrete aggregate
GHG	Greenhouse gas
HAS	Heating-air classification system
LCA	Life cycle assessment
LCC	Life cycle costing
MFA	Material flow analysis
nZEB	Nearly zero energy building
PCE	Prefabricated concrete element
PCE-new	Prefabricated concrete element for new building construction
PCE-refurb	Prefabricated concrete element for existing building refurbishment
REB	Demolition and rebuilding/reconstruction (scenario)
REN	Renovation with PCE (scenario)
RFUA	Recycled fiber wool ultrafine admixture
RGUA	Recycled glass ultrafine admixture
SI	Supporting information
URLWCA	Ultrafine recycled light-weight concrete aggregate
URSCA	Ultrafine recycled siliceous concrete aggregate
VEEP	European Union Horizon 2020 project “Cost-effective recycling of C&DW in high added-value, energy-efficient prefabricated concrete components for the massive retrofitting of our built environment”

6.1 Introduction

The Dutch construction sector is a priority with respect to transitioning towards a circular and carbon-neutral society, because it is responsible for 50% of raw material use, 40% of total energy use, 40% of waste generation, and 35% of greenhouse gas (GHG) emissions in the country (Dijksma and Kamp 2016). The Netherlands already achieved 92% recovery rates for CDW in 1995 and reached 97% recovery in 2018 (CLO 2021). However, most stony CDW is downcycled as back filler for road base construction and site elevation. Only 5% is recycled, in e.g. new concrete production (Zhang et al. 2020b). To improve the material efficiency within the construction sector, the Netherlands launched the program “A Circular Economy in the Netherlands by 2050”, aiming to achieve an interim target of 50% less use of primary raw materials (minerals, fossil fuels, and metals) by 2030, and a long-term goal of a fully circular economy by 2050: efficiently using and reusing materials without any damage entering the environment (Dijksma and Kamp 2016).

Most buildings in the Netherlands remains a poor insulation level. Around half of the buildings in the Netherlands were constructed between the 1950s and 1970s, before the introduction of minimum energy performance requirements in 1995 (Staniaszek 2015). In 2018, the Netherlands realized a reduction of GHG emissions of 17% compared to emissions in 1990 (PBL et al. 2020). In the Climate Act, the Netherlands set an ambitious interim target with a total GHG emission reduction of 49% by 2030 and a long-term target of 95% by 2050 compared to 1990 (Government of the Netherlands 2019). For the residential sector, an operational GHG emission reduction of approximately 10 Mt and 20 Mt by 2030 and 2050 is expected, respectively (PBL et al. 2020; EZK 2019). Given these GHG mitigation ambitions, the Netherlands committed to energy neutrality target for the built environment by 2050. This means of 7.5 million dwellings, 80% are to be renovated to energy-neutral levels by 2050, which implies that 170,000 homes are to be renovated per annum until 2050 (Staniaszek 2015). Crucially, making the Dutch housing stock energy-neutral has significant implications for material requirements and waste generation, which may interfere with realizing the aforementioned circularity targets.

Based on the building circularity and energy efficiency framework shown in Figure 1.3 and Figure 1.4, we investigated the potential trade-offs between GHG emission, capital expanse, and material use in the upcoming energy renovation of the Dutch residential sector regarding achieving the circularity and energy neutrality targets. We set a business-as-usual (BAU) scenario (no additional effort to achieve energy neutrality), a rebuilding (REB) scenario (achieving an energy-neutral built environment by demolition and rebuilding energy-inefficient stock), and a renovation scenario (REN). For REN, we focused on the use of recently developed advanced technological solutions, including prefabricated concrete elements (PCEs) made with recycled concrete, glass, and insulating mineral wool.

We analyzed the scenarios via a combination of a dynamic material flow analysis (MFA) of the Dutch building stock and environmental life cycle assessment (LCA) and costing (LCC). This provides insights into prospective GHG reductions, material requirements and waste flows, and investment costs for the scenarios in the Dutch context for the period from 2015 to 2050. By doing so, we explore whether the PCE system would bring about economic and ecological wins. To reduce complexity, the approach had to apply several simplifications. Illustrations are the following. First, we used a limited number of ideal-typical residential building types. Second, we assumed that in the REN scenario refurbishing with PCEs would be the main approach and feasible for all building types. Third, we had to make our assumptions about starting years of large-scale renovation efforts and implementation of nearly zero energy building (nZEB) for new build houses. Fourth, we did not consider the carbon uptake by cement carbonation (Xi et al. 2016; Cao et al. 2020). Fifth, as operational energy accounts for most of the life cycle energy use of a building, we only modeled the development of renewable electricity for the operation phase and electricity used in manufacturing processing was assumed to remain at the 2015 level. Sixth, we used a steady-state costing system and assumed a zero interest rate. Seventh, we did not consider the impacts of building maintenance. Finally, we assumed all old houses are supposed to be renovated or demolished by 2050. However, there is approximately 3% of historical dwellings will never be demolished or renovated for heritage reasons (Sandberg et al. 2016). Our study is a simplified reflection of reality. But despite such limitations, the evaluation still can provide a clear illustration of the main trade-offs between scenarios in realizing the interim and long-term dematerialization and decarbonization ambitions for the Netherlands.

6.2 Materials and Methods

6.2.1 Goal and scope

The goal of the analysis is twofold: (i) to identify the trade-offs between GHG emission, capital expanse, and material use incurred in energy renovation; (ii) to explore the potential pathways in building energy renovation towards the circularity and decarbonization ambitions. By doing this, this study assesses climate impacts (in kg CO_2 eq), costs (in Euros), and material footprints (in kg of primary material extraction) for different scenarios with regards to the energy neutrality of the Dutch housing stock for the period 2015–2050. Here we discuss the goals and scope of the study.

6.2.1.1 Methodological overview

The methodological framework for integrating LCA and LCC with the dynamic MFA is presented in Figure 6.1. Buildings constructed at different times are categorized into five vintage cohorts: (i) up to 1960, (ii) 1961–1980, (iii) 1981–2000, (iv) 2001–2014, (v) 2014-2050. Different cohorts of houses vary in insulation levels of envelopes and heating efficiency, resulting in different operational costs, energy use, and GHG emission performance. The Dutch houses are divided into five archetypical types: detached houses, semi-detached houses, terraced houses, maisonettes, and apartments (Agentschap NL 2011). Apartments include gallery apartments, porch apartments, and other apartments. Terraced houses account for the biggest share of the gross Dutch houses (33.60%), followed by maisonettes (24.38%), detached houses (15.98%), apartments (14.65%), and semi-detached houses (11.39%) (Zhang et al. 2021c). We assumed this division to be constant over time, with a slight expansion of the overall housing stock till 2050 according to the building stock model (Zhang et al. 2021c). Three scenarios were considered: BAU, REB, and REN, which represent different circularity and energy-efficiency strategies. Given the availability of data, 2015 was chosen as the starting year of the assessment.

<u>*Assessment at product and country level*</u>

The assessment is conducted at both the product and country levels. LCA and LCC are suitable for investigating product-oriented life cycle environmental impacts and costs (Hunkeler et al. 2003). LCA and LCC have been widely used in areas of CDW management (Hossain et al. 2016; Zhang et al. 2020c; Pavlu et al. 2019; Massarutto et al. 2011; Mah et al. 2018; Di Maria et al. 2018) and building energy efficiency (Almutairi et al. 2015; Rodrigues et al. 2018; Minne and Crittenden 2015; Günkaya 2020; Pedinotti-Castelle et al. 2019; Atmaca 2016a, 2016b). At the product level, the LCA and LCC were primarily applied to evaluate the (re)construction, renovation, demolition activities in three scenarios. The life cycle of the PCE system consists of three stages: construction, operation, and demolition. However, LCA and LCC are product-oriented approaches and are not directly able to investigate issues at a regional or national scale. To overcome this drawback, we employed a dynamic MFA model to simulate the turnover of building stock in the Netherlands from 2015 to 2050.

At the country level, a dynamic MFA was conducted to explore the characteristics of the Dutch housing stock. Müller developed a dynamic stock-driven building stock model for simulating inflows and outflows of the Dutch housing stock till 2100 (Müller 2006). In the model, turnover of the housing stock is driven by population, floor area per capita, and building lifetime probability distribution. We adjusted Müller's model with more recent data and additionally established REB and REN scenarios (as illustrated in 5.3.3). The outcome of this model was used as an implementation size factor to scale up the environmental and economic impacts of the PCE system, and the requirements for new build houses in the different scenarios. Based on the flows of the model, we estimated the CDW generation and material requirement in three scenarios. In addition, according to the stock information, we simulated the operational energy use according to the vintage cohorts of the housing stock. At the country level, we also additionally considered changes over time in GHG emissions and material footprints of electricity supply, and the operational energy use for heating and cooling, cooking, hot water supply, electrical appliance use, and lighting.

Calculations of actual-value approach and annualized-value approach

We express these indicators using an actual value approach and an annualized-value approach. At the product level, the actual-value approach calculates the GHG emissions, costs, and material footprints of (re)constructing and operating a new or existing house for 36 years (2015–2050); at the country-level, it adds up actual GHG emissions, costs, and material footprints of the energy use of the Dutch housing stock and construction and demolition activities each year to a total in that year for the period 2015–2050.

At the product level, the annualized-value approach calculates the operational energy use the same way as the actual-value approach but distributes GHG emissions, costs, and material footprints, which occur in the construction and demolition phases, as annualized values per year per m^2 per housing type over the lifetime of a house. The annualized country-level results in a year between 2015 and 2050 can be obtained by multiplying annualized GHG emissions, costs, and material footprints of construction, and demolition activities per m^2 in that year with the total m^2 per type of building present in that year to a total. In the annualized-value approach, the annualized construction and demolition impacts and costs of an existing house were also accounted for till this house is demolished.

The annualized-value approach is more suitable for comparing scores for a sound functional unit, such as the impacts and costs per m^2 floor space in use per year, including impacts and costs of the construction phase and benefits of recycling and reuse at the end-of-life (EoL) stage. The actual-value approach provides insights regarding when impacts and costs occur and when initial impacts and costs of refurbishing are compensated by lower impacts and costs in the use phase. Illustrations of these two approaches can be found in the supporting information (SI).

Evaluations of the 2030 and 2050 circularity goals

Based on the country-level actual values, we further analyzed whether the proposed PCE system can realize the circularity goals. Regarding the circularity goal, the REN scenario assumes that sufficient secondary materials are available. Realistically, however, the supply of secondary raw materials might be insufficient to support extensive energy renovation, especially secondary raw materials made from emerging materials, such as lightweight concrete waste. We further considered the supply-demand conditions of secondary raw materials in the realization of the circularity goal. The Circular Economy Program for the Netherlands 2050 proposed a preliminary interim target of 50% less use of material footprints (minerals, fossil fuels, and metals) by 2030 and a long-term goal of a fully circular economy by 2050 (Dijksma and Kamp 2016). This program is rather general in its formulation. For instance, it does not specify a base year, nor does it quantify target volumes for the construction sector for the 2030 and 2050 goals. In consultation with the Ministries of Infrastructure and Environment and Economic Affairs and Climate Policy, Potting et al. (2018) set 2014 as the base year. Considering the temporal scope of our system, we set 2015 as the base year. For the 2050 circularity goal, the Netherlands is supposed to achieve a fully circular economy, qualitatively defined as 'efficient use of material and high-grade reuse of secondary materials while without yielding any harmful emissions into the environment (Dijksma and Kamp 2016). In the EU's Waste Hierarchy, recovery comprises recycling and downcycling, in which recycling represents a high value-added treatment method (EC 2008a). Therefore, we measured the 2050 goal in a simple way via recycling rate and material carbon footprint. The REB and BAU scenarios were assumed to have an identical recycling system, which is in accordance with the real situation of how each fraction in CDW is treatment in the Netherlands (given in the supporting information(SI)). In the REN, concretes waste, glass waste, and insulation mineral wool waste were assumed cost-effectively recycled through advanced ADR, HAS, and DGR systems.

Evaluations of the 2030 and 2050 decarbonization goals

The country-level actual-value results were also used to discuss the realization of the circularity goals. The Netherlands and the EU both set decarbonization goals to facilitate the transition to a carbon-neutral society. The EU represented its member states to commit to reducing GHG emissions by at least 40% by 2030 and by 80%–95% by 2050 compared to 1990 (EZK 2019). In the Climate Act, the Netherlands set an ambitious interim target with a reduction by 49% of gross national emission by 2030 and a long-term goal of 95% by 2050 (Government of the Netherlands 2019). The associated carbon budget was allocated to the Dutch residential built environment. To comprehensively reflect the thermal insulating features of the panels, ideally one would consider both heating and cooling demand for each house. However, regarding the real household cooling demand in the Netherlands, air conditioners are usually installed in the non-residential sector and only 6% of all dwellings have an air conditioning system (Menkveld and Beurskens 2009). In addition, energy demand for cooling accounts for less than 1% of electricity use in the Dutch housing sector (PBL 2018). Therefore, cooling was not taken into account in the evaluation of realizing the carbon-neutral goal.

The emissions from the built environment only include emissions from natural gas and electricity use for heating, cooking, hot water supply, electric appliance use, and lighting. Moreover, in the EU, new buildings constructed after 2020 must meet the standards of nZEB (EC 2010). Two nZEB solutions, in which thermal transmittance of each housing element were introduced, were simulated in three scenarios to explore how the passive energy efficiency approach can influence GHG emissions.

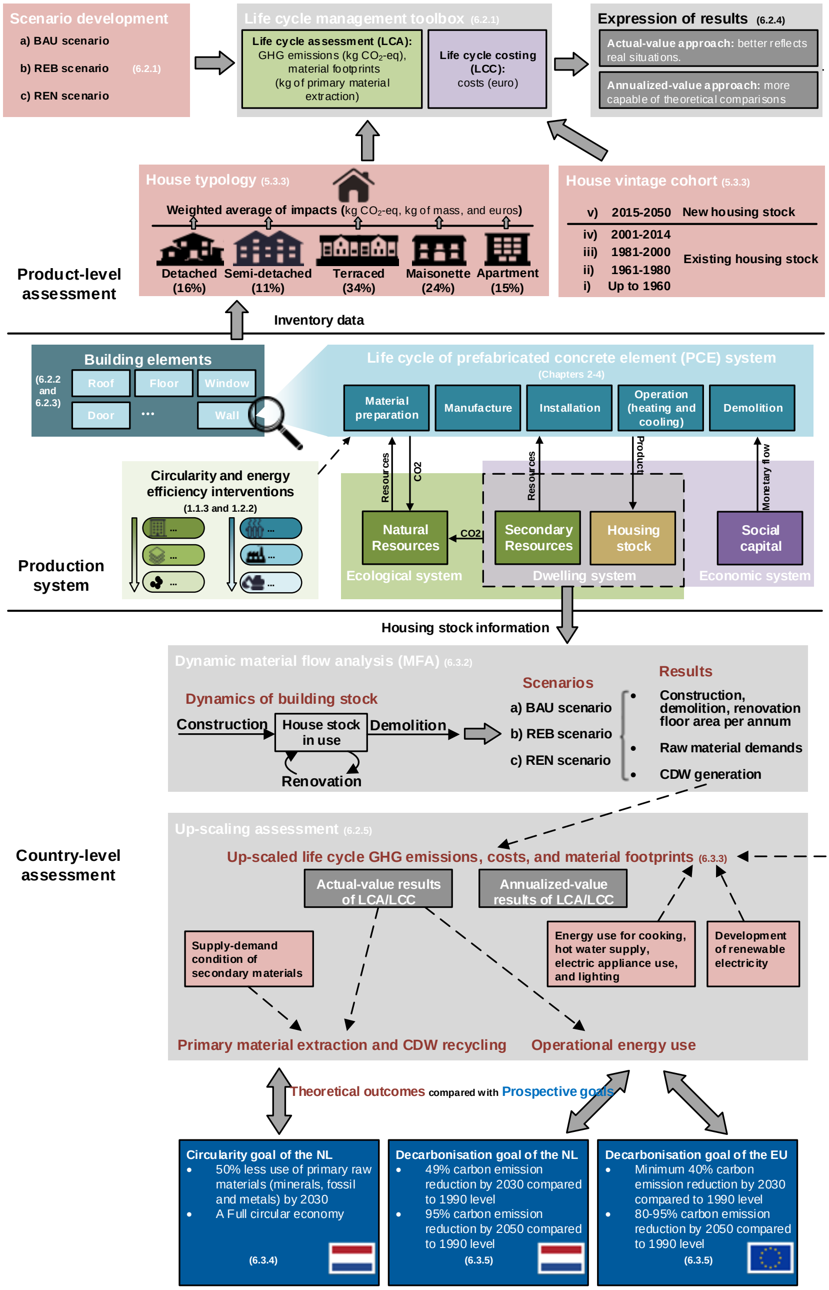


Figure 6.1 Methodological framework of integrated life cycle assessment and life cycle costing with dynamic material flow analysis. BAU: business-as-usual, EU: European Union, NL: The Netherlands, REB: demolition and rebuilding, REN: refurbishment with the PCEs.

6.2.1.2 Scenario development

Three scenarios were developed in this study: BAU, REB, and REN. The characteristics of the three scenarios are presented in Table 6.1. The BAU scenario (i) uses baseline PCE-new, which only includes virgin materials, for constructing walls of new buildings; (ii) does not implement any renovation method for existing buildings; and (iii) does not implement any reuse strategies.

The REB scenario is similar to the BAU scenario. The difference is that in the REB scenario old buildings are demolished early and replaced with new buildings with higher energy efficiency performance, so that by 2050 all houses are constructed after 2014, with equal amounts of houses rebuilt per year starting with the oldest vintage cohorts.

The REN scenario assumes that all existing buildings will be refurbished by 2050 using green PCE-refurb, with equal amounts of houses being refurbished per year starting with the oldest vintage cohorts. The scenario assumes (i) that an innovative recycling system is applied to produce high-quality secondary materials; (ii) application of green PCE-new for building new houses and green PCE-refurb for refurbishing old buildings and (iii) that green PCE-refurb is reused at its EoL stage.

Table 6.1 Characteristics of three scenarios for product- and country-level assessment

Scenario	At building level	At element level	At material level
BAU	**At product level:** New building: (i) is implemented with the baseline PCE-new, which do not contain secondary raw materials and use expanded polystyrene as insulation in walls; (ii) of which other envelopes (floors, roofs, windows) have high insulation level; (iii) uses air-conditioning and heat pump for space cooking and heating. Existing building: (i) with poor insulation level envelopes; (ii) uses air-conditioning and condensing boiler for space cooling and heating; (iii) is demolished when become obsolete; **At country level:** The up-scaled assessment additionally considered (i) the influence of developing renewable electricity over time; (ii) operational energy use for	**At product level/ country level:** Each component of the baseline PCE-new is recovered through traditional waste treatment systems.	**At product level:** Only virgin raw materials are used. **At country level:** Only primary raw materials are used.

	cooking, hot water supply, electric appliance use, and lighting.		
REB	**At product level:** New building: same as the BAU. Existing building: (i) in poor insulation level of envelopes; (ii) uses air-conditioning and condensing boiler for space cooling and heating; (iii) is demolished early and then to rebuild conforming to the requirement of a new building. **At country level:** Same as the BAU.	**At product level/ country level:** Same as the BAU.	**At product level/ country level:** Same as the BAU.
REN	**At product level:** New building: (i) is implemented with green PCE-new, which contain high volumes of secondary raw materials and use aerogel as insulation in walls; (ii) of which other envelopes have high insulation level; (iii) uses air-conditioning and heat pump for space cooking and heating. Existing building: (i) in poor insulation level of envelopes; (ii) uses air-conditioning and condensing boiler for space cooling and heating; (iii) is implemented with green PCE-refurb to over-clad the wall for optimizing the thermal performance as well as prolonging the lifetime. **At country level:** Same as the BAU.	**At product/country level:** PCE-refurb is dismantled for reuse; each component of a green PCE-new is recovered through advanced waste treatment systems.	**At product level:** Secondary raw materials are used to substitute certain portions of primary raw materials in the manufacturing of green PCE-new and PCE-refurb. **At country level:** Secondary raw materials are assumed sufficiently supplied.

6.2.1.3 Functional unit of the LCA and LCC

The functional units for the LCA and LCC were: provisioning the service of habitable floor area. The product-level assessment aims to evaluate how the PCE system improves the thermal performance of a building. Therefore, only the energy savings for heating and cooling were considered at the product level. At the country level, the habitable surface not only comprises heating and cooling comfort but also includes cooking, hot water supply, electrical appliance use, and lighting.

- The product-level functional unit: maintaining the indoor thermal comfort by active heating and cooling of 1 m^2 habitable floor area per year of (i) existing houses per vintage group (BAU scenario), (ii) demolished and rebuilt houses (REB scenario), and (iii) refurbished houses with green PCE-new and green PCE-refurb (REN scenario) in the Netherlands.

- The country-level functional unit: providing the total amount required habitable floor area in the Netherlands while maintaining the indoor thermal comfort by active heating and cooling per year for the period 2015-2050. As indicated, we added also the energy use for cooking, hot water supply, electrical appliance use, and lighting in the country-level assessment.

6.2.2 Life cycle inventory of GHG emissions, costs, and material footprints

In this section, we discuss the method used to collect data for the central parameters in this study: GHG emissions, costs, and material footprints. Given the scenarios presented previously, the following elements must be considered:

1. Three types of PCE panels: (embodied) GHG emissions, costs, and material footprints for (i) baseline PCE-new, used for constructing the wall of new houses in the BAU and REB scenario; (ii) green PCE-new, used for new houses in the REN scenario; (iii) green PCE-refurb, used for refurbishing existing houses in the REN scenario;

2. Existing houses: (embodied) GHG emissions, costs, and material footprints for constructing per m^2 of existing house, and gas and electricity use per vintage per m^2/year in the operation phase;

3. New and rebuild houses: (embodied) GHG emissions, costs, and material footprints per m^2 of rebuild houses in the production phase, and gas and electricity use per m^2/year in the operation phase;

4. Refurbished houses: (embodied) GHG emissions, costs and material footprints per m^2 of refurbishing with PCE-refurb in the production phase, and gas and electricity use per m^2/year in the operation phase;

5. Other unit processes: costs related to electricity and gas use, house demolition, and CDW generation and recycling, nZEB standards.

Below, we explain the approach for obtaining this inventory information. Detailed inventory data are presented in the SI. For the heating and cooling of Dutch houses, we assumed the use of gas boilers or electric heat pumps. For electricity, we assumed changes in the electricity mix for the Netherlands, leading to carbon-neutral electricity provision in 2050. The Supporting Information provides more detail.

6.2.2.1 PCE panels: baseline PCE-new, green PCE-new and green PCE-refurb

The proposed PCE system presents a potential solution for simultaneously realizing the circularity and carbon neutrality goals. From the waste prevention aspect, prefabricated construction has been seen as a solution to minimize CDW arising in the construction phase (Tam et al. 2006), as well as during the EoL phase owing to the dismantlability of the precast components. In the REN scenario, we discern two PCE systems: green PCE-new and PCE-refurb. was assumed as the insulating layer for green PCE-refurb and green PCE-new in the REN scenario. The main difference is that the green PCE-new is implemented in new buildings and uses normal-weight concrete as a component and cannot be reused, whereas green PCE-refurb is implemented in existing buildings, use lightweight concrete as a component, and can be reused (see Figure 6.2). For new buildings in the BAU and REB scenarios, a reference baseline PCE-new was used, which has the same dimensions as the green PCE-new but only contains primary raw materials and uses expanded polystyrene as insulation materials.

The hypothetical lifetime of the baseline/green PCE-new is determined by the average lifetime of a new house, which is 120 years (Zhang et al. 2021c). It is not yet clear whether the PCE-new can be reused. The green PCE-refurb are reusable after the end of life of a refurbished house (Zhang et al. 2021a). Hence, in this study, we assumed that only green PCE-refurb were reused when the refurbished houses were demolished. At the material level, recycled materials are assumed to be used to manufacture the green PCE-new and green PCE-refurb in the REN scenario to reduce the depletion of raw materials, as described in earlier studies (Moreno-Juez et al. 2020; Zhang et al. 2019a; Gebremariam et al. 2020; Zhang et al. 2020a, 2021a, 2021b). The GHG emissions, costs, material footprints related to the three PCEs, baseline PCE-new, green PCE-refurb, and green PCE-new are all based on this earlier work and illustrated in the SI.

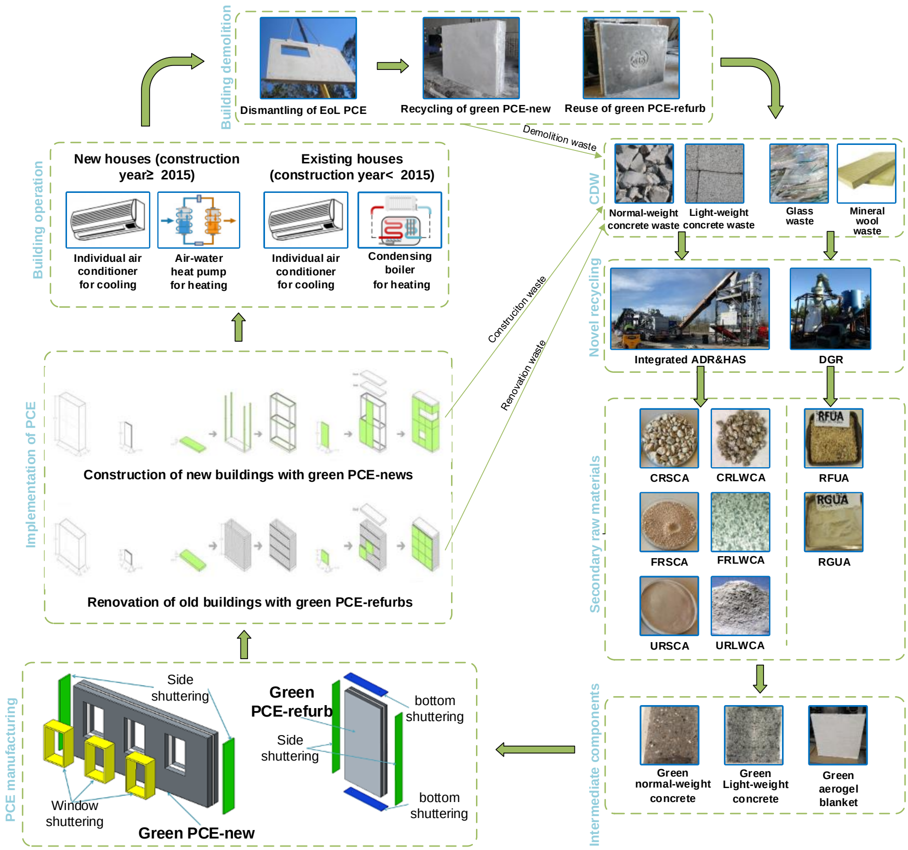


Figure 6.2 Technological route of the novel PCE system for the REN scenario. ADR: Advanced dry recovery system; CDW: construction and demolition waste; CRLWCA: coarse recycled light-weight concrete aggregate; CRSCA: coarse recycled normal-weight concrete aggregate; DGR: Dry grinding and refining system; FRLWCA: fine light-weight recycled concrete aggregate; FRSCA: fine siliceous normal-weight recycled concrete aggregate; HAS: Heating-Air Classification System; PCE-new: prefabricated concrete element for new building construction; PCE-refurb: prefabricated concrete element for existing building renovation; RFUA: recycled fiber wool ultrafine aggregate; RGUA: recycled glass ultrafine aggregate; URLWCA: ultrafine recycled light-weight concrete aggregate; URSCA: ultrafine recycled siliceous normal-weight concrete aggregate.

6.2.2.2 Existing houses: construction, and operation phase data

For existing houses, we need to have insight into the GHG emissions, costs, and material footprints of construction, and energy requirements in the operation phase. The GHG emissions and material footprints of constructing per 1 m^2 of existing houses were assumed the same as that of new houses. Construction costs of existing houses is 711.60

€/m^2 with an uncertainty range of (541.20, 881.56). It was adjusted based on the current construction costs and historical house price index (Arcadis 2017; CBS 2021a).

The operational energy for heating (assumed by condensing gas boilers) and cooling (assumed by individual air conditioners) by vintage per m^2/year, and energy for cooking, hot water supply, electric appliance use, and lighting in general. We assumed that the insulation level of the walls of a house was determined by the vintage of construction. Following a report of PBIE, we divided the Dutch houses into five cohorts based on construction vintages of houses: (i) up to 1960, (ii) 1961–1980, (iii) 1981–2000, (iv) 2001–2014, and (v) since 2015 (Economidou 2011). In general, the thermal performance of walls of residential buildings improves over time. The thermal transmittance of the wall presents an increasing trend from 2.70 W/(m ^{2}K) in the 1960s to 0.40 W/(m ^{2}K) in the 2000s (Economidou 2011). The thermal transmittance of the wall of existing buildings constructed up to 1960 is 2.70 W/(m ^{2}K); buildings constructed in 1961 to 1980 is 1.00 W/(m ^{2}K); that in 1981 to 2000 is 0.60 W/(m ^{2}K); that in 2000 to 2014 is 0.40 W/(m ^{2}K).

The operational energy demand for space heating and cooling of houses of different vintage cohorts was estimated. We estimated the yearly heating and cooling requirements per m^2, assumed to be provided by a traditional condensing gas boiler and an individual air conditioner using electrical power (TABULA 2017). We conducted dynamic thermal simulations to obtain the yearly heating and cooling demand of houses in the climate condition of the Netherlands. The yearly heating and cooling demands for exiting housing cohorts range from 52.17–99.80 and 1.31–1.85 kWh/(m ^{2}annum), respectively. The gas and electricity consumption for cooking, hot water supply, electric appliance use, and lighting are 83.92 MJ/(m ^{2}annum) and 0.0241 kWh /(m ^{2}annum), respectively, according to Statistics Netherlands (PBL 2018). We further linked the yearly energy uses with the unit processes to estimate the consequent GHG emissions, costs, and material footprints. We refer to the detailed inventory data in the SI.

6.2.2.3 New and rebuild houses: production (construction) and operation phase data

The EU Directive 2010/31/EU required that the member states must ensure that new buildings constructed after 2020 meet the requirements of nZEB (EC 2010). As there is no evidence of large-scale renovation in the Netherlands in the past, for the sake of simplicity, we assumed that the Netherlands could start with renovation for low-energy use in 2015 and build new houses according to nZEB standards in 2015. Thus, new houses in this assessment refer to houses that have been constructed since 2015. These houses have excellent insulation levels and use efficient heating systems. Houses constructed since 2015 were assumed to be equipped with gas-free heating systems, i.e. an air-water heat pump (TABULA 2017).

Construction phase: GHG emissions, costs and primary material use per m^2

For material requirements regarding the construction of new houses, Arnoldussen et al. reported the material intensity for constructing 1 m^2 of different types of houses in the Netherlands (Arnoldussen et al. 2020). Based on the stock share of each type of house and emission intensities, the weighted material intensities for constructing a habitable floor area are as follows, 1,198.52 kg/m^2, 36.57 kg/m^2 of wood, 62.60 kg/m^2 of steel, 25.20 kg/m^2 of glass, 18.57 kg/m^2 of insulation, and 12.27 kg/m^2 of gypsum, amounting to 1,353.74 kg/m^2 in gross. We linked these material intensities with primary material extraction indicators and unit emission indicators from the Ecoinvent database to obtain the GHG emission and material footprint per m^2. The weighted emission and material footprints for constructing a habitable floor area are 338.87 kg CO_2 eq/m^2 and 6342.27 kg/m^2. We assumed that the manufacturing impacts of heat pumps were included in these emissions and resource uses. Arcadis NV reported construction costs for different types of houses in the Netherlands (Arcadis 2017). From this, a weighted average-based construction costs according to the share per housing type to stock is estimated at 904.18 euros/m^2 with an uncertainty range of 687.66–1120.14 euros/m^2. We assumed that the costs of heat pumps and air conditioners were included in these costs.

