Life cycle assessment of material footprint in recycling
A case of concrete recycling

Zhang, Chunbo; Hu, Mingming; van der Meide, Marc; Di Maio, Francesco ; Yang, Xining; Gao, Xiaofeng ; Li, Kai; Zhao, Hailong; Li, Chen

DOI
10.1016/j.wasman.2022.10.035

Publication date
2023

Document Version
Final published version

Published in
Waste Management

Waste Management 155 (2023) 311–319

Contents lists available at ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Life cycle assessment of material footprint in recycling: A case of concrete recycling

Chunbo Zhang [a,b], Mingming Hu [a,*], Marc van der Meide [a], Francesco Di Maio [c], Xining Yang [a], Xiaofeng Gao [d], Kai Li [a], Hailong Zhao [e], Chen Li [a]

[a] Institute of Environmental Sciences, Leiden University, Leiden 2300RA, Netherlands
[b] College of Engineering, Cornell University, Ithaca, NY 14853, USA
[c] Faculty of Civil Engineering and Geosciences, Delft University of Technology, Delft 2628CN, Netherlands
[d] State Key Laboratory of the Three Gorges Reservoir Region's Eco-Environment, Ministry of Education, Chongqing University, Chongqing 400045, China
[e] State Key Joint Laboratory of Environment Simulation and Pollution Control (SKLESPC), School of Environment, Tsinghua University, Beijing 100084, China

ARTICLE INFO

Keywords:
Concrete
Recycling
Life cycle assessment
Material footprint
Renewable energy
Construction and demolition waste

ABSTRACT

Meeting the current demand for concrete requires not only mining tons of gravel and sand, but also burning large amounts of fossil fuel resources in cement kilning. Consequently, concrete recycling is crucial to achieving a material-efficient society, especially with the application of various categories of concrete and the goal of phasing out fossil fuels. A comparative life cycle assessment (LCA) is used to assess the engineering material footprint (EMF) and the fossil fuel material footprint (FMF) in closed-loop recycling of three types of concrete: siliceous concrete, limestone concrete, and lightweight aggregate concrete. This study aims to investigate the impact of (i) concrete categories, (ii) methods to model recycling, and (iii) using renewable energy sources on the material footprint in concrete recycling. The results highlight that the concrete recycling system can reduce 99% of the EMF and 66–93% of the FMF compared with the baseline system, in which concrete waste is landfilled. All three recycling modeling approaches indicate that concrete recycling can considerably reduce EMF and FMF compared with the baseline system, primarily resulting from the displacement of virgin raw materials. Using alternative diesels is more sensitive than adopting renewable electricity in reduction of the FMF in concrete recycling. Replacing diesel with electrolysis- and coal-based synthetic diesel for concrete recycling could even increase the FMF, while using biodiesel made from rapeseed and wood-based synthetic diesel can reduce 47–51% and 84–89% of the FMF, respectively, compared to the virgin diesel-based recycling system. Finally, we discussed the multifunctionality and rebound effects of recycling, and double-counting risk in material and energy accounting.

1. Introduction

As the skeleton of modern cities, concrete is the most consumed material worldwide (Miller et al., 2018). The global concrete consumption increased from around 900 Mt in 1950 to approximately 30 Gt in 2020 (Miller et al., 2018), leading to 4.3 Gt of cement and 19.4 Gt of aggregate requirements for the concrete industry (Monteiro et al., 2017). Cement, the binder of concrete aggregate, is the most energy-intensive material for concrete production (Habert et al., 2020). Cement manufacturing currently accounts for 3% of global energy consumption (Miller and Moore, 2020). It is forecast that the yearly cement production will rise by 50% by 2050, which results in 420–505 TJ of energy demand (Monteiro et al., 2017). Therefore, the surge in concrete consumption will exacerbate fossil fuel depletion and the resulted greenhouse gas emissions. With such a massive amount of raw material input, the output of concrete waste also cannot be neglected. Concrete waste accounts for 15–28% of total solid waste (Jin and Chen, 2019). This ratio estimates that global concrete waste will reach

Abbreviations: ADR, Advanced dry recovery; CED, Cumulative Energy Demand; CRA, Coarse recycled aggregate; EMF, Engineering material footprint; FMF, Fossil fuel material footprint; FRA, Fine recycled aggregate; HAS, Heat air classification system; LCA, Life cycle assessment; LCI, Life cycle inventory; RCP, Representative concentration pathway; SI, Supporting information; SSP, Shared socioeconomic pathway; URA, Ultrafine recycled aggregate; WFD, Waste Framework Directive.

* Corresponding author.
E-mail address: hu@cml.leidenuniv.nl (M. Hu).

https://doi.org/10.1016/j.wasman.2022.10.035
Received 8 March 2022; Received in revised form 13 October 2022; Accepted 26 October 2022
Available online 19 November 2022

383–715 Mt in 2025 (Hoornweg and Bhada-Tata, 2012). However, concrete is still being processed in an open-loop industrial operation. In Europe, most concrete waste is discarded through landfilling or downcycling as roadbase filler, which is regarded as a low material-efficient route (Zhang et al., 2020).

Circular economy has been proposed as a potential pathway to boosting prosperity while alleviating the dependence on primary materials extraction (Ellen MacArthur Foundation, 2015). As one of the three pillars in the 3R principle (Reduce-Reuse-Recycle) of a circular economy, recycling has been widely embraced for sustainable material management. Wet process is the most widespread technology to recycle concrete, regenerating secondary concrete aggregate by crushing, sieving and washing (Purnell and Dunster, 2010). However, this route is costly and generates by-products of sludge and fine sieve sand (Zhang et al., 2019). An advanced dry recovery (ADR) system was developed to process concrete waste under an anhydrate condition, yet, it still yields by-product sieve sand (Gebremariam et al., 2020). To completely close the concrete loop, the fine fraction is reprocessed by a thermal treatment process. A regular treatment process involves a rotary kiln at a temperature of approximately 700 °C, through which waste concrete can be almost completely recycled by separating 98 % of the hardened cement paste from sand and gravel grains (Mulder et al., 2007). An innovative thermal treatment technology heating air classification system (HAS) was developed to further process the by-product sieve sand. This integrated ADR-HAS system simultaneously implements a combination of mechanical and thermal processes to fully recycle waste concrete (Gebremariam et al., 2020). However, realizing a circular economy in concrete recycling by thermal treatment relies on the combustion of fossil fuels, which may conflict with a low-carbon economy aiming to decouple economic growth from fossil energy usage (Yuan et al., 2011). A low-carbon economy focuses on replacing fossil fuels to generate power and heat. In contrast, a circular economy concerns about Reducing, Reusing and Recycling engineering materials used to construct artificial components and structures. The primary function of engineering materials is to withstand applied loading without exhibiting excessive deflection and breaking. Recycling concrete could lead to a potential trade-off between retrieving engineering materials (e.g., gravel, sand, and cement) that would otherwise end as waste and consuming fossil fuel materials (e.g., diesel, and coal). Therefore, it is necessary to analyze both the engineering material footprint (EMF) and the fossil fuel material footprint (FMF) in concrete recycling.

