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).

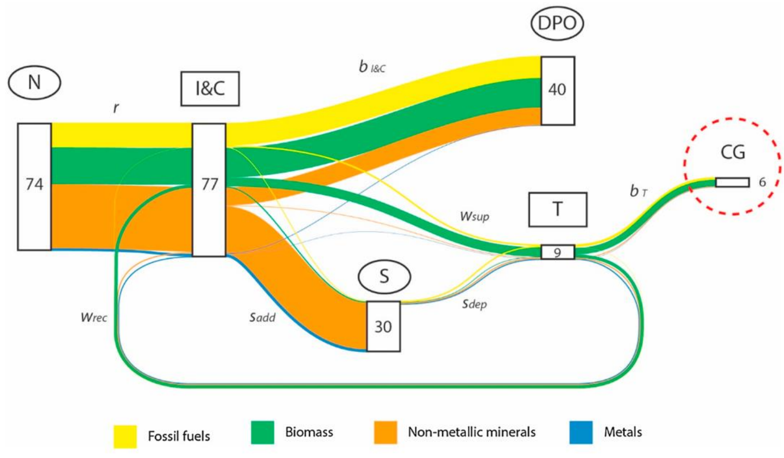

Figure 1.1 Sankey diagram of global material extraction, waste, and emission flows in 2011 (Aguilar-Hernandez et al. 2019). Numbers indicate the size of flows in Gt. Solid blocks indicate economic activities of: I&C=Intermediate sectors and final demand; T = waste treatment activities. Solid circles

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).

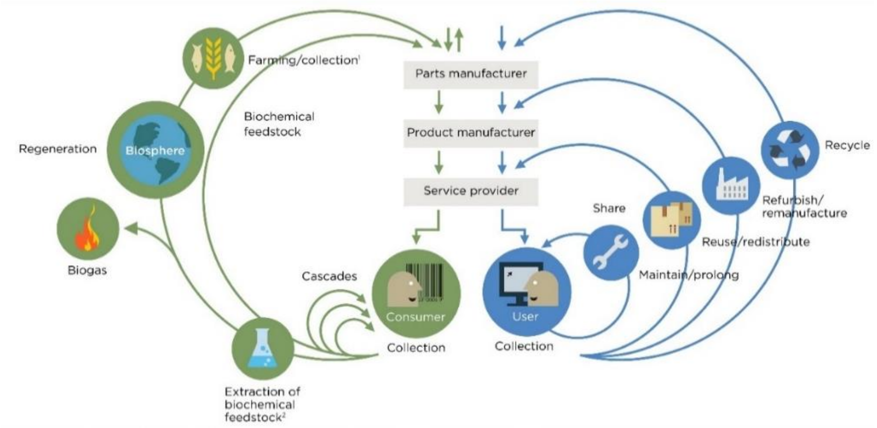

Figure 1.2 Outline of a circular economy framework by the Ellen MacArthur Foundation (2013)

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.

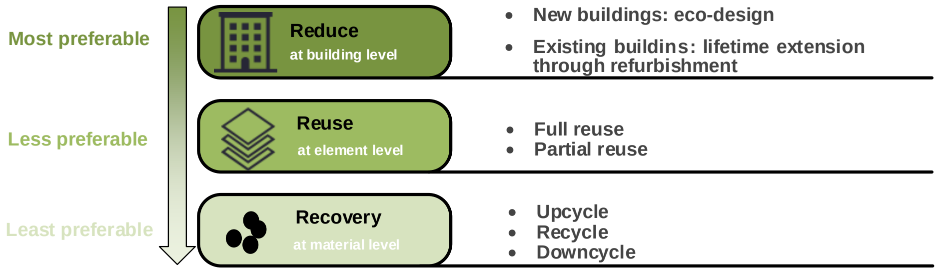

Figure 1.3 Building circularity framework based on 3R concept. 3R: reduce, reuse, and recycle. The circularity interventions are presumed based on (EC 2008a; Gharfalkar et al. 2015; Aguilar-Hernandez et al. 2018; Eberhardt et al. 2019).