In the BAU and REB scenarios, we assumed that the impacts and costs of the PCE-new facades are included in the aforementioned data. The REN scenario however uses green PCE new instead of baseline PCE new for new build houses. For new build houses in the REN scenario hence we added the difference in impacts and costs between baseline PCE new and green PCE new. This data was taken from previous work (Zhang et al. 2020a). In short, each m^2 of habitable floor area requires a weighted average-based share in gross house stock across the different housing types (0.57 m^2) of baseline/green PCE-new for cladding the fa çade of new houses (Zhang et al. 2021c). The production emission, cost, and material footprint of baseline PCE-new for per m^2 floor area are 98.03 kg CO_2 eq/m^2, 285.40 euros/m^2, and 463.85 kg/m^2, respectively. For the green PCE-new, 142.95 kg CO_2 eq/m^2, 286.97 euros/m^2, and 415.69 kg/m^2, respectively. In the REN scenario, the impacts and costs of construction and demolition of 1 m^2 new house are hence modified by adding the differences (44.92 kg CO_2 eq/m^2 of GHG emissions, 1.57 euros/m^2 of costs, and –47.89 kg/m^2 of primary material extractions) of the green PCE-new from the baseline PCE-new. Consequently, the GHG emissions, costs, and primary material extractions of constructing per m^2 of new habitable floor area in the REN scenario is 383.79 kg CO_2 eq/m^2, 937.58 euros/m^2, and 6294.10 kg/m^2.

Use phase: energy requirements per m^2/year

Regarding energy requirements of new houses, the yearly heating and cooling demands for new houses constructed with baseline PCE-new are 29.25 kWh/(m^2annum) and 3.35 kWh/(m^2annum), respectively. For houses constructed with green PCE-new, the heating and cooling demands are 25.90 kWh/(m²annum) and 3.84 kWh/(m²annum), respectively. For new houses, energy cooking, hot water supply, electric appliance use, and lighting were assumed using electricity, which is 23.33 kWh/(m²annum) (PBL 2018). Associated GHG emissions, costs, and primary material depletions were calculated based on energy uses. The material intensities, yearly electricity use, and related costs per m^2/year for newly built houses in detail are presented in the SI.

6.2.2.4 Refurbished houses: production (refurbishing) and operation phase data

We refer to existing buildings as houses built before 2015. Compared with new buildings, elements of existing buildings have a lower insulation level.

Production phase: GHG emissions, costs and material use per m^2

Refurbishment assumes that the exterior walls are over-cladded by green PCE-refurb. The GHG emissions, costs, material footprints for producing the amount of green PCE-refurb (0.57 m^2) for refurbishing per floor area are modified based on our previous study Zhang et al. (2021a), which are 101.28 kg CO_2 eq/m^2, 119.77 euros/m^2, and 387.86 kg/m^2, respectively.

For annualized values calculations, the green PCE-refurb are assumed to be reused when the refurbished houses are demolished in the annualized-value approach. The reused green PCE-refurb are assumed to be repaired and then used to refurbish other houses. We assumed a 10% loss of mass when a green PCE-refurb is dismantled and reused (Zhang et al. 2021a). Following this, the reuse of green PCE-refurb in the REN scenario was modeled as reducing 90% of the impacts from material production, PCE-refurb manufacture, and PCE-refurb disposal. We further assume that refurbishing will extend the lifetime of the building by 15 years (Duurzaam Gebouwd 2014).

Use phase: energy requirements per m^2/year

After refurbishment, the thermal insulation level of the walls was improved with a range of 0.14–0.20 W/(m²K)), leading to lower heating and cooling requirements per m^2/year. The thermal transmittance of the refurbished wall of existing buildings constructed up to 1960 is 0.203 W/(m²K); that in 1961 to 1980 is 0.180 W/(m²K); that in 1981 to 2000 is 0.161 W/(m²K); that in 2000 to 2014 is 0.142 W/(m²K). The consequent heating and cooling demands per annum are 44.67–46.68 and 2.29–3.35 kWh/(m²annum), respectively. The energy requirements for cooking, hot water supply, electric appliance use, and lighting remain the same. The associated GHG emissions, costs, and material footprint were estimated according to yearly energy use. Detailed environmental and economic inventory data for refurbishing are given in the SI.

6.2.2.5 Other inventory data

The Dutch housing stock, consisting of a mix of existing, refurbished and new build houses need a mix of gas and electricity for heating and cooling and other purposes. We used Ecoinvent 3.4 to assess GHG emissions, costs, and primary material uses. The costs of space heating by gas boilers were estimated at 0.010 euro/MJ and heat pumps were estimated at 0.011 euro/MJ, respectively, and assumed to be constant. For electricity supply, we assumed changes in the electricity mix for the Netherlands, leading to carbon-neutral electricity provision in 2050 (see the SI). Costs for electricity were estimated at 0.21 euro/kWh and assumed constant (Eurostat 2020).

In the different scenarios, at different moments in time, houses are demolished. In this study, we only consider the demolition impacts of new houses. For demolition, the carbon emission at the demolition stage is usually assumed 10% of the construction emission (Lu and Wang 2019). Thus, the demolition emission is 33.89 kg CO_2 eq/m^2. The demolition costs of a house are dependent on the scale of the demolition project and the amount of hazardous asbestos. The average demolition cost of buildings constructed after 2000 is 39.50 euro/m^2 with an uncertainty range of 31.00–48.00 euro/m^2; buildings constructed before 2001 have a demolition cost of 72.00 euro/m^2 with a range of 40.00–104.00 euro/m^2. Primary material use of demolition is 1.94 kg/m^2, excluding disposal of CDW.

Regarding the circularity goal, we estimated the generation of construction waste, demolition waste, and renovation waste from the Dutch housing sector and associated secondary raw materials. The waste intensities for per m^2 of construction, demolition, renovation per floor are 40.61 kg/m^2, 1,208.27 kg/m^2, and 30.61 kg/m^2, respectively (Zhang et al. 2021c). The recycling rate of each CDW in the Netherlands was estimated based on the study (Mulders 2013).

As for the decarbonization goals, there is however no defined nZEB standard. Two databases ZEBRA2020 (2020) and TABULA (2017) proposed preliminary nZEB standards for the Netherlands (given in the SI), including the requirement of thermal transmittance and heating system. To reflect the extent to which the nZEB solution can facilitate the residential sector to achieve the decarbonization goals, we compared new buildings implemented with the PCE-new system from this study with the new buildings with the insulation requirement of the two databases.

6.2.3 Life cycle environmental and economic impact assessment

In terms of impact assessment, we further evaluated GHG emissions, costs, and primary material extraction incurred from the construction, operation, and demolition phases at a temporal and country-wise scale. GHG emissions were expressed in GWP100 according to the report of IPCC (2007). Material footprints were measured in kg of total primary material requirement (Mostert and Bringezu 2019). The costs were calculated in euros. Given the current near-zero interest rates (De Nederlandsche Bank 2020), future costs were not discounted. The LCC was conducted from a hybrid perspective (combined perspective of a consumer and a manufacturer) to investigate the real cash flows incurred (Zhang et al. 2021a).

6.2.3.1 Impacts over time of (re)building, refurbishing, demolishing, and energy use

Because we calculated both environmental and economic parameters via an actual-value and an annualized-value approaches, in Figure 6.3, we illustrate the timing of activities in the different scenarios. This study takes 2015 as the time for prospective policy interventions, thus impacts and costs that occurred before 2015 were seen as "sunk costs" and were not considered (Fuller and Petersen 1995). The REN scenario assumes that all existing buildings will be refurbished by 2050, with equal amounts of houses being refurbished per year. Refurbishment is assumed to start with the building cohort constructed in the oldest vintage. The REB scenario follows a similar pattern, with demolition and reconstruction rather than refurbishment. Further, in all scenarios, including BAU, some housing cohorts reach their natural end of life and are demolished and rebuilt. From these findings, we derived three phases characterized by different activities: (re)building or refurbishment, use/operations, and demolition. The longest period is the operation phase with operational impacts due to heating, cooling, etc. Construction and demolition activities usually require 1–2 years. The construction impacts of a new building are accounted for at the beginning of each year, and demolition impacts are incurred at the end of each year. Operational costs and impacts incurred for the period are always accounted for. While the costs and impacts of construction and demolition were handled differently by using actual-value and annualized-value approaches, as shown in Table 6.2. The differences are further elaborated in the next section.

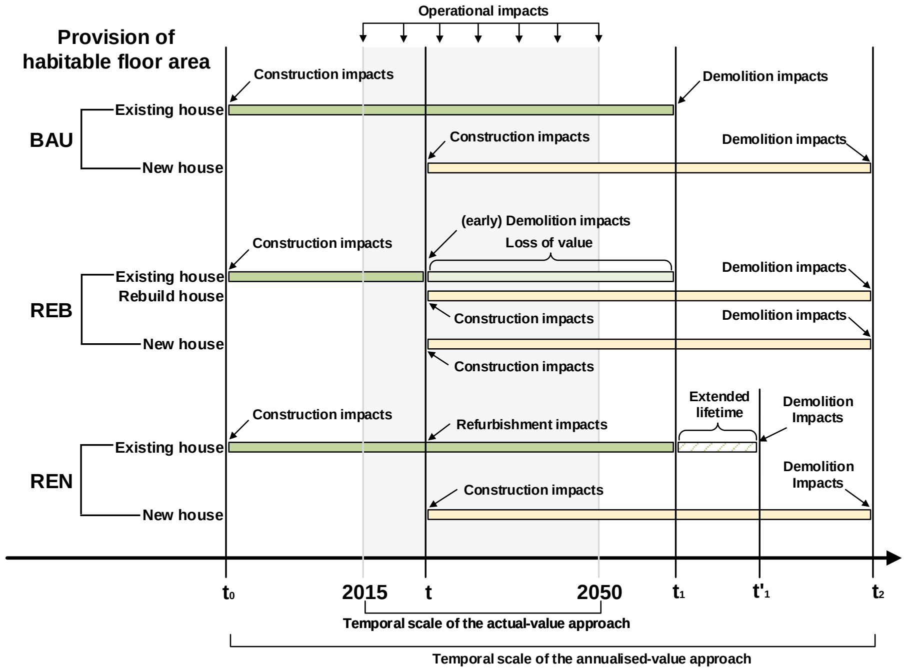


Figure 6.3 Illustration of actual and annualized impacts (GHG emissions, costs, and material footprints) of existing buildings and new buildings in three scenarios. Construction impacts include material production and use of labor, energy, facilities. In the figure, *t* is a snapshot of a discrete year between the temporal scale (2015, 2050), t_0 is the discrete year of construction of the hypothetic existing house, t_1 is the discrete year of demolition of a hypothetic existing house, t_1' is the discrete year of demolition of the hypothetic existing house after refurbishment, t_2 is the discrete year of demolition of a hypothetical new house.

Table 6.2 Costs and impacts of construction and demolition in the actual-value approach and annualized-value approach

	Actual-value approach	Annualized-value approach
Construction	Costs & impacts < 2015 were neglected/irrelevant.	Costs & impacts < 2015 were considered.
	Costs & impacts within 2015-2050 were considered.	Costs & impacts within 2015-2050 were considered.
	Costs & impacts >2050 were neglected/irrelevant.	Costs & impacts >2050 were neglected/irrelevant.
Demolition	Costs & impacts < 2015 were neglected/irrelevant.	Costs & impacts within < 2015 were neglected/irrelevant.
	Costs & impacts within 2015-2050 were considered.	Costs & impacts within 2015-2050 were considered.

	Costs & impacts > 2050 were neglected/irrelevant.	Costs & impacts > 2050 were considered.

6.2.4 Actual-value and annualized-value approach at a product level

The results of the LCA and LCC were expressed in actual and annualized values. The actual-value approach was used to analyze whether the circularity and decarbonization goals at specific moments in time are realized, while the annualized-value approach was applied for comparing different production systems. Based on Figure 6.3 and Table 6.2, the actual product-level impacts of the three scenarios are estimated using Eq. (1). As it can be seen, only costs and impacts of (re)construction, operation, and demolition activities incurred for the period 2015-2050 were considered in the actual-value approach.

The annualized-value approach is more complex. Until 2014, the stocks of existing houses in the three scenarios were the same. Consequently, the construction impacts and costs of existing buildings were incurred at the same time points in the three scenarios. Due to rebuilding and refurbishment, the construction and demolition impacts of some existing houses in the REB and REN scenarios would occur earlier and later, compared with the BAU scenario. Because the time value of money was not considered, the gross impacts of demolition of existing housing stock are identical. However, if annualized-value approach accounting was applied, lifetime extension leads to lower annualized impacts of construction and demolition of existing houses. Similarly, early demolition leads to higher annualized impacts and costs of construction and demolition of existing houses. Construction costs and impacts of existing houses were regarded as sunk ones in the actual-value approach but were considered in the annualized-value approach using Eq. (2).

$$Actual\ product-level\ impacts\ =\begin{cases} I_{O(an)}^{(exi)} + I_{D}^{(exi)}, & existing\ house\ in\ BAU \\ I_{O(an)}^{(exi)} + I_{D}^{(exi)}, & exsting\ house\ in\ REB \\ I_{O(an)}^{(ren)} + I_{R}^{(exi)}, & exsiting\ house\ in\ REN \\ I_{C}^{(new)} + I_{O(an)}^{(new)}, & new\ house\ in\ BAU, REB, REN \end{cases} \quad (1)$$

$$Annualised\ product-level\ impacts\ =\begin{cases} I_{O(an)}^{(exi)} + \frac{I_{C}^{(exi)} + I_{D}^{(exi)}}{t_1 - t_0}, & existing\ house\ in\ BAU \\ I_{O(an)}^{(exi)} + \frac{I_{C}^{(exi)} + I_{D}^{(exi)}}{t - t_0}, & exsting\ house\ in\ REB \\ I_{O(an)}^{ref} + \frac{I_{R}^{(exi)}}{\left(t_1' - t\right)} + \frac{I_{C}^{(exi)} + I_{D}^{(exi)}}{t_1' - t_0}, & exsiting\ house\ in\ REN \\ I_{O(an)}^{(new)} + \frac{I_{C}^{(new)} + I_{D}^{(new)}}{t_2 - t}, & new\ house\ in\ BAU, REB, REN \end{cases} \quad (2)$$

where t is a snapshot of a year between the temporal scale (2015, 2050), t_0 is the year of constructing an existing building, t_1 is the year of demolishing an existing building, t_1' is the year of demolishing an existing building after refurbishment, t_2 is the year of demolishing a new building, $I_C^{(exi)}$ is the impacts (both GHG emissions, costs, and material footprints) of constructing an existing building, $I_D^{(exi)}$ is the impact of demolishing an existing building, $I_C^{(new)}$ is the impact of constructing a new building, $I_D^{(new)}$ is the impact of demolishing a new building, $I_R^{(exi)}$ is the impact of production, installation, EoL treatment of the PCE-refurb, $I_{O(an)}^{(new)}$ is the impact of operating a new building per year, $I_{O(an)}^{(exi)}$ is the impact of operating an existing building, and $I_{O(an)}^{(ren)}$ is the impact of operating a refurbished existing building per year.

6.2.5 Up-scaling both actual and annualized values at a country level

Eq. (1) and Eq. (2) indicate how the life cycle environmental and economic impacts of construction, refurbishment, operation, and demolition activities at the product level. The yielded actual values and annualized values were further up-scaled by combining them with building stock information. The up-scaled actual-value impacts of in year *t* in the BAU, REB, and REN scenarios are estimated using Eq. (3). The up-scaled annualized-value impacts in year *t* in the BAU, REB, and REN scenarios are formulated as Eq. (4).

$$Actual\ country-level\ impacts=\begin{cases}\sum_{V=1}^{V=5} S_{(BAU,V)}(t) I_{O(an)}^{(BAU,V)}+F_C(t) I_C^{(new)}+F_D(t) I_D^{(exi)}, & BAU\\ \sum_{V=1}^{V=5} S_{(REB,V)}(t) I_{O(an)}^{(REB,V)}+\left(F_C(t)+F_R(t)\right) I_C^{(new)}+\left(F_D(t)+F_R(t)\right) I_D^{(exi)}, & REB\\ \sum_{V=1}^{V=5} S_{(REF,V)}(t) I_{O(an)}^{(REF,V)}+F_C(t) I_C^{(new)}+I_R^{(exi)} F_R(t)+F_D(t) I_D^{(exi)}, & REN\end{cases} \tag{3}$$

$$Annualised\ country-level\ impacts=$$
$$\begin{cases}\sum_{V=1}^{V=5} S_{(BAU,V)}(t) I_{O(an)}^{(BAU,V)}+\sum_{V=1}^{V=4} S_{(BAU,V)}(t) \frac{(I_C^{(exi)}+I_D^{(exi)})}{t_1-t_0}+\sum_{2015}^{t} F_C(t) \frac{I_C^{(new)}+I_D^{(new)}}{t_2-t}, & BAU\\ \sum_{V=1}^{V=5} S_{(REB,V)}(t) I_{O(an)}^{(REB,V)}+\sum_{2015}^{t}\left(F_R(t)+F_C(t)\right) \frac{I_C^{(new)}+I_D^{(new)}}{t_2-t}+\sum_{V=1}^{V=4} S_{(REB,V)} \frac{I_C^{(exi)}+I_D^{(exi)}}{t-t_0}, & REB\\ \sum_{V=1}^{V=5} S_{(REN,V)}(t) I_{O(an)}^{(REN,V)}+\sum_{2015}^{t} F_R(t) \frac{I_R^{(exi)}}{(t_1'-t)}+F_C(t) \frac{I_D^{(new)}}{t_2-t}+\sum_{V=1}^{V=4} S_{(REN,V)}(t) \frac{(I_C^{(exi)}+I_D^{(exi)})}{t_1-t_0}+\sum_{2015}^{t} F_R(t) \frac{I_C^{(exi)}+I_D^{(exi)}}{t_1'-t_0}, & REN\end{cases} \tag{4}$$

where v (=1,2,3,4,5) represents five vintage cohorts of houses in the Netherlands (up to 1960, 1961–1980, 1981–2000, 2001–2014, 2014-2050), $I_{O(an)}^{(BAU,V)}, I_{O(an)}^{(REB,V)}, I_{O(an)}^{(REN,V)}$ are operation impacts of a house in vintage cohort *v* in different scenarios, $F_R(t)$ is refurbished/rebuilt floor area in year *t*, $F_C(t)$ is newly constructed floor area in year *t*, $F_D(t)$ is demolition floor area in year *t*, $S_{(BAU,V)}(t), S_{(REB,V)}(t),$ and $S_{(REF,V)}(t)$ are housing stocks of vintage cohorts *v* in year *t* in different scenarios.

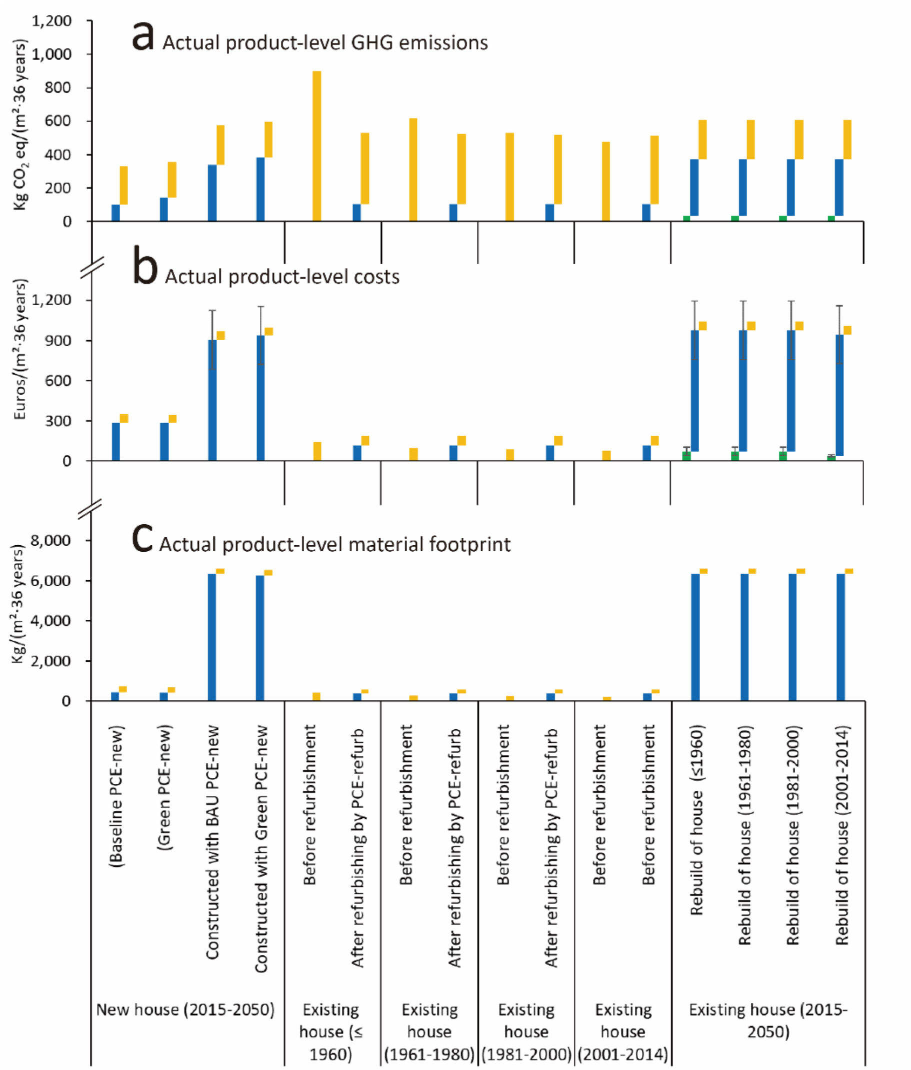
6.3 Results

6.3.1 Product-level GHG emissions, costs, and material footprints

The actual-value and annualized-value GHG emissions, costs, and material footprints of each system are presented in Figure 6.4. The temporal scale of the actual-value approach is 2015-2050. Results of actual-value and annualized-value approaches are analogous to some degree but also could lead to different conclusions. The green PCE-new system does not show an apparent economic and environmental advantage over the baseline PCE-new system, using both actual-value and annualized-value modeling.

The GHG mitigation, cost saving, and material footprint reduction of green PCE-refurb are more significant than those of the green PCE-new. However, the performance of the PCE-refurb system is highly dependent on the vintage of the building that is to be refurbished. Implementing PCE-refurb in buildings constructed before 1960 leads to significant GHG mitigations, cost savings, and resource use reductions. These benefits decrease if the building is in a newer cohort. From the economic point of view, the green PCE-refurb solution is costlier than walls without refurbishment using an actual-value approach but shows lower life cycle costs using an actual-value approach. This is because reuse is expected to yield considerable financial benefits using an actual-value approach.

In addition, rebuild of house emits more GHG than refurbishment using an actual-value approach whereas has lower life cycle GHG emissions using an annualized-value approach. However, rebuilding is significantly costly and resource-intensive in both actual-value and annualized approaches, therefore, not economically and material-efficiently feasible.


a Actual product-level GHG emissions
Kg CO_2 eq/(m²·36 years)
1,200
1,000
800
600
400
200
0
b Actual product-level costs
Euros/(m²·36 years)
1,200
900
600
300
0
c Actual product-level material footprint
Kg/(m²·36 years)
8,000
6,000
4,000
2,000
0
(Baseline PCE-new)
(Green PCE-new)
Constructed with BAU PCE-new
Constructed with Green PCE-new
Before refurbishment
After refurbishing by PCE-refurb
Before refurbishment
After refurbishing by PCE-refurb
Before refurbishment
After refurbishing by PCE-refurb
Before refurbishment
Before refurbishment
Rebuild of house (≤1960)
Rebuild of house (1961-1980)
Rebuild of house (1981-2000)
Rebuild of house (2001-2014)
New house (2015-2050)
Existing house (≤ 1960)
Existing house (1961-1980)
Existing house (1981-2000)
Existing house (2001-2014)
Existing house (2015-2050)

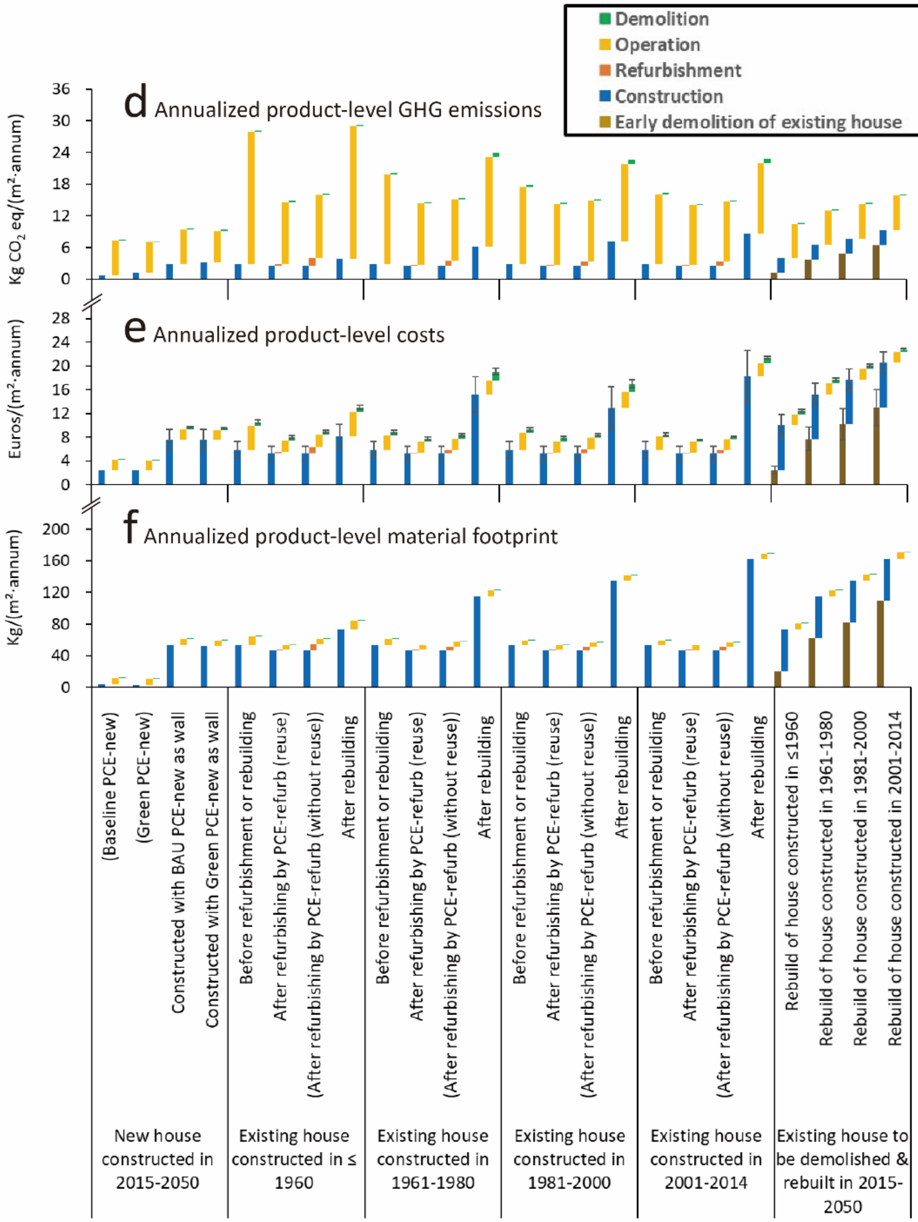


Figure 6.4 Actual-value (a) GHG emissions, (b) life cycle costs, and (c) life cycle material footprints and annualized-value (d) GHG emissions, (e) life cycle costs, and (f) life cycle material footprints of each technological system. Note: At a product level, GHG emissions, costs, and material footprints from cooking, hot water supply, electric appliance use, and lighting were not included. The electricity mix remains at the 2015 level for the period 2015–2050. The operation impacts of the actual-value

approach in (a)-(c) were accounted for the period 2015-2050. The "early demolition of existing house" in Panel (d)-(f) is the difference between "before refurbishment and rebuilding" and "after rebuilding". The "36 years" mean the time span from 2015 to 2050.

6.3.2 Dynamics of building stock and material requirements

Based on the dynamic MFA model, we investigated the inflow and outflow of the stock per annum, turnover of the stock, and material use from 2015 to 2050 as shown in Figure 6.5. Building stocks, inflows and outflows, and material use were modeled at the end of the discrete-time interval. The BAU and REN scenarios have the same construction and demolition floor area per year, based on the expected end-of-life of houses in the existing building stock of 2015. In the REN scenario, the annual floor area that is renovated is 16.95 million m^2, amounting to 593.10 million m^2 in total. The REB scenario has far more construction and demolition floor areas owing to extensive reconstruction. In the REB scenario, 16.95 million m^2 of old buildings are demolished and then rebuilt as opposed to being renovated. However, three scenarios have an identical gross habitable surface per year, as shown in panels d, e, and f of Figure 6.5. Panels g, h, and i of Figure 6.5 show the main materials used in the three scenarios. The material use in the REN scenario was slightly higher than that in the BAU scenario. The material use in the REB scenario was approximately two times more than that in the BAU and REN scenarios. Details of the dynamic MFA model are presented in the SI.

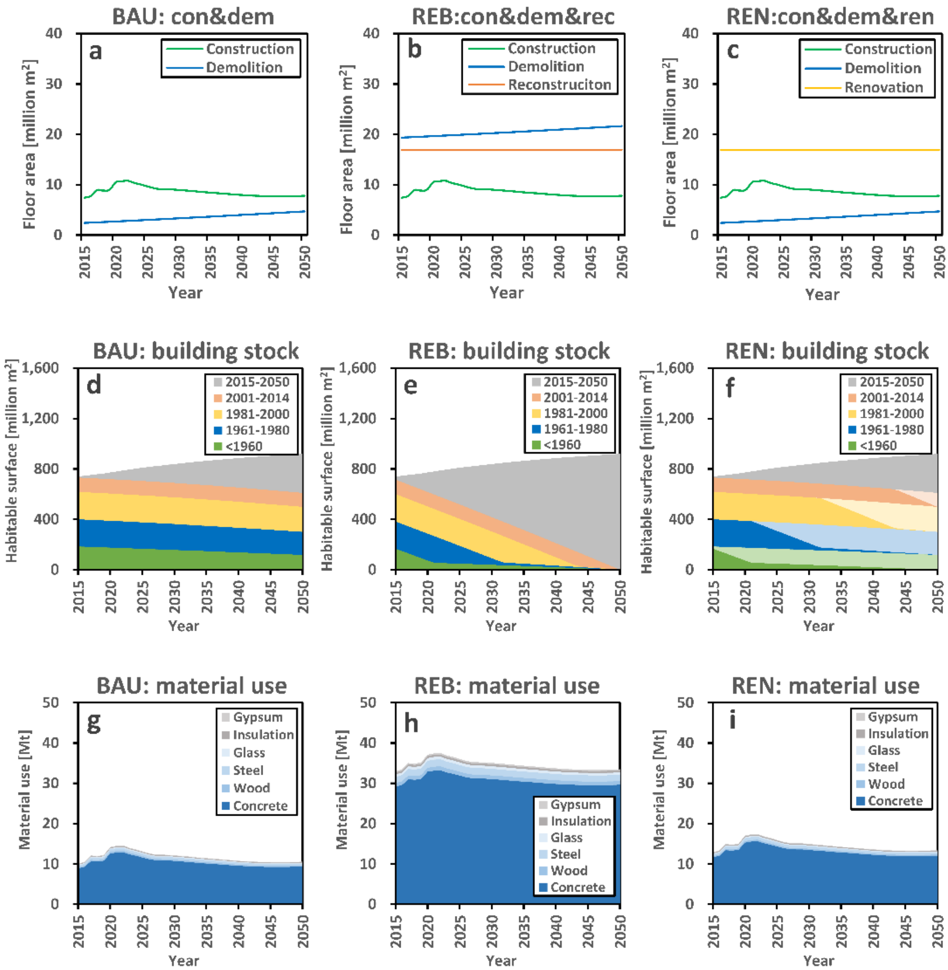


Figure 6.5 Dynamics of construction, demolition, and renovation floor area and residential building stocks of three scenarios for 2015–2050. The slightly lighter-colored areas in each construction cohort area in panel (f) represent refurbished building stocks. “Material use” in panels (g)-(i) does not mean primary raw material use but the use of products.