A life cycle perspective is needed for analyzing concrete recycling to avoid strategies that reduce material consumption in one stage but may lead to more material use in other life stages (Huang et al., 2020). Applying a life cycle perspective can include the direct impacts of recovery and the hidden impacts embedded in upstream and downstream processes. Therefore, life cycle assessment (LCA) has been widely employed to evaluate the environmental impacts of waste management (Laurent et al., 2014). We reviewed those studies related to LCA of waste management, then selected and summarized some typical LCA studies related to concrete recycling, as shown in the supporting information (SI). Despite research efforts placed on evaluating the environmental impacts of concrete recycling, some knowledge gaps are still worth further discussion at least from the following aspects. First, previous studies just modeled CDW or concrete waste as feedstocks for recycling in general but did not specific the exact categories of concrete waste. In a comparative system, secondary products made from different types of concrete waste are related with various primary production systems. There are mainly 23 types of concrete that use different raw materials as aggregates and binders, as illustrated in the SI. Therefore, treating different types of concrete waste may lead to different costs and gains from a same recycling system. Then, recycling is a multifunctional process with dual benefits of waste treatment and secondary material production (Ackerman, 1997). The outcomes of an LCA may also vary regarding different modeling approaches for handling the multi-functionality of recycling. Previous studies used substitution (Waskow et al., 2021), allocation (Mostert et al., 2021), and system expansion (Moreno-Juez et al., 2020) to model the multifunctionality of recycling. However, it is not clear whether the selection of multifunctionality modeling approaches would lead to different conclusions. Moreover, concrete recycling possesses, especially thermal treatment, is energy intensive. Previous LCA studies did not consider whether the use of renewable energy, especially alternative diesels, could reduce the consumption of fossil fuels. Last but not least, previous studies looked into a wide range of impact categories, mostly global warming potential, and cumulative energy use; while analyses from a perspective of material use are limited. Mostert et al. (2021) accounted for the material footprint in concrete recycling by summing up both engineering and energy materials extractions in kg, and it is unclear to what extent concrete recycling can reduce engineering material consumption. Finally, applying impact assessment derived from the inventory analysis could lead to double counting with regard to fossil fuel-based resource depletion and other energy-related impact categories (Klöpffer, 1997). This indicates that if we weight material footprint and energy footprint into a single indicator, fossil fuel materials such as coal will count twice as material in kg and as energy in MJ.

Based on those research gaps, the main goal of this study is to unveil how concrete categories, recycling modelling methods, and the adoption of renewable energy can affect the EMF and the FMF in concrete recycling. The Ellen MacArthur Foundation (2015) defined two types of circular economies—biological cycles that focus on biodegradable materials and technical cycles that focus on non-biodegradable materials. This study focuses on the material footprint assessment of the technical cycle. A process-based LCA is conducted to evaluate the EMF and the FMF in concrete recycling. The proposed EMF could be a potential solution to avoid double counting concerning material and energy accounting schemes. Due to data availability, this study will not investigate a full range of concrete types but focuses on the three most used ones: normal-weight siliceous concrete (hereafter siliceous concrete), normal-weight limestone concrete (hereafter limestone concrete), and lightweight aggregate concrete (hereafter lightweight concrete). Standard ready-mix concrete is the most common form, so standard siliceous and limestone concrete are selected. Besides, to support the extensive building energy efficiency ambition in Europe, lightweight concrete is increasingly used owing to its low thermal conductivity (Zhang et al., 2021a,c). Therefore, lightweight concrete is also included in this study. In addition, three typical solutions to dealing with the multifunctionality of recycling in an LCA study are applied: allocation, substitution, and system expansion. Finally, the benefits of using alternative diesels and renewable electricity are identified.

The paper is structured as follows: we introduce the methods and materials in Section 2; the results are presented in Section 3; we further illustrate the results in Section 4; finally, we conclude our findings, shortcomings, and further directions in Section 5.

2. Methods and materials

2.1. Goal and scope definition

This LCA study aims to evaluate the material footprint in the recycling process of concrete. According to the Waste Framework Directive (WFD) of the European Union (EC, 2008), recycling can be understood as “an operation that feeds waste materials back into the economy for their original purpose and avoids waste being backfilled, landfilled, or incinerated”. In this study, we followed the definition of “recycling” in the WFD and defined concrete recycling as “an operation that feeds waste concrete back into the economy for its original purpose and avoids waste being backfilled and landfilled”. The geographical boundary for recycling operations was assumed to be the border of the Netherlands. Market prices of virgin and recycled materials, landfill gate fees, and recycling gate fees were collected based on the Dutch market for economic allocation. The conceptual diagram for this assessment is shown in Fig. 1. Three factors that may affect the material footprint of concrete recycling are examined: (i) categories of concrete waste, (ii) recycling modeling approaches, and (iii) the development of renewable energy.

The aforementioned integrated ADR-HAS system is the target concrete recycling technology to be assessed. This technological system mainly consists of three components: (i) a crushing set that encompasses a Keestrack Destroyer 1313 crusher, a CX350D excavator, and a 921E rubber-wheel loader; (ii) an ADR facility, (iii) and an HAS facility, as illustrated in Fig. 1a. Three fractions are yielded from the system, coarse recycled aggregate (CRA) (4–22 mm), fine recycled aggregate (FRA) (0.125–4 mm), and ultrafine recycled aggregate (URA) (0–0.125 mm). The transport and depreciation of the recycling facility are omitted. A reference baseline system is established, in which concrete waste is landfilled and virgin raw materials are produced, as shown in Fig. 1b.

Three types of concrete waste are considered as feedstock for the recycling system: siliceous concrete, limestone concrete, and lightweight concrete. The differences between these concrete wastes are explained in the SI. The functional unit of the assessment varies regarding the objective and feedstock of the recycling system, as illustrated in Table 1.

2.2. Life cycle inventory analysis

To make the recycling system and the baseline system comparable, it is imperative to render the target subsystems (as demonstrated in Table 1) with identical functions. Multiple potential solutions exist for coping with the multifunctionality problems of a unit process, such as subdivision, substitution, allocation, and system expansion (Heijungs et al., 2021). A subdivision approach is inoperable because the three unit processes in the product system (as shown in Fig. 1a) are supposed to be considered simultaneously to ensure closed-loop recycling. In this study, we examined how multifunctionality solutions of allocation, substitution, and system expansion affect the material use of concrete recycling. A 5% uncertainty was assumed for input data for inventory modeling (Huijbregts et al., 2003).

2.2.1. Multifunctionality modeling approaches

Allocation is a longstanding but important issue for life cycle inventory analysis (Heijungs and Frischknecht, 1998). Recycling has been considered to present clear allocation problems that need a separate treatment (Weidema, 2000). Diverse allocation methods exist, such as economic, mass-based, and energy-based allocations (Guinée et al., 2004). We examined both economic and mass-based allocations for recycling three types of waste concrete. Detailed information on each allocation method is shown in the SI. According to the Handbook on LCA (Guinée et al., 2001), economic allocation is advised as the baseline approach for most allocation situations in an LCA, as market prices determine the qualitative description of the degradation of a product (Werner and Richter, 2000). In this study, the market prices of emerging recycling materials (CRA, FRA, and URA) have not stabilized commercially. Therefore, their market prices were estimated through field investigation and literature research. Based on this price information, the process-based economic allocation was performed at a unit process level in the recycling system. The mass balance of recycling different concretes was measured based on experimental trials in Hoorn, the Netherlands. Regarding siliceous concrete waste, the shares of the consecutive recycled materials CRA, FRA, and URA by weight are 68.00%, 25.60%, and 6.40%, respectively. As for the limestone concrete waste, the shares are 45.00%, 43.90%, and 11.10%, respectively. The shares of recycled materials made from lightweight concrete are 52.00%, 38.40%, and 9.60%, respectively. The mass-based allocation was also conducted at a process level.

Substitution is not mentioned in ISO 14044 (2006b) but is generally recognized as a valid method for handling multifunctionality in an LCA (Brander and Wylie, 2011). Amendment 2 of the ISO 14044 (2020) added substitution as a means of system expansion. Therefore, interpretations of the term "system expansion" from ISO are twofold: (i) the system expansion method that includes all functions in a functional