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.

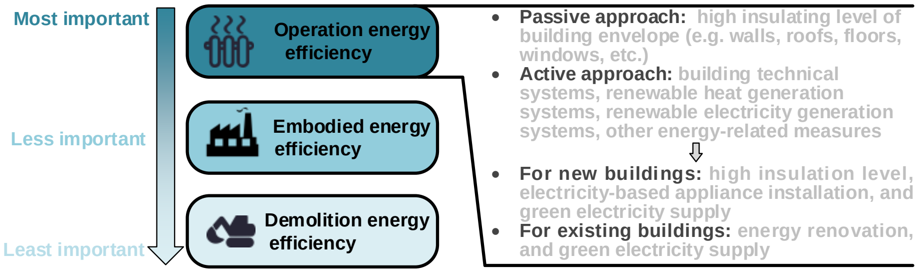

Figure 1.4 Building energy-efficiency framework from a life cycle perspective. Note: energy efficiency interventions are gathered from the European Energy Efficiency Platform (Economidou 2021).

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.

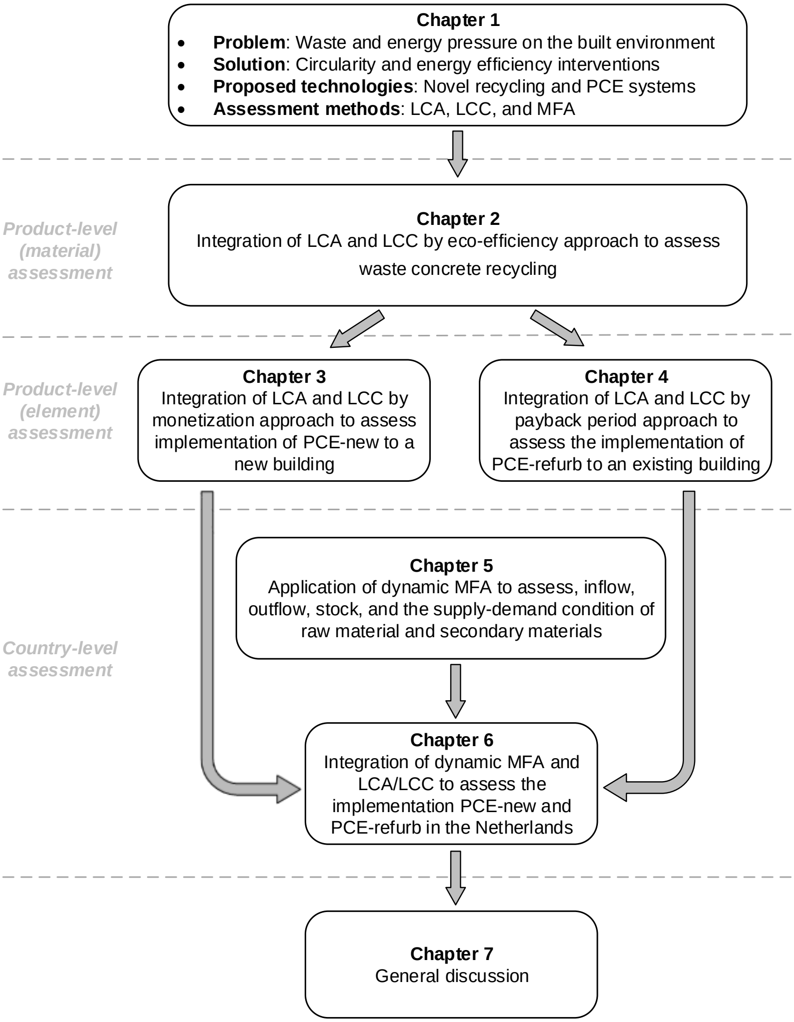

Figure 1.5 Conceptual scheme of the thesis and the topic of each chapter

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.