6.3.3 Country-level GHG emissions, costs, and material footprints

The environmental and economic information from the LCA and LCC are scaled up to the total flooring area in the Netherlands by combining them with stock information from the building stock model. We compared the 2015 electricity mix with a dynamic electricity mix with an increasing share of renewables up to 2050, in each scenario. The share of renewable sources is 10% in 2015 and will increase to 50% in 2030 (EZK 2015) and 100% in 2050 (EZK 2019). The up-scaled actual-value and annualized-value life cycle of GHG emissions, costs, and material footprints for the BAU, REB, and REN scenarios are shown in Figure 6.6. Details of the renewable electricity modeling are presented in the SI.

Our actual-value approach (Figure 6.6a) shows a clear trade-off between embodied emissions and operational emissions. Owing to additional material use, the REB and REN scenarios had higher emissions in 2015, compared with the BAU scenario. These embodied emissions from the REN scenario are paid back by 2027, after which there is a net environmental benefit. The REB scenario emits much more GHG emissions compared to the BAU, and cannot compensate for the increased embodied emissions during the modeled time. This is because heat pumps and other appliances using grey electricity cannot create gains over efficient gas boilers. However, this situation is reversed when the development of renewable electricity is considered. The payback periods of the REN scenario remain the same. The REB scenarios can return in 2038 and will yield less GHG emissions than the REN scenario from 2039 onwards. From an economic perspective, although the REN scenario has much lower life cycle costs than the REB scenario, both REB and REN cannot reduce costs compared to the BAU scenario (see Figure 6.6b). Material footprints of three scenarios (Figure 6.6c) present the same trend as the cost results. However, it is noteworthy that the supply of cleaner electricity considerably reduces GHG emissions but slightly mitigates the material footprints.

The annualized-value approach provides additional insights compared to the actual-value approach. With this approach, the REB scenario shows a similar emission tendency compared to the BAU scenario if the electricity mix remains steady (see Figure 6.6d). Emissions from the REN scenario will be noticeably less than that of the other two scenarios. Considering the development of renewable electricity, the emissions of the three scenarios would be reduced to different extents, most significantly for the REB scenario. From an economic perspective, the REN scenario can save expenses to some degree compared with the BAU scenario (Figure 6.6e). However, the REB scenario is more expensive and not a cost-effective option. From the material footprint perspective, the REB scenario requires noticeably more primary materials than the other two scenarios (Figure 6.6f). The REN scenario has lower material footprints compared to with BAU scenario using an annualized-value approach. Note that material footprints of the REB and REN scenarios (Figure 6.6f) both show a descending trend, which does not indicate that construction and renovation activities in the future are less material-intensive. For the REB scenario, the decreasing material footprints result from additional demolition of existing houses, which leads to a cessation of the annualized construction and demolition material use of existing houses. While as for the REN scenario, it is because that refurbishment reduced the annualized construction and demolition material use of existing houses.

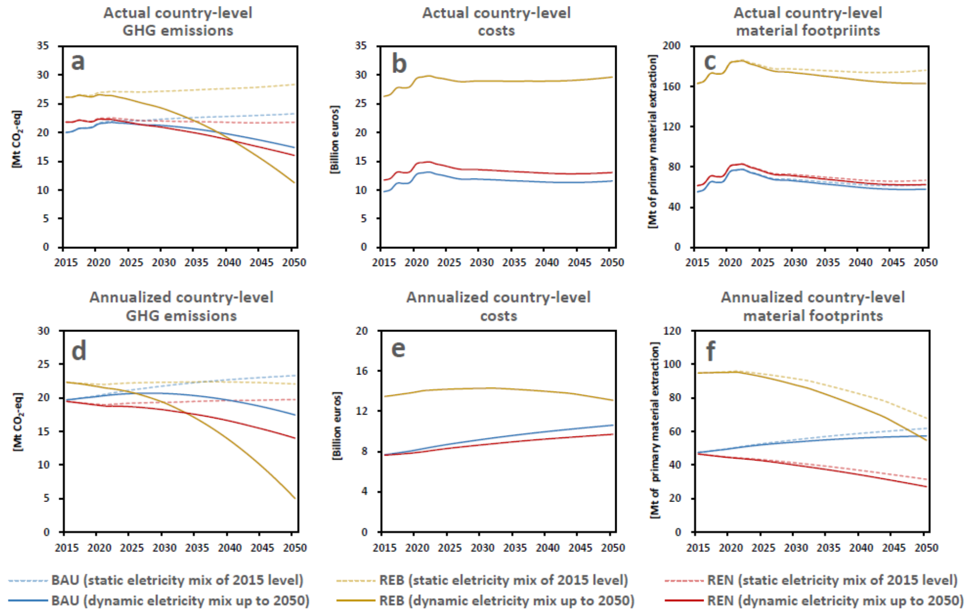


Figure 6.6 (a-c) Actual and (d-f) annualized country-level GHG emissions in [kg CO_2 eq], costs in [billion euros], and material footprints in [Mt of primary material extraction] of the housing sector in the Netherlands for 2015–2050. Note: impacts of cooking, hot water supply, electric appliance use, and lighting were included. It is assumed the costs of electricity and gas remain steady-state of the 2015 level.

6.3.4 Circularity goal: halving material footprint by 2030 and achieving full circularity by 2050

For the 2030 circularity goal, the 50% reduction in material footprints can be measured either in general or for each resource individually (Potting et al. 2018; Bastein et al. 2017). We measured the material footprints in general as shown in Figure 6.7a. None of the three scenarios can reduce the overall material footprint. Given that building stock undergoing an extensive energy renovation, the Netherlands is less likely to achieve the circularity goal by 2030, even with advanced recycling systems in place.

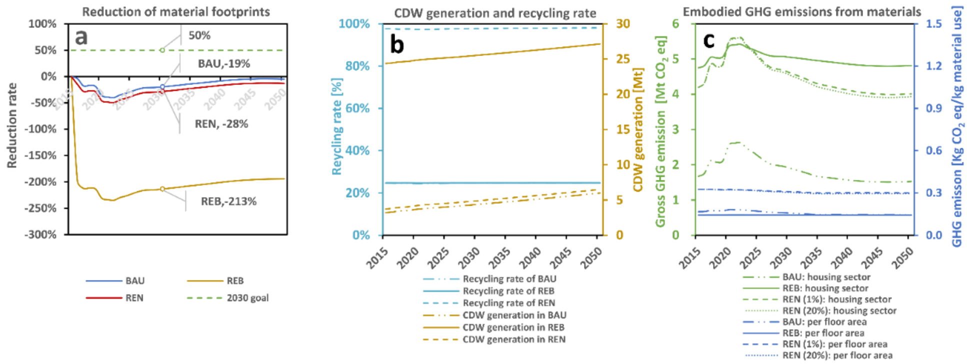


Figure 6.7 (a) Reduction rate of material footprints of three scenarios compared with 2015 level of the BAU scenario, (b) the generation of construction and demolition waste and recycling rate of three scenarios, (c) embodied GHG emissions of material uses of three scenarios. Note: The “(1%)” in (c) represents the light-weight concrete waste accounts for 1% in the gross normal-weight concrete waste from 2015 to 2050; the “(20%)” indicates the share of the light-weight concrete waste linearly increases from 1% in 2015 to 20% in 2050.

Figure 6.7b shows the waste generations and recycling rates of three scenarios. Owing to intensive demolition and reconstruction, the REB scenario has a relatively large amount of CDW generation compared to the other two scenarios. The recycling rates of the BAU and REB scenarios remain stable at approximately 25% till 2050 if no additional efforts to upgrade the current CDW treatment system are undertaken. These scenarios assume a recycling rate of concrete waste less than 5%. The REN scenario has a significantly higher recycling rate, approximately 98% because the ADR and HAS technological systems can fully recycle concrete waste.

Figure 6.7c shows the embodied GHG emissions associated with materials in the three scenarios. For each scenario, the gross embodied GHG emissions account for both materials for PCE and also other materials used for constructing new buildings. The gross GHG emissions from material use show an ascending trend until 2023 and then a long continuous descending trend. This relates to vast construction activities in the period from 2017-2037, as shown in Panel a, b, and c of Figure 6.5. The REN scenario generally has lower GHG emissions than the REB scenario, but is over two times higher than that of the BAU scenario. When the share of lightweight concrete waste remains steady at 1% of gross concrete waste, the secondary materials are not sufficient for the production of green PCE-refurb. The deficit was assumed substituted by virgin raw materials. When the share increase to 20% in 2050, the GHG emissions of the REN scenario would just be slightly reduced. We further investigated the GHG emissions per kg material use of three scenarios by dividing the gross GHG emissions by gross material uses (see Panel g, h, and i of Figure 6.5). As shown in Figure 6.7c, the carbon footprint per kg material use of the REN scenario is twice that of the REB and BAU scenarios. This is because concrete is the primary material for rebuilding, and energy renovation depends heavily on synthetic insulation materials such as expanded polystyrene and aerogel that have a higher carbon footprint compared to concrete.

6.3.5 Decarbonization goal: reduction of 49% GHG emissions by 2030 and 95% by 2050

In this section, we evaluated whether the three scenarios BAU, REB, and REN can achieve the interim and long-term decarbonization goals. The effects of adopting green electricity and applying different insulation levels are assessed. The energy mix for electricity production in the Netherlands over time was presented in Figure 6.8a. The share of renewable sources is 10% in 2015 and will increase to 50% in 2030 (EZK 2015), and will achieve 100% renewable electricity production by 2050 (EZK 2019).

We additionally added two scenarios in which the thermal insulation performance of new houses meets the insulation requirement of nZEB. The “BAU/REB/REN-a” scenario represents the insulation level of the new houses in those scenarios that meet the nZEB. insulation requirement from the ZEBRA2020 database (2020). The “BAU/REB/REN-b” denotes the insulation level of the new houses in those scenarios that meet the nZEB insulation requirement from the TABULA database (2017). Figure 6.8b shows the historical and prospective emissions from the residential sector of the three previously defined scenarios and six additional scenarios. For BAU-related scenarios, without renovation and reconstruction and with the implementation of high-insulation facades and electrical appliances for new buildings, the residential sector would be unable to achieve the carbon-neutral goal in 2030 and 2050. The three REN scenarios have the potential to fulfil the decarbonization goal by 2030 but fail to reach a carbon-neutral level by 2050. Only the REB scenario appears suitable for achieving both mid-term and long-term goals.

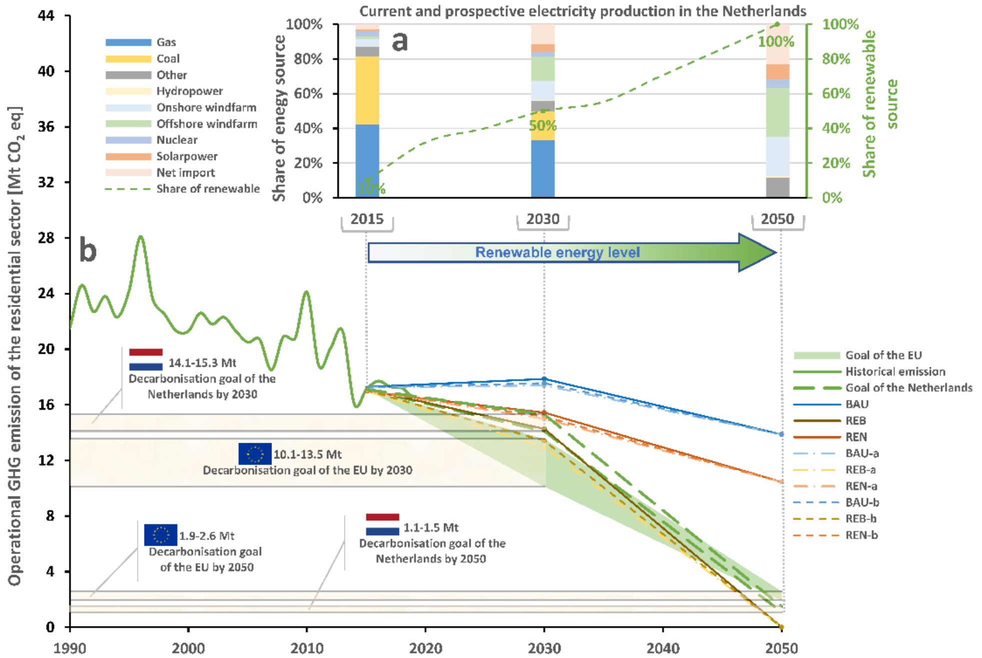


Figure 6.8 (a) Current and prospective electricity production and (b) greenhouse gas emission of the residential sector in the Netherlands. Note: Only operational emissions were considered. The "BAU/REB/REN-a" represents the insulation level of the new houses that meet the nearly zero energy building (nZEB) insulation requirement from the ZEBRA2020 database (2020). The "BAU/REB/REN-b" denotes the insulation level of the new houses that meet the nZEB insulation requirement from the TABULA database (2017).

6.4 Discussion

The discussion section mainly focuses on the application of the actual-value and annualized-value modeling, the challenges and opportunities in realizing the circularity and decarbonization goals of the built environment. First, we expressed the results of LCA and LCA. This results from the mismatch of temporality between LCA/LCC and MFA, as shown in Figure 6.9. Synchronic life cycle time was assumed in the LCA and LCC (120 years), while in MFA an exact diachronic time of 2015-2050 was explored. Therefore, under the temporal scope of the MFA, circularity interventions, such as building lifetime extension, recycling, and reuse of materials that occur after 2050, cannot be included. To overcome this drawback, an annualized-value approach was introduced to take into account the benefits from PCE reuse and building lifetime extension.

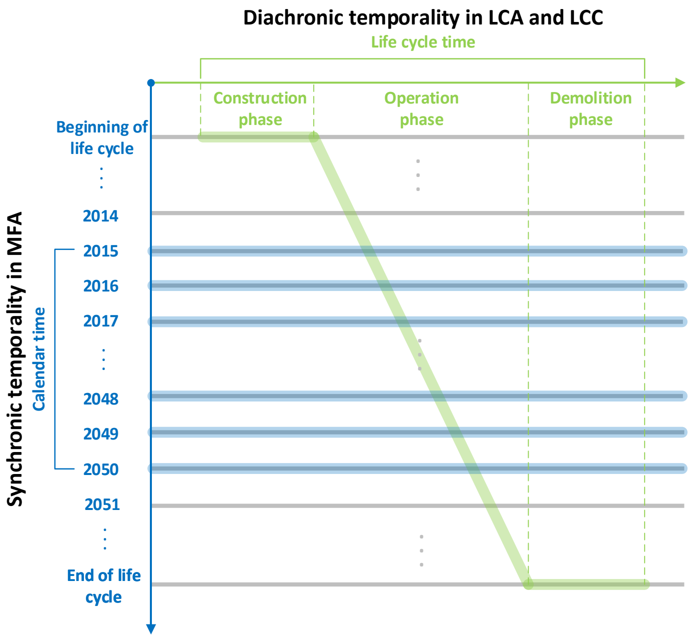


Figure 6.9 Temporality in material flow analysis, and life cycle assessment/life cycle costing. Modified based on the research (Birat 2015).

It is noteworthy that the selection of different estimation approaches led to different conclusions. The annualized-value approach is more capable of theoretical comparison as it can take into account the sunken costs and future benefits; while the actual-value approach better reflects real situations. The annualized-value approach shows that the REN scenario leads to lower GHG emissions, costs, and material footprint compared to the BAU scenario. The REB scenario shows a greater GHG mitigation potential than the other two scenarios but also leads to higher costs and material footprint, using the annualized-value approach. The actual-value approach reveals a clearer trade-off between GHG emission and capital investment and between energy use and material depletion in energy renovation. Therefore, the essence of energy renovation (especially deep renovation) is to sacrifice capital investment and material for lower GHG emissions and energy use. Moreover, combining use of actual-value and annualized-value approaches can provide comprehensive insights that help policymakers, for instance, in assessing how the use of carbon budgets and cost-effective strategies evolves over time.

In our assessment of the 2030 circularity goal, reduction of material footprints was achieved through recycling, lightweight design such as using lightweight concrete and aerogel in green PCE-refurb, and using renewable energy carriers. However, lightweight design was only applied to the green PCE-refurb whereas will lead to a more noticeably less use of material if implemented on new buildings. For example, by applying lightweight design in all steel and aluminum products, the material required can be reduced by up to 30% (Carruth et al. 2011). These factors induced the overestimation of material footprints. Note that we did not consider the development of cleaner electricity in the manufacturing sector as operational energy use dominates the life cycle energy use of a building.

For the 2050 circularity goal, our results imply that the REN scenario has the potential to boost the recycling rate to almost 100%, but unfortunately, we also find this scenario brings with it more embodied GHG than the BAU scenario. In the REN scenario, green aerogel is extensively used in green PCEs. However, the technological maturity of the aerogel is still at a lab scale, production per unit of aerogel needs a large amount of grey electricity and raw materials compared with producing per unit of baseline insulation expanded polystyrene. Thus, the primary focus of future innovation in the PCE system should be a cleaner production process for green aerogel. In addition, secondary materials yielded from the ADR, HAS, and DGR technological systems have relatively lower life cycle environmental impacts compared to virgin material, although we note that these results are highly sensitive to choices in allocation method and impact category (Zhang et al. 2019a, 2020a). Currently, these recycling systems still rely on diesel to operate. At a commercialized scale, fossil fuels will have to be replaced by cleaner energy sources.

Evaluation of the decarbonization goals reveals the roadmap towards a low-carbon and energy-efficient built environment. With regard to the passive approach, improving the insulation of building elements in the REN proved to prevent heat loss compared to the BAU scenario. The thermal transmittance of new building elements (walls, floors, glazing, roofs) in the REN is higher than the requirement of nZEB for houses in the Netherlands. This is because the PCE-refurb is specific for implementation in cold zones such as the Netherlands, while the PCE-new system is currently designed for all climatic areas in the EU. Therefore, the insulating performance of the PCE-new should be further specified in cold, cool, and warm zones. Conversely, although the thermal insulation level of the façade in the nZEB requirement is superior to the baseline and green PCE-new, the BAU/REB/REN-a and -b scenarios still cannot noticeably reduce GHG emissions compared with the BAU/REB/REN scenarios. As of the active approach, the use of renewable energy is a crucial factor affecting the decarbonization of the residential sector. Refurbishment by the PCE system can realize the interim goal of carbon mitigation of 49% by 2030 but cannot further support the realization of the long-term goal. A nearly net-zero goal by 2050 can be achieved if nationwide reconstruction is implemented. Reconstruction intervention is theoretically feasible. However, it reflects a crucial point that implementing energy renovation only by a passive approach, such as refurbishment with PCE-refurb, is insufficient to achieve the long-term decarbonization goal. The replacement of traditional gas-based systems (such as boilers, water heaters, and gas stoves) with electricity-based systems and the supply of 100% renewable electricity are two decisive conditions for shifting to a carbon-neutral society.

6.5 Conclusions

Regarding the issues of energy inefficiency and the surging volumes of CDW in Europe, the PCE system delivers an energy-efficient improvement in new building construction and existing building renovation by high-value recycling CDW as secondary raw material in the production of PCE-new and PCE-refurb. Our assessment integrated a top-down, stock-driven building stock model with LCA and LCC to explore the product-level and up-scaled performance of the proposed PCE solutions for new building construction and existing building renovation in the Netherlands from 2015 to 2050. Three scenarios, BAU, REB, and REN, were established. The BAU scenario does not apply a baseline PCE-new or include any renovation strategies for old buildings. The REN scenario implements the green PCE-new and green PCE-refurb system for material-efficient energy renovation. The REB scenario demolishes and rebuilds old buildings instead of renovating them. The building stock model was used to simulate the construction, demolition, renovation floor area per annum, and stock cohorts of the three scenarios. The LCA and LCC were deployed to estimate the life cycle environmental and economic impacts of the proposed PCE system, as well as building reconstruction. The results of the LCC and LCA were up-scaled using the stock and flow information from the dynamic building stock model.

The product-level and up-scaled results were expressed as both actual values and annualized values. The annualized-value approach is more capable of theoretical comparison, while the actual-value approach better reflects real situations. The selection of different estimation approaches led to different conclusions. The actual-value approach revealed the trade-offs between GHG emissions and investment/material use in energy renovation. However, the annualized-value approach proved the PCE system achieved comprehensive benefits compared to the BAU scenario.

We further evaluated the extent to which the circularity and decarbonization goals can be realized by up-scaling the proposed PCE system to the residential building sector. The Netherlands aims to achieve a reduction of 50% material footprints by 2030 compared to the 2015 level. The results show the BAU and REN scenarios cannot reduce resource uses till 2050. Regarding the circularity goal by 2050, advanced technological systems in the REN scenario can raise the recycling rate from the current 25% to 98%. However, the associated embodied emissions of the REN scenario is twice that of the BAU scenario, due to the use of carbon intensive aerogels as insulation. Therefore, the Netherlands is less likely to achieve the circularity goal by 2030 and 2050 with the proposed PCE system, unless a cleaner production process for aerogel is in place. It should be noted that we did not consider the development of renewable electricity for manufacturing processes in the study.

Furthermore, operational GHG emissions of the residential building stock in 2030 and 2050 were quantified under different scenarios. We found that if no energy renovation strategies are applied, the baseline BAU scenario cannot achieve the decarbonization goal by 2030 or 2050. REN implementing the PCE system for extensive energy renovation can achieve an interim goal of 49% carbon mitigation by 2030 but still cannot realize the carbon-neutral level by 2050. Only our REB scenario, in which traditional gas-based heating systems are replaced with electricity-based systems and supplied with renewable electricity, can reach the long-term carbon goals.

Acknowledgements

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Appendix

Table A6.1 Material intensity for new houses (2015-2050) in the Netherlands. Note: As for new houses that are constructed after 2015, Arnoldussen et al. (2020) reported the material intensity for constructing 1 m^2 of a new detached house, semi-detached house, terraced house, maisonette, and apartment in the Netherlands. Concrete accounts for more than 80% of material use by weight in new buildings. Since "other" fraction makes up around 1% of the gross material requirement, its GHG emissions were not considered. Regarding the material intensity of existing houses that were constructed from 1900 to 2014, we assumed it is the same as the material intensity of new houses (Zhang et al. 2020b).

Building type	Share of house	Material intensity [Kg/m^2]	Concrete [Kg/m^2]	Wood [Kg/m^2]	Steel [Kg/m^2]	Glass [Kg/m^2]	Insulation [Kg/m^2]	Gypsum [Kg/m^2]	Other [Kg/m^2]
Detached house	15.98%	1,200	1,048.60	42.06	33.64	23.83	19.63	19.63	12.62
Semi-detached house	11.39%	1,430	1,238.99	53.35	51.62	25.81	18.93	27.53	13.77
Terraced house	33.60%	1,120	965.01	51.22	40.43	25.61	17.52	8.09	12.13
Apartment and maisonette	39.03%	1,630	1,449.12	16.83	96.75	25.24	18.93	8.41	14.72

Table A6.2 Weighted material intensities (in Table A6.1) and their GHG emission indicators

Mateiral	Weighted material intensity [Kg/m^2]	Emission indicator [kg CO_2 eq/kg]	Reference process in Ecoinvent 3.4
Concrete	1198.52	0.14791	BAU siliceous concrete production
Wood	36.57	-1.68752	"planing, board, softwood, u=20%-GLO"
Steel	62.60	2.41765	"steel production, converter, low-alloyed-ReR"
Glass	25.20	0.99312	"flat glass production, uncoated-GLO"
Insulation	18.57	0.69539	"polystyrene foam slab production, 100% recycled | polystyrene foam slab | Cutoff, U – RoW"
Gypsum	12.27	0.01063	"gypsum production, mineral | gypsum, mineral | Cutoff, U-GLO"

Table A6.3 GHG emission indicators for construction and demolition of houses. Note: materials account for around 90% of gross carbon emission in the construction stage of a building, in contrast to manpower, energy and equipment around 8%, 1% and 0.1% (Shao et al. 2014). Therefore, it was assumed the emission of manpower, energy and equipment account for 10% of the material emission. The carbon emission at the demolition stage is deemed equal to 10% of the construction stage (Lu and Wang 2019).

	Construction emission (kg CO_2 eq/m^2)		Demolition(kg CO_2 eq/m^2)
	Material	Energy and equipment	
Emission indicator	304.98	33.89	33.89

Table A6.4 Construction costs of different types of new houses after 2015 in the Netherlands [€/m^2] (Arcadis 2017)

Building type	Share of housing	Lowest level	Highest level	Average level
Detached house	15.98%	600	1,175	888
Semi-detached house	11.39%	650	1,080	865
Terraced house	33.60%	600	800	700
Apartment and maisonette	39.03%	810	1,385	1,098

Table A6.5 Price index of house construction costs for the period 1995 to 2020 (CBS 2021a).Note: Due to lack of data, we simplified the estimation of the construction costs for existing houses from 1900 to 2014 by using the average house price of the period 1995 to 2014. This may result in overestimation as prices of an old house are lower. However, we also noticed from the reports that the real house prices in the Netherlands did not dramatically adjust from 1965 to 1995 (Vries 2010). The estimated construction costs of houses from 1900 to 2014 is 711.60 €/m^2 with an uncertainty range of (541.20, 881.56).

Years	Price index of house construction costs	Estimated construction costs [€/m^2]
1995	90.1	558.37
1996	90.7	562.09
1997	92.4	572.63
1998	94.9	588.12
1999	96.5	598.04
2000	100.0	619.73
2001	105.2	651.95
2002	109.3	677.36
2003	111.5	690.99
2004	113.7	704.63
2005	115.9	718.26
2006	119.3	739.33
2007	124.1	769.08
2008	129.8	804.40
2009	130.0	805.64
2010	130.8	810.60
2011	133.4	826.71
2012	135.7	840.97
2013	136.0	842.83
2014	137.2	850.26
2015	139.8	866.38

2016	142.6	883.73
2017	145.9	904.18
2018	149.6	927.11
2019	153.8	953.14
2020	157.3	974.83

Table A6.6 Demolition costs of buildings in the Netherlands [euro/m^2](Arcadis 2017). Note: the mean value of those costs is selected to estimate the costs of demolition of old houses. The Netherlands in 1993 and the EU in 2005 banned the use of asbestine materials in construction (Government of the Netherlands 2020), the asbestos is only included in the demolition waste of old buildings. Hence, we assumed demolition costs of buildings that were constructed before 2000 needs to consider costs for asbestos removal. The average demolition costs of buildings constructed after 2000 is 39.50 euro/m^2 with an uncertainty range of (31.00, 48.00); buildings constructed before 2001 is 72.00 euro/m^2 with an uncertainty range of (40.00, 104.00).

	Lowest level	Highest level
Demolition costs for small scale project	31.00	38.00
Demolition costs for large scale project	36.00	48.00
Costs for asbestos removal	9.00	56.00

Table A6.7 Thermal transmittance [$W/(m^2 K)$] of each building element. Note: assumptions based on TABULA database (TABULA 2017); b data from BPIE report (Economidou 2011); c calculated results based on ISO 6946 (ISO 2017b); d data from ZEBRA2020 database (2020); e data from the TABULA database from the EPISCOPE project (TABULA 2017).

Element	Existing building	Existing building with PCE-refurb	New building with PCE-new	nZEB-ZEBRA-NL [d]	nZEB-EPISCOPE - NL [e]	nZEB-EPISCOPE for EU Member States [e]
Floor	3.370 [a]	3.370 [a]	0.181[a]	0.130	0.110	0.060-0.320
Roof	0.520 [a]	0.520 [a]	0.118 [a]	0.110	0.080	0.060-0.480
Window	1.300 [a]	1.300 [a]	1.300 [a]	0.780	1.000	0.500-2.800
Wall in general	/	/	/	0.110	0.100	0.090-0.480
Wall (≤ 1960)	2.700 [b]	0.203 [c]	/			/
Wall (1961-1980)	1.000 [b]	0.180 [c]	/	/	/	/
Wall (1981-2000)	0.600 [b]	0.161 [c]	/	/	/	/

Wall (2001-2014)	0.400 [b]	0.142 [c]	/	/	/	/
Wall-baseline PCE-new(⩾2015)	/	/	0.317 [a]	/	/	/
Wall-green PCE-new(⩾2015)	/	/	0.190 [a]	/	/	

Table A6.8 Energy demand per annum for space heating and cooling of different envelope systems. Note: According to Norm DIN V 18599(DIN 2011) for split air conditioning (>12kW), the seasonal energy efficiency ratio is around 4.7. This indicated that 4.7 kWh of cooling is supplied with 1 kWh of electricity.

House	Heating demand [kWh/(m²·annum)]	Cooling demand [kWh/(m²·annum)]	Electricity used for cooling [kWh/(m²·annum)]
Existing house (⩽1960)	99.80	1.31	0.28
Existing house (1961-1980)	67.74	1.40	0.30
Existing house (1981-2000)	58.03	1.64	0.35
Existing house (2001-2014)	52.17	1.85	0.39
Existing house (⩽1960) refurbished with PCE-refurb	46.68	2.29	0.49
Existing house (1961-1980) refurbished with PCE-refurb	45.85	2.07	0.44
Existing house (1981-2000) refurbished with PCE-refurb	45.26	2.07	0.44
Existing house (2001-2014) refurbished with PCE-refurb	44.67	2.08	0.44
New house (⩾2015) with baseline PCE-new	29.25	3.35	0.71
New house (⩾2015) with green PEC-new (⩾2015)	25.90	3.84	0.82

Table A6.9 Reference processes for the production of primary and secondary material

Primary material	Remark and sources
Concrete	Foreground process “BAU normal-weight siliceous concrete production”
wood	“planing, board, softwood, u=20% | sawnwood, board, softwood, dried (u=20%), planed | Cutoff, U-RoW”
Steel	“steel production, converter, low-alloyed | steel, low-alloyed | Cutoff, U-RER”
Glass	“flat glass production, uncoated | flat glass, uncoated | Cutoff, U-RER”
Insulation-EPS	To use EPS to represent the insulating material in a new building: “polystyrene foam slab production, 100% recycled | polystyrene foam slab | Cutoff, U - RoW”
Insulation-aerogel	Foreground process ”green aerogel production”
Gypsum	“market for gypsum, mineral | gypsum, mineral | Cutoff, U-GLO”
Secondary material	**Remark and sources**
BAU recycled concrete	The secondary coarse aggregate was reproduced by the wet recycling process (Zhang et al. 2019a) to replace the primary coarse gravel in “BAU normal-weight siliceous concrete production”.
VEEP green concrete	Foreground process “VEEP normal-weight siliceous concrete production”, “VEEP lightweight concrete production”
Wood	“planing, board, softwood, u=20% | sawnwood, board, softwood, dried (u=20%), planed | Cutoff, U-RoW”
steel	“steel production, electric, low-alloyed | steel, low-alloyed | Cutoff, U-RER”
Glass	to use the secondary silica sand recycled by DGR system to replace the primary sand in process “flat glass production, uncoated | flat glass, uncoated | Cutoff, U-RER”
Insulation-EPS	The recycling process for EPS is currently unknown, to use the impact of primary EPS to replace secondary EPS.
Insulation-aerogel	The recycling process for aerogel is currently unknown, to use the impact of primary aerogel to replace secondary aerogel.