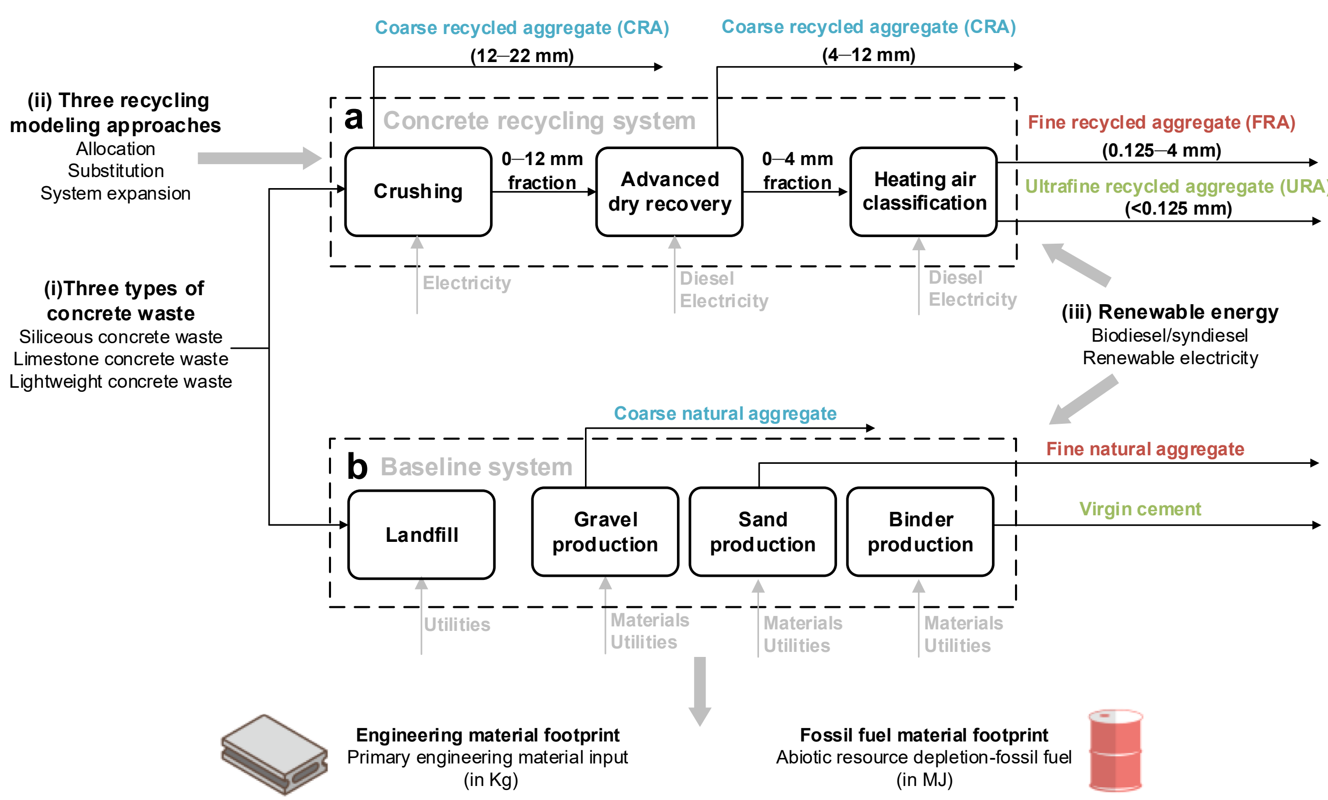

Fig. 1. Conceptual diagram of the (a) recycling system and the (b) baseline system and the three factors that may affect the material footprint in concrete recycling: (i) concrete waste categories, (ii) recycling modeling approaches, and (iii) use of renewable energy.

Table 1
Functional units for the recycling system and baseline system.

		Siliceous concrete waste	Limestone concrete waste	Lightweight concrete waste
Functional unit of waste treatment	Recycling system	Treating 1 ton of siliceous concrete waste	Treating 1 ton of limestone concrete waste	Treating 1 ton of lightweight concrete waste
	Baseline system	Landfilling 1 ton of siliceous concrete waste	Landfilling 1 ton of limestone concrete waste	Landfilling 1 ton of lightweight concrete waste
Functional unit of secondary co-production	Recycling system	Producing 0.680 tons of CRA, 0.250 tons of FRA, and 0.640 tons of URA from siliceous concrete waste	Producing 0.450 tons of CRA, 0.439 tons of FRA, and 0.111 tons of URA from limestone concrete waste	Producing 0.520 tons of CRA, 0.384 tons of FRA, and 0.960 tons of URA from lightweight concrete waste
	Baseline system	Producing 0.680 tons of siliceous gravel, 0.250 tons of siliceous sand, and 0.640 tons of cement (CEM III/A 42.5R)	Producing 0.450 tons of lime gravel, 0.439 tons of lime sand, and 0.111 tons of cement (CEM II/A-LL 42.5R)	Producing 0.520 tons of expanded clay, 0.384 ton of lime sand, and 0.960 ton of cement (CEM III/A 42.5 N/SRC)
Functional unit of recycling	Recycling system	Recycling 1 ton of siliceous concrete waste	Recycling 1 ton of limestone concrete waste	Recycling 1 ton of lightweight concrete waste
	Baseline system	Landfilling 1 ton of siliceous concrete waste; and producing 0.680 tons of siliceous gravel, 0.250 tons of siliceous sand, and 0.640 tons of cement (CEM III/A 42.5R)	Landfilling 1 ton of limestone concrete waste; and producing 0.450 tons of lime gravel, 0.439 tons of lime sand, and 0.111 tons of cement (CEM II/A-LL 42.5R)	Landfilling 1 ton of lightweight concrete waste; and producing 0.520 tons of expanded clay, 0.384 tons of lime sand, and 0.960 tons of cement (CEM III/A 42.5 N/SRC)

unit; (ii) the substitution method that subtracts avoided burdens. In this study, we regard substitution as the subtraction approach. When only considering the function of waste treatment, the impacts of corresponding primary material production are subtracted from the recycling system. On the other hand, if the system is intended for secondary material production only, the impacts of waste treatment are subtracted. Details of the substitution approach is illustrated in the SI.

System expansion is a preferable option for dealing with co-production systems, and allocation shall always be avoided by using system expansion (Weidema, 2000). When applying a system expansion approach, the concrete recycling system includes waste treatment and material production functions and will account for all materials and energy input. For the baseline system, impacts of landfilling and virgin raw material production are assessed accordingly. More information on the system expansion approach can be found in the SI.

2.2.2. Use of renewable energy in concrete recycling

The studied recycling system primarily uses diesel and electricity to process concrete waste and produce new concrete raw materials. The current recycling set of HAS is fueled by virgin diesel, which would be replaced by alternative diesels such as biodiesel and synthetic diesel (syndiesel hereafter) at an industrial scale conforming to the requirement of the clean energy transition. Biodiesel in this study is assumed to be produced from rapeseed, and syndiesel is manufactured through the Fischer-Tropsch process that converts carbon monoxide and hydrogen (Schulz, 1999). Three hydrogen sources are considered—electrolysis, wood gasification, and coal gasification. Moreover, electricity is supposed to be generated from more renewable sources such as wind, hydro, solar, and geothermal in future. The Netherlands Environmental Assessment Agency proposed an integrated assessment model IMAGE 3.2 to assess global environmental issues (PBL, 2021). Different shared socioeconomic pathways (SSPs) (O'Neill et al., 2014) were established in the IMAGE 3.2 framework to reveal possible future developmental trajectories (PBL, 2021). The intermediate-challenging scenario (SSP2) was selected under two representative concentration pathways (RCPs): SSP2-RCP6 and SSP2-RCP26. The more ambitious SSP2-RCP26 represents a higher level of renewable energy penetration in the local power grid compared with the baseline scenario SSP2-RCP6. We used the superstructure approach proposed by Steubing and de Koning (2021) to implement future production of electricity and the impacts it has on further production of virgin diesel, biodiesel, and syndiesel in inventorying modeling from 2020 to 2050. The advantage of using the superstructure approach is that a single life cycle inventory (LCI) database can be used to represent multiple background systems under different scenarios that change technological circumstances, market mixes, and temporal horizons, eliminating the need to reconnect a foreground system to multiple LCI databases. The market background data used for the superstructure approach to model the future electricity mix are derived from the premise 1.2.6 database (Sacchi et al., 2022). An advanced LCA software, Activity Browser 2.6.5 (Steubing et al., 2020), was employed to simulate the superstructure-based comparative LCA with the database ecoinvent 3.8 (cut-off) (ecoinvent, 2021).

2.3. Life cycle impact assessment

In this study, we used a relative indicator of Abiotic Resource Depletion-Fossil Fuel (expressed in MJ, or megajoule) by van Oers et al. (2002) to assess the FMF. An absolute indicator of Engineering Material footprint (expressed in Kg, or kilogram), adapted from the impact assessment method Raw Material Input (Mostert and Bringezu, 2019), was used to measure the EMF of the target concrete recycling system. The characterization factors of the EMF impact method are given in the SI.