Table A6.10 Share of source for electricity production in 2015 in the Netherlands. Note: the energy mix of electricity production in 2015 is referred to the process “market for electricity, high voltage | electricity, high voltage | Cutoff, U-NL” in the Ecoinvent 3.4.

Energy resource	Share	Remark
Solar power	0.01%	46.15% from, 3kWp slanted-roof installation, single-Si, photovoltaic panel; 53.85% from 3kWp slanted-roof installation, multi-Si, photovoltaic panel
Import	30.30%	Amongst the electricity import, 0.03% from the UK, 9.26% from Belgium; 74.08% from Germany, 16.63% from Norway
Nuclear	3.55%	Pressure waste reactor

Offshore windfarm	0.34%	1-3 MW offshore turbine
Onshore windfarm	4.02%	28.69% from <1 MW turbine, 53.18% from 1-3 MW turbine, 18.14%f from >3 MW turbine
Hydropower	0.10%	Run-of river
Other	1.55%	96.42% from heat and power co-generation by wood chips (6667 kW), 3.58% from heat and power co-generation by biogas
Coal	26.11%	74.59% from electricity production by hard coal, 25.41% from heat and power co-generation by hard coal
Gas	34.02%	17.62% from electricity production by natural gas with conventional, 24.62% from heat and power co-generation by natural gas with combined cycle power plant (400MW), 34.97% from electricity production by natural gas with the combined cycle power plant, 2.37% from heat and power co-generation by oil, 20.41% from heat and power co-generation by natural gas with the conventional power plant (100MW)

Table A6.11 Energy mix for electricity production in 2030 and 2050. Note: The source of energy mix share in 2030 is (EZK 2015); the energy mix share in 2050 is assumed based on the situation of 2030: the share of the renewable remain the same; the coal and gas are replaced by renewable sources. Those years between 2015 and 2030, and between 2030 and 2050 were through linear interpolation. The costs for heating and electricity were assumed to remain static.

Energy resource	Share in 2030	Share in 2050	Remark
Solar power	4.46%	8.88%	Same as 2015
Import	11.50%	22.90%	Same as 2015
Nuclear	2.35%	4.67%	Same as 2015
Offshore windfarm	14.32%	28.50%	Same as 2015
Onshore windfarm	11.27%	22.43%	Same as 2015
Hydropower	0.47%	0.93%	Same as 2015
Other	5.87%	11.68%	Same as 2015
Coal	16.43%	0.00%	/
Gas	13.62%	0.00%	/

Table A6.12 Construction/demolition/renovation waste intensity of the Netherlands. Note: The intensity of construction waste and demolition waste in the Netherlands was collected and modified based on the study (Zhang et al. 2020b, 2021c). The renovation waste from the cladding system for refurbishment is from the study (Villoria Sáez et al. 2018). It is assumed the lightweight concrete waste accounts for 1% of the gross normal-weight concrete waste. Besides, asbestos is assumed to only exist in the demolition waste of old buildings.

Waste intensity	Construction waste intensity for construction of new buildings [kg/m^2]	Demolition waste intensity for end-of-life buildings [kg/m^2]	Renovation waste intensity for renovation of old buildings [kg/m^2]
Concrete and other stony wastes	26.00	901.66	29.94
Ferrous and in non-ferrous metal	5.23	181.40	0.00
Wood	2.48	85.91	0.12
Glass	0.13	4.51	0.00
Plastic	0.31	10.70	0.07
Paper	0.09	3.10	0.19
Insulation	0.03	0.99	0.04
Asbestos	0.58	20.00	0.00
Sorting residue& mixed waste	5.77	200.13	0.25

Table A6.13 Recycling rate of each CDW (Mulders 2013). Note: the "Recycling rate within the Netherlands" means the geographic border of the Netherlands is the boundary for recycling rate statistics, and "export unknown recycling" is not included. In this assessment, the CDW for "export unknown recycling is assumed to be recycled in the Netherlands. The "*" represents recycling of this waste is considered for the calculation of gross CDW recycling rate in the Netherlands but the associated recycling impact is not considered. The shares of plastic and paper are negligible so the recycling impacts are not considered. Moreover, the recycling processes of insulation, gypsum, sorting residue& mixed waste are unknown, therefore, the recycling impacts of those waste are not considered.

Building materials	Recycling rate within the Netherlands	Recycling rate within the residential sector in the BAU and REB scenario	Recycling rate within the residential sector in the REN scenario
Concrete (and other stony wastes)	2%	2%	100%
Metal (ferrous and in non-ferrous)	78%	95%	95%
Wood	13%	23%	23%
Glass	100%	100%	100%
Plastics	17%	50%*	50%*

Paper	100%	100%*	100%*
Insulation	37%	37%*	37%*
Asbestos	0%	0%	0%
Gypsum	40%	40%*	40%*
Sorting residue& mixed waste	43%	43%*	43%*

Table A6.14 Space heating demand per annum of the ZABRA and TABULA nZEB standard. Note: The heating demand of the nZEB- EPISCOPE -NL is a net heating demand that excludes heat recovered by the ventilation system.

House type	Stock share	nZEB-ZEBRA-NL [kWh/(m^2·annum)]	nZEB-EPISCOPE-NL [kWh/(m^2·annum)]
Detached house	15.98%	/	12.30
Semi-detached	11.39%	/	9.20
Terraced house	33.60%	/	7.80
Apartment and maisonette	39.03%	/	7.40
In general		15.73	8.52

Table A6.15 The decarbonization goal of the residential built environment by 2030 and by 2050. Note: a) data from (CBS 2021b); b) data from (PBL et al. 2020) c) data from (EZK 2019); d) reduction by 95% based on 1990 level (21.5 Mt); e) data from the “A Roadmap for moving to a competitive low carbon economy in 2050” (EC 2011c), the emission of the residential and service sector is supposed to be reduced by 37%–53% by 2030 and reduced by 88%–91% by 2050.

	Historical emission of the residential built environment [a]	Decarbonization goal of the Netherlands	Decarbonization goal of the EU[e]
1990	21.5	/	/
1991	24.6	/	/
1992	22.7	/	/
1993	23.8	/	/
1994	22.3	/	/
1995	24.2	/	/
1996	28.1	/	/
1997	23.6	/	/
1998	22.5	/	/
1999	21.3	/	/
2000	21.3	/	/
2001	22.6	/	/
2002	21.8	/	/
2003	22.3	/	/

2004	21.3	/	/
2005	20.5	/	/
2006	20.7	/	/
2007	18.5	/	/
2008	20.9	/	/
2009	20.8	/	/
2010	24.1	/	/
2011	18.8	/	/
2012	20.1	/	/
2013	21.3	/	/
2014	16	/	/
2015	17.1	/	/
2016	17.7	/	/
2017	17.3	/	/
2018	17.2	/	/
2019	16.4	/	/
2030	/	14.1[b] -15.3[c]	10.1-13.5
2050	/	1.1[d] -1.5[c]	1.1-1.5

Chapter 7

General discussion

7.1 Introduction

The overall aim of this thesis was to understand the potentials to enhance the material circularity and energy efficiency of the Dutch building stock through circularity strategies at different levels: lifetime extension of buildings, reuse and production of building components with recycled materials, and recycling of materials, all with a focus on concrete. For the analysis of functional units at the level of specific products and materials, the analytical tools life cycle assessment (LCA) and life cycle costing (LCC) have been applied. To scale up the analysis to flows and stocks of certain construction materials and buildings in the Netherlands as a whole, LCA and LCC were combined with dynamic MFA (material flow analysis). This chapter synthesized the main findings and provided general discussions. Having these insights can help developers and policymakers make better-informed decisions in the sustainable design of construction products and political interventions.

This thesis started by exploring the life cycle environmental and economic impacts of four technologies (as examples for circularity interventions at the material level) for high-quality concrete recycling (Chapter 2). The recycled concrete aggregates are used to produce prefabricated concrete elements (PCE), PCE-new for new construction and PCE-refurb for energy refurbishment. Two separate LCA-LCC studies were carried out to capture the life cycle environmental and economic profile of the PCE-new and PCE-refurb (as examples for circularity interventions at element level; Chapters 3 and 4). To understand the future application limits of PCEs, a dynamic building stock model was constructed to estimate the future supply of construction and demolition waste (CDW) (e.g. concrete aggregates) and the demand of secondary raw materials for PCEs in new construction and building energy renovation (Chapter 5). As a final step, the product-level assessments were scaled up to a country-level by connecting the LCC and LCA studies with the dynamic MFA study (Chapter 6). Based on the findings reported in Chapters 2 through Chapter 6, this concluding chapter will provide answers to the research questions posed in Chapter 1.

7.2 Answers to research questions

This section presents answers to the five research questions and the overall question.

7.2.1 Assessment of concrete recycling at the product (material) level

Research question 1 was formulated as follows:

RQ1. Is it possible to achieve environmental-economic win-win situation in high-grade concrete recycling? Would the innovations trigger any potential problem-shifts between different impact categories?

High-value recycling is the circular strategy at the material level. Waste concrete is the most voluminous constituent of CDW generated in the EU member states. In most cases, CDW is crushed and downcycled to low-grade applications such as road foundations and filler under buildings. There are technologies that lead to high-grade recycling of concrete. The case study in Chapter 2 investigated the eco-efficiencies of four technological systems for the high-value-added recycling of concrete. It concluded that it is possible to achieve an environmental and economic win-win situation by recycling the waste concrete as local as possible and producing as much as possible high-value secondary products (e.g. cementitious fines). We found the most environmentally sound and cost-saving way for concrete recycling is the integrated advanced dry recovery (ADR) and heating air classification system (HAS), which had the highest eco-efficiency score.

Potential problem shifts between different impact categories were noticed. In order not to leave out any impact category that may have a significant impact, a comprehensive ILCD method recommended by the Joint Research Centre of the European Commission was selected as the impact assessment method for LCA. The integrated ADR and HAS technology is superior to the other technological system for 10 out of 15 investigated environmental impact indicators. At the same time, it has the highest environmental impact in photochemical ozone formation, particulate matter, acidification, terrestrial eutrophication, and marine eutrophication. These drawbacks are due to the fact that the pilot-scale HAS still uses fossil fuel (diesel) to thermally decompose the concrete waste. An industrial-scale HAS using cleaner energy such as renewable electricity could be a direction to go. This case study evidenced the need for a broader environmental perspective.

7.2.2 Assessment of the PCE-new system for new building construction at the product (element) level

Research question 2 was formulated as follows:

RQ2. What are the environmental and economic implications of using the prefabricated element system (PCE-new) for cladding walls to improve building energy efficiency in new constructions in the Netherlands?

Improving the thermal insulation level of façades is seen as an essential method in the proposed energy-efficiency interventions to upgrade the energy performance of building stock. This can be achieved by using the high thermal performance PCE systems to construct the wall of new buildings, either "green PCE-new" made of recycled aggregates and aerogel or "baseline PCE-new" made of primary aggregates and EPS for cladding walls. A combined LCA and LCC study (see Chapter 3) was carried out to assess the carbon mitigation potential and the life cycle costs of using the green PCE-new as façade for a new building over 120 years, in comparison to those of manufacturing and using a baseline PCE-new. The results show that the GHG mitigation potential of the green PCE-new is only slightly better than the baseline PCE-new. This is because the implementation of the experimental thermal material – aerogel, which although achieves a lower thermal transmittance than the conventional insulation EPS, the production of aerogel is much more energy-intensive and emits more GHG. From a life cycle point of view, using green PCE-new is slightly cheaper if it considers the internal costs only. If on top of this the external costs of GHG emissions are included, avoided emissions lead to an additional cost advantage. By internalizing the environmental revenue, the extended life cycle costs enhance the economic advantage of green PCE-new, though a wider uncertainty in cost estimation needs to be considered.

7.2.3 Assessment of the PCE-refurb system for existing building renovation at the product (element) level

Research question 3 was formulated as follows:

RQ3. What are the environmental and economic implications of using the prefabricated element system (PCE-refurb) to over-clad existing buildings for energy refurbishment in the Netherlands? How is the applicability of the PCE-refurb under different climatic conditions?

Design for reuse is an important circular strategy for new buildings at the element level. The majority of existing houses in the Netherlands and Europe are still energy inefficient. Therefore, renovating those old houses is of great significance towards a low-carbon built environment. Chapter 4 presents an LCA-LCC study that assesses the potential life cycle impacts of using 'green PCE-refurb' for energy refurbishment of existing houses on energy conservation, carbon mitigation, and cost reduction in the Netherlands. As the remaining life spans of old buildings in the Netherlands vary widely, the life spans for the assessment cannot be directly determined. Therefore, the payback period is used to combine LCA and LCC to investigate a breakeven point. We analyzed if the carbon emission, investment, and energy use of the PCE-refurb embodiment can be paid back by the savings in operational energy use, costs, and emissions of the building during this period. It appeared indeed to be positive energy and carbon payback (approximately 17 years), however not to be the case for economic payback (more than 100 years). This indicates that manufacturing and using a PCE-refurb for energy renovation will reduce energy use and GHG emissions but it is costly. The PCE refurb can however be designed in a reusable way by applying easy-to-disassemble joints. Such design allows them to be reused again after one service life, which helps to overcome this cost gap. It was found that using secondary materials in the PCE-refurb can just slightly reduce the three types of payback periods, while the reuse of the PCE-refurb can considerably shorten the three types of payback periods in three cases. To comprehensively examine the applicability of the PCE-refurb system, two additional Swedish and Spanish cases were conducted. Results show that implementing the PCE-refurb in a colder zone would result in shorter payback periods for energy conservation, carbon mitigation, and costs.

7.2.4 Assessment of turnover of housing stock at the country level

Research question 4 was formulated as follows:

*RQ4**.** How much CDW from the construction, demolition, and renovation of the Dutch housing stock will arise from 2015 to 2050? To which extent the CDW can be recycled as a feedstock in building energy renovation in the Netherlands?*

Implementing circularity interventions at a country level needs to additionally consider the local market condition and material constraints. Enhancing building energy efficiency by energy renovation will lead to structural changes in demands for construction materials and CDW generation in the Netherlands. An illustration is the use of novel insulation for better thermal insulation performance and that of non-structural lightweight concrete for weight load reduction. To explore to which extent the use of recycling materials in the PCE system for building energy renovation can improve the recycling potential, a case study on the Dutch residential building stock was carried out (see Chapter 5). Using a dynamic material flow analysis, this work explores the supply-demand balance of secondary raw materials made from CDW (including normal-weight and lightweight concrete, glass, insulation mineral wool, steel) and the secondary raw materials demanded for manufacturing PCEs in building energy renovation from 2015 to 2050. The main results show: with the advanced recycling system ADR and HAS, the secondary raw materials recovered from normal-weight concrete waste, glass waste, and insulation mineral wool waste will be more than sufficient to support the manufacturing of PCE-new walls. However, secondary material made from lightweight concrete will not be sufficient in a near future to meet the raw material demand. For large scale refurbishment with PCE-refurb walls, virgin expanded clay, sand and cement would be needed to complement the available amounts of recycled materials. It was also found that the Netherlands still will have to rely on imports to satisfy its demand of specific mineral materials (e.g. gravel, expanded clay, cement, and limestone). Using the novel technological system ADR and HAS to recycle concrete waste has the potential to supply all the normal concrete aggregates needed for housing energy renovation in the Netherlands. Additionally, the Dutch case study projected the amount of CDW generated from the construction, demolition, and renovation activities for the period 2015 to 2050.

We found that CDW generated from demolition activities is still the primary waste stream; while the amount of CDW from renovation activities even exceeds the amount of CDW from construction activities. If the Netherlands and the EU hence will embark on an extensive renovation of houses to make them energy-efficient, it is hence important to pay more attention to the management of renovation waste.

7.2.5 Assessment of implementing the recycling and the prefabrication systems at the country level

Research question 5 was formulated as follows:

RQ5. What are the up-scaled environmental benefits and economic consequences of implementing the recycling and prefabrication systems in the Netherlands? To which extent can the proposed recycling and prefabrication system achieve the prospective circularity goal and decarbonization goal of the Netherlands?

An integrated framework was proposed to combine a top-down dynamic building stock model with the LCA and LCC models to evaluate the scaled-up impacts of applying the proposed PCE solutions to improve the thermal performance of the residential building stock in the Netherlands from 2015 to 2050. We considered three scenarios: (i) business-as-usual (BAU) scenario that which does not apply any renovation strategy, (ii) (rebuilding) REB scenario in which old houses are demolished and reconstructed, and (iii) Renovation (REN) scenario in which old houses are demolished and refurbished instead of reconstruction. The building stock model was used to simulate the sizes of construction, demolition, renovation floor area per annum, and stock cohorts in the three scenarios. The LCA and LCC models were deployed to estimate per unit life cycle GHG emissions, costs, and material footprints of the proposed PCE system. Then the unit results were up-scaled using the sizes of the stocks and flows estimated from the dynamic building stock model.

The results were expressed via an annualized-value approach and an actual-value approach. The actual-value approach adds up actual costs and impacts of the energy use of the Dutch housing stock and construction and demolition activities each year to a total in that year. The annualized-value approach calculates the operational energy use the same way as the actual-value approach but distributes costs and impacts, which occur in the construction and demolition phases, as annualized values per year per m^2 per housing type over the lifetime of a house. The selection of different estimation approaches led to different conclusions. The actual-value approach suggests the use of the PCE system for Dutch housing energy renovation would achieve GHG mitigation with increased material use and financial cost. However, the annualized-value approach suggests the overall benefits of the PCE system on all three aspects.

This study also evaluated the extent to which the circularity and decarbonization goals can be realized by applying the proposed PCE system to the Dutch residential building stock. The Netherlands aims to achieve a reduction of 50% of primary material use by 2030 compared to the 2015 level. The results show all three scenarios cannot reduce resource use till 2050. Regarding the circularity goal by 2050, advanced technological systems in the REN scenario can raise the recycling rate from the current 25% to 98%. However, the associated embodied emissions of the REN scenario is twice as the BAU scenario, resulting from using aerogel as insulation. Therefore, applying the PCE system alone is less likely to let the Netherlands reach the circularity goal by 2030 and 2050.

Furthermore, operational GHG emissions of the residential building stock in 2030 and 2050 were quantified under different scenarios. We found that the BAU scenario which does not have any energy renovation strategy, will not achieve the decarbonization goals for 2030 or 2050. In the REN scenario, implementing the PCE system for extensive energy renovation in the Netherlands can achieve the interim goal of 49% carbon mitigation by 2030, but still cannot realize the carbon-neutral level by 2050. However, the REN scenario can possibly reach the long-term goal by replacing traditional gas-based systems with electricity-based systems and supplying renewable electricity nationwide. This study hence shows too that the development from the energy sector would significantly influence the realization of the carbon neutrality goal in the Dutch housing sector.

7.2.6 Answer to the overall question

The overarching research question posed in this thesis is: *what are the potential impacts of the application of selected novel technological systems to enhance circular use of materials, energy efficiency and carbon neutrality in the residential sector of the Netherlands?*

Since the need to realize a carbon-neutral environment will be a dominant factor in the adjustment of the Dutch residential sector in the next decades, we first look at energy efficiency. Chapters 3 and 4 demonstrated that improving the insulation level of existing houses is more necessary than upgrading the insulation level of new houses. This is not surprising given the large number of existing houses, usually with low energy efficiency, compared to the number of houses that will be newly built until 2050. The case in Chapter 6 shows that demolition of old energy-inefficient houses followed by rebuilding new houses (REB scenario) in combination with the decarbonisation of the energy system results in the lowest carbon footprint. This comes however at very high financial costs and at significant growth in material footprint. The renovation (REN) scenario has much lower costs and material footprints (although still higher than the BAU scenario), but can realise significant carbon emission reduction by the installation of electrical space heating systems and supply of renewable electricity. In short, the REB scenario has much more material use and capital costs, while the REN scenario can only realize carbon reduction goals by switching to electrical heating systems and renewable electricity.

This result illustrates also another conclusion that can be drawn from this research: i.e. that to realise circularity goals, lifetime extension is to be preferred over component reuse, which in turn is to be preferred over material reuse. As indicated the **lifetime extension** in the REN scenario in Chapter 6 has much lower costs and material footprints than the REB scenario. The PCE-refurb case (Chapter 4) shows **element reuse,** i.e. taking out PCE elements from houses that are to be demolished and using them to renovate other houses, is a more preferable option to material recovery regarding GHG mitigation, cost saving, and energy use reduction. With regard to material recycling of concrete, we found that technological systems, such as transportable ADR-HAS, have the likelihood to achieve environmental-economic co-benefits compared to downcycling. But when such materials are incorporated into manufacturing the green PCE-new and green PCE-refurb we did not find obvious environmental and economic benefits from a life cycle perspective of a PCE (Chapter 3 and 4). The life cycle impacts of material production (and hence potential benefits of recycling) are relatively low compared to the life cycle impacts of the PCEs.

We see however too, particularly in Chapter 6, that even if implementing three circularity interventions simultaneously, this is still unlikely to reduce the primary material extraction by 2030. This is due to the contradiction between achieving circularity goals and energy efficiency ambitions. Recovery, reuse, and lifetime extension can reduce material use, while constructing new houses and renovating old houses to achieve carbon-neutrality require a large amount of raw materials. Therefore, the Netherlands needs to seek more ambitious and advanced technologies and policies to realize circularity goals, such as more lightweight design more efficient production processes of materials.

7.3 Methodology discussion

7.3.1 Integration of LCA with LCC

Integrated LCA-LCC assessments have become a widely used approach for analyzing the environmental pressure and economic costs of specific products. Researchers and practitioners have been developing and implementing different methods to integrate LCA and LCC. Miah et al. (2017) reviewed frameworks for the combination of LCA and LCC and classified them into the following six categories:

(i) Independent LCA and LCC: LCA and LCC are conducted independently for the same product system.

(ii) Independent LCA and LCC as part of an overarching framework: LCA and LCC are conducted independently to form important dimensions within the overarching framework and the final output is a portfolio of results similar to Type i.

(iii) Independent LCA and LCC integrated by multi-criteria decision analysis: LCA and LCC are conducted independently as part of an overarching framework for the same product system. There is some overlap with type ii, however, the key difference lies in the multiple-criteria decision analysis to determine the "best" alternative from a group of options.

(iv) Optimization of LCA and LCC analysis: it is similar to type iii, however, the key difference addresses the optimization of environmental and financial parameters to find the optimal solution via multiple-objective decision-making methods.

(v) Environmental LCC: LCA-based costing method where the functional unit, scope, and system boundary are the same.

(vi) eco-efficiency assessment: an assessment that evaluates "the aspect of sustainability relating the environmental performance of a product system to its product system value" that was standardized in ISO 14045 (2012).

This thesis contains three cases where LCA and LCC have been applied, which cover three of the approaches mentioned above:

1. a case of recycling waste concrete in which an eco-efficiency assessment and diagram was applied (Chapter 2; approach vi),
2. a case of the PCE-new system with a method that monetizes the external costs of environmental impacts from the LCA and combines them with the internal costs from the LCA (Chapter 3; approach v),
3. a case of the PCE-refurb system, using independent LCA and LCC as part of an overarching framework based on a payback period (Chapter 4, approach ii).

In the first case where the eco-efficiency approach was applied, we visualized an eco-efficiency index through a two-dimensional diagram. This gives an easy to interpret insights into the trade-offs between environmental and cost aspects of multiple systems, which is suitable to make a quick comparison between multiple alternatives. However, if one system is better than another for some environmental impact categories but poorer for others, it is difficult to figure out whether the total environmental performance was improved or deteriorated, so does the eco-efficiency. The second case used monetarization of environmental impacts which allows expressing monetary costs and impacts in the same unit. The case shows that when environmental externalities are taken into account, the (economic) performance of green products improves. Such monetary results on internal and external costs can also support policy-making with regard to e.g. taxation of environmental impacts and resource use. However, monetization of environmental impacts will lead to greater uncertainty in results because monetary costs of environmental impacts are difficult to quantify, especially if such impacts take place far in the future. Such future impacts often are discounted, a practice which is still heavily debated since even at a moderate discount rate potentially large impacts only taking place in e.g. a century have a negligible net present value. In the third case, LCA and LCC were conducted independently but under a joint payback framework. This case study showed a payback method is suitable to explore an LCA or LCC study of which the temporal scope cannot be directly defined.

These case studies illustrate some methods of integrating how LCA and LCC, and how this helps to support the strategy exploration in material circularity and building energy efficiency projects. However, a large number of methodological inconsistencies between LCA and LCC still need to be solved by the scientific community. For example, discount rates are usually applied in LCC and not in LCA; the concept of "value-added" in LCC has no counterpart in an LCA; LCC is conducted from a perspective of an actor to clarify costs, prices, and revenues while an LCA focuses on a functional unit.

With the growing concerns about sustainability and its environmental, social, and economic pillars, the emerging overarching framework of life cycle sustainability assessment (LCSA) could provide a direction of how to improve integrating LCA and LCC. LCSA is an umbrella concept that combines life cycle thinking with the concept of sustainability. At a product level, LCA was seen as the environmental dimension and LCC was formulated as the economic dimension of LCSA via a conceptual equation (Kloepffer 2008, 2003): LCSA = LCA + environmental LCC + social LCA. A broader conceptualization of LCSA even shows the potentials of analyzing meso-level and economic-wide issues (Guinée et al. 2011). However, LCSA still has not provided solutions for the mentioned inconsistencies between LCA and LCC. The development of LCSA may contribute to a more consistent integration of LCA and LCC.

7.3.2 Integration of MFA with LCA/LCC

MFA and LCA/LCC are two core analytical methods in the field of industrial ecology. In ISO 14040 (2006a) and 14044 (2006b), the LCA was defined as a product-oriented assessment of the inputs and outputs and the associated potential environmental impacts of a product or service system during its life cycle. The MFA approach is based on the law of conservation of mass as aiming to assess the metabolism of materials in a system (Brunner and Rechberger 2004). Similarities between LCA and MFA can be identified, such as both are systemic approaches, quantification of material/substance/pollutant flows, distinguishing between direct and indirect flows, etc. (Azapagic et al. 2007).

Meanwhile, clear differences exist. First, regarding the system scale in the definitions, LCA mainly focus on micro-level assessment and the MFA primarily aims to investigate regional or nationwide issues. However, in some minor cases, LCA can also estimate the total life cycle environmental impacts of the total use of concrete products (Arrigoni et al. 2020) in a region, while MFA has also been used to analyze the material flows of CDW at a unit process level (Dahlbo et al. 2015). This flexibility of the system scale used in LCA and MFA shows the potential of combining these two methods. Second, the object to be traced in an MFA has to be concrete substances and materials, while an LCA can explore not only materials but also abstract objects, such as a service and a technology. Third, MFA uses mass-based indicators; LCA uses both mass-based and impact-based indicators. Finally, an MFA system must be determined with regard to four basic variables: space, function, time and materials (van der Voet 1996). However, specific time and space are not necessary for an LCA system.

In general, LCA has a broader scope than MFA—LCA encompasses sources, receptors, pollutants, and impacts, while MFA focuses only on material inputs and outputs and stocks and flows in the economic system. Therefore, MFA can be incorporated into the overarching framework of life cycle management. Hu (2013) proposed an approach to LCSA by combining MFA with LCA (or LCC or SLCA) with a case of concrete recycling. In the LCA-based overarching framework, the functional unit of an LCA is the critical connection to bridge the LCA and results from MFA can be used to build up-scaled life cycle inventory.

Chapter 6 presented an up-scaling study that combined dynamic MFA with LCA and LCC. LCA and LCC were used to estimate the unit benefits of the PCE system at the product level. The Dynamic MFA was conducted to evaluate the material inflow and outflow of the Dutch residential building sector at the country level. The housing stock from the MFA study was used as a size factor for scaling up environmental and economic impacts per functional unit from LCA and LCC. In addition, the temporal inconsistency between LCA/LCC and MFA were addressed. Generally, an MFA is conducted through the synchronic-temporality approach (i.e. all processes that take place in a specific year), while an LCA is modeled from the diachronic-temporality approach (i.e. all processes related to a life cycle of a product, independent of the moment in time that they take place) (Birat 2015). Therefore, in our case study in Chapter 6 in the LCA/LCC calculations the lifetime of buildings was taken into account (assumed to be 120 years), while MFA analysis was done for each year in a clearly defined period (for which we chose the period between 2015-2050). To show both the time-specific impacts that usually are provided by MFA and the lifetime impacts that are usually provided in LCA/LCC, we developed the concept of actual-time value and annual value. This is an innovation in the way of how results of combined MFA and LCA/LCC studies can be presented.

The actual-time value approach provides insight regarding in which moment in time impacts and costs occur. This approach allowed us to show for instance in which year(s) the actual impacts and costs of rebuilding or refurbishing take place, but also at what moment in time the reduction of energy use and GHG emissions in the use phase of buildings starts to become higher as such impacts and costs. This approach is hence highly suitable to assess if e.g. carbon emissions (both operational carbon emissions in the use phase and embodied carbon generated for building materials used in refurbishing) and material use will comply with the 2030 and 2050 targets of the Dutch government. This approach is hence highly suitable to assess if e.g. carbon emissions (both operational carbon emissions in the use phase and embodied carbon generated for building materials used in refurbishing) and material use will comply with the 2030 and 2050 targets of the Dutch government. The annualized-value approach is more suitable for comparing scores for a sound functional unit, such as the impacts and costs per m^2 floor space in use per year, including impacts and costs of the construction phase and benefits of recycling and reuse at the end-of-life (EoL) stage. The annualized-value approach is more capable of a theoretical comparison of life cycle impacts and costs, while the actual time value better reflects the actual emissions and costs incurred in specific moments in time. Such insights can help policymakers, for instance, in assessing how the use of carbon budgets evolves over time.

7.4 Policy implications

7.4.1 Building material circularity

In this thesis, multiple cases have been conducted to evaluate the proposed building circularity framework. We illustrated the potential policies from the three layers of the circularity intervention framework: resource use reduction, element reuse, and material recovery.

Resource use reduction

Waste prevention has the highest priority. CDW can be prevented (i) by adopting ecodesign for constructing new buildings and (ii) by extending the lifetime of existing buildings. For new building construction, the government is supposed to encourage the application of ecodesign. **First**, prefabrication design has been identified as a promising solution to prevent CDW generation (Tam et al. 2006). The practice of prefabricating buildings is well established in the Netherlands, with prefabricated elements used in 55% of all Dutch construction projects in 2016 (de Gruijl 2018). Upscaling the implementation of prefabrication design can further improve the circularity by component reuse in the building sector. **Second**, lightweight design is also a promising solution to reduce material requirements and prevent CDW. It can be applied by either adjusting the structure and decreasing material demand or using lighter materials.

Chapter 5 shows an example of using lightweight concrete that if 1% of normal-weight concrete is replaced by ultra-lightweight concrete, 28 Kt of concrete and associated waste concrete would be avoided. **Third,** waste prevention can also be achieved through long-lasting design (Geissdoerfer et al. 2017), which is also termed ‘durability’ by the EU’s Principles for Building Design (EC 2020b). The EU Ecodesign Directive regulated the minimum guaranteed lifetime, the minimum time for availability of spare parts, modularity, upgradeability, and reparability must be considered for energy-related products (EC 2009). Compared to a normal product, newly constructed houses in the Netherlands have a much longer lifespan, which was assumed 120 years in this thesis. Therefore, the major elements of a building are supposed to have the same or longer service life planning as well as their associated maintenance and replacement cycles.

Extending the lifetime of existing buildings is also an important approach to CDW prevention in Europe. The ‘Renovation Wave’ initiative claimed in the European Green Deal to lead to significant improvements in energy efficiency in the EU will be implemented in line with circular economy principles, notably, building assets that have a longer life expectancy (EC 2020c). The Principles for Building Design addressed the importance of ‘adaptability’ that refers to extending the service life of the building as a whole with a focus on replacement and refurbishment (EC 2020b). In this thesis, prolonging the lifetime of existing buildings is realized by adopting the PCE-refurb system. In the Netherlands, it is expected that the lifetime of houses can be extended by at least 15 years after renovation (Duurzaam Gebouwd 2014). Chapter 6 shows that extending the lifetime of existing houses through refurbishing their facades can noticeably reduce the annualized GHG emissions, costs, and material footprints of material use incurred at the construction stage. Refurbishing older houses would yield more economic and environmental benefits. Therefore, renovation activities should start with those houses that were constructed before the 1960s.

Element reuse

Reuse is a second important strategy to realise a circular economy. This thesis explores the example of full reuse of concrete façade elements.

In the EU, such reuse of building elements is not yet a common practice. Although the EU’s Waste Framework Directive (WFD) (EC 2008a) addresses the importance of reuse, it did not set a quantitative target for reuse. For instance, the WFD requires member states to (i) take any necessary measures including (reuse, recycling, and other material recovery) to prepare for material recovery of at least 70% (by weight) of CDW by 2020 and to (ii) increase the reuse and recycling of waste materials from households to a minimum of overall 50 % by weight (EC 2008a). Therefore, a quantified target of reuse should be set, such as “10% of material recovery should be realized by reuse” to stimulate the reuse of building components.

In addition, adoption of reuse largely depends on the design of a product at the concept stage. In the Netherlands, waste management transitions did not radically alter product features regarding design for disassembly and reuse (Kemp and Van Lente 2011). In order to strengthen reuse, the WFD also encourages member states to take measures to apply appropriate designs that promote reuse (EC 2008a), such as design for dismantling, and design for deconstruction. Moreover, it is also important to establish procurement criteria and quality standards and to develop repair networks to support reuse.

Material recovery

Goals should be set for material recovery to promote the recycling of CDW. These goals should be challenging. For instance, the Netherlands realised already a much higher recycling rate of 70% of CDW in the WFD in 1995 (92%; (CLO 2021). However, in 2015, only 3% of waste concrete was recycled back into the concrete cycle with the rest being used as filler in e.g. road foundation. While the amount of CDW will continue to rise till 2025 (Zuidema et al. 2016), the Netherlands is already facing a problem since road construction is not growing anymore and the requirement for filler is hence limited (Hu et al. 2013; Bio Intelligence Service 2011). The case study in the Netherlands (Chapter 2) suggests that the EU could set more ambitious goals to encourage the building sector to shift from downcycling to recycling or upcycling. For example, a circular goal could be set as “those member states who already achieved the goal of recovering 70% CDW, are encouraged to upcycle 20% of the recycled volume of CDW”.

Setting more ambitious goals at the EU level is, however, only possible if a clear definition of recycling or upcycling (as opposed to downcycling, or energy recovery) is given, which is currently lacking. Furthermore, the definition of "backfilling" should be strictly clarified to avoid landfilling without useful application operations in this definition. Unfortunately, current waste registration systems and databases are not suitable for estimating end-of-life flows of CDW, and in particular concrete. It is, therefore, necessary to develop a more systematic waste registration system that includes quantities CDW is generated, and how it is treated.

Furthermore, the massive need for housing renovation to meet GHG emission targets in the Netherlands will not only increase the need for resources but raises new problems surrounding the disposal of new types of CDW in the future, particularly posed by new insulation materials used in renovation (Chapter 5). It is urgent for the Dutch government to promote the development of novel reuse technology and standardization of secondary materials made from insulation waste, and to foster a secondary market.

Finally, sustainable public procurement can be a strong potential driver for CDW recycling, but at this stage sustainable public procurement criteria in general do not yet require minimum use of recycled materials. Standards for building materials are based on virgin materials and it is not always easy for secondary materials to comply with them. The VEEP project has demonstrated that with proper quality control of secondary materials, the recycled concrete aggregate will not be noticeably different in terms of quality and strength, compared to concrete made with primary aggregate. A minimum required share of recycled aggregates and cement could be introduced in national and local Sustainable Public Procurement criteria.

7.4.2 Building energy renovation

Building energy renovation is seen by the EU as an essential measure to shift to a carbon-neutral built environment. This thesis proposed a building energy efficiency framework that prioritizes energy efficiency at the operation stage. The energy efficiency of a building can be realized through an active and a passive approach (compare IEA (2021) who mention two main strategies to realize decarbonization of existing building stock: enhancing energy efficiency by refurbishing and electrifying heating and cooling). Via the passive approach, heat transfer through the building envelope can be reduced by optimizing the insulating degree of walls, roofs, floors, and windows. Using the active approach, GHG emission reductions can be realized by adopting efficient installations for building technical systems, renewable heat generation systems, renewable electricity generation systems, and other energy-related measures. For new buildings, the EU Energy Performance of Buildings Directive (EPDB, 2010/31/EU) requires that the Member States must ensure new buildings constructed after 2020 should be nearly zero-energy buildings (nZEBs), which have high insulation level and are highly electrified (EC 2010). Existing buildings are supposed to be renovated so that better insulation and more energy-efficient heating system is realized, ideally based on the electricity that can be sourced from renewable sources, are used.

Passive energy efficiency approach

With regard to the passive approach, improving the thermal insulating level of building elements can effectively reduce heat loss. The country-level study case in Chapter 6 shows that approximately 3.3 Mt of GHG emission can be mitigated by upgrading the thermal insulation level of façades of existing housing stock. Different degrees of energy renovation lead to different ranges of energy savings. In one EU report (Esser et al. 2019), four depths of energy renovation were defined regarding energy savings: (i) below threshold (< 3% savings), (ii) light renovations (3% $\leqslant x \leqslant$ 30% savings), (iii) medium renovations (30% < x $\leqslant$ 60% savings), (iv) deep renovations (x > 60% savings). Renovation to an nZEB level can achieve over 90% of energy savings (Economidou 2011). The EPDB also request EU Member States to set up their own definitions of nZEB that reflect their national, regional or local conditions and national plans to stimulate the transformation of buildings that are refurbished into nZEBs (EC 2010).

In response to the EU, in 2017 the Dutch standard NEN 7120 used an Energy Performance Coefficient (EPC) to define an nZEB, whose EPC is close to 0 (Government of the Netherlands 2012). The Netherlands implemented an energy label certificate for houses when they are being built, sold or rented. The labels range from the worst ‘G’ level to the best “A++++” level, determined on the basis of fossil energy use in [kWh/(m^2·yr)] (Netherlands Enterprise Agency 2021a). The National Energy Agreement aims to achieve at least an average energy label A in 2030 for all buildings in the Netherlands (MENCP and MIKR 2017). The Netherlands also set standards and target housing insulation requirements (Netherlands Enterprise Agency 2021b). However, the connection between nZEB levels and the current energy label is weak, and thermal transmittance benchmarks of nZEB envelope elements in the Netherlands are still not clear. Therefore, policies and measures for promoting the nZEB solution should be more specific.

High-performance insulation material is the key for housing façade refurbishment. Aerogel is a promising insulation material that has a significantly lower thermal conductivity value (0.0120-0.0157 W/(m·K)), compared to conventional insulation expanded polystyrene (0.0310 W/(m·K)). The aerogel is however too costly to be extensively implemented at the current stage (Chapter 3). Under the EPBD, cost-optimality has been emphasized to improve the energy performance with the lowest life cycle cost. Therefore, more information on the life cycle costs of insulating materials should be generated.

Active energy efficiency approach

The Dutch study case in Chapter 6 shows that the active approach of electrification and the use of renewable energy is at least as important as the passive approach, i.e. insulation. By adopting electrification strategies, traditional gas-based systems (such as boilers, water heaters, and gas stoves) would gradually be replaced with electricity-based systems.

Space heating system is the most important part of electrification. Gas-based heating is currently the primary source of household space heat, accounting for 93% of the heat generated (EZK 2015). The gas is being deployed with increasing efficiency devices in houses, such as high-efficiency boilers and cogeneration. To support electrification, economic policy tools need to be considered. For example, the 2016 Tax Plan increased the rate for natural gas by about 5 cents and decreased that for electricity by about 2 cents to support the process of electrification (EZK 2015). Apart from tax measures, extra encouragements in the form of a subsidy for gas heating replacement can also facilitate the transition.

Moreover, in this thesis, we only considered electricity-based heat pumps, while there are other alternative heating options, such as geothermal heating networks, waste combustion heat work, and locally available biomass. Integrated design of the regional structure of the heat supply should be made to meet both the cost-optimal level and lowest energy requirement of nZEBs.

The development of renewable electricity is the prerequisite for electrification. The structural change in energy supply caused by electrification may also lead to an increase in electricity consumption (EZK 2015). The Dutch case in Chapter 6 shows that electrification may also backfire if the development of renewable electricity cannot keep up the pace of electrification. Therefore, a particularly important task is planning and integrating renewable energy sources in national nZEB implementation. For the Netherlands, offshore and land-based wind farms are expected to be the biggest contributor to renewable electricity generation by 2035 (EZK 2015). Wind power is promising for generating renewable energy but the opportunities are also limited due to the complications involved in its spatial accommodation, particularly land-based windfarm. Off-shore wind power is more productive compared with land-based one. This will probably lead to offshore windfarm expansion in the North Sea. Hence, it is important to identify how more international cooperation in the North Sea region can contribute to more growth of the offshore wind industry. The offshore wind turbine system is also relatively expensive, technological developments are in an urgent need towards a cost-effective wind energy industry.

7.5 Recommendation for further research

This thesis aims to propose an integrated method that combines LCA, LCC and MFA to explore the intertwined issue of material circularity and energy efficiency by taking the housing stock in the Netherlands as a case study. However, there remain many gaps that require further research. It concerns both methodological as case-related topics. Methodological issues include:

- Creating more consistency in the integration of LCC and LCA. This includes issues such as that discount rates are usually applied in LCC and not in LCA; that the concept of “value-added” in LCC has no counterpart in an LCA; and that LCC is conducted from a perspective of an actor to clarify costs, prices, and revenues while an LCA focuses on a functional unit.

- Creating more consistency when combining LCA/LCC and MFA. These include alternative methods to harmonize the temporality mismatch between LCA/LCC and MFA; specifying the sources of imported materials in MFA so that country-specific environmental pressures and cost can be included in LCA/LCC; and developing approaches for uncertainty and sensitivity analysis of an LCA/LCC-MFA upscaling assessment.

Practical and case study related issues include:

- Analysing benefits of other strategies and technologies as researched here in terms of costs, carbon emissions and circularity/resource use (for instance lightweight design and prefabrication).
- Doing similar analyses for the housing sector in other regions, that use different building techniques or have a different historical building stock, especially major economies such as the U.S. and China.

References

Afrane, G. and A. Ntiamoah. 2012. Analysis of the life-cycle costs and environmental impacts of cooking fuels used in Ghana. *Applied Energy* 98: 301–306. http://dx.doi.org/10.1016/j.apenergy.2012.03.041.

Agentschap NL. 2011. *Voorbeeldwoningen 2011 - Bestaande bouw (No.2KPWB1034)*. https://www.rvo.nl/sites/default/files/bijlagen/4. Brochure Voorbeeldwoningen 2011 bestaande bouw.pdf.

Aguilar-Hernandez, G.A., C.P. Sigüenza-Sanchez, F. Donati, S. Merciai, J. Schmidt, J.F.D. Rodrigues, and A. Tukker. 2019. The circularity gap of nations: A multiregional analysis of waste generation, recovery, and stock depletion in 2011. *Resources, Conservation and Recycling* 151: 104452. https://doi.org/10.1016/j.resconrec.2019.104452.

Aguilar-Hernandez, G.A., C.P. Sigüenza-Sanchez, F. Donati, J.F.D. Rodrigues, and A. Tukker. 2018. Assessing circularity interventions: a review of EEIOA-based studies. *Journal of Economic Structures* 7(1): 14. https://doi.org/10.1186/s40008-018-0113-3.

Akanbi, L.A., L.O. Oyedele, O.O. Akinade, A.O. Ajayi, M. Davila Delgado, M. Bilal, and S.A. Bello. 2018. Salvaging building materials in a circular economy: A BIM-based whole-life performance estimator. *Resources, Conservation and Recycling* 129(May 2017): 175–186. http://dx.doi.org/10.1016/j.resconrec.2017.10.026.

Akhlaghi, F. 1987. Life cycle costing — a tool for decision making. *Facilities* 5(8): 4–10. https://www.emerald.com/insight/content/doi/10.1108/eb006413/full/html.

Albuquerque, T.L.M., C.A. Mattos, G. Scur, and K. Kissimoto. 2019. Life cycle costing and externalities to analyze circular economy strategy: Comparison between aluminum packaging and tinplate. *Journal of Cleaner Production* 234: 477–486. https://doi.org/10.1016/j.jclepro.2019.06.091.

Almutairi, K., G. Thoma, J. Burek, S. Algarni, and D. Nutter. 2015. Life cycle assessment and economic analysis of residential air conditioning in Saudi Arabia. *Energy and Buildings* 102: 370–379. http://dx.doi.org/10.1016/j.enbuild.2015.06.004.

Anand, C.K. and B. Amor. 2017. Recent developments, future challenges and new research directions in LCA of buildings: A critical review. *Renewable and Sustainable Energy Reviews* 67: 408–416. http://dx.doi.org/10.1016/j.rser.2016.09.058.

Anastasiou, E., K.G. Filikas, and M. Stefanidou. 2014. Utilization of fine recycled aggregates in concrete with fly ash and steel slag. *Construction and Building Materials* 50: 154–161.

Arcadis. 2017. *Element ratio & cost indicators Dutch assets 2017*. Arnhem. https://www.google.com/search?rlz=1C1CHWL_zh-CNNL831NL831&sxsrf=ALeKk02o7zy5qC0a02C01BO94D_Wr_j8Yg%3A160285

8691314&ei=w66JX4zgEsO6kwW_5qjQAQ&q=demolition+cost+netherlands&oq=demolition+cost&gs_lcp=CgZwc3ktYWIQARgCMgQIIxAnMgQIIxAnMgQIIxAnMgIIADICCAAyAggAMg.

Ardente, F., M. Beccali, M. Cellura, and M. Mistretta. 2011. Energy and environmental benefits in public buildings as a result of retrofit actions. *Renewable and Sustainable Energy Reviews* 15(1): 460–470. http://dx.doi.org/10.1016/j.rser.2010.09.022.

Arnoldussen, J., S. Errami, R. Semenov, G. Roemers, M. Blok, M. Kamps, and K. Faes. 2020. *Materiaalstromen, milieu-impact en energieverbruik in de woning-en utiliteitsbouw (Material flows, environmental impact and energy consumption in residential and non-residential construction）.*

Arrigoni, A., I.D. Posen, H.L. MacLean, M. Duhamel, and D.K. Panesar. 2020. Optimizing the Use of a Constrained Resource to Minimize Regional Greenhouse Gas Emissions: The Case Study of Slag in Ontario's Concrete. *Environmental Science and Technology* 54(20): 12840–12849.

Arrow, K.J., M.L. Cropper, C. Gollier, B. Groom, G.M. Heal, R.G. Newell, W.D. Nordhaus, et al. 2014. Should governments use a declining discount rate in project analysis? *Review of Environmental Economics and Policy*.

Asdrubali, F., I. Ballarini, V. Corrado, L. Evangelisti, G. Grazieschi, and C. Guattari. 2019. Energy and environmental payback times for an NZEB retrofit. *Building and Environment* 147(August 2018): 461–472. https://doi.org/10.1016/j.buildenv.2018.10.047.

Atici, K.B. and V. V. Podinovski. 2015. Using data envelopment analysis for the assessment of technical efficiency of units with different specialisations: An application to agriculture. *Omega* 54: 72–83. https://linkinghub.elsevier.com/retrieve/pii/S0305048315000171.

Atmaca, A. 2016a. Life-cycle assessment and cost analysis of residential buildings in South East of Turkey: part 2—a case study. *International Journal of Life Cycle Assessment* 21(7): 925–942. http://dx.doi.org/10.1007/s11367-016-1051-7.

Atmaca, A. 2016b. Life cycle assessment and cost analysis of residential buildings in south east of Turkey: part 1—review and methodology. *The International Journal of Life Cycle Assessment* 21(6): 831–846. http://dx.doi.org/10.1007/s11367-016-1050-8.

Auer, J., N. Bey, and J.M. Schäfer. 2017. Combined Life Cycle Assessment and Life Cycle Costing in the Eco-Care-Matrix: A case study on the performance of a modernized manufacturing system for glass containers. *Journal of Cleaner Production* 141: 99–109.

Augiseau, V. and S. Barles. 2017. Studying construction materials flows and stock: A review. *Resources, Conservation and Recycling* 123: 153–164. http://dx.doi.org/10.1016/j.resconrec.2016.09.002.

Azapagic, A., C. Pettit, and P. Sinclair. 2007. A life cycle methodology for mapping the flows of pollutants in the urban environment. *Clean Technologies and Environmental Policy* 9(3): 199–214.

Baccini, P. and P.H. Btunner. 2012. *Metabolism of the anthroposphere-analysis, evaluation, design*. Second edi. Cambridge, Massachusetts,and London England: The MIT Press.

Bastein, T., E. Rietveld, and E. Keijzer. 2017. *Ex-ante evaluatie van het Rijksbrede Programma Circulaire Economie*. Hague. https://repository.tno.nl/islandora/object/uuid:5cc51c6c-f078-4226-b5fd-feb1fab0b073.

Benachio, G.L.F., M. do C.D. Freitas, and S.F. Tavares. 2020. Circular economy in the construction industry: A systematic literature review. *Journal of Cleaner Production* 260: 121046. https://doi.org/10.1016/j.jclepro.2020.121046.

Berggren, B., M. Hall, and M. Wall. 2013. LCE analysis of buildings – Taking the step towards Net Zero Energy Buildings. *Energy and Buildings* 62: 381–391. https://linkinghub.elsevier.com/retrieve/pii/S0378778813001709.

Bergsdal, H., R.A. Bohne, and H. Brattebø. 2008. Projection of Construction and Demolition Waste in Norway. *Journal of Industrial Ecology* 11(3): 27–39. http://doi.wiley.com/10.1162/jiec.2007.1149.

Bertram, M., T.. Graedel, H. Rechberger, and S. Spatari. 2002. The contemporary European copper cycle: waste management subsystem. *Ecological Economics* 42(1–2): 43–57. https://linkinghub.elsevier.com/retrieve/pii/S0921800902001003.

Bio Intelligence Service. 2011. *Service Contract on Management of Construction and Demolition Waste – SR1, Final Report Task 2. European Commission (DG ENV) ENV.G.4/FRA/2008/0112*. https://ec.europa.eu/environment/waste/pdf/2011_CDW_Report.pdf.

Birat, J.-P. 2015. Life-cycle assessment, resource efficiency and recycling. *Metallurgical Research & Technology* 112(2): 206. http://www.metallurgical-research.org/10.1051/metal/2015009.

Blomsma, F. and G. Brennan. 2017. The Emergence of Circular Economy: A New Framing Around Prolonging Resource Productivity. *Journal of Industrial Ecology* 21(3): 603–614.

Bohne, R.A., H. Brattebø, and H. Bergsdal. 2008. Dynamic Eco-Efficiency Projections for Construction and Demolition Waste Recycling Strategies at the City Level. *Journal of Industrial Ecology* 12(1): 52–68. http://doi.wiley.com/10.1111/j.1530-9290.2008.00013.x.

BPIE. 2017. Factsheet 97% of Buildings in the EU Need to be Upgraded. http://bpie.eu/wp-content/uploads/2017/12/State-of-the-building-stock-briefing_Dic6.pdf. Accessed March 26, 2020.

Brunner, P.H. and H. Rechberger. 2004. *Practical handbook of material flow analysis*. *Lewis Publishers*. Washington DC.

Bull, J., A. Gupta, D. Mumovic, and J. Kimpian. 2014. Life cycle cost and carbon footprint of energy efficient refurbishments to 20th century UK school buildings. *International Journal of Sustainable Built Environment* 3(1): 1–17. http://dx.doi.org/10.1016/j.ijsbe.2014.07.002.

Burchart-Korol, D., P. Krawczyk, K. Czaplicka-Kolarz, and A. Smoliński. 2016. Eco-efficiency of underground coal gasification (UCG) for electricity production. *Fuel* 173: 239–246.

Buyle, M., J. Braet, and A. Audenaert. 2013. Life cycle assessment in the construction sector: A review. *Renewable and Sustainable Energy Reviews* 26: 379–388. http://dx.doi.org/10.1016/j.rser.2013.05.001.

Cabeza, L.F., L. Rincón, V. Vilariño, G. Pérez, and A. Castell. 2014. Life cycle assessment (LCA) and life cycle energy analysis (LCEA) of buildings and the building sector: A review. *Renewable and Sustainable Energy Reviews* 29: 394–416. http://dx.doi.org/10.1016/j.rser.2013.08.037.

Cao, Z., R.J. Myers, R.C. Lupton, H. Duan, R. Sacchi, N. Zhou, T. Reed Miller, J.M. Cullen, Q. Ge, and G. Liu. 2020. The sponge effect and carbon emission mitigation potentials of the global cement cycle. *Nature Communications* 11(1): 3777. http://dx.doi.org/10.1038/s41467-020-17583-w.

Carruth, M.A., J.M. Allwood, and M.C. Moynihan. 2011. The technical potential for reducing metal requirements through lightweight product design. *Resources, Conservation and Recycling* 57: 48–60. http://dx.doi.org/10.1016/j.resconrec.2011.09.018.

CBS. 2019a. Population, households and population development;from 1899. https://opendata.cbs.nl/statline/#/CBS/nl/dataset/37556/table?ts=1603787196611. Accessed March 24, 2020.

CBS. 2019b. Population forecast: gender and age, 2020-2060. https://opendata.cbs.nl/statline/#/CBS/nl/dataset/84646NED/table?ts=1579182479470. Accessed April 8, 2020.

CBS. 2021a. New-build homes: input price index construction costs 2000=100, from 1990. https://opendata.cbs.nl/statline/#/CBS/nl/dataset/80444ned/table?fromstatweb.

CBS. 2021b. Which sectors emit greenhouse gases? https://www.cbs.nl/nl-nl/dossier/dossier-broeikasgassen/hoofdcategorieen/welke-sectoren-stoten-broeikasgassen-uit-. Accessed February 24, 2021.

CEN. 2012. EN 15804:Sustainability of construction works - Environmental product declarations - Core rules for the product category of construction products.

Chalmers, P. 2014. Climate change: Implications for buildings. http://bpie.eu/publication/climate-change-implications-for-buildings/. Accessed March 26, 2020.

Chastas, P., T. Theodosiou, K.J. Kontoleon, and D. Bikas. 2018. Normalising and assessing carbon emissions in the building sector: A review on the embodied CO 2 emissions of residential buildings. *Building and Environment* 130: 212–226. https://doi.org/10.1016/j.buildenv.2017.12.032.

Chau, C.K., T.M. Leung, and W.Y. Ng. 2015. A review on life cycle assessment, life cycle energy assessment and life cycle carbon emissions assessment on buildings. *Applied Energy* 143(1): 395–413.

Chen, L. and G. Jia. 2017. Environmental efficiency analysis of China's regional industry: a data envelopment analysis (DEA) based approach. *Journal of Cleaner Production* 142: 846–853. https://linkinghub.elsevier.com/retrieve/pii/S0959652616000809.

Chen, W.Q. 2018. Dynamic Product-Level Analysis of In-Use Aluminum Stocks in the United States. *Journal of Industrial Ecology*.

Chen, W.Q. and T.E. Graedel. 2012. Dynamic analysis of aluminum stocks and flows in the United States: 1900-2009. *Ecological Economics*.

Chen, W.Q. and L. Shi. 2012. Analysis of aluminum stocks and flows in mainland China from 1950 to 2009: Exploring the dynamics driving the rapid increase in China's aluminum production. *Resources, Conservation and Recycling*.

Cheng, J.C.P. and L.Y.H. Ma. 2013. A BIM-based system for demolition and renovation waste estimation and planning. *Waste Management* 33(6): 1539–1551. http://dx.doi.org/10.1016/j.wasman.2013.01.001.

Chong, W.K. and C. Hermreck. 2011. Modeling the use of transportation energy for recycling construction steel. *Clean Technologies and Environmental Policy* 13(2): 317–330.

Ciroth, A., D. Hunkeler, G. Huppes, K. Lichtenvort, G. Rebitzer, I. Rüdenauer, and B. Steen. 2008. Environmental Life Cycle Costing. In *Environmental Life Cycle Costing*. SETAC, CRC Press.

CLO. 2021. Bouw-en sloopafval: vrijkomen en verwerking, 1985-2018(14 February 2020). https://www.clo.nl/indicatoren/nl0147-bouw--en-sloopafval. Accessed March 2, 2021.

Cochran, K., T. Townsend, D. Reinhart, and H. Heck. 2007. Estimation of regional building-related C&D debris generation and composition: Case study for

Florida, US. *Waste Management* 27(7): 921–931. https://linkinghub.elsevier.com/retrieve/pii/S0956053X06001449.

Coelho, A. and J. De Brito. 2011. Generation of construction and demolition waste in Portugal. *Waste Management and Research* 29(7): 739–750.

Comodi, G., M. Bevilacqua, F. Caresana, C. Paciarotti, L. Pelagalli, and P. Venella. 2016. Life cycle assessment and energy-CO2-economic payback analyses of renewable domestic hot water systems with unglazed and glazed solar thermal panels. *Applied Energy* 164: 944–955. http://dx.doi.org/10.1016/j.apenergy.2015.08.036.

Condeixa, K., A. Haddad, and D. Boer. 2017. Material flow analysis of the residential building stock at the city of Rio de Janeiro. *Journal of Cleaner Production* 149: 1249–1267. http://dx.doi.org/10.1016/j.jclepro.2017.02.080.

Cucurachi, S., C. Van Der Giesen, and J. Guinée. 2018. Ex-ante LCA of Emerging Technologies. *Procedia CIRP* 69(May): 463–468. http://dx.doi.org/10.1016/j.procir.2017.11.005.

D'Oca, S., T. Hong, and J. Langevin. 2018. The human dimensions of energy use in buildings: A review. *Renewable and Sustainable Energy Reviews* 81: 731–742. https://linkinghub.elsevier.com/retrieve/pii/S1364032117311632.

Dahlbo, H., J. Bachér, K. Lähtinen, T. Jouttijärvi, P. Suoheimo, T. Mattila, S. Sironen, T. Myllymaa, and K. Saramäki. 2015. Construction and demolition waste management - A holistic evaluation of environmental performance. *Journal of Cleaner Production* 107: 333–341.

Danish Energy Agency. 2015. *Energy and Climate Policies beyond 2020 in Europe - Overall and selected countries*. https://ens.dk/sites/ens.dk/files/Globalcooperation/eu_energy_and_climate_policy_overview.pdf.

Deetman, S., S. Marinova, E. van der Voet, D.P. van Vuuren, O. Edelenbosch, and R. Heijungs. 2020. Modelling global material stocks and flows for residential and service sector buildings towards 2050. *Journal of Cleaner Production* 245: 118658. https://doi.org/10.1016/j.jclepro.2019.118658.

Deng, S., R.Z. Wang, and Y.J. Dai. 2014. How to evaluate performance of net zero energy building - A literature research. *Energy* 71(2014): 1–16. http://dx.doi.org/10.1016/j.energy.2014.05.007.

Dijksma, S.A.M. and H.G.J. Kamp. 2016. *A circular economy in the Netherlands by 2050*. https://www.government.nl/documents/policy-notes/2016/09/14/a-circular-economy-in-the-netherlands-by-2050.

DIN. 2011. *DIN V 18599 - Energy Performance of Buildings*.

Ding, Z., W. Gong, V.W.Y. Tam, and I.M.C.S. Illankoon. 2019a. Conceptual framework for renovation waste management based on renovation waste generation rates in residential buildings: An empirical study in China. *Journal of Cleaner Production* 228: 284–293. https://doi.org/10.1016/j.jclepro.2019.04.153.

Ding, Z., M. Shi, C. Lu, Z. Wu, D. Chong, and W. Gong. 2019b. Predicting Renovation Waste Generation Based on Grey System Theory: A Case Study of Shenzhen. *Sustainability* 11(16): 4326. https://www.mdpi.com/2071-1050/11/16/4326.

Dissanayake, D.M.K.W., C. Jayasinghe, and M.T.R. Jayasinghe. 2017. A comparative embodied energy analysis of a house with recycled expanded polystyrene (EPS) based foam concrete wall panels. *Energy and Buildings* 135: 85–94. http://dx.doi.org/10.1016/j.enbuild.2016.11.044.

Dodoo, A., L. Gustavsson, and R. Sathre. 2010. Life cycle primary energy implication of retrofitting a wood-framed apartment building to passive house standard. *Resources, Conservation and Recycling* 54(12): 1152–1160. http://dx.doi.org/10.1016/j.resconrec.2010.03.010.

Dong, B., C. Kennedy, and K. Pressnail. 2005. Comparing life cycle implications of building retrofit and replacement options. *Canadian Journal of Civil Engineering* 32(6): 1051–1063.

Dong, D., L.A.T. Espinoza, A. Loibl, M. Pfaff, A. Tukker, and E. Van der Voet. 2020. Scenarios for anthropogenic copper demand and supply in China: implications of a scrap import ban and a circular economy transition. *Resources, Conservation and Recycling* 161(January).

Dong, D., A. Tukker, and E. Van der Voet. 2019. Modeling copper demand in China up to 2050: A business-as-usual scenario based on dynamic stock and flow analysis. *Journal of Industrial Ecology* 23(6): 1363–1380. https://onlinelibrary.wiley.com/doi/abs/10.1111/jiec.12926.

Duurzaam Gebouwd. 2014. Renovatie buitengevel zorgt voor win-winsituatie. https://www.duurzaamgebouwd.nl/project/20141016-renovatie-buitengevel-zorgt-voor-win-winsituatie. Accessed November 22, 2020.

Dylewski, R. and J. Adamczyk. 2014. Life cycle assessment (LCA) of building thermal insulation materials. In *Eco-Efficient Construction and Building Materials*, 267–286. Elsevier. http://dx.doi.org/10.1533/9780857097729.2.267.

Eberhardt, L.C.M., H. Birgisdottir, and M. Birkved. 2019. Potential of Circular Economy in Sustainable Buildings. In *IOP Conference Series: Materials Science and Engineering*, 471:. February 24. https://iopscience.iop.org/article/10.1088/1757-899X/471/9/092051.

EC-JRC. 2016. *Guidance for the implementation of the EU Product Environmental Footprint (PEF) during the Environmental Footprint (EF) pilot phase*.

http://ec.europa.eu/environment/eussd/smgp/pdf/Guidance_products.pdf (accessed 11.12.2018).

EC. 2002. Directive 2002/91/EC of the European Parliament and of the Council of 16 December 2002 on the energy performance of buildings. https://eur-lex.europa.eu/legal-content/en/ALL/?uri=CELEX:32002L0091. Accessed March 16, 2020.

EC. 2008a. Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives (Text with EEA relevance). https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0098. Accessed April 13, 2018.

EC. 2008b. *DIRECTIVE 2008/98/EC*.

EC. 2009. Directive 2009/125/EC of the European Parliament and of the council of 21 october 2009-establishing a framework for the setting of ecodesign requirements for energy-related products. https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX%3A32009L0125.

EC. 2010. Directive 2010/31/EU of the European Parliament and of the Council of 19 May 2010 on the energy performance of buildings. https://eur-lex.europa.eu/eli/dir/2010/31/oj. Accessed March 1, 2020.