3. Results

3.1. Engineering and fossil fuel material footprint of concrete recycling

The results of the EMF and the FMF of concrete recycling using different multifunctionality solutions are illustrated in this section. The allocation approach was used to attribute the EMF and the FMF of the waste treatment function and material production function, as shown in Fig. 2. The impacts of the waste treatment for each type of concrete are the same in both the recycling and the baseline systems (see Fig. 2a and Fig. 2b) because the waste status of each concrete waste ends after the preconditioning procedure in the recycling system (the crushing process in Fig. 1a) and the utilities for crushing different concrete are the same. As of the baseline system, the waste treatment process for three types of concrete is assumed through an identical landfill process. However, the EMF (0.02 Kg/t) and the FMF (2.60–3.40 Kg/t) of the recycling system are much lower compared to the baseline system (1.22 Kg/t and 37.73 MJ/t, respectively), via either an economic or a mass-based approach. Regarding the function of secondary material production, the difference is even more significant. The amount of the EMF and the FMF of the recycling systems ranges from 0.77 to 1.27 Kg/t and 106.49–179.41 MJ/t, respectively. While the EMF and the FMF of the baseline system are significantly higher, with a range of 1086.47–1109.26 Kg/t and 499.35–2358.93 MJ/t, respectively. Furthermore, for the baseline system, expanded clay production has much higher FMF than the production of lime/siliceous coarse aggregate. This indicates that substituting virgin expanded clay with the CRA made from lightweight concrete waste could lead to a more noticeable reduction of fossil fuel consumption.

The substitution approach was then employed to identify the

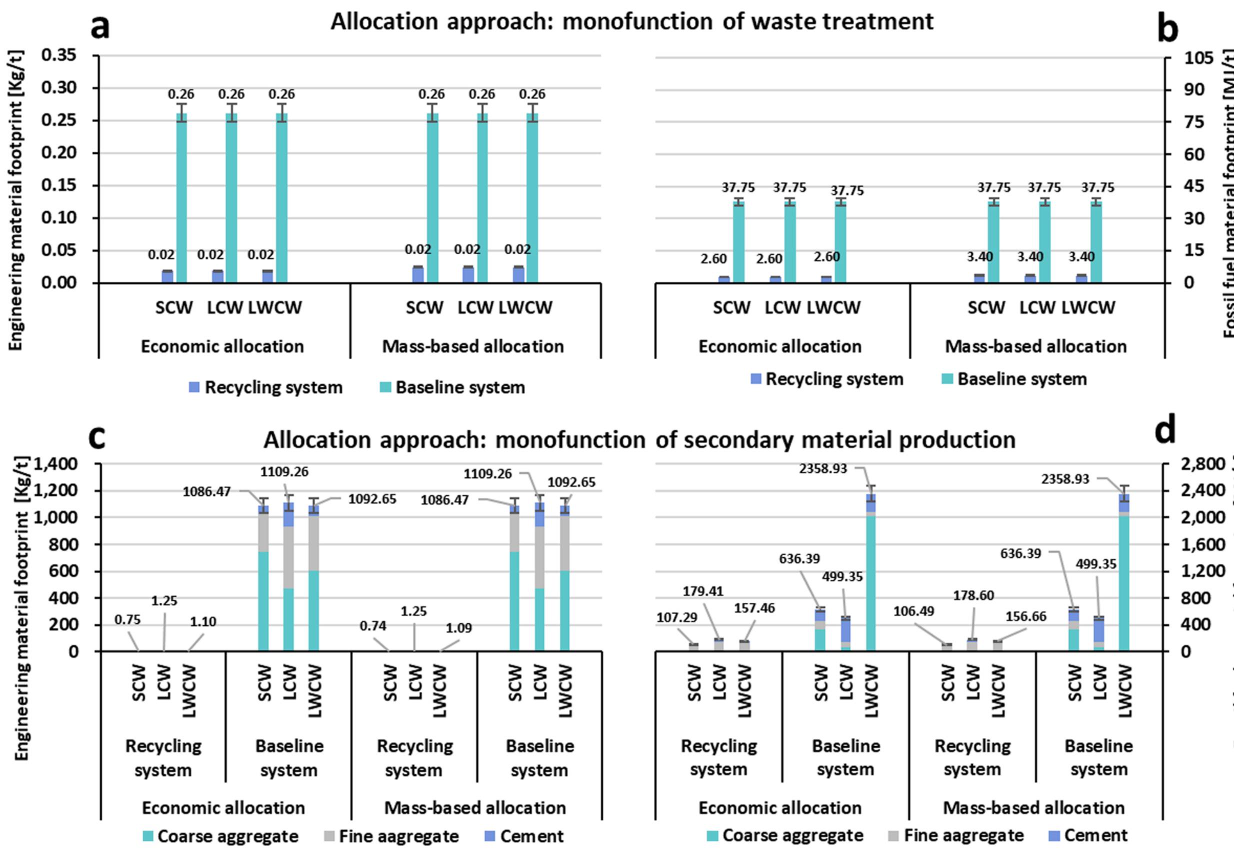

Fig. 2. Engineering material footprint (EMF) and fossil fuel material footprint (FMF) of the recycling system and baseline system by using the allocation approach. (a) EMF and (b) FMF of the waste management function and (c) EMF and (d) FMF of the material production function of both systems. SCW represents siliceous concrete waste; LCW denotes limestone concrete waste; LWCW means lightweight concrete waste.

material footprint of the waste treatment and the secondary material production functions of the recycling system. Regarding the function of waste treatment, the results of the substitution approach (Fig. 3) show the same trend as that of the allocation approach. The difference is that the results of the waste treatment of the recycling system are negative (Fig. 3a–b), which indicates the benefits gained from avoided flows.

Finally, the system expansion approach was applied to assess the material footprint of recycling three types of concrete waste compared to the baseline system. The EMF and the FMF of the recycling and baseline systems using the system expansion approach (in Fig. 4) also demonstrate the same trend as the allocation and substitution approaches—the recycling system has much lower EMF and FMF than the baseline system.

3.2. Adopting renewable energy for concrete recycling

In this section, we introduce the benefits of adopting alternative diesels and renewable electricity in the concrete recycling system on reducing the FMF. To comprehensively reflect the influence of using renewable energy on concrete recycling, the results of the system expansion approach were examined. Fig. 5a–b compares the FMF of the baseline system and the recycling system using alternative diesels under two energy transition scenarios—SSP2-RCP6 and SSP2-RCP26. The electricity mix of these two energy transition scenarios is shown in Fig. 5c. Generally, with the development of renewable electricity, the FMF of baseline scenarios slightly declines. The recycling system using biodiesel, coal- and wood-based syndiesel seems not to be affected by the adoption of renewable electricity. However, the FMF of the recycling system using electrolysis-based syndiesel can be noticeably reduced in the SSP2-RCP26 scenario compared with the SSP2-RCP6 scenario, as electrolysis is a power-intensive process. It is also noteworthy that the recycling system using coal- and electrolysis-based syndiesel have higher FMF compared with that using virgin diesel. This is because that coal-based syndiesel is produced by gasifying fossil fuel, and electrolysis still is powered by electricity with higher proportion of fossil-based sources in the SSP2-RCP6 scenario. In contrast, using biodiesel for recycling those three types of concrete wastes can reduce 47–51% of the FMF compared with virgin diesel-based recycling system; an 84–89% reduction can be achieved if shifting to wood-based syndiesel. Therefore, regarding concrete recycling, reduction of the FMF is more sensitive to the use of alternative diesels than adoption of renewable electricity.