EC. 2011a. *European Commission-DG ENV 2011 Management of C&DW in EU, Final Report elaborated by BIO Intelligence Service.*

EC. 2011b. A roadmap to a resource efficient Europe (COM(2011) 571). https://ec.europa.eu/environment/resource_efficiency/about/roadmap/index_en.htm. Accessed April 11, 2020.

EC. 2011c. *SEC/2011/0288 final-A Roadmap for moving to a competitive low carbon economy in 2050*. Reussel. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52011SC0288.

EC. 2012a. *Use of Economic Instruments & Waste Management Performances - Final Report*. http://ec.europa.eu/environment/waste/pdf/final_report_10042012.pdf.

EC. 2012b. Directive 2012/27/EU of the European Parliament and of the Council of 25 October 2012 on energy efficiency, amending Directives 2009/125/EC and 2010/30/EU and repealing Directives 2004/8/EC and 2006/32/EC (Text with EEA relevance). https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32012L0027. Accessed March 1, 2020.

EC. 2014a. Towards a circular economy: A zero waste programme for Europe c. https://eur-lex.europa.eu/resource.html?uri=cellar:aa88c66d-4553-11e4-a0cb-01aa75ed71a1.0022.03/DOC_1&format=PDF. Accessed April 11, 2020.

EC. 2014b. 2030 climate and energy goals for a competitive , secure and low-carbon EU economy. *Press Release*.

EC. 2014c. COM/2014/0445 final of Communication from the commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the regions on resource efficiency opportunities in the building sector. https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1411482206636&uri=CELEX:52014DC0445. Accessed March 1, 2020.

EC. 2014d. Resource efficiency opportunities in the building sector (COM/2014/0445 final). https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1411482206636&uri=CELEX:52014DC0445. Accessed April 11, 2020.

EC. 2015a. COM/2015/0614: Closing the loop - An EU action plan for the Circular Economy. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52015DC0614. Accessed August 13, 2021.

EC. 2015b. *COM/C(2015) 6317 final: Towards an Integrated Strategic Energy Technology (SET) Plan: Accelerating the European Energy System Transformation*. https://ec.europa.eu/energy/sites/ener/files/publication/Complete-A4-setplan.pdf.

EC. 2018. Energy efficiency： Buildings. https://ec.europa.eu/energy/en/topics/energy-efficiency/buildings.

EC. 2020a. Waste: construction and Demolition Waste. http://ec.europa.eu/environment/waste/construction_demolition.htm. Accessed March 1, 2020.

EC. 2020b. Circular Economy - Principles for Building Design. https://ec.europa.eu/docsroom/documents/39984.

EC. 2020c. A new Circular Economy Action Plan:For a cleaner and more competitive Europe. https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1583933814386&uri=COM:2020:98:FIN. Accessed August 10, 2021.

Economidou, Marina. 2011. *Europe's buildings under the microscope: A country by country review of the energy performance of buildings*. Brussel. http://bpie.eu/wp-content/uploads/2015/10/HR_EU_B_under_microscope_study.pdf.

Economidou, M. 2021. European energy efficiency platform: energy renovation. https://e3p.jrc.ec.europa.eu/articles/energy-renovation. Accessed November 28, 2020.

Ellen MacArthur Foundation. 2013. *Towards the circular economy: economic and business rationale for an accelerated transition*. https://www.ellenmacarthurfoundation.org/assets/downloads/publications/Ellen-MacArthur-Foundation-Towards-the-Circular-Economy-vol.1.pdf.

Ellen MacArthur Foundation. 2015a. *Towards a Circular Economy: Business Rationale for an Accelerated Transition*. https://www.ellenmacarthurfoundation.org/assets/downloads/TCE_Ellen-MacArthur-Foundation_9-Dec-2015.pdf.

Ellen MacArthur Foundation. 2015b. Growth within: a circular economy vision for a competitive europe: 100. https://www.ellenmacarthurfoundation.org/assets/downloads/publications/EllenMacArthurFoundation_Growth-Within_July15.pdf.

EN 15459. 2008. Energy Performance of Buildings. Economic Evaluation Procedure for Energy Systems in Buildings.

ESCAP. 2009. *Eco-efficiency Indicators: Measuring Resource-use Efficiency and the Impact of Economic Activities on the Environment*. Bangkok, Thailand.

Esser, A., A. Dunne, T. Meeusen, S. Quaschning, and W. Denis. 2019. *Comprehensive study of building energy renovation activities and the uptake of nearly zero-energy buildings in the EU(Final report)*. https://ec.europa.eu/energy/sites/ener/files/documents/1.final_report.pdf.

Eurostat. 2018. *Economy-wide material flow accounts handbook*. Luxembourg: Publications Office of the European Union. http://ec.europa.eu/eurostat/about/policies/copyright%0Ahttps://ec.europa.eu/eurostat/documents/3859598/9117556/KS-GQ-18-006-EN-N.pdf/b621b8ce-2792-47ff-9d10-067d2b8aac4b%0Ahttp://epp.eurostat.ec.europa.eu/portal/page/portal/product_details/publication?p_.

Eurostat. 2020. Electricity prices by type of user. https://ec.europa.eu/eurostat/tgm/table.do?tab=table&init=1&language=en&pcode=ten00117&plugin=1. Accessed April 13, 2020.

Eurostat. 2021. Recovery rate of construction and demolition waste. https://ec.europa.eu/eurostat/databrowser/view/cei_wm040/default/table?lang=en. Accessed April 5, 2021.

EZK. 2015. *Energy report-Transition to sustainable energy*. Hague. https://www.government.nl/documents/reports/2016/04/28/energy-report-transition-tot-sustainable-energy.

EZK. 2019. *Integrated National Energy and Climate Plan 2021-2030:the Netherlands*. Hague |. https://ec.europa.eu/energy/sites/ener/files/documents/nl_final_necp_main_en.pdf.

Faludi, J. and M. Lepech. 2012. Ecological payback time of an energy-efficient modular building. *Journal of Green Building* 7(1): 100–119.

Fernández-García, A., V. Ibáñez-Forés, and M.D. Bovea. 2016. Eco-efficiency analysis of the life cycle of interior partition walls: A comparison of alternative solutions. *Journal of Cleaner Production* 112(1): 649–665.

Friedlingstein, P., R.M. Andrew, J. Rogelj, G.P. Peters, J.G. Canadell, R. Knutti, G. Luderer, et al. 2014. Persistent growth of CO2emissions and implications for reaching climate targets. *Nature Geoscience* 7(10): 709–715. http://dx.doi.org/10.1038/ngeo2248.

Frischknecht, R. 1997. The seductive effect of identical physical units. *The International Journal of Life Cycle Assessment* 2(3): 125–126. http://link.springer.com/10.1007/BF02978800.

Frischknecht, R., N. Jungbluth, and H.J. Althaus. 2003. *Implementation of life cycle impact assessment methods (Final report Ecoinvent 2000)*. Duebendorf.

Frischknecht, R., F. Wyss, S. Büsser Knöpfel, T. Lützkendorf, and M. Balouktsi. 2015. Cumulative energy demand in LCA: the energy harvested approach. *International Journal of Life Cycle Assessment* 20(7): 957–969.

Fuller, S.K. and S.R. Petersen. 1995. *Life-Cycle Costing Manual of the US Federal Energy Management Program*. *NIST Handbook 135*. 1995th ed. Gaithersburg.

Gagg, C.R. 2014. Cement and concrete as an engineering material: An historic appraisal and case study analysis. *Engineering Failure Analysis* 40: 114–140. http://dx.doi.org/10.1016/j.engfailanal.2014.02.004.

Gálvez-Martos, J.L., D. Styles, H. Schoenberger, and B. Zeschmar-Lahl. 2018. Construction and demolition waste best management practice in Europe. *Resources, Conservation and Recycling* 136(April): 166–178. https://doi.org/10.1016/j.resconrec.2018.04.016.

Gassner, A., J. Lederer, and J. Fellner. 2020. Material stock development of the transport sector in the city of Vienna. *Journal of Industrial Ecology*: jiec.13024. https://onlinelibrary.wiley.com/doi/abs/10.1111/jiec.13024.

Gebremariam, A.T., F. Di Maio, A. Vahidi, and P. Rem. 2020. Innovative technologies for recycling End-of-Life concrete waste in the built environment. *Resources, Conservation and Recycling* 163: 104911. https://doi.org/10.1016/j.resconrec.2020.104911.

Geissdoerfer, M., P. Savaget, N.M.P. Bocken, and E.J. Hultink. 2017. The Circular Economy – A new sustainability paradigm? *Journal of Cleaner Production* 143: 757–768. http://dx.doi.org/10.1016/j.jclepro.2016.12.048.

Gerdessen, J.C. and S. Pascucci. 2013. Data Envelopment Analysis of sustainability indicators of European agricultural systems at regional level. *Agricultural Systems* 118: 78–90. https://linkinghub.elsevier.com/retrieve/pii/S0308521X13000371.

Ghaffar, S.H., M. Burman, and N. Braimah. 2020. Pathways to circular construction: An integrated management of construction and demolition waste for resource recovery. *Journal of Cleaner Production* 244: 118710. https://doi.org/10.1016/j.jclepro.2019.118710.

Gharfalkar, M., R. Court, C. Campbell, Z. Ali, and G. Hillier. 2015. Analysis of waste hierarchy in the European waste directive 2008/98/EC. *Waste Management* 39: 305–313. http://dx.doi.org/10.1016/j.wasman.2015.02.007.

Ghisellini, P., C. Cialani, and S. Ulgiati. 2016. A review on circular economy: The expected transition to a balanced interplay of environmental and economic systems. *Journal of Cleaner Production* 114: 11–32. http://dx.doi.org/10.1016/j.jclepro.2015.09.007.

Gluch, P. and H. Baumann. 2004. The life cycle costing (LCC) approach: a conceptual discussion of its usefulness for environmental decision-making. *Building and Environment* 39(5): 571–580. https://linkinghub.elsevier.com/retrieve/pii/S0360132303002610.

Goldstein, B., M. Birkved, M.B. Quitzau, and M. Hauschild. 2013. Quantification of urban metabolism through coupling with the life cycle assessment framework: Concept development and case study. *Environmental Research Letters* 8(3).

Göswein, V., A.B. Gonçalves, J.D. Silvestre, F. Freire, G. Habert, and R. Kurda. 2018. Transportation matters – Does it? GIS-based comparative environmental assessment of concrete mixes with cement, fly ash, natural and recycled aggregates. *Resources, Conservation and Recycling* 137: 1–10.

Government of the Netherlands. 2012. Netherlands nearly zero-energy buildings national plans.

Government of the Netherlands. 2019. *Climate Agreement*. Hague. https://www.government.nl/documents/reports/2019/06/28/climate-agreement.

Government of the Netherlands. 2020. Main asbestos regulations. https://www.government.nl/topics/asbestos/regulations. Accessed December 7, 2020.

Graedel, T. and B. Allenby. 2003. *Industrial Ecology*. New Jersy: Prentice Hall.

Groleau, G., F. Tanguay-Rioux, L. Spreutels, M. Heroux, and R. Legros. 2018. Development of a msw gasification model for flexible integration into a mfa-lca framework. *Detritus* 7: 44–54. https://digital.detritusjournal.com/doi/13850-245.

Gruijl, J. de. 2018. The future of prefab: what was the situation in 2016 and where is it going? https://www.usp-mc.nl/nl/artikel/667/the-future-of-prefab-what-was-the-situation-in-2016-and-where-is-it-going/.

Guinée, J.B., R. Heijungs, G. Huppes, R. Kleijn, A. de Koning, L. van Oers, A. Wegener Sleeswijk, et al. 2001. *Handbook on life cycle assessment: operational guide to the ISO standards*. *Kluwer Academic Publishing*. Dordrencht.

Guinée, J.B., R. Heijungs, G. Huppes, A. Zamagni, P. Masoni, R. Buonamici, T. Ekvall, and T. Rydberg. 2011. Life cycle assessment: Past, present, and future. *Environmental Science and Technology* 45(1): 90–96.

Günkaya, Z. 2020. A Comparison of the Environmental Impact of Different Heating Systems Using the Life Cycle Approach. *Environmental Engineering Science* 37(3): 214–228. https://www.liebertpub.com/doi/10.1089/ees.2019.0272.

Haes, H.U. de, E. van der Voet, and R. Kleijn. 1997. From Quality to Quantity: Substance Flow Analysis (SFA), an analytical tool for integrated chain management. In *Proceedings of the ConAccount Workshop*. Wuppertal:Institute for Climate, Environmental and Energy.

Heeren, N. and S. Hellweg. 2019. Tracking Construction Material over Space and Time: Prospective and Geo-referenced Modeling of Building Stocks and Construction Material Flows. *Journal of Industrial Ecology* 23(1): 253–267.

Hincapié, I., A. Caballero-Guzman, D. Hiltbrunner, and B. Nowack. 2015. Use of engineered nanomaterials in the construction industry with specific emphasis on paints and their flows in construction and demolition waste in Switzerland. *Waste Management* 43: 398–406. https://linkinghub.elsevier.com/retrieve/pii/S0956053X15300222.

Hoornweg, D. and P. Bhada-Tata. 2012. *What a waste: a global review of solid waste management*.

Hoque, M.R., X.G. i. Durany, C.S. i. Sala, G.V. Méndez, L.T. Peiró, and T.V. Huguet. 2012. Energy Intensity of the Catalan Construction Sector: An Application of Material and Exergy Flow Analysis. *Journal of Industrial Ecology* 16(5): 699–709.

Hossain, M.U., C.S. Poon, I.M.C. Lo, and J.C.P. Cheng. 2016. Comparative environmental evaluation of aggregate production from recycled waste materials and virgin sources by LCA. *Resources, Conservation and Recycling*.

Hossain, Y. and T. Marsik. 2019. Conducting Life Cycle Assessments (LCAs) to Determine Carbon Payback: A Case Study of a Highly Energy-Efficient House in Rural Alaska. *Energies* 12(9): 1732. https://www.mdpi.com/1996-1073/12/9/1732.

Hsiao, T.Y., Y.T. Huang, Y.H. Yu, and I.K. Wernick. 2002. Modeling materials flow of waste concrete from construction and demolition wastes in Taiwan. *Resources Policy* 28(1–2): 39–47.

Hu, M., R. Kleijn, K.P. Bozhilova-Kisheva, and F. Di Maio. 2013. An approach to LCSA: the case of concrete recycling. *The International Journal of Life Cycle Assessment* 18(9): 1793–1803. http://link.springer.com/10.1007/s11367-013-0599-8.

Hu, M., F. Di Maio, X. Lin, and E. Van Roekel. 2012. Fates of end-of-life concrete and their economic implications. In *Proceedings of the 27th International Conference on Solid Waste Technology and Management.*, 1–9. Philadelphia. http://www.c2ca.eu/custom/page/page_block/icsw2012c2cacdwcost.pdf.

Hu, M., S. Pauliuk, T. Wang, G. Huppes, E. van der Voet, and D.B. Müller. 2010a. Iron and steel in Chinese residential buildings: A dynamic analysis. *Resources, Conservation and Recycling* 54(9): 591–600.

Hu, M., E. van der Voet, and G. Huppes. 2010b. Dynamic Material Flow Analysis for Strategic Construction and Demolition Waste Management in Beijing. *Journal of Industrial Ecology* 14(3): 440–456.

Huang, B., X. Gao, X. Xu, J. Song, Y. Geng, J. Sarkis, T. Fishman, H. Kua, and J. Nakatani. 2020. A Life Cycle Thinking Framework to Mitigate the Environmental Impact of Building Materials. *One Earth* 3(5): 564–573. https://doi.org/10.1016/j.oneear.2020.10.010.

Huang, B., X. Wang, H. Kua, Y. Geng, R. Bleischwitz, and J. Ren. 2018a. Construction and demolition waste management in China through the 3R principle. *Resources, Conservation and Recycling* 129: 36–44. https://doi.org/10.1016/j.resconrec.2017.09.029.

Huang, B., F. Zhao, T. Fishman, W.Q. Chen, N. Heeren, and E.G. Hertwich. 2018b. Building Material Use and Associated Environmental Impacts in China 2000-2015. Research-article. *Environmental Science and Technology* 52(23): 14006–14014.

Huang, C.L., J. Vause, H.W. Ma, and C.P. Yu. 2012a. Using material/substance flow analysis to support sustainable development assessment: A literature review and outlook. *Resources, Conservation and Recycling* 68: 104–116. http://dx.doi.org/10.1016/j.resconrec.2012.08.012.

Huang, Y., J.L. Niu, and T.M. Chung. 2012b. Energy and carbon emission payback analysis for energy-efficient retrofitting in buildings - Overhang shading option. *Energy and Buildings* 44(1): 94–103. http://dx.doi.org/10.1016/j.enbuild.2011.10.027.

Huijbregts, M.A.J., G.A. Norris, R. Bretz, A. Ciroth, B. Maurice, N. Mahasenan, B. von Bahr, and B.. Weidema. 2003. *Code of Life-Cycle Inventory Practice*. Ed. by A. S. H. de Baufort-Langeveld, R. Bretz, R. Hishier, M. A. J. Huibregts, P. Jean, T. Tanner, and G. van Hoof. Brussel: Society of Environmental Toxicology and Chemistry (SETAC).

Hunkeler, D., K. Lichtenvort, and G. Rebitzer. 2008. *Environmental life cycle costing*. SETAC Publications.

Hunkeler, D., K. Saur, H. Stranddorf, G. Rebitzer, M. Finkbeiner, W. Schmidt, A.A. Jensen, and K. Christiansen. 2003. *Life cycle management*. Pensacola, Florida, USA: SETAC Press.

Huppes, G. and M. Ishikawa. 2005. A Framework for Quantified Eco-efficiency Analysis. *Journal of Industrial Ecology* 9(4): 25–41. http://doi.wiley.com/10.1162/108819805775247882.

Ibbotson, S., T. Dettmer, S. Kara, and C. Herrmann. 2013. Eco-efficiency of disposable and reusable surgical instruments - A scissors case. *International Journal of Life Cycle Assessment* 18(5): 1137–1148.

Ibn-Mohammed, T., S.C.L. Koh, I.M. Reaney, A. Acquaye, D. Wang, S. Taylor, and A. Genovese. 2016. Integrated hybrid life cycle assessment and supply chain environmental profile evaluations of lead-based (lead zirconate titanate) versus lead-free (potassium sodium niobate) piezoelectric ceramics. *Energy & Environmental Science* 9(11): 3495–3520. http://dx.doi.org/10.1039/C6EE02429G.

IEA. 2020. *Energy Technology Perspectives 2020*. *Energy Technology Perspectives 2020*.

IEA. 2021. *Net Zero by 2050-A Roadmap for the Global Energy Sector*.

IPCC. 2007. *Climate Change 2007: Synthesis Report*. Geneva, Switzerland. https://www.ipcc.ch/site/assets/uploads/2018/02/ar4_syr_full_report.pdf.

IRP. 2019. *Global Resources Outlook 2019: Natural resources for the future we want*. *Global Resources Outlook 2019*. Nairobi. https://www.resourcepanel.org/reports/global-resources-outlook.

Islam, H., M. Jollands, and S. Setunge. 2015. Life cycle assessment and life cycle cost implication of residential buildings - A review. *Renewable and Sustainable Energy Reviews* 42: 129–140. http://dx.doi.org/10.1016/j.rser.2014.10.006.

Islam, H., M. Jollands, S. Setunge, I. Ahmed, and N. Haque. 2014. Life cycle assessment and life cycle cost implications of wall assemblages designs. *Energy and Buildings* 84: 33–45. http://dx.doi.org/10.1016/j.enbuild.2014.07.041.

ISO. 2006a. *ISO 14040: Environmental management — Life cycle assessment — Principles and framework*. Geneva.

ISO. 2006b. *ISO 14044: Environmental management — Life cycle assessment — Requirements and guidelines*. Geneva.

ISO. 2012. *ISO 14045: Environmental management — Eco-efficiency assessment of product systems — Principles, requirements and guidelines*. Geneva.

ISO. 2017a. *ISO 15686-5:Buildings and Constructed Assets – Service-Life Planning – Part 5: Life-Cycle Costing*. Geneva.

ISO. 2017b. *ISO 6946:Building components and building elements - Thermal resistance and thermal transmittance - Calculation methods*. Geneva.

Jaillon, L., C.S. Poon, and Y.H. Chiang. 2009. Quantifying the waste reduction potential of using prefabrication in building construction in Hong Kong. *Waste Management* 29(1): 309–320. http://dx.doi.org/10.1016/j.wasman.2008.02.015.

Jain, S., S. Singhal, and N.K. Jain. 2019. Construction and demolition waste generation in cities in India: an integrated approach. *International Journal of Sustainable Engineering* 12(5): 333–340. https://doi.org/10.1080/19397038.2019.1612967.

Jeswani, H.K., A. Azapagic, P. Schepelmann, and M. Ritthoff. 2010. Options for broadening and deepening the LCA approaches. *Journal of Cleaner Production* 18(2): 120–127. http://dx.doi.org/10.1016/j.jclepro.2009.09.023.

Jin, R. and Q. Chen. 2019. Overview of Concrete Recycling Legislation and Practice in the United States. *Journal of Construction Engineering and Management* 145(4).

Kalmykova, Y., M. Sadagopan, and L. Rosado. 2018. Circular economy – From review of theories and practices to development of implementation tools. *Resources, Conservation and Recycling* 135: 190–201. https://doi.org/10.1016/j.resconrec.2017.10.034.

Kapur, A., H.G. Van Oss, G. Keoleian, S.E. Kesler, and A. Kendall. 2009. The contemporary cement cycle of the United States. *Journal of Material Cycles and Waste Management* 11(2): 155–165.

Karatum, O., M.M.H. Bhuiya, M.K. Carroll, A.M. Anderson, and D.L. Plata. 2018. Life Cycle Assessment of Aerogel Manufacture on Small and Large Scales: Weighing the Use of Advanced Materials in Oil Spill Remediation. *Journal of Industrial Ecology* 22(6): 1365–1377.

Kayo, C., S.M.R. Dente, C. Aoki-Suzuki, D. Tanaka, S. Murakami, and S. Hashimoto. 2019. Environmental Impact Assessment of Wood Use in Japan through 2050 Using Material Flow Analysis and Life Cycle Assessment. *Journal of Industrial Ecology* 23(3): 635–648. https://onlinelibrary.wiley.com/doi/abs/10.1111/jiec.12766.

Keffer, C., R. Shimp, and M. Lehni. 1999. *Eco-efficiency Indicators & Reporting*. Geneva.

Kemp, R. and H. Van Lente. 2011. The dual challenge of sustainability transitions. *Environmental Innovation and Societal Transitions* 1(1): 121–124.

Kicherer, A., S. Schaltegger, H. Tschochohei, and B.F. Pozo. 2007. Eco-efficiency: Combining life cycle assessment and life cycle costs via normalization. *International Journal of Life Cycle Assessment* 12(7): 537–543.

Kirchherr, J., D. Reike, and M. Hekkert. 2017. Conceptualizing the circular economy: An analysis of 114 definitions. *Resources, Conservation and Recycling* 127: 221–232. https://linkinghub.elsevier.com/retrieve/pii/S0921344917302835.

Kleijn, R. 2000. Adding it all up: The sense and non-sense of bulk-MFA. *Journal of Industrial Ecology*.

Klinglmair, M., C. Vadenbo, T.F. Astrup, and C. Scheutz. 2017. An MFA-based optimization model for increased resource efficiency: Phosphorus flows in Denmark. *Resources, Conservation and Recycling*.

Kloepffer, W. 2003. Life-cycle based methods for sustainable product development. *International Journal of Life Cycle Assessment* 8(3): 157–159.

Kloepffer, W. 2008. Life cycle sustainability assessment of products. *The International Journal of Life Cycle Assessment* 13(2): 89–95. http://link.springer.com/10.1065/lca2008.02.376.

Klöpffer, W. 1997. In defense of the cumulative energy demand. *The International Journal of Life Cycle Assessment* 2(2): 61–61. http://link.springer.com/10.1007/BF02978754.

Klostermann, J.E.M. and A. Tukker. 1998. *Product Innovation and Eco-efficiency*. Kluwer Academic Publishers.

Kneifel, J. 2010. Life-cycle carbon and cost analysis of energy efficiency measures in new commercial buildings. *Energy and Buildings* 42(3): 333–340.

Knoeri, C., E. Sanyé-Mengual, and H.J. Althaus. 2013. Comparative LCA of recycled and conventional concrete for structural applications. *International Journal of Life Cycle Assessment* 18(5): 909–918.

Korhonen, J., A. Honkasalo, and J. Seppälä. 2018. Circular Economy: The Concept and its Limitations. *Ecological Economics* 143: 37–46. https://doi.org/10.1016/j.ecolecon.2017.06.041.

Korhonen, P.J. and M. Luptacik. 2004. Eco-efficiency analysis of power plants: An extension of data envelopment analysis. *European Journal of Operational Research* 154(2): 437–446. https://linkinghub.elsevier.com/retrieve/pii/S0377221703001802.

Korpi, E. and T. Ala-Risku. 2008. Life cycle costing: a review of published case studies. *Managerial Auditing Journal* 23(3): 240–261.

Koutamanis, A., B. van Reijn, and E. van Bueren. 2018. Urban mining and buildings: A review of possibilities and limitations. *Resources, Conservation and Recycling* 138(June): 32–39. https://doi.org/10.1016/j.resconrec.2018.06.024.

Krausmann, F., D. Wiedenhofer, C. Lauk, W. Haas, H. Tanikawa, T. Fishman, A. Miatto, H. Schandl, and H. Haberl. 2017. Global socioeconomic material stocks rise 23-fold over the 20th century and require half of annual resource use. *Proceedings of the National Academy of Sciences* 114(8): 1880–1885. http://www.pnas.org/lookup/doi/10.1073/pnas.1613773114.

Lavers Westin, A., Y. Kalmykova, L. Rosado, F. Oliveira, R. Laurenti, and T. Rydberg. 2019. Combining material flow analysis with life cycle assessment to identify environmental hotspots of urban consumption. *Journal of Cleaner Production* 226: 526–539. https://doi.org/10.1016/j.jclepro.2019.04.036.

Leadership, U. of C.I. for S. 2015. *10 years of carbon pricing in Europe-a business perspective*. https://www.cisl.cam.ac.uk/resources/publication-pdfs/10-years-carbon-pricing-europe.pdf.

Leckner, M. and R. Zmeureanu. 2011. Life cycle cost and energy analysis of a Net Zero Energy House with solar combisystem. *Applied Energy* 88(1): 232–241. http://dx.doi.org/10.1016/j.apenergy.2010.07.031.

Lederer, J., A. Gassner, F. Kleemann, and J. Fellner. 2020. Potentials for a circular economy of mineral construction materials and demolition waste in urban areas: a case study from Vienna. *Resources, Conservation and Recycling* 161(May): 104942. https://doi.org/10.1016/j.resconrec.2020.104942.

Lee, J.Y., K.H. Cha, T.W. Lim, and T. Hur. 2011. Eco-efficiency of H2 and fuel cell buses. *International Journal of Hydrogen Energy* 36(2): 1754–1765. http://dx.doi.org/10.1016/j.ijhydene.2010.10.074.

Li, Y., X. Zhang, G. Ding, and Z. Feng. 2016. Developing a quantitative construction waste estimation model for building construction projects. *Resources, Conservation and Recycling* 106(January): 9–20. http://dx.doi.org/10.1016/j.resconrec.2015.11.001.

Liu, Z., M. Adams, R.P. Cote, Y. Geng, Q. Chen, W. Liu, L. Sun, and X. Yu. 2017. Comprehensive development of industrial symbiosis for the response of greenhouse gases emission mitigation: Challenges and opportunities in China. *Energy Policy* 102: 88–95. http://dx.doi.org/10.1016/j.enpol.2016.12.013.

Lockrey, S., H. Nguyen, E. Crossin, and K. Verghese. 2016. Recycling the construction and demolition waste in Vietnam: opportunities and challenges in practice. *Journal of Cleaner Production* 133: 757–766. https://linkinghub.elsevier.com/retrieve/pii/S0959652616306540.

Lopes Silva, D.A., J.A. De Oliveira, Y.M.B. Saavedra, A.R. Ometto, J. Rieradevall I Pons, and X. Gabarrell Durany. 2015. Combined MFA and LCA approach to evaluate the metabolism of service polygons: A case study on a university campus. *Resources, Conservation and Recycling*.

Lorenzo-Toja, Y., I. Vázquez-Rowe, M.J. Amores, M. Termes-Rifé, D. Marín-Navarro, M.T. Moreira, and G. Feijoo. 2016. Benchmarking wastewater treatment plants under an eco-efficiency perspective. *Science of The Total Environment* 566–567: 468–479. http://dx.doi.org/10.1016/j.scitotenv.2016.05.110.

Lotfi, S., J. Deja, P. Rem, R. Mróz, E. Van Roekel, and H. Van Der Stelt. 2014. Mechanical recycling of EOL concrete into high-grade aggregates. *Resources, Conservation and Recycling* 87: 117–125.

Lotfi, S., M. Eggimann, E. Wagner, R. Mróz, and J. Deja. 2015. Performance of recycled aggregate concrete based on a new concrete recycling technology. *Construction and Building Materials* 95: 243–256.

Lu, K., X. Jiang, J. Yu, V.W.Y. Tam, and M. Skitmore. 2021. Integration of life cycle assessment and life cycle cost using building information modeling: A critical review. *Journal of Cleaner Production* 285: 125438. https://linkinghub.elsevier.com/retrieve/pii/S0959652620354846.

Lu, K. and H. Wang. 2019. Estimation of Building's Life Cycle Carbon Emissions Based on Life Cycle Assessment and Building Information Modeling: A Case Study of a Hospital Building in China. *Journal of Geoscience and Environment Protection* 07(06): 147–165.

Lu, L. and H.X. Yang. 2010. Environmental payback time analysis of a roof-mounted building-integrated photovoltaic (BIPV) system in Hong Kong. *Applied Energy* 87(12): 3625–3631.

Luo, L., E. van der Voet, and G. Huppes. 2009. Life cycle assessment and life cycle costing of bioethanol from sugarcane in Brazil. *Renewable and Sustainable Energy Reviews* 13(6–7): 1613–1619.

Mah, C.M., T. Fujiwara, and C.S. Ho. 2018. Life cycle assessment and life cycle costing toward eco-efficiency concrete waste management in Malaysia. *Journal of Cleaner Production* 172: 3415–3427.

Mália, M., J. de Brito, M.D. Pinheiro, and M. Bravo. 2013. Construction and demolition waste indicators. *Waste Management & Research* 3: 241–255.

Marcellus-Zamora, K.A., P.M. Gallagher, and S. Spatari. 2020. Can Public Construction and Demolition Data Describe Trends in Building Material Recycling? Observations From Philadelphia. *Frontiers in Built Environment* 6: 1–7. https://www.frontiersin.org/article/10.3389/fbuil.2020.00131/full.

Mardani, A., E.K. Zavadskas, D. Streimikiene, A. Jusoh, and M. Khoshnoudi. 2017. A comprehensive review of data envelopment analysis (DEA) approach in energy efficiency. *Renewable and Sustainable Energy Reviews* 70: 1298–1322. https://linkinghub.elsevier.com/retrieve/pii/S1364032116310875.

Maria, A. Di, J. Eyckmans, and K. Van Acker. 2018. Downcycling versus recycling of construction and demolition waste: Combining LCA and LCC to support sustainable policy making. *Waste Management* 75: 3–21. https://doi.org/10.1016/j.wasman.2018.01.028.

Marinova, S., S. Deetman, E. van der Voet, and V. Daioglou. 2020. Global construction materials database and stock analysis of residential buildings between 1970-2050. *Journal of Cleaner Production* 247: 119146. https://doi.org/10.1016/j.jclepro.2019.119146.