4. Discussion

4.1. The myth of dual merits of recycling

Proponents usually endorse recycling for its two merits: primary material displacement and disposal elimination (Ackerman, 1997). Recycling of concrete can produce secondary raw materials and avoid landfilling. However, Zink and Geyer (2019) mathematically demonstrated that primary material displacement is the major function of recycling. Hence, the "dual merits" of recycling are just one, and the environmental benefits of recycling just lie in primary material displacement. This can also be noticed from the findings of this LCA case study. The mass-based and economic allocation approaches demonstrate that more than 98% of the EMF and the FMF are attributed to the function of secondary raw material production, while the impact of waste treatment is negligible. However, this conclusion may be reversed in other cases. First, a debated assumption for modeling recycling is that recycled material can displace primary production (Zink and Geyer, 2019). Concrete recycling can indeed reduce the use of virgin sand,

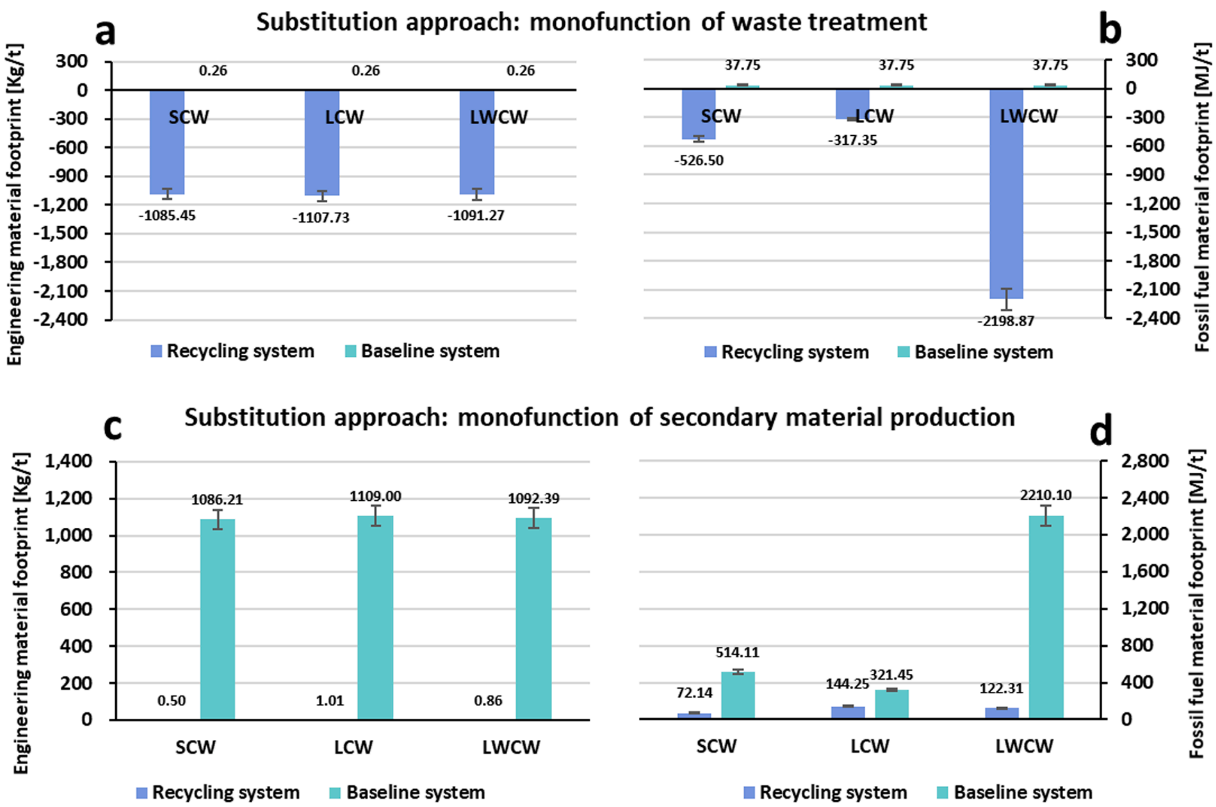

Fig. 3. Engineering material footprint (EMF) and fossil fuel material footprint (FMF) of the recycling and baseline systems by using a substitution approach. (a) EMF and (b) FMF of the waste management function and (c) EMF and (d) FMF of the material production function of both systems. SCW represents siliceous concrete waste; LCW denotes limestone concrete waste; LWCW means lightweight concrete waste.

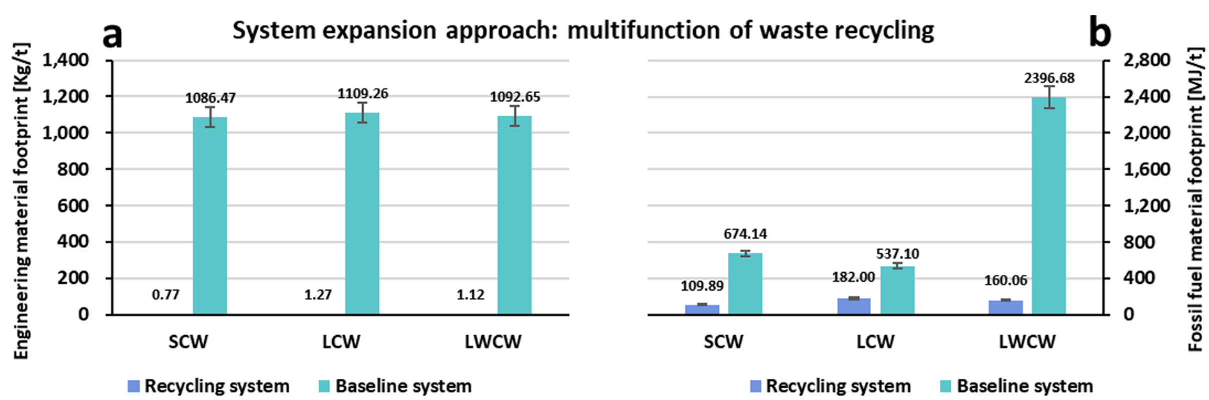

Fig. 4. Engineering material footprint (a) and fossil fuel material footprint (b) of the recycling and baseline systems by using the system expansion approach. SCW: siliceous concrete waste; LCW: limestone concrete waste; LWCW: lightweight concrete waste.

gravel, and cement. While recycled materials cannot entirely substitute all virgin materials in concrete as the recycled materials' physico-chemical properties cannot fully match those of virgin materials at current stage. Therefore, the function of the recycling system and the baseline system is not rigorously the same. Second, in general, the end-of-waste status of the waste concrete is hard to determine in a recycling system. For instance, in this study (see Fig. 1), only one process—crushing—is modeled as a recycling-type of multifuntional process, while the subsequent ADR and HAS are modeled as production-based processes. The semi-product sieve sand (0–12 mm and 0–4 mm fractions) is reckoned as a product in this study as it can be sold as backfilling material for 2.50 €/t, yet, its price varies according to the market. Therefore, the waste treatment function is fulfilled after waste concrete is processed by a crusher. While if the market is not available, sieve sand has to be disposed of through elevating sites without useful applications; then, sieve sand can be seen as a waste and all three processes in Fig. 1a become recycling processes. In this case, additional impacts will be attributed to the function of waste treatment. Third, the primary purpose of recycling some wastes might be just to avoid the adverse impacts of wastes instead of producing secondary material. Concrete waste is inert and non-hazardous, thus, the gate fee charged by the waste treatment plant remains relatively low (2.50 €/t). While the gate fees for containing hazardous waste could be much higher, resulting in that the waste treatment function could be the main contributor to its life cycle impacts based on an economic allocation.

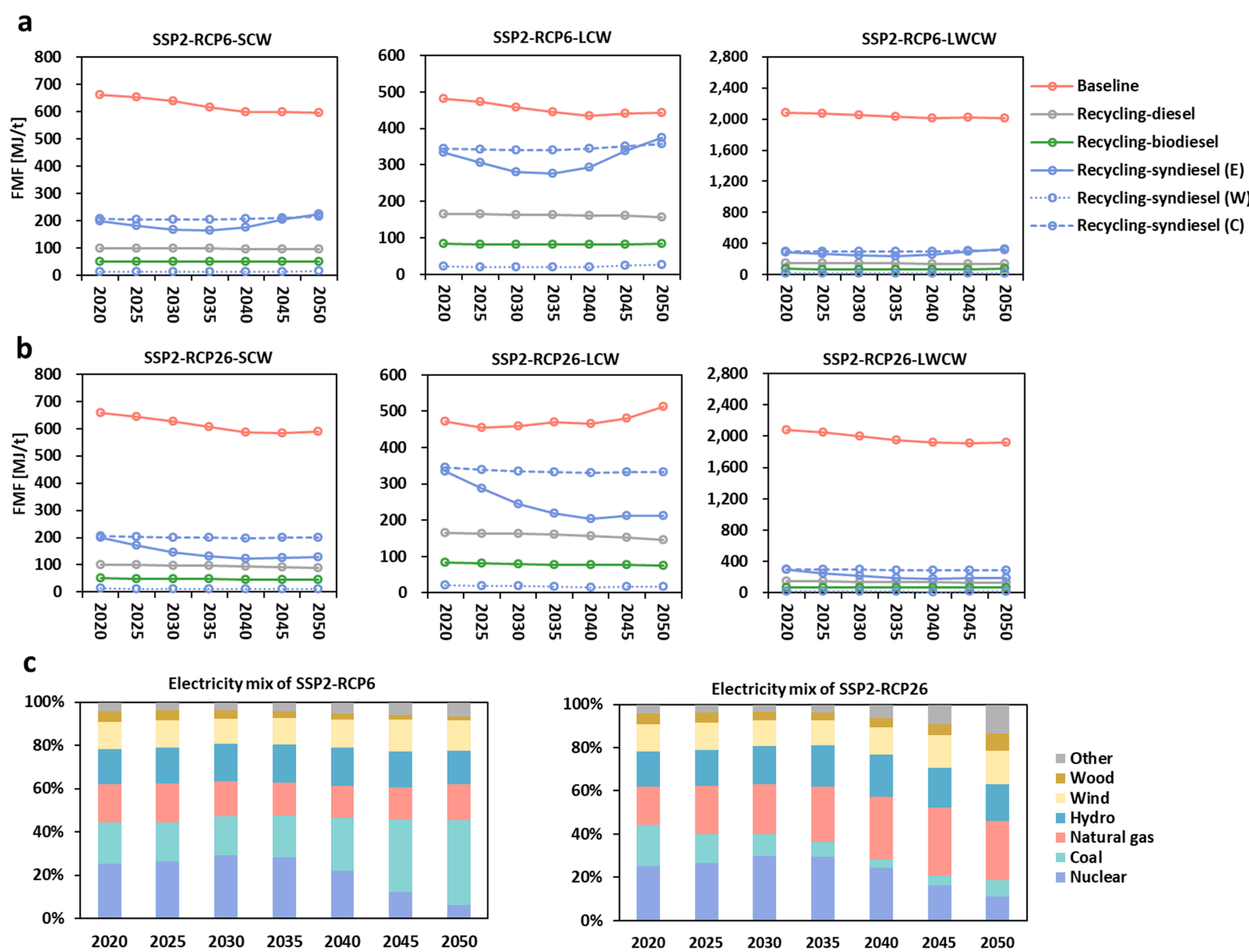

Fig. 5. Comparison of fossil fuel material footprint (FMF) of recycling siliceous, limestone, and lightweight concrete wastes across different energy carriers (a, and b), and electricity mix under two socioeconomic transitions (c). SSP2: Shared Socioeconomic Pathway under the intermediate challenge; RCP: Representative Concentration Pathway; SCW: siliceous concrete waste; LCW: limestone concrete waste; LWCW: lightweight concrete waste; biodiesel means biodiesel made from rapeseed; syndiesel (E): synthetic diesel made from hydrogen that is produced from water electrolysis; syndiesel (W): synthetic diesel made from hydrogen that is produced from wood gasification; syndiesel (C): synthetic diesel made from coal gasification that is produced from coal gasification. The electricity from other sources includes oil, photovoltaic, geothermal, and biomass.

4.2. The potential rebound effect of concrete recycling

Recycling should also be applied with caution. First, not every waste is technically recyclable, and the recyclability largely depends on the condition and status of the waste (Nash, 2009). Heavily contaminated concrete cannot re-enter a new concrete cycle and is usually sanitarily landfilled. More importantly, not all wastes are worth recycling. Abundant evidence has indicated that recycling is usually costly and inefficient (Tierney, 2015). Optimists believe challenges like increasing resource depletion can be overcome by human ingenuity, such as political interventions and technological improvement (Nordhaus, 1992). To ensure the quality of concrete waste at its source, the EU proposed selective demolition and dismantling regulations for end-of-life buildings (EC, 2018, 2016). However, selective demolition and sound dismantling of buildings also lead to higher costs and longer time. Technological innovation can significantly improve the cost-effectiveness and productive efficiency of recycling operations (Zhang et al., 2019). Nevertheless, novel recycling methods may also bring about new issues when dealing with the existing ones, and could even be counterproductive, leading to higher material input (Zink and Geyer, 2017). Researchers and practitioners have pointed out the importance of considering the rebound effects of recycling (Birat, 2015; Chen, 2021; Lonca et al., 2018; Plank et al., 2020; Zink and Geyer, 2019, 2017).

The rebound effect was first described by the English economist William Jevons (1865) in his well-known work "The Coal Question". The rebound effect can be generally understood as a change in overall consumption and production induced by a change in the efficiency of a technology (Font Vivanco and van der Voet, 2014). For concrete recycling, this study demonstrates that the emerging thermal treatment powered by alternative diesels can fully recycle each composite of concrete waste in a higher material-efficient way. However, closed-loop recycling is not necessarily superior to open-loop recycling or downcycling (Geyer et al., 2016). Di Maria et al. (2018) illustrated that concrete recycling is more expensive than downcycling and landfilling if the landfill tax were not included. Therefore, greater recycling might lead to greater loss, compared to downcycling or under a broader evaluation criterion.

4.3. Potential double counting in energy footprint and material footprint

The product material footprint impact category Raw Material Input (Mostert and Bringezu, 2019) adds up different material sources in terms of their mass equivalents, which is analogous to the Cumulative Energy Demand (CED) method. One may argue that material accounting schemes do not reflect the environmental impacts caused by material extraction. Therefore, reckoning material footprint as an impact assessment method may lead to the same debate as to the CED: is it just an inventory indicator that belongs to life cycle inventory analysis or an impact category that goes to the life cycle impact assessment (Frischknecht et al., 2015)? This study does not aim to clarify this intertwined issue but tries to explain what kind of materials leave the footprint in material footprint accounting.

Materials accounted for in the material footprint can be divided into three categories: (i) engineering material (e.g., gravel and iron), (ii) fossil fuel material (oil, natural gas, and coal), and (iii) fissionable energy material (e.g., uranium). The gross product material footprint could lead to double counting concerning resource depletion and other energy consumption impacts. This is because the depletion of fossil fuel materials such as coal can be quantified twice in mass value (kg) and caloric value (MJ), either of which could indicate the depletion of fossil fuel material. Therefore, we came up with the concepts of the EMF to differentiate energetic resources from engineering material to avoid double counting.

On the other hand, the boundary of energy and material becomes even blurrier when referring to Einstein's formula ($E = mc^2$), which reveals the mass-energy equivalence (Bodanis, 2009). Therefore, each resource input can theoretically be expressed in mass terms or full energy content. Either option can identically represent a resource input and avoid double counting. However, this conversion may be "physically unambiguous" but useless for the characterization or inventorying in an LCA (Frischknecht et al., 1998). For example, in almost every case, sand can always be considered an engineering material. It is also pointless to convert sand into energy content as we cannot utilize the 9^{10} MJ of energy embedded in 1 Kg of sand based on the current technologies. Moreover, the term "material circularity" only addresses the recyclability of engineering materials. This is because the energy in fossil fuel material cannot be circularized without violating the second law of thermodynamics. At the same time, some materials can indeed be used for either engineering purposes or as energy sources. Crude oil can be used to produce plastics for engineering purposes or to be combusted as fuel. Additionally, uranium is used as the main resource to fuel nuclear power plants, but it was also already used as an engineering material in some containers used to store and transport radioactive materials in the pre-nuclear era (Frischknecht et al., 1998). Therefore, it is necessary to identify the functions and purposes of the investigated material in a production system and adopt the impact method accordingly, which is how the EME is proposed in this study. The purpose of this study is not to provide an accurate assessment of the EMF or the FMF. Instead, it sheds light on the potential double-counting risks in material and energy accounting systems. When assessing the use of material and energy of a product system in a multiple-criteria decision analysis, possible solutions to avoid double couting are: (i) to account for the engineering and energy materials separately; (ii) to avoid weighting and merging the final results into a single indicator.