Marszal, A.J. and P. Heiselberg. 2011. Life cycle cost analysis of a multi-storey residential Net Zero Energy Building in Denmark. *Energy*.

Martinez-Sanchez, V., M.A. Kromann, and T.F. Astrup. 2015. Life cycle costing of waste management systems: Overview, calculation principles and case studies. *Waste Management* 36: 343–355.

Massarutto, A., A. de Carli, and M. Graffi. 2011. Material and energy recovery in integrated waste management systems: A life-cycle costing approach. *Waste Management* 31(9–10): 2102–2111. http://dx.doi.org/10.1016/j.wasman.2011.05.017.

Mehr, J., C. Vadenbo, B. Steubing, and S. Hellweg. 2018. Environmentally optimal wood use in Switzerland—Investigating the relevance of material cascades. *Resources, Conservation and Recycling* 131: 181–191. https://doi.org/10.1016/j.resconrec.2017.12.026.

MENCP and MIKR. 2017. Fourth National Energy Efficiency Action Plan for the Netherlands: 1–121. https://ec.europa.eu/energy/sites/ener/files/nl_neeap_2017_en.pdf.

Menikpura, S.N.M., J. Sang-Arun, and M. Bengtsson. 2016. Assessment of environmental and economic performance of Waste-to-Energy facilities in Thai cities. *Renewable Energy* 86: 576–584. http://dx.doi.org/10.1016/j.renene.2015.08.054.

Menkveld, M. and L. Beurskens. 2009. *Renewable heating and cooling in the Netherlands*. https://waermepumpe-izw.de/wp-content/uploads/2020/05/090600-Renewable-Heat-Cooling-NL.pdf.

Mesta, C., R. Kahhat, and S. Santa-Cruz. 2017. Quantification of lost material stock of buildings after an earthquake. *16th World Conference on Earthquake*.

Miah, J.H., S.C.L. Koh, and D. Stone. 2017. A hybridised framework combining integrated methods for environmental Life Cycle Assessment and Life Cycle Costing. *Journal of Cleaner Production* 168: 846–866. https://doi.org/10.1016/j.jclepro.2017.08.187.

Miatto, A., H. Schandl, T. Fishman, and H. Tanikawa. 2017a. Global Patterns and Trends for Non-Metallic Minerals used for Construction. *Journal of Industrial Ecology* 21(4): 924–937.

Miatto, A., H. Schandl, and H. Tanikawa. 2017b. How important are realistic building lifespan assumptions for material stock and demolition waste accounts? *Resources, Conservation and Recycling* 122: 143–154. https://linkinghub.elsevier.com/retrieve/pii/S0921344917300265.

Miatto, A., H. Schandl, D. Wiedenhofer, F. Krausmann, and H. Tanikawa. 2017c. Modeling material flows and stocks of the road network in the United States 1905–2015. *Resources, Conservation and Recycling* 127: 168–178. https://linkinghub.elsevier.com/retrieve/pii/S0921344917302720.

Miller, S.A., A. Horvath, and P.J.M. Monteiro. 2016. Readily implementable techniques can cut annual CO2 emissions from the production of concrete by over 20%. *Environmental Research Letters*.

Miller, S.A., A. Horvath, and P.J.M. Monteiro. 2018. Impacts of booming concrete production on water resources worldwide. *Nature Sustainability* 1(1): 69–76. http://www.nature.com/articles/s41893-017-0009-5.

Millward-Hopkins, J., J. Busch, P. Purnell, O. Zwirner, C.A. Velis, A. Brown, J. Hahladakis, and E. Iacovidou. 2018. Fully integrated modelling for sustainability assessment of resource recovery from waste. *Science of the Total Environment* 612: 613–624. https://doi.org/10.1016/j.scitotenv.2017.08.211.

Minne, E. and J.C. Crittenden. 2015. Impact of maintenance on life cycle impact and cost assessment for residential flooring options. *International Journal of Life Cycle Assessment* 20(1): 36–45.

Mohammadpourkarbasi, H. and S. Sharples. 2013. Eco-Retrofitting very old dwellings: current and future energy and carbon performance for two uk cities. In *PLEA2013 - 29th Conference, Ustainable Architecture for a Renewable Future*. Munich, Germany.

Monier, V., M. Hestin, A.-C. Impériale, G. Hobbs, K. Adams, M. Pairon, M.R. de Winghe, et al. 2017. *Resource efficient use of mixed wastes:improving management of construction and demolition waste*.

Monteiro, P.J.M., S.A. Miller, and A. Horvath. 2017. Towards sustainable concrete. *Nature Materials* 16(7): 698–699. http://dx.doi.org/10.1038/nmat4930.

Moore, T. and J. Morrissey. 2014. Lifecycle costing sensitivities for zero energy housing in Melbourne , Australia. *Energy & Buildings* 79: 1–11. http://dx.doi.org/10.1016/j.enbuild.2014.04.050.

Moreno-Juez, J., I.J. Vegas, A.T. Gebremariam, V. García-Cortés, and F. Di Maio. 2020. Treatment of end-of-life concrete in an innovative heating-air classification system for circular cement-based products. *Journal of Cleaner Production* 263.

Morison, R. and J. Hodges. 2018. Carbon reaches 10-year high, pushing up european power prices. *Bloomberg*. https://www.bloomberg.com/news/articles/2018-08-23/carbon-reaching-20-euros-a-ton-in-europe-raises-price-for-power.

Morrissey, J. and R.E. Horne. 2011. Life cycle cost implications of energy efficiency measures in new residential buildings. *Energy and Buildings* 43(4): 915–924. http://dx.doi.org/10.1016/j.enbuild.2010.12.013.

Mostert, C. and S. Bringezu. 2019. Measuring Product Material Footprint as New Life Cycle Impact Assessment Method: Indicators and Abiotic Characterization Factors. *Resources* 8(2): 61. https://www.mdpi.com/2079-9276/8/2/61.

Mulders, L. 2013. High quality recycling of construction and demolition waste in the Netherlands. Utrencht University.

Müller, D. 2006. Stock dynamics for forecasting material flows—Case study for housing in The Netherlands. *Ecological Economics* 59(1): 142–156. http://www.sciencedirect.com/science/article/pii/S092180090500460X.

Nakamura, S. and G. Rebitzer. 2008. Environmental Life Cycle Costing. In *Environmental Life Cycle Costing*, 35–57. CRC Press, May 29. http://www.crcnetbase.com/doi/10.1201/9781420054736.ch3.

Nederlandsche Bank, De. 2020. 10-year interest rates. https://statistiek.dnb.nl/dashboards/rente/index.aspx. Accessed December 17, 2020.

Netherlands Enterprise Agency. 2021a. Energy labels. https://business.gov.nl/regulation/energy-labels/.

Netherlands Enterprise Agency. 2021b. Standard and target values for home insulation. https://www.rvo.nl/onderwerpen/duurzaam-ondernemen/gebouwen/wetten-en-regels/standaard-en-streefwaarden-voor-woningisolatie.

Ning, Z. 2012. Thermal Treatment of Recycled Concrete Fines. Eindhoven University of Technology.

Nocker, L. De and W. Debacker. 2017. *Annex: Monetisation of the MMG method (update 2017)*. Mechelen. https://www.ovam.be/sites/default/files/atoms/files/Monetisation of the MMG method - Anex - update 2017.pdf.

Noll, D., D. Wiedenhofer, A. Miatto, and S.J. Singh. 2019. The expansion of the built environment, waste generation and EU recycling targets on Samothraki, Greece: An island's dilemma. *Resources, Conservation and Recycling* 150: 104405. https://doi.org/10.1016/j.resconrec.2019.104405.

Nørup, N., K. Pihl, A. Damgaard, and C. Scheutz. 2019. Evaluation of a European textile sorting centre: Material flow analysis and life cycle inventory. *Resources, Conservation and Recycling* 143(September 2018): 310–319. https://doi.org/10.1016/j.resconrec.2019.01.010.

Novick, D. 1959. *The federal budget as an indicator of government intentions*. Santa Monica. https://www.rand.org/content/dam/rand/pubs/papers/2008/P1803.pdf.

O'Sullivan, D.T.J., M.M. Keane, D. Kelliher, and R.J. Hitchcock. 2004. Improving building operation by tracking performance metrics throughout the building lifecycle (BLC). *Energy and Buildings*.

OECD. 2008. *Measuring Material Flows and Resource Productivity-Volume I. The OECD Guide*. Vol. Volume 1. http://www.oecd.org/dataoecd/46/48/40485853.pdf%5Cnpapers2://publication/uuid/686D0BB7-44D3-454F-8D23-02FE7A24E1E8.

Ottelé, M., K. Perini, A.L.A. Fraaij, E.M. Haas, and R. Raiteri. 2011. Comparative life cycle analysis for green façades and living wall systems. *Energy and Buildings*.

Papaefthimiou, S., E. Syrrakou, and P. Yianoulis. 2006. Energy performance assessment of an electrochromic window. *Thin Solid Films* 502(1–2): 257–264.

Pauliuk, S. and N. Heeren. 2020. ODYM—An open software framework for studying dynamic material systems: Principles, implementation, and data structures. *Journal of Industrial Ecology* 24(3): 446–458.

Pavel, C.C. and D.T. Blagoeva. 2018. *Competitive landscape of the EU's insulation materials industry for energy-efficient buildings*. Luxembourg. https://publications.jrc.ec.europa.eu/repository/bitstream/JRC108692/kj1a28816enn.pdf.

Pavlu, T., V. Kočí, and P. Hájek. 2019. Environmental assessment of two use cycles of recycled aggregate concrete. *Sustainability (Switzerland)* 11(21).

PBL. 2018. Energy consumption of private households. https://www.cbs.nl/nl-nl/achtergrond/2018/14/energieverbruik-van-particuliere-huishoudens. Accessed February 2, 2021.

PBL, CBS, TNO, and RIVM. 2020. *Netherlands Climate and Energy Outlook 2020 Summary*. Hague. https://www.pbl.nl/sites/default/files/downloads/pbl-2020-netherlands-climate-and-energy-outlook-2020-summary-4299.pdf.

Pedinotti-Castelle, M., M.F. Astudillo, P.-O. Pineau, and B. Amor. 2019. Is the environmental opportunity of retrofitting the residential sector worth the life cycle cost? A consequential assessment of a typical house in Quebec. *Renewable and Sustainable Energy Reviews* 101: 428–439. https://doi.org/10.1016/j.rser.2018.11.021.

Peng, J., L. Lu, and H. Yang. 2013. Review on life cycle assessment of energy payback and greenhouse gas emission of solar photovoltaic systems. *Renewable and Sustainable Energy Reviews* 19: 255–274. http://dx.doi.org/10.1016/j.rser.2012.11.035.

Pfaff, M., S. Glöser-Chahoud, L. Chrubasik, and R. Walz. 2018. Resource efficiency in the German copper cycle: Analysis of stock and flow dynamics resulting from different efficiency measures. *Resources, Conservation and Recycling* 139: 205–218.

Portney, P.R. and J.P. Weyant. 2013. *Discounting and Intergenerational Equity*. 1st Editio. New York: Routledge, October. https://www.taylorfrancis.com/books/9781135891947.

Potting, J., A. Hanemaaijer, R. Delhaye, J. Ganzevles, R. Hoekstra, and J. Lijzen. 2018. *Circular economy: what we want to know and can measure-framework and baseline assessment for monitoring the progress of the circular economy in the Netherlands*. *PBL Policy Report*. Vol. 3217. Hague.

PRC. 2011. *The Lead Market Initiative and Sustainable Construction: Lot 1, Screening of National Building Regulations (Final Report).*

Pretel, R., A. Robles, M. V. Ruano, A. Seco, and J. Ferrer. 2016. Economic and environmental sustainability of submerged anaerobic MBR-based (AnMBR-based) technology as compared to aerobic-based technologies for moderate-/high-loaded urban wastewater treatment. *Journal of Environmental Management* 166: 45–54. http://dx.doi.org/10.1016/j.jenvman.2015.10.004.

Purnell, P. and A. Dunster. 2010. Recycling of concrete. In *Management, Recycling and Reuse of Waste Composites*, 569–591. Elsevier, June 10. https://www.taylorfrancis.com/books/9780203927571.

Ramesh, T., R. Prakash, and K.K. Shukla. 2010. Life cycle energy analysis of buildings: An overview. *Energy and Buildings* 42(10): 1592–1600.

Rebitzer, G. and D. Hunkeler. 2003. Life cycle costing in LCM: Ambitions, opportunities, and limitations - Discussing a framework. In *International Journal of Life Cycle Assessment*.

Reports and Data. 2020. Lightweight Aggregate Concrete Market Analysis, By Material Type, By Air Dry Unit Weight, By Application, By Regional Outlook And Forecasts To 2026. https://www.reportsanddata.com/report-detail/lightweight-aggregate-concrete-market. Accessed April 10, 2020.

Rincón, L., A. Castell, G. Pérez, C. Solé, D. Boer, and L.F. Cabeza. 2013. Evaluation of the environmental impact of experimental buildings with different constructive systems using Material Flow Analysis and Life Cycle Assessment. *Applied Energy* 109: 544–552. http://dx.doi.org/10.1016/j.apenergy.2013.02.038.

Ristimäki, M., A. Säynäjoki, J. Heinonen, and S. Junnila. 2013. Combining life cycle costing and life cycle assessment for an analysis of a new residential district energy system design. *Energy*.

Rochat, D., C.R. Binder, J. Diaz, and O. Jolliet. 2013. Combining Material Flow Analysis, Life Cycle Assessment, and Multiattribute Utility Theory. *Journal of Industrial Ecology* 17(5): n/a-n/a. http://doi.wiley.com/10.1111/jiec.12025.

Rodr, G. and F.J. Alegre. 2007. The contribution of environmental management systems to the management of construction and demolition waste : The case of the Autonomous Community of Madrid (Spain). *Resources, Conservation and Recycling* 50(3): 334–349.

Rodrigues, C., R. Kirchain, F. Freire, and J. Gregory. 2018. Streamlined environmental and cost life-cycle approach for building thermal retrofits: A case of residential buildings in South European climates. *Journal of Cleaner Production*.

Ruukki. 2011. *Environmental product declaration and safety information sheet：hot-rolled steel plates, sheets and coils*. Helsinki, Finland. http://resource.jerei.com/11468/13102913452850_0.pdf.

Sadineni, S.B., S. Madala, and R.F. Boehm. 2011. Passive building energy savings: A review of building envelope components. *Renewable and Sustainable Energy Reviews* 15(8): 3617–3631. https://linkinghub.elsevier.com/retrieve/pii/S1364032111002504.

Sakai, S. ichi, H. Yoshida, Y. Hirai, M. Asari, H. Takigami, S. Takahashi, K. Tomoda, et al. 2011. International comparative study of 3R and waste management policy developments. *Journal of Material Cycles and Waste Management* 13(2): 86–102.

Saling, P., A. Kicherer, B. Dittrich-Krämer, R. Wittlinger, W. Zombik, I. Schmidt, W. Schrott, and S. Schmidt. 2002. Eco-efficiency analysis by basf: the method. *The International Journal of Life Cycle Assessment* 7(4): 203–218. http://link.springer.com/10.1007/BF02978875.

Sandberg, N.H., I. Sartori, O. Heidrich, R. Dawson, E. Dascalaki, S. Dimitriou, T. Vimm-r, et al. 2016. Dynamic building stock modelling: Application to 11 European countries to support the energy efficiency and retrofit ambitions of the EU. *Energy and Buildings* 132: 26–38. http://linkinghub.elsevier.com/retrieve/pii/S0378778816304893.

Sartori, I. and A.G. Hestnes. 2007. Energy use in the life cycle of conventional and low-energy buildings: A review article. *Energy and Buildings* 39(3): 249–257.

Säynäjoki, A., J. Heinonen, and S. Junnila. 2012. A scenario analysis of the life cycle greenhouse gas emissions of a new residential area. *Environmental Research Letters* 7(3).

Scharff, H. 2014. Landfill reduction experience in The Netherlands. *Waste Management* 34(11): 2218–2224. http://dx.doi.org/10.1016/j.wasman.2014.05.019.

Schiller, G., K. Gruhler, and R. Ortlepp. 2017a. Continuous Material Flow Analysis Approach for Bulk Nonmetallic Mineral Building Materials Applied to the German Building Sector. *Journal of Industrial Ecology* 21(3): 673–688.

Schiller, G., F. Müller, and R. Ortlepp. 2017b. Mapping the anthropogenic stock in Germany: Metabolic evidence for a circular economy. *Resources, Conservation and Recycling* 123: 93–107. http://dx.doi.org/10.1016/j.resconrec.2016.08.007.

Schwartz, Y., R. Raslan, and D. Mumovic. 2016. Implementing multi objective genetic algorithm for life cycle carbon footprint and life cycle cost minimisation: A building refurbishment case study. *Energy* 97: 58–68. http://dx.doi.org/10.1016/j.energy.2015.11.056.

Seppälä, J., M. Melanen, I. Mäenpää, S. Koskela, J. Tenhunen, and M.R. Hiltunen. 2005. How can the eco-efficiency of a region be measured and monitored? *Journal of Industrial Ecology* 9(4): 117–130.

SER. 2013. Energy agreement for sustainable growth. https://www.ser.nl/nl/thema/energie-en-duurzaamheid/energieakkoord. Accessed November 2, 2020.

Sesana, M.M. and G. Salvalai. 2013. Overview on life cycle methodologies and economic feasibility for nZEBs. *Building and Environment* 67: 211–216. http://dx.doi.org/10.1016/j.buildenv.2013.05.022.

Sevigné-Itoiz, E., C.M. Gasol, J. Rieradevall, and X. Gabarrell. 2014. Environmental consequences of recycling aluminum old scrap in a global market. *Resources, Conservation and Recycling* 89: 94–103. http://dx.doi.org/10.1016/j.resconrec.2014.05.002.

Sevigné-Itoiz, E., C.M. Gasol, J. Rieradevall, and X. Gabarrell. 2015. Contribution of plastic waste recovery to greenhouse gas (GHG) savings in Spain. *Waste Management* 46(2015): 557–567.

Shao, L., G.Q. Chen, Z.M. Chen, S. Guo, M.Y. Han, B. Zhang, T. Hayat, A. Alsaedi, and B. Ahmad. 2014. Systems accounting for energy consumption and carbon emission by building. *Communications in Nonlinear Science and Numerical Simulation* 19(6): 1859–1873. http://dx.doi.org/10.1016/j.cnsns.2013.10.003.

Sharma, A., A. Saxena, M. Sethi, V. Shree, and Varun. 2011. Life cycle assessment of buildings: A review. *Renewable and Sustainable Energy Reviews* 15(1): 871–875.

Singh, A., G. Berghorn, S. Joshi, and M. Syal. 2011. Review of Life-Cycle Assessment Applications in Building Construction. *Journal of Architectural Engineering* 17(1): 15–23. http://ascelibrary.org/doi/10.1061/%28ASCE%29AE.1943-5568.0000026.

Song, M. and S. Wang. 2018. Market competition, green technology progress and comparative advantages in China. *Management Decision* 56(1): 188–203.

Stahel, W.R. 2016. The circular economy. *Nature* 531(7595): 435–438.

Staniaszek, D. 2014. *Renovation Strategies of Selected EU Countries*. http://bpie.eu/wp-content/uploads/2015/10/Renovation-Strategies-EU-BPIE-2014.pdf. Accessed March 3, 2020.

Staniaszek, D. 2015. *Renovation in practice: best practice examples of voluntary and mandatory initiatives across Europe*. Brussels. http://bpie.eu/wp-content/uploads/2015/12/Renovation-in-practice_08.pdf. Accessed March 26, 2020.

Statistics Netherlands. 2020. Voorraad woningen en niet-woningen; mutaties, gebruiksfunctie, regio. https://opendata.cbs.nl/statline/#/CBS/nl/dataset/81955NED/table?ts=1603383709104. Accessed November 2, 2020.

Sterner, E. 2000. Life-cycle costing and its use in the Swedish building sector. *Building Research and Information*.

Suzuki, S., F. Part, Y. Matsufuji, and M. Huber-Humer. 2018. Modeling the fate and end-of-life phase of engineered nanomaterials in the Japanese construction sector. *Waste Management* 72: 389–398. https://doi.org/10.1016/j.wasman.2017.11.037.

Swarr, T.E., D. Hunkeler, W. Klöpffer, H.L. Pesonen, A. Ciroth, A.C. Brent, and R. Pagan. 2011. *Environmental life-cycle costing: A code of practice*. *SETAC Press*.

TABULA. 2017. National building typologies-TABULA WebTool. http://webtool.building-typology.eu/#bm. Accessed September 22, 2020.

Tajbakhsh, A. and E. Hassini. 2015. A data envelopment analysis approach to evaluate sustainability in supply chain networks. *Journal of Cleaner Production* 105: 74–85. https://linkinghub.elsevier.com/retrieve/pii/S0959652614007732.

Tam, C.M., V.W.Y. Tam, J.K.W. Chan, and W.C.Y. Ng. 2005. Use of Prefabrication to Minimize Construction Waste - A Case Study Approach. *International Journal of Construction Management* 5(1): 91–101.

Tam, V.W.Y. 2009. Comparing the implementation of concrete recycling in the Australian and Japanese construction industries. *Journal of Cleaner Production* 17(7): 688–702. http://dx.doi.org/10.1016/j.jclepro.2008.11.015.

Tam, V.W.Y., C.M. Tam, J.K.W. Chan, and W.C.Y. Ng. 2006. Cutting Construction Wastes by Prefabrication. *International Journal of Construction Management* 6(1): 15–25.

Tam, V.W.Y., L. Tam, and K.N. Le. 2010. Cross-cultural comparison of concrete recycling decision-making and implementation in construction industry. *Waste Management* 30(2): 291–297.

Tangtinthai, N., O. Heidrich, and D.A.C. Manning. 2019. Role of policy in managing mined resources for construction in Europe and emerging economies. *Journal of Environmental Management* 236: 613–621. https://doi.org/10.1016/j.jenvman.2018.11.141.

Technalia. 2018. *VEEP project report-WP2: new concrete formulations containing high share of Construction and demolition waste recycled material*.

Thienel, K.-C., T. Haller, and N. Beuntner. 2020. Lightweight Concrete—From Basics to Innovations. *Materials* 13(5): 1120. https://www.mdpi.com/1996-1944/13/5/1120.

Thorpe, W. 2008. *Refurbishment Waste Benchmarking Report*. http://www.wrap.org.uk/sites/files/wrap/Refurbishment-waste-benchmarking-report.pdf.

Tol, R.S.J. 2008. The Social Cost of Carbon: Trends, Outliers and Catastrophes. *Economics: The Open-Access, Open-Assessment E-Journal*.

UN Comtrade database. 2020. Commodities import and export in the Netherlands. https://comtrade.un.org/data/. Accessed April 12, 2020.

Utne, I.B. 2009. Life cycle cost (LCC) as a tool for improving sustainability in the Norwegian fishing fleet. *Journal of Cleaner Production* 17(3): 335–344. http://dx.doi.org/10.1016/j.jclepro.2008.08.009.

Vegas, I., K. Broos, P. Nielsen, O. Lambertz, and A. Lisbona. 2015. Upgrading the quality of mixed recycled aggregates from construction and demolition waste by using near-infrared sorting technology. *Construction and Building Materials* 75: 121–128.

Vercalsteren, A., C. Spirinckx, and T. Geerken. 2010. Life cycle assessment and eco-efficiency analysis of drinking cups used at public events. *International Journal of Life Cycle Assessment* 15(2): 221–230.

Vilches, A., A. Garcia-Martinez, and B. Sanchez-Montañes. 2017. Life cycle assessment (LCA) of building refurbishment: A literature review. *Energy and Buildings* 135: 286–301.

Villoria Sáez, P., J. Santa Cruz Astorqui, M. del Río Merino, M. del P. Mercader Moyano, and A. Rodríguez Sánchez. 2018. Estimation of construction and demolition waste in building energy efficiency retrofitting works of the vertical envelope. *Journal of Cleaner Production* 172: 2978–2985. https://linkinghub.elsevier.com/retrieve/pii/S0959652617327865.

Voet, E. van der. 1996. Substances from cradle to grave: development of a methodology for the analysis of substance flows through the economy and the environment of a region. Leiden University.

Vries, P. De. 2010. *Prijsbewegingen op de koopwoningmarkt*.

Wäger, P.A., R. Hischier, and M. Eugster. 2011. Environmental impacts of the Swiss collection and recovery systems for Waste Electrical and Electronic Equipment (WEEE): A follow-up. *Science of the Total Environment* 409(10): 1746–1756. http://dx.doi.org/10.1016/j.scitotenv.2011.01.050.

Wan Omar, W.M.S., J.H. Doh, K. Panuwatwanich, and D. Miller. 2014. Assessment of the embodied carbon in precast concrete wall panels using a hybrid life cycle assessment approach in Malaysia. *Sustainable Cities and Society* 10: 101–111. http://dx.doi.org/10.1016/j.scs.2013.06.002.

Weil, M., U. Jeske, and L. Schebek. 2006. Closed-loop recycling of construction and demolition waste in Germany in view of stricter environmental threshold values. *Waste Management and Research* 24(3): 197–206.

Weingartner, H.M. 1969. Some New Views on the Payback Period and Capital Budgeting Decisions. *Management Science* 15(12): B-594-B-607. http://pubsonline.informs.org/doi/abs/10.1287/mnsc.15.12.B594.

Weitzman, M.L. 1998. Why the far-distant future should be discounted at its lowest possible rate. *Journal of Environmental Economics and Management* 36(3): 201–208.

Weitzman, M.L. 2011. Fat-tailed uncertainty in the economics of catastrophic climate change. *Review of Environmental Economics and Policy* 5(2): 275–292.

Wiedenhofer, D., T. Fishman, C. Lauk, W. Haas, and F. Krausmann. 2019. Integrating Material Stock Dynamics Into Economy-Wide Material Flow Accounting: Concepts, Modelling, and Global Application for 1900–2050. *Ecological Economics* 156: 121–133. https://doi.org/10.1016/j.ecolecon.2018.09.010.

Wiedenhofer, D., J.K. Steinberger, N. Eisenmenger, and W. Haas. 2015. Maintenance and Expansion: Modeling Material Stocks and Flows for Residential Buildings and Transportation Networks in the EU25. *Journal of Industrial Ecology* 19(4): 538–551.

Wikipedia. 2020a. Madrid. https://en.wikipedia.org/wiki/Madrid. Accessed December 9, 2020.

Wikipedia. 2020b. Amsterdam. https://en.wikipedia.org/wiki/Amsterdam. Accessed December 9, 2020.

Wikipedia. 2020c. Stockholm. https://en.wikipedia.org/wiki/Stockholm. Accessed December 9, 2020.

Wit, M. de, J. Verstraeten-Jochemsen, J. Hoogzaad, and B. Kubbinga. 2019. *The Circularity Gap Report: Closing the Circularity Gap in a 9% World*. https://bfc732f7-80e9-4ba1-b429-7f76cf51627b.filesusr.com/ugd/ad6e59_ba1e4d16c64f44fa94fbd8708eae8e34.pdf.

Woon, K.S. and I.M.C. Lo. 2016. An integrated life cycle costing and human health impact analysis of municipal solid waste management options in Hong Kong using modified eco-efficiency indicator. *Resources, Conservation and Recycling* 108: 104–114.

Wu, H., J. Zuo, H. Yuan, G. Zillante, and J. Wang. 2020. Cross-regional mobility of construction and demolition waste in Australia: An exploratory study. *Resources, Conservation and Recycling* 156: 104710. https://doi.org/10.1016/j.resconrec.2020.104710.

Xi, F., S.J. Davis, P. Ciais, D. Crawford-Brown, D. Guan, C. Pade, T. Shi, et al. 2016. Substantial global carbon uptake by cement carbonation. *Nature Geoscience* 9(12): 880–883.

Yang, X., M. Hu, N. Heeren, C. Zhang, T. Verhagen, A. Tukker, and B. Steubing. 2020. A combined GIS-archetype approach to model residential space heating energy: A case study for the Netherlands including validation. *Applied Energy* 280: 115953. https://linkinghub.elsevier.com/retrieve/pii/S0306261920314082.

Yang, Z., X. Zhou, and L. Xu. 2008. Eco-efficiency optimization for municipal solid waste management. *Journal of Cleaner Production* 28(3): 179. http://dx.doi.org/10.1016/j.jclepro.2014.09.091.

Yard, S. 2000. Developments of the payback method. *International Journal of Production Economics* 67(2): 155–167.

Yeheyis, M., K. Hewage, M.S. Alam, C. Eskicioglu, and R. Sadiq. 2013. An overview of construction and demolition waste management in Canada : a lifecycle analysis approach to sustainability. *Clean Technologies and Environmental Policy* 15(1): 81–91.

Yoon, P.H. and T.-M. Fang. 2009. Proton heating by parallel Alfvén wave cascade. *Physics of Plasmas* 16(6): 062314. http://aip.scitation.org/doi/10.1063/1.3159605.

Zakeri, B. and S. Syri. 2015. Electrical energy storage systems: A comparative life cycle cost analysis. *Renewable and Sustainable Energy Reviews*.

ZEBRA2020. 2020. Energy efficiency trends in buildings: Equivalent major renovation rate. http://www.zebra-monitoring.enerdata.eu/overall-building-activities/equivalent-major-renovation-rate.html#equivalent-major-renovation-rate.html. Accessed March 26, 2020.

Zhang, C., M. Hu, L. Dong, A. Gebremariam, B. Miranda-Xicotencatl, F. Di Maio, and A. Tukker. 2019a. Eco-efficiency assessment of technological innovations in high-grade concrete recycling. *Resources, Conservation and Recycling* 149: 649–663. https://linkinghub.elsevier.com/retrieve/pii/S0921344919302848.

Zhang, C., M. Hu, L. Dong, P. Xiang, Q. Zhang, J. Wu, B. Li, and S. Shi. 2018. Co-benefits of urban concrete recycling on the mitigation of greenhouse gas emissions and land use change: A case in Chongqing metropolis, China. *Journal of Cleaner Production* 201: 481–498. https://linkinghub.elsevier.com/retrieve/pii/S0959652618322273.

Zhang, C., M. Hu, B. Laclau, T. Garnesson, X. Yang, C. Li, and A. Tukker. 2021a. Environmental life cycle costing at the early stage for supporting cost optimization of precast concrete panel for energy renovation of existing buildings. *Journal of Building Engineering* 35: 102002. https://linkinghub.elsevier.com/retrieve/pii/S2352710220336342.

Zhang, C., M. Hu, B. Laclau, T. Garnesson, X. Yang, and A. Tukker. 2021b. Energy-carbon-investment payback analysis of prefabricated envelope-cladding system for building energy renovation: cases in Spain, the Netherlands, and Sweden. *Renewable and Sustainable Energy Reviews* 145: 111077.

Zhang, C., M. Hu, F. Di Maio, B. Sprecher, X. Yang, and A. Tukker. 2022. An overview of the waste hierarchy framework for analyzing the circularity in construction and demolition waste management in Europe. *Science of The Total Environment* 803: 149892. https://doi.org/10.1016/j.scitotenv.2021.149892.

Zhang, C., M. Hu, B. Sprecher, X. Yang, X. Zhong, C. Li, and A. Tukker. 2021c. Recycling potential in building energy renovation: A prospective study of the Dutch residential building stock up to 2050. *Journal of Cleaner Production* 301: 126835. https://linkinghub.elsevier.com/retrieve/pii/S0959652621010544.

Zhang, C., M. Hu, X. Yang, A. Amati, and A. Tukker. 2020a. Life cycle greenhouse gas emission and cost analysis of prefabricated concrete building façade elements. *Journal of Industrial Ecology* 24(5): 1016–1030.