5. Conclusions

This study conducts a comparative LCA to explore the material footprint in concrete recycling. Three factors that may affect the material footprint assessment of concrete recycling were considered. First, three different categories of waste concrete were assessed, namely siliceous concrete, limestone concrete, and lightweight concrete. Second, three different recycling modeling approaches, allocation, substitution, and system expansion, were examined. Finally, how the deployment of renewable energy including electricity, biodiesel, and syndiesel would affect the concrete recycling was evaluated. The key conclusions we obtained from the analysis are that concrete recycling can noticeably reduce the EMF by 99% and the FMF by 66–93%. However, the types of waste concrete can lead to different levels of the FMF reduction. Recycling lightweight concrete can alleviate significantly more fossil fuel consumption than the other two types of concrete. Regarding the use of renewable energy, adopting alternative diesel is more sensitive than using renewable electricity regarding the reduction of FMF in concrete recycling. Replacing virgin diesel with electrolysis- and coal-based diesels in the thermal treatment of the recycling system could even increase the FMF, while using biodiesel made from rapeseed and wood-based syndiesel can reduce 47–51% and 84–89%, respectively, compared to virgin diesel-based recycling system.

Based on the results, we further discussed the bi-functionality, potential rebound effects of recycling, and double-counting risk in material and energy accounting. First, regarding concrete recycling, secondary raw material production and the subsequent primary production displacement is the main function of the recycling system. Second, with respect to the rebound effect, the advanced recycling system can fully recycle each composite of waste concrete in a higher material-efficient way. However, it might also lead to a greater loss compared to downcycling or under a broader evaluation criterion. Finally, the potential double counting risk in material and energy assessment system was discussed. The outcome of this study can not only provide insights into transitioning to a dematerialized concrete industry and avoiding problem shifts in adopting a circular economy but also help better to understand the existing material and energy accounting schemes.

Declaration of Competing Interest

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

Data availability

Data will be made available on request.

Acknowledgements

The authors thank the support of the EU Horizon2020 project VEEP "Cost-Effective Recycling of C&DW in High Added Value Energy Efficient Prefabricated Concrete Components for Massive Retrofitting of our Built Environment" (No. 723582), and the EU Horizon2020 project ICEBERG "Innovative Circular Economy Based solutions demonstrating the Efficient recovery of valuable material Resources from the Generation of representative End-of-Life building materials" (No. 869336). The authors also sincerely thank Dr. Clemens Mostert of the University of Kassel for his support and comments on the Material Footprint method.

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2022.10.035.

References

Ackerman, F., 1997. Why do we recycle? Markets, values, and public policy. Island Press, Washington D.C.

Birat, J.-P., 2015. Life-cycle assessment, resource efficiency and recycling. Metall. Res. Technol. 112, 206. https://doi.org/10.1051/metal/2015009.

Bodanis, D., 2009. E=mc2: A Biography of the World's Most Famous Equation. Bloomsbury Publishing.

Brander, M., Wylie, C., 2011. The use of substitution in attributional life cycle assessment. Greenh. Gas Meas. Manag. 1, 161–166. https://doi.org/10.1080/20430779.2011.637670.

Chen, C.W., 2021. Clarifying rebound effects of the circular economy in the context of sustainable cities. Sustain. Cities Soc. 66, 102622 https://doi.org/10.1016/j.scs.2020.102622.

Di Maria, A., Eyckmans, J., Van Acker, K., 2018. Downcycling versus recycling of construction and demolition waste: Combining LCA and LCC to support sustainable policy making. Waste Manag. 75, 3–21. https://doi.org/10.1016/j.wasman.2018.01.028.

EC, 2008. 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)

[WWW Document]. URL https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0098 (accessed 4.13.18).
EC, 2018. Guidelines for the waste audits before demolition and renovation works of buildings. UE Construction and Demolition Waste Management.
EC, 2016. EU Construction & Demolition Waste Management Protocol.
Ecoinvent, 2021. ecoinvent 3.8 cut-off database [WWW Document]. URL https://ecoinvent.org/.
Ellen MacArthur Foundation, 2015. Towards a Circular Economy: Business Rationale for an Accelerated Transition.
Font Vivanco, D., van der Voet, E., 2014. The rebound effect through industrial ecology's eyes: a review of LCA-based studies. Int. J. Life Cycle Assess. 19, 1933–1947. https://doi.org/10.1007/s11367-014-0802-6.
Frischknecht, R., Heijungs, R., Hofstetter, P., 1998. Einstein's lessons for energy accounting in LCA. Int. J. Life Cycle Assess. 3, 266–272. https://doi.org/10.1007/BF02979833.
Frischknecht, R., Wyss, F., Büsser Knöpfel, S., Lützkendorf, T., Balouktsi, M., 2015. Cumulative energy demand in LCA: the energy harvested approach. Int. J. Life Cycle Assess. 20, 957–969. https://doi.org/10.1007/s11367-015-0897-4.
Gebremariam, A.T., Di Maio, F., Vahidi, A., Rem, P., 2020. Innovative technologies for recycling End-of-Life concrete waste in the built environment. Resour. Conserv. Recycl. 163, 104911 https://doi.org/10.1016/j.resconrec.2020.104911.
Geyer, R., Kuczenski, B., Zink, T., Henderson, A., 2016. Common Misconceptions about Recycling. J. Ind. Ecol. 20 (5), 1010–1017.
Guinée, J.B., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H. a., de Bruijn, H., van Duin, R., Huijbregts, M. a. J., Gorrée, M., 2001. Handbook on life cycle assessment: operational guide to the ISO standards, Kluwer Academic Publishing. Dordrencht.
Guinée, J.B., Heijungs, R., Huppes, G., 2004. Economic allocation: Examples and derived decision tree. Int. J. Life Cycle Assess. 9, 23. https://doi.org/10.1007/BF02978533.
Habert, G., Miller, S.A., John, V.M., Provis, J.L., Favier, A., Horvath, A., Scrivener, K.L., 2020. Environmental impacts and decarbonization strategies in the cement and concrete industries. Nat. Rev. Earth Environ. 1, 559–573. https://doi.org/10.1038/s43017-020-0093-3.
Heijungs, R., Allacker, K., Benetto, E., Brandão, M., Guinée, J., Schaubroeck, S., Schaubroeck, T., Zamagni, A., 2021. System Expansion and Substitution in LCA: A Lost Opportunity of ISO 14044 Amendment 2. Front. Sustain. 2, 1–3. https://doi.org/10.3389/frsus.2021.692055.
Heijungs, R., Frischknecht, R., 1998. The Nature of the Allocation Problem LCA Methodology A Special View on the Nature of the Allocation Problem. Lca 3 (6), 321–332.
Hoornweg, D., Bhada-Tata, P., 2012. What a waste: a global review of solid waste management. https://doi.org/10.1111/febs.13058.
Huang, B., Gao, X., Xu, X., Song, J., Geng, Y., Sarkis, J., Fishman, T., Kua, H., Nakatani, J., 2020. A Life Cycle Thinking Framework to Mitigate the Environmental Impact of Building Materials. One Earth 3, 564–573. https://doi.org/10.1016/j.oneear.2020.10.010.
Huijbregts, M.A.J., Norris, G.A., Bretz, R., Ciroth, A., Maurice, B., Mahasenan, N., Bahr, B. von, Weidema, B.., 2003. Code of Life-Cycle Inventory Practice. Society of Environmental Toxicology and Chemistry (SETAC), Brussel.
ISO, 2006. ISO 14044: Environmental management — Life cycle assessment — Requirements and guidelines. Geneva.
ISO, 2020. ISO 14044:Environmental Management - Life Cycle Assessment - Requirements and Guidelines - Amendment 2. Geneva.
Jevons, W.S., 1865. The Coal Question: An Inquiry Concerning the Progress of the Nation, and the Probable Exhaustion of Our Coal Mines. Macmillan Publishers, London.
Jin, R., Chen, Q., 2019. Overview of Concrete Recycling Legislation and Practice in the United States. J. Constr. Eng. Manag. 145 https://doi.org/10.1061/(ASCE)CO.1943-7862.0001630.
Klöpffer, W., 1997. In defense of the cumulative energy demand. Int. J. Life Cycle Assess. 2, 61–61. https://doi.org/10.1007/BF02978754.
Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., Hauschild, M.Z., Christensen, T.H., 2014. Review of LCA studies of solid waste management systems - Part I: Lessons learned and perspectives. Waste Manag. 34, 573–588. https://doi.org/10.1016/j.wasman.2013.10.045.
Lonca, G., Muggéo, R., Imbeault-Tétreault, H., Bernard, S., Margni, M., 2018. Does material circularity rhyme with environmental efficiency? Case studies on used tires. J. Clean. Prod. 183, 424–435. https://doi.org/10.1016/j.jclepro.2018.02.108.
Miller, S.A., Horvath, A., Monteiro, P.J.M., 2018. Impacts of booming concrete production on water resources worldwide. Nat. Sustain. 1, 69–76. https://doi.org/10.1038/s41893-017-0009-5.
Miller, S.A., Moore, F.C., 2020. Climate and health damages from global concrete production. Nat. Clim. Chang. 10, 439–443. https://doi.org/10.1038/s41558-020-0733-0.
Monteiro, P.J.M., Miller, S.A., Horvath, A., 2017. Towards sustainable concrete. Nat. Mater. 16, 698–699. https://doi.org/10.1038/nmat4930.
Moreno-Juez, J., Vegas, I.J., Gebremariam, A.T., García-Cortés, V., Di Maio, F., 2020. Treatment of end-of-life concrete in an innovative heating-air classification system for circular cement-based products. J. Clean. Prod. 263 https://doi.org/10.1016/j.jclepro.2020.121515.
Mostert, C., Bringezu, S., 2019. Measuring Product Material Footprint as New Life Cycle Impact Assessment Method: Indicators and Abiotic Characterization Factors. Resources 8, 61. https://doi.org/10.3390/resources8020061.
Mostert, C., Sameer, H., Glanz, D., Bringezu, S., 2021. Climate and resource footprint assessment and visualization of recycled concrete for circular economy. Resour. Conserv. Recycl. 174, 105767 https://doi.org/10.1016/j.resconrec.2021.105767.
Mulder, E., de Jong, T.P.R., Feenstra, L., 2007. Closed Cycle Construction: An integrated process for the separation and reuse of C&D waste. Waste Manag. 27, 1408–1415. https://doi.org/10.1016/j.wasman.2007.03.013.
Nash, H.A., 2009. The revised directive on waste: Resolving legislative tensions in waste management? J. Environ. Law 21, 139–149. https://doi.org/10.1093/jel/eqp001.
Nordhaus, W.D., 1992. The ecology of markets. Proc. Natl. Acad. Sci. 89, 843–850. https://doi.org/10.1073/pnas.89.3.843.
O'Neill, B.C., Kriegler, E., Riahi, K., Ebi, K.L., Hallegatte, S., Carter, T.R., Mathur, R., van Vuuren, D.P., 2014. A new scenario framework for climate change research: The concept of shared socioeconomic pathways. Clim. Change 122, 387–400. https://doi.org/10.1007/s10584-013-0905-2.
PBL, 2021. Integrated Model to Assess the Global Environment (IMAGE) 3.2 [WWW Document]. URL https://models.pbl.nl/image/index.php/Welcome_to_IMAGE_3.2_Documentation.
Plank, B., Eisenmenger, N., Schaffartzik, A., 2021. Do material efficiency improvements backfire?: Insights from an index decomposition analysis about the link between CO2 emissions and material use for Austria. J. Ind. Ecol. 25 (2), 511–522.
Purnell, P., Dunster, A., 2010. Recycling of concrete, in: Management, Recycling and Reuse of Waste Composites. Elsevier, pp. 569–591. https://doi.org/10.1533/9781845697662.5.569.
Sacchi, R., Terlouw, T., Siala, K., Dirnaichner, A., Bauer, C., Cox, B., Mutel, C., Daioglou, V., Luderer, G., 2022. PRospective EnvironMental Impact asSEment (premise): A streamlined approach to producing databases for prospective life cycle assessment using integrated assessment models. Renew. Sustain. Energy Rev. 160, 112311 https://doi.org/10.1016/j.rser.2022.112311.
Schulz, H., 1999. Short history and present trends of Fischer-Tropsch synthesis. Appl. Catal. A Gen. 186, 3–12. https://doi.org/10.1016/S0926-860X(99)00160-X.
Steubing, B, de Koning, D, 2021. Making the use of scenarios in LCA easier: the superstructure approach. Int. J. Life Cycle Assess. 26 (11), 2248–2262. https://doi.org/10.1007/s11367-021-01974-2.
Steubing, B., de Koning, D., Haas, A., Mutel, C.L., 2020. The Activity Browser — An open source LCA software building on top of the brightway framework. Softw. Impacts 3, 100012. https://doi.org/10.1016/j.simpa.2019.100012.
Tierney, J., 2015. The reign of recycling [WWW Document]. New York Times. URL https://www.nytimes.com/2015/10/04/opinion/sunday/the-reign-of-recycling.html (accessed 6.4.20).
van Oers, L., de Koning, A., Guinée, J.B., Huppes, G., 2002. Abiotic resource depletion in LCA.
Waskow, R., Gonçalves Maciel, V., Tubino, R., Passuello, A., 2021. Environmental performance of construction and demolition waste management strategies for valorization of recycled coarse aggregate. J. Environ. Manage. 295, 113094 https://doi.org/10.1016/j.jenvman.2021.113094.
Weidema, B., 2000. Avoiding Co-Product Allocation in Life-Cycle Assessment. J. Ind. Ecol. 4, 11–33. https://doi.org/10.1162/108819800300106366.
Werner, F., Richter, K., 2000. Economic allocation in LCA: A case study about aluminium window frames. Int. J. Life Cycle Assess. 5, 79–83. https://doi.org/10.1007/BF02979727.
Yuan, H., Zhou, P., Zhou, D., 2011. What is Low-Carbon Development? A Conceptual Analysis. Energy Procedia 5, 1706–1712. https://doi.org/10.1016/j.egypro.2011.03.290.
Zhang, C., Hu, M., Dong, L., Gebremariam, A., Miranda-Xicotencatl, B., Di Maio, F., Tukker, A., 2019. Eco-efficiency assessment of technological innovations in high-grade concrete recycling. Resour. Conserv. Recycl. 149, 649–663. https://doi.org/10.1016/j.resconrec.2019.06.023.
Zhang, C., Hu, M., Yang, X., Miranda-Xicotencatl, B., Sprecher, B., Di Maio, F., Zhong, X., Tukker, A., 2020. Upgrading construction and demolition waste management from downcycling to recycling in the Netherlands. J. Clean. Prod. 266, 121718 https://doi.org/10.1016/j.jclepro.2020.121718.
Zhang, C., Hu, M., Laclau, B., Garnesson, T., Yang, X., Li, C., Tukker, A., 2021a. Environmental life cycle costing at the early stage for supporting cost optimization of precast concrete panel for energy renovation of existing buildings. J. Build. Eng. 35, 102002 https://doi.org/10.1016/j.jobe.2020.102002.
Zhang, C., Hu, M., Laclau, B., Garnesson, T., Yang, X., Tukker, A., 2021c. Energy-carbon-investment payback analysis of prefabricated envelope-cladding system for building energy renovation: Cases in Spain, the Netherlands, and Sweden. Renew. Sustain. Energy Rev 145, 111077. https://doi.org/10.1016/j.rser.2021.111077.
Zhang, C., Hu, M., Sprecher, B., Yang, X., Zhong, X., Li, C., Tukker, A., 2021b. Recycling potential in building energy renovation: A prospective study of the Dutch residential building stock up to 2050. J. Clean. Prod. 301, 126835 https://doi.org/10.1016/j.jclepro.2021.126835.
Zink, T., Geyer, R., 2017. Circular Economy Rebound. J. Ind. Ecol. 21, 593–602. https://doi.org/10.1111/jiec.12545.
Zink, T., Geyer, R., 2019. Material Recycling and the Myth of Landfill Diversion. J. Ind. Ecol. 23, 541–548. https://doi.org/10.1111/jiec.12808.