Zhang, C., M. Hu, X. Yang, B. Miranda-Xicotencatl, B. Sprecher, F. Di Maio, X. Zhong, and A. Tukker. 2020b. Upgrading construction and demolition waste management from downcycling to recycling in the Netherlands. *Journal of Cleaner Production* 266: 121718. https://linkinghub.elsevier.com/retrieve/pii/S0959652620317650.

Zhang, J., L. Ding, F. Li, and J. Peng. 2020c. Recycled aggregates from construction and demolition wastes as alternative filling materials for highway subgrades in China. *Journal of Cleaner Production* 255.

Zhang, Q., B.L. Oo, and B.T.H. Lim. 2019b. Drivers, motivations, and barriers to the implementation of corporate social responsibility practices by construction enterprises： A review. *Journal of Cleaner Production* 210: 563–584. https://doi.org/10.1016/j.jclepro.2018.11.050.

Zhao, W., G. Huppes, and E. van der Voet. 2011. Eco-efficiency for greenhouse gas emissions mitigation of municipal solid waste management: A case study of Tianjin, China. *Waste Management* 31(6): 1407–1415. http://dx.doi.org/10.1016/j.wasman.2011.01.013.

Zuidema, M., R. Saitua, and N. Smit. 2016. *Macroeconomic exploration concrete agreement (in Dutch: Macro-economische verkenning betonakkoord)*. Netherlands: Amsterdam. https://www.eib.nl/pdf/Macro-economische verkenning betonakkoord.pdf.

Summary

Material circularity and energy efficiency are highly relevant and intertwined issues for the transition towards a carbon-neutral and circular built environment. In the Netherlands, the building sector has been rendered a priority towards a circular and low-carbon society. Regarding material circularity, the Netherlands has the best practice of construction and demolition waste (CDW) management over the world, achieving an almost 100% recovery rate. However, most CDW is downcycled for road base backfilling, which is considered as a low value-added treatment. Moreover, the Netherlands is already facing the problem of saturation of road construction. It is urgent to explore an outlet for the vast surplus of waste concrete to make the construction supply chain more circular. To improve the material efficiency within the construction sector, the Netherlands launched the "A circular economy in the Netherlands by 2050", aiming to achieve an interim target of 50% less use of primary raw materials by 2030 and a long-term goal of a fully circular economy by 2050. On the other hand, the housing stock in the Netherlands is poorly insulated and obsolete; about half of the building stock was constructed between the 1950s and 1970s before minimum energy performance requirements were introduced in 1995. The government of the Netherlands has set up ambitious energy renovation goals that aim to achieve the decarbonization goal by 2030 and 2050. The upcoming extensive renovation wave in the Netherlands not only in return increases the burden of resources but also raises new problems surrounding their disposal. This thesis explored potential solutions for these twin issues in light of a novel technological system. This system presents an energy–material efficiency solution for energy renovation of building stocks with prefabricated concrete elements (PCEs) with recycled CDW as feedstock. Life cycle assessment (LCA) and life cycle costing (LCC) were combined with dynamic material flow analysis (MFA) to estimate the economic and environmental implications at both a product level and a national level. Therefore, this thesis puts forward the following overarching research question: "*what are the potential impacts of the application of selected novel technological systems to enhance circular use of materials, energy efficiency and carbon neutrality in the residential sector of the Netherlands?*". Based on the main research question, five sub-questions were investigated in the thesis:

RQ1. Assessment of concrete recycling at the product (material) level

Is it possible to achieve environmental-economic win-win situation in high-grade concrete recycling? Would the innovations trigger any potential problem-shifts between different impact categories? **(Chapter 2)**

RQ2. Assessment of the PCE-new system for new building construction at a product (element) level

What are the environmental and economic implications of using the prefabricated element system (PCE-new) for cladding walls to improve building energy efficiency in new constructions in the Netherlands? **(Chapter 3)**

RQ3. Assessment of the PCE-refurb system for existing building renovation at a product (element) level

What are the environmental and economic implications of using the prefabricated element system (PCE-refurb) to over-clad existing buildings for energy refurbishment in the Netherlands? How is the applicability of the PCE-refurb under different climatic conditions? **(Chapter 4)**

RQ4. Assessment of turnover of the housing stock at the country level

How much CDW from the construction, demolition, and renovation of the Dutch housing stock will arise from 2015 to 2050? To which extent the CDW can be recycled as a feedstock in building energy renovation in the Netherlands? **(Chapter 5)**

RQ5. Assessment of implementing the recycling and prefabrication systems at the country level

What are the up-scaled environmental benefits and economic consequences of implementing the recycling and prefabrication systems in the Netherlands? To which extent can the proposed recycling and prefabrication system achieve the prospective circularity goal and decarbonization goal of the Netherlands? **(Chapter 6)**

To answer the first sub-question, Chapter 2 proposed a framework for LCA/LCC-type eco-efficiency assessment to investigate multiple technological systems for high-value-added recycling concrete and to identify key factors for cost-effective concrete recycling. This case study concluded that the most eco-efficient technological routes for recycling waste concrete are technologies that recycle the waste concrete on-site and produce high-value secondary products. We found the most environmentally sound and cost-saving way for concrete recycling is the integrated mobile Advanced dry recovery (ADR) and Heating air classification system (HAS), which had the highest eco-efficiency score. Regarding individual impact categories, 10 out of 15 environmental impact indicators show that the integrated ADR and HAS is superior to the other technological systems. At the same time, it has the highest environmental impact in photochemical ozone formation, particulate matter, acidification, terrestrial eutrophication, and marine eutrophication. These drawbacks are due to the fact that the pilot-scale HAS the pilot-scale HAS is still uses fossil fuel diesel to thermally decompose the concrete waste. An industrial-scale HAS will use cleaner energy such as renewable electricity.

Regarding the second sub-question, Chapter 3 used a monetary method to combine LCA and LCC to assess the life cycle performance of the green PCE-new as façade for a new building over 120 years in comparison to those of manufacturing and using a baseline PCE-new. The main difference is that the green PCE-new uses an aerogel that isolates better as expanded polystyrene (EPS) used in the baseline PCE-new. Moreover, the green PCE-new comprises recycled materials made from CDW while the baseline PCE only uses virgin materials. The production of the aerogel is much more energy-intensive as EPS, the difference over the life cycle is however limited. If we look only at direct costs, green PCE-new is slightly cheaper. If on top of this the external costs of greenhouse gas (GHG) emissions are included, avoided emissions lead to an additional cost advantage. Considerable uncertainties exist however in such calculations.

As of the third sub-question, Chapter 4 conducted a payback-based LCC and LCA to assess the life cycle performance of energy conservation, carbon mitigation, and cost reduction of the green PCE-refurb for refurbishing existing buildings in the Netherlands. We analyzed how reuse and recovery of the PCE-refurb influence the carbon-energy-investment payback periods. The results of the Dutch case indicate an apparent environment-economic trade-off. There appeared indeed to be a positive energy and carbon payback (within approximately 17 years). There was however not a good economic payback (more than 100 years). This indicates that manufacturing and using a PCE-refurb for energy renovation will reduce energy use and GHG emissions but it is costly. The PCE refurb can however be designed in a way that it can be reused, by applying easy-to-disassemble joints. Such design allows them to be reused again after one service life, which helps to overcome this cost disadvantage. It was found that using secondary materials in the PCE-refurb can just slightly reduce the three types of payback periods, while the reuse of the PCE-refurb can considerably shorten the three types of payback periods in three cases. To comprehensively examine the applicability of the PCE-refurb system, two additional cases were analysed for the Swedish and Spanish situations. Results show that implementing the green PCE-refurb in a colder zone would result in shorter payback periods for energy conservation, carbon mitigation, and costs.

With respect to the fourth sub-question, Chapter 5 applied dynamic MFA to explore the supply-demand balance of secondary raw materials made from CDW (including normal-weight and lightweight concrete, glass, insulation mineral wool, steel) and the secondary raw materials demanded for manufacturing PCEs in building energy renovation from 2015 to 2050. The main results show that with the advanced recycling system ADR and HAS, the secondary raw materials recovered from normal-weight concrete waste, glass waste, and insulation mineral wool waste will be more than sufficient to support the manufacturing of PCE-new walls. However, for emerging materials – such as lightweight concrete – the related waste will not be sufficient in a near future to meet the raw material demand for large scale refurbishment with PCE-refurb walls. The

Netherlands will further stay dependent on imports for certain building materials. Use of recycled materials will reduce such imports. In short, ADR and HAS support upcycling of CDW for use in PCEs, but primary raw materials are still needed for emerging materials such as lightweight concrete. Additionally, this case study also demonstrates that the projected amount of renovation waste exceeds the amount of construction waste. Under the circumstance of extensive building energy renovation in the EU, it is important to take into account the management of renovation waste.

For the last sub-question, Chapter 6 proposed an integrated framework that combines the dynamic MFA with LCA and LCC to explore the product-level and up-scaled performance of the PCE solutions for constructing new building construction and renovating existing buildings in the Netherlands from 2015 to 2050. The product-level and up-scaled results were expressed via an annual value approach and an actual-time value approach. The selection of different estimation approaches led to different conclusions. The actual-time value reveals the trade-offs of energy renovation between GHG emissions and material extraction reduction/cost saving. However, the annual value approach proves the comprehensive benefits of the PCE-refurb system compared with the business-as-usual (BAU) scenario in which no renovation (REN-scenario) or (accelerated) rebuilding (REB-scenario) takes place to realise targets with regard to energy efficiency. This study also evaluated the extent to which the circularity and decarbonization goals can be realized by up-scaling the proposed PCE system to the residential building sector in a scenario of renovating existing buildings (REN) and accelerated demolition and rebuilding (REB). The results show the BAU and REN scenarios cannot reduce resource uses till 2050. Accelerated demolition and rebuilding (the REB scenario) scores even worse on material use. Advanced technological systems in the REN scenario can however realise a shift from the current dominant route of downcycling of CDW to 98% recycling. Overall it hence does not seem possible to realise the Dutch circularity goals for 2030 and 2050 for the built environment in the BAU and REN scenarios. Furthermore, operational GHG emissions of the residential building stock in 2030 and 2050 were quantified under the three scenarios. We found that if not any energy renovation strategy is applied as in the baseline BAU scenario, the decarbonization goal by 2030 or 2050 cannot be realised. In the REN scenario, implementing the PCE system for extensive energy renovation in the Netherlands can achieve the interim goal of 49% carbon mitigation by 2030, but still cannot realize carbon-neutrality by 2050. The REB scenario can achieve this since it only builds only near zero energy houses, and replaces traditional gas-based systems with electricity-based systems. It does so however at tremendous costs and with very high material requirements.

Overall, Chapters 2 to 6 evaluated the novel PCE technological system from both economic and environmental perspectives and at both product level and country level.

From the **material circularity** perspective, we examined the priority of building lifetime extension, element reuse, and material recovery in the three-layer framework. We found that (i) building lifetime extension is the most desirable option for material circularity, ii) element reuse is less preferable but still show noticeably environmental and economic benefits, and iii) the benefits of material recovery are almost negligible compared with the first two options. From the **energy efficiency** and carbon-neutrality perspective, this thesis illustrated that passively improving the insulation level of the housing façade is necessary, while electrification and the use of renewable electricity is another key factor that contributes to realising a carbon-neutral built environment. The results provide insights into the opportunities and challenges of realising a circular and carbon-neutral built environment. The thesis also illustrates how LCA, LCC and MFA can be integrated for an assessment that scales up analyses at the product level to a national level. Further research could focus on exploring a broader scope of innovations for circularity and carbon-neutrality in the built environment in different regions, and further work on overcoming some remaining inconsistencies between LCA, LCC and MFA.

Samenvatting

Materiaalcirculariteit en energie-efficiëntie zijn zeer relevante en met elkaar verweven onderwerpen die van belang zijn voor de beoogde transitie naar een CO_2-neutrale en circulaire gebouwde omgeving in Nederland. De gebouwde omgeving is wegens zijn omvang heel belangrijk in de transitie naar een circulaire en koolstofarme samenleving. Wat betreft materiaalcirculariteit staat Nederland wereldwijd aan kop met het verwerken van bouw- en sloopafval (BSA), waarbij een percentage recycling van bijna 100% wordt gehaald. Het meeste BSA wordt echter gedowncycled als wegfundering, wat wordt beschouwd als een behandeling met een lage toegevoegde waarde. Bovendien kampt Nederland nu al met het probleem dat de wegenbouw niet al het BSA niet meer kan opnemen. Er is dringend behoefte aan een manier om het enorme overschot aan afvalbeton hoogwaardig in te zetten en zo de bouwketen circulair te maken.

Om de materiaalefficiëntie binnen de bouwsector te verbeteren, lanceerde Nederland het ‘Een circulaire economie in Nederland in 2050’, met een tussentijdse doelstelling van 50% minder gebruik van primaire grondstoffen in 2030 en een lange termijn doel van een in 2050 volledig circulaire economie. Tegelijk is de woningvoorraad in Nederland relatief slecht geïsoleerd en verouderd; ongeveer de helft van de woningvoorraad is gebouwd tussen de jaren vijftig en zeventig voordat in 1995 een energieprestatienormering werd werden ingevoerd. Renovatie van dergelijke woningen in Nederland om het energieverbruik te verlagen vergt niet alleen het gebruik van grondstoffen, maar kan ook tot nieuwe problemen leiden in de afvalfase. Dit proefschrift onderzocht mogelijke oplossingen voor deze onderling verbonden problemen rond energieneutraliteit en grondstofefficiëntie. Het proefschrift kijkt naar nieuwe technologische systemen die deze twee problemen aanpakken. Het betref het gebruik van geprefabriceerde betonnen elementen met gerecycled BSA als grondstof. Levenscyclusanalyse (LCA) en levenscycluskostenberekening (Life cycle costing, ofwel LCC) werden gecombineerd met dynamische materiaalstroomanalyse (MFA) om de economische en ecologische implicaties op zowel productniveau als nationaal niveau in te schatten. Daarom stelt dit proefschrift de volgende overkoepelende onderzoeksvraag: "Wat zijn de potentiële effecten van de toepassing van geselecteerde nieuwe technologische systemen om circulair materiaalgebruik, energie-efficiëntie en CO_2-neutraliteit in de gebouwde omgeving in Nederland te verbeteren?". Op basis van de hoofdonderzoeksvraag zijn in het proefschrift vijf subvragen (SV) onderzocht:

SV1. Beoordeling van betonrecycling op product- en materiaal niveau

Is het mogelijk om bij hoogwaardige betonrecycling een milieu-economische win-winsituatie te realiseren? of kunnen leiden nieuwe technieken mogelijk tot probleemverschuivingen tussen verschillende impactcategorieën ? (Hoofdstuk 2)

SV2. Beoordeling van het Geprefabriceerde Element (Prefabricated Construction Element, PCE) systeem voor nieuwbouw op product(element)niveau

Wat zijn de ecologische en economische implicaties van het gebruik voor gevelbekleding voor nieuwe gebouwen van PCE (‘PCE-nieuw’) om de energie-efficiëntie Nederland te verbeteren? (Hoofdstuk 3)

SV3. Beoordeling van het PCE-systeem voor renovatie van bestaande gebouwen (‘PCE refurb’) op product(element)niveau

Wat zijn de ecologische en economische implicaties van het gebruik van het geprefabriceerde elementensysteem (PCE-refurb) om de gevels van bestaande gebouwen in Nederland opnieuw te bekleden zodat gebouwen energie-efficienter worden? Hoe is hangen de ecologisch en economische voor- en nadelen af van de klimatologische omstandigheden in verschillende landen? (Hoofdstuk 4)

SV4. Analyse van de ontwikkeling van de woningvoorraad op landniveau

Hoeveel BSA komt er uit de bouw, sloop en renovatie van de Nederlandse woningvoorraad van 2015 tot 2050? In hoeverre kan het BSA worden hergebruikt als grondstof in de renovatie voor energie-neutraliteit van gebouwen in Nederland? (Hoofdstuk 5)

SV5. Analyse van de implementatie van de recycling- en prefabricagesystemen op landniveau

Wat zijn de milieu-implicaties en economische gevolgen van de implementatie van de recycling- en prefabricagesystemen in Nederland? In hoeverre kan met het voorgestelde recycling- en prefabricagesysteem Nederland de beoogde circulariteits- en CO_2-neutraliteitsdoelen bereiken? (Hoofdstuk 6)

Om de eerste subvraag te beantwoorden, beschrijft Hoofdstuk 2 een aanpak voor de analyse van eco-efficiëntie, gebaseerd op LCA en LCC. Dit raamwerk wordt gebruikt diverse technologische systemen voor hoogwaardig betonrecycling te onderzoeken en om sleutelfactoren te identificeren voor hun kosten- en milieueffectiviteit. Deze casestudy concludeert dat de meest eco-efficiënte technologische routes voor het recyclen van afvalbeton technologieën zijn die het afvalbeton ter plaatse recyclen en hoogwaardige secundaire producten produceren. We ontdekten dat de meest milieuvriendelijke en kostenbesparende manier voor betonrecycling de geïntegreerde mobiele ADR en HAS is. Wat de individuele effectcategorieën betreft, blijkt uit 10 van de 15 milieueffectindicatoren dat het geïntegreerde Advanced dry recovery (ADR) en Heating air classification system (HAS) superieur is aan andere onderzochte opties. Tegelijkertijd scoort dit systeem slechter op het gebied van fotochemische ozonvorming, fijn stof, verzuring, terrestrische eutrofiëring en mariene eutrofiëring. Deze nadelen zijn te wijten aan het feit dat de HAS op pilootschaal nog steeds fossiele brandstof (diesel) gebruikt om het afvalbeton thermisch af te breken. Een HAS op industriële schaal zal schonere energiebronnen gebruiken, zoals hernieuwbare elektriciteit.

Met betrekking tot de tweede subvraag, gebruikte hoofdstuk 3 een monetaire methode om LCA en LCC te combineren om de te analyseren hoe een groene PCE-nieuw als gevel voor een nieuw gebouw over 120 jaar scoort in vergelijking met die van productie en het gebruik van een baseline PCE -nieuw. Het belangrijkste verschil is dat een groene PCE nieuw een aerogel gebruikt die beter isoleert als geëxpandeerd polystyreen (EPS) dat in gewone PCE nieuw wordt gebruikt. Verder gebruikt de groene PCE nieuw gerecyled BSA terwijl gewone PCE primaire materialen gebruikt. Omdat de productie van aerogel echter veel energie-intensiever is dan van EPS, is het verschil over de levenscyclus echter beperkt. Als alleen naar de directe kosten wordt gekeken valt groene PCE-nieuw iets goedkoper uit. Als ook de externe kosten van de uitstoot van broeikasgassen worden meegerekend, leidt de vermeden uitstoot tot een extra kostenvoordeel. Er moet echter rekening worden gehouden met een grote onzekerheid in de kostenraming.

Voor wat betreft de derde subvraag voerde hoofdstuk 4 een analyse uit om terugverdientijden te berekenen voor CO_2 emissies, energieverbruik, en kosten van renovatie van woningen in Nederland met PCE-refurb. De analyse werd uitgevoerd met behulp van LCC en LCA. Verder is geanalyseerd hoe product- en materiaalhergebruik van PCE-refurb de terugverdientijden beïnvloeden. Er bleek sprake te zijn van een positieve terugverdientijd van energieverbruik en CO_2 emissies (ongeveer 17 jaar). De terugverdientijd van de investering bleek heel lang (meer dan 100 jaar). Dit geeft aan dat de productie en het gebruik van een PCE in renovatie het energieverbruik en de uitstoot van broeikasgassen zal verminderen, maar dat de kosten relatief hoog zijn. De PCE-refurb kan echter herbruikbaar worden ontworpen door eenvoudig te demonteren verbindingen toe te passen. Door een dergelijk ontwerp kunnen ze na één levensduur opnieuw worden gebruikt, wat helpt om deze kostenkloof te overbruggen. Dit producthergebruik blijkt veel zinvoller dan materiaalhergebruik. Het gebruik van secundaire materialen in de PCE-refurb verkort de terugverdientijd voor energie, CO_2 emissies en kosten maar beperkt. Naast een analyse voor de Nederlandse context is ook nog een analyse voor de Zweedse en Spaanse situatie uitgevoerd. Daaruit blijkt dat het toepassen van PCE-refurb in koude landen resulteert in kortere terugverdientijden voor energiegebruik, CO_2-emissies en kosten.

Met betrekking tot de vierde subvraag, heeft hoofdstuk 5 een dynamische MFA toegepast om de balans in vraag en aanbod te onderzoeken van secundaire grondstoffen die vrijkomen uit de gebouwde omgeving (o.a. normaal gewicht en lichtgewicht beton, glas, minerale isolatiewol, staal) en de behoefte aan secundaire materialen die nodig zijn voor de productie van PCE's voor renovatie van gebouwen voor het verhogen van energie-efficiëntie van 2015 tot 2050 in Nederland. De belangrijkste resultaten laten zien dat met het geavanceerde recyclingsysteem ADR en HAS de secundaire grondstoffen die worden teruggewonnen uit normaal gewicht betonafval, glasafval en afval van minerale isolatiewol ruim voldoende om de fabricage van PCE-nieuw te ondersteunen. Voor nieuwe typen van materialen – lichtgewicht beton – zal het in de nabije toekomst de hoeveelheid die gewonnen kan worden uit BSA nog niet voldoende zijn om te voldoen aan de grondstofvraag voor grootschalige renovatie met PCE-refurb-wanden. Nederland blijft verder nog afhankelijk voor import van diverse bouwmaterialen. Gebruik van gerecycleerde materialen beperkt die import wel. Kort gezegd helpen ADR en HAS bij een hoogwaardige verwerking van BSA in Nederland in PCE's, maar zijn er nog steeds primaire grondstoffen nodig voor gebruik in nieuwe, opkomende materialen, zoals b.v. lichtgewicht beton. Daarnaast toont deze case studie ook aan dat de verwachte hoeveelheid renovatieafval groter is dan de hoeveelheid afval die ontstaat bij nieuwbouw. Gezien de grootschalige opgave ten aanzien van energierenovatie van gebouwen in de EU, is het belangrijk om rekening te houden met het beheer van dit renovatieafval.

Voor de laatste subvraag combineert Hoofdstuk 6 de dynamische MFA met LCA en LCC. Hiermee kunnen we de prestaties van de PCE oplossingen onderzoeken op productniveau, maar ook opgeschaald naar landelijk niveau, bij toepassing in nieuwbouw en renovatie in Nederland van 2015 tot 2050. De resultaten op productniveau en opgeschaald zijn uitgedrukt via een jaarlijkse waardebenadering en een actuele waardebenadering. De actuele waarde laat per jaar zien hoeveel uitgaven, koolstofemissies en materiaalextractie in dat jaar plaats vindt als gevolg van renovatie in dat jaar en (verminderd) energiegebruik door renovatie in voorgaande jaren. De jaarlijkse waardebenadering kijkt naar de totale kosten, materiaalbehoefte en koolstofemissies over de levensduur en berekent vervolgens een gemiddelde per jaar. Deze aanpak laat zien dat het PCE-refurb systeem duidelijke voordelen heeft boven het business-as-usual (BAU) scenario, waarin geen renovatie (REN-scenario) of versnelde nieuwbouw (Rebuild of REB-scenario) plaatsvindt om doelen ten aanzien van energie-efficiëntie te halen. Deze studie evalueerde ook de mate waarin de circulariteits- en decarbonisatiedoelen kunnen worden gerealiseerd door implementatie van PCE-refurb. De resultaten laten zien dat de BAU- en REN-scenario's nauwelijks leiden tot een vermindering van extractie van grondstoffen tot 2050. Versnelde afbraak en nieuwbouw (het REB scenario) scoort echter nog veel slechter qua materiaalgebruik. Geavanceerde technologische systemen in het REN-scenario kunnen echter bewerkstelligen dat CDW voor 98% gaat worden gerecycled, in plaats van zoals nu nog grotendeels gedowncycled als wegfundering. Kort gezegd lijkt er geen mogelijkheid om voor de gebouwde omgeving, noch in het BAU als REN scenario met PCE's, de circulariteitsdoelstellingen in 2030 en 2050 te halen. Hiernaast zijn de operationele kooflstofemissies van de woningvoorraad in 2030 en 2050 gekwantificeerd onder de drie verschillende scenario's. Als er zoals in het BAU scenario geen energierenovatie strategie wordt toegepast, blijken de decarbonisatiedoelstellingen voor 2030 of 2050 onhaalbaar zijn. In het renovatie (REN) scenario kan met de implementatie van PCE-systemen het tussendoel van 49 % CO_2-reductie in 2030 worden gehaald, maar niet het doel om in 2050 een energie-neutrale gebouwde omgeving te realiseren. Het versnelde herbouw (REB)-scenario kan het lange termijn doel echter wel bereiken, doordat alle nieuwbouw energie-neutraal is en verder het gebruik van gas wordt vervangen door (groene) elektriciteit. Het REB scenario brengt echter enorme kosten en materiaalbehoeftes met zich mee

Samenvattend evalueerden de hoofdstukken 2 tot en met 6 het nieuwe PCE-systeem vanuit zowel economische als milieuperspectief en zowel op productniveau als op landniveau. Bezien vanuit het perspectief van materiaalcirculariteit onderzochten we de rol van verlenging van de levensduur van gebouwen, hergebruik van elementen en materiaalterugwinning. We vonden dat (i) het verlengen van de levensduur van gebouwen de meest wenselijke optie is vanuit het perspectief van circulariteit, (ii) hergebruik van elementen minder de voorkeur heeft, maar nog steeds aanzienlijke ecologische en economische voordelen oplevert, en (iii) de voordelen van materiaalterugwinning bijna te verwaarlozen zijn in vergelijking met de eerste twee opties. Bezien vanuit het perspectief van energie-efficiëntie illustreerde dit proefschrift verbetering van het isolatieniveau van de gevel van de woning noodzakelijk is, terwijl elektrificatie en het gebruik van hernieuwbare elektriciteit een andere cruciale factor is voor het realiseren van een koolstofneutrale gebouwde omgeving. De resultaten dragen bij aan het inzichtelijk maken van de kansen en uitdagingen om een circulaire en CO_2-neutrale gebouwde omgeving ter realiseren. Het proefschrift geeft verder aan hoe LCA en LCC en MFA kunnen worden geïntegreerd tot een analyse waarin beoordelingen op productniveau kunnen worden opgeschaald tot beoordelingen op landelijk niveau. Verder onderzoek zou zich kunnen richten op het analyseren van een bredere set van innovaties voor circulariteit en koolstofneutraliteit in de gebouwde omgeving in verschillende regio’s, en verder werk ten aanzien van het wegnemen van een aantal resterende inconsistenties tussen LCA, LCA en MFA.

List of publications

1. **Zhang, C.**, Hu, M.*, Dong, L., Xiang, P., Zhang, Q., Wu, J., Li, B., Shi, S. Co-benefits of urban concrete recycling on mitigation of greenhouse gas emissions and land use change: A case in Chongqing metropolis, China. Journal of Cleaner Production, 2018, 201, 481–498.

2. **Zhang, C**., Hu, M.*, Dong, L.**, Gebremariam, A., Miranda-Xicotencatl, B., Di Maio, F., Tukker, A. Eco-efficiency assessment of technological innovations in high-grade concrete recycling. Resource, Recovery & Conservation, 2019, 149, 649–663.

3. **Zhang, C.**, Hu, M.*, Yang, X., Amati, A., Tukker, A. Life cycle greenhouse gas emission and cost analysis of prefabricated concrete building façade elements. Journal of Industrial Ecology, 2020, 24, 1016–1030.

4. **Zhang, C.**, Hu, M.*, Yang, X., Miranda-Xicotencatl, B., Sprecher, B., Di Maio, F., Zhong, X., Tukker, A. Upgrading construction and demolition waste management from downcycling to recycling in the Netherlands. Journal of Cleaner Production, 2020, 266, 121718.

5. **Zhang, C.**, Hu, M.*, Laclau, B., Garnesson, T., Yang, X., Li, C., Tukker, A. Environmental life cycle costing at the early stage for supporting cost optimization of precast concrete panel for existing building retrofit. Journal of Building Engineering, 2021, 35, 102002.

6. **Zhang, C.**, Hu, M.*, Sprecher, B., Yang, X., Zhong, X., Li, C., Tukker, A. Recycling potential in building energy renovation: a prospective study of the Dutch residential building stock till 2050. Journal of Cleaner Production, 2021, 301, 126835.

7. **Zhang, C.**, Hu, M.*, Laclau, B., Garnesson, T., Yang, X., Tukker, A. Investment-energy-carbon payback period analysis on prefabricated cladding system for building energy renovation: cross-state cases of Spain, the Netherlands, and Sweden. Renewable and Sustainable Energy Reviews, 2021, 145, 111077.

8. **Zhang, C.,** Hu, M.*, Di Maio, F., Sprecher, B., Yang, X., Tukker, A. An overview of waste hierarchy framework for analyzing the circularity of construction and demolition waste management in Europe. Science of the Total Environment, 2022,803,149892.

9. **Zhang, C.**, Hu, M.*, Sprecher, B., Yang, X., Verhagen, T., Tukker, A. Towards the 2050 circularity and decarbonization goals: Economic and environmental

implication of integrated material-energy efficiency renovation of residential building stock in the Netherlands. (In review)

10. Li, B., Xiang, P.*, Hu, M., **Zhang, C.**, Dong, L. The vulnerability of industrial symbiosis: a case study of Qijiang Industrial Park, China. Journal of Cleaner Production, 2017, 157, 267-277.

11. Yang, X.*, Hu, M., Heeren, N., **Zhang, C.**, Verhagen, T., Tukker, A. A combined GIS-archetype approach to model residential space heating energy: A case study for the Netherlands including validation. Applied Energy, 2020, 280, 115953.

12. Yang, X.*, Hu, M., Tukker, A., **Zhang, C.**, Huo, T., Steubing, B. A bottom-up dynamic building stock model for residential energy transition : A case study for the Netherlands. Applied Energy, 2022, 206, 118060.

13. Zhong, X.*, Hu, M., Deetman, S., Steubing, B., Lin, H., Aguilar-Hernandez, G., Harpprecht, C., **Zhang**, C., Tukker, A., Behrens, P. Global greenhouse gas emissions from housing materials and mitigation strategies to 2060. **Nature communications**, 2021, 12(1), 6126.

Conference/Forum

1. **Chunbo Zhang**, Hu Mingming. Study on the mechanism of public-private partnership for concrete recycling. Chinese Sand & Rock Aggregate Industry Technological Innovation Conference. Chongqing, 28-29, June, 2016. Oral presentation

2. **Chunbo Zhang**, Mingming Hu. Study on benefits of concrete recycling on carbon emissions and land use reduction, with case in Chongqing, China. ISIE-ISSST 2017: Science in Support of Sustainable and Resilient Communities. Chicago, 25-29, June, 2017. Oral presentation

3. **Chunbo Zhang**, Mingming Hu. Life cycle costing on emerging technology for high quality recycling of concrete in Europe: from lab-scale to industrial-scale. 2018 International Workshop on Environmental Management, Science and Engineering. Fujian, 16-17, June, 2018. Oral presentation

4. **Chunbo Zhang**, Mingming Hu. Life cycle costing of construction and demolition waste recycling for cost effective building retrofits -- A case study in the Netherlands. International Conference on Resource Sustainability. Beijing, 27-29, June, 2018. Oral presentation

5. **Chunbo Zhang**, Mingming Hu. Eco-efficiency analysis of technological innovations in high-grade concrete recycling in Europe. ISIE 2019 10th International Conference on Industrial Ecology, Beijing, 7-11, July, 2019. Oral presentation

6. **Chunbo Zhang**, Mingming Hu. Construction and demolition waste use and the situation in Asia: case of China. Virtual Conference: Promoting circular economy for buildings: the experience of VEEP, ICEBERG, SeRaMCo projects (organized by the EU Build Up), 9-10, March, 2021. Oral presentation

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Curriculum vitae

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