Towards circular and energy-efficient management of building stock: an analysis of the residential sector of the Netherlands

Zhang, C.

Chapter 2

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

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Published in Resources, Conservation and Recycling, 2019, 149, 1016-1030

Abstract

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

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

Abbreviations

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

2.1 Introduction

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

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

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

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

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

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

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

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

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

2.2 Literature review of eco-efficiency analysis

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

$$Eco-efficiency = \frac{Value\ added}{Environmental\ impact} \quad (1)$$

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

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

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

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

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

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

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

2.3 Methods

2.3.1 Framework for integrating LCC and LCA for eco-efficiency

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

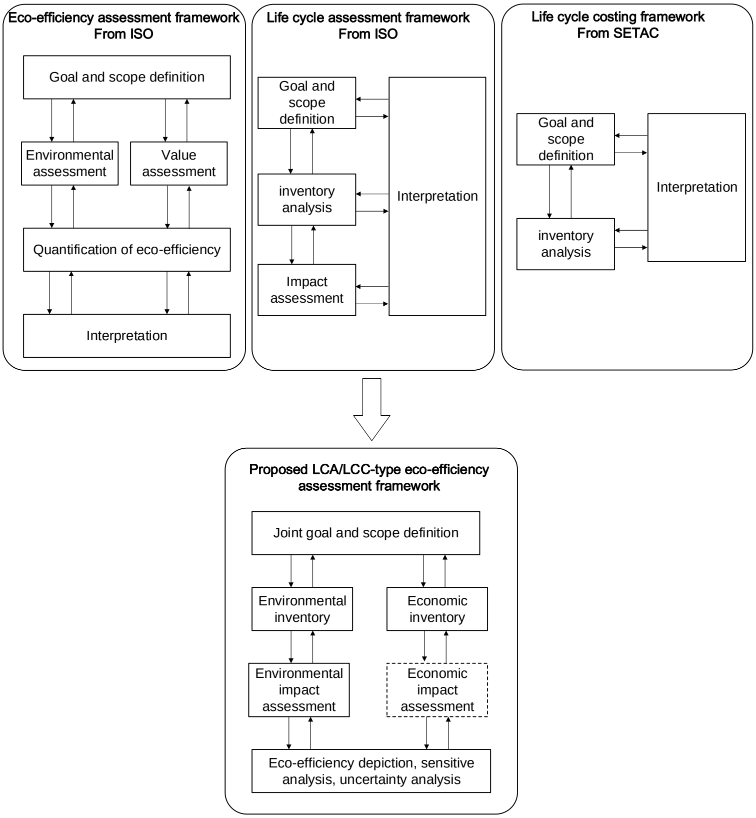

Figure 2.1 Proposed LCA/LCC-type eco-efficiency assessment framework, based on eco-efficiency assessment framework (ISO 2012), life cycle assessment framework (ISO 2006b), and SETAC environmental life cycle costing framework (Swarr et al. 2011; Ciroth et al. 2008)

2.3.2 Goal and scope definition

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

2.3.2.1 Description of the innovative technologies

Wet process

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

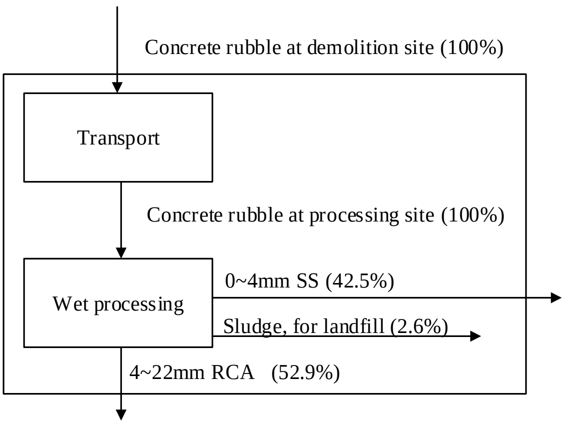

Figure 2.2 Mass balance of wet process

Advanced dry recovery (ADR)

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

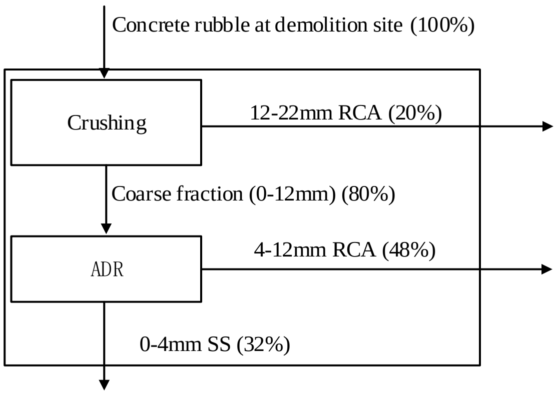

Figure 2.3 Mass balance of the Advance dry recovery (ADR)

Heating air classification system (HAS)

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

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

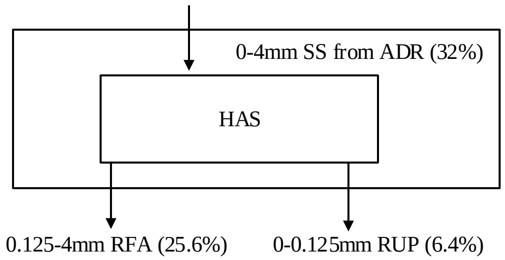

Figure 2.4 Mass balance of Heat air classification system (HAS)

Recycling systems used in the comparative evaluation

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

Table 2.2 Four technological systems for concrete recycling in this study

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

2.3.2.2 Functional unit

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

- ***a.*** *treatment of 100 tons of EoL concrete,*
- ***b.*** *68 tons of 4–22mm coarse aggregate for concrete;*
- ***c.*** *25.6 tons of 0.125–4mm fine aggregate for concrete;*
- ***d.*** *6.4 tons of cementitious material for concrete.*

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

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

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

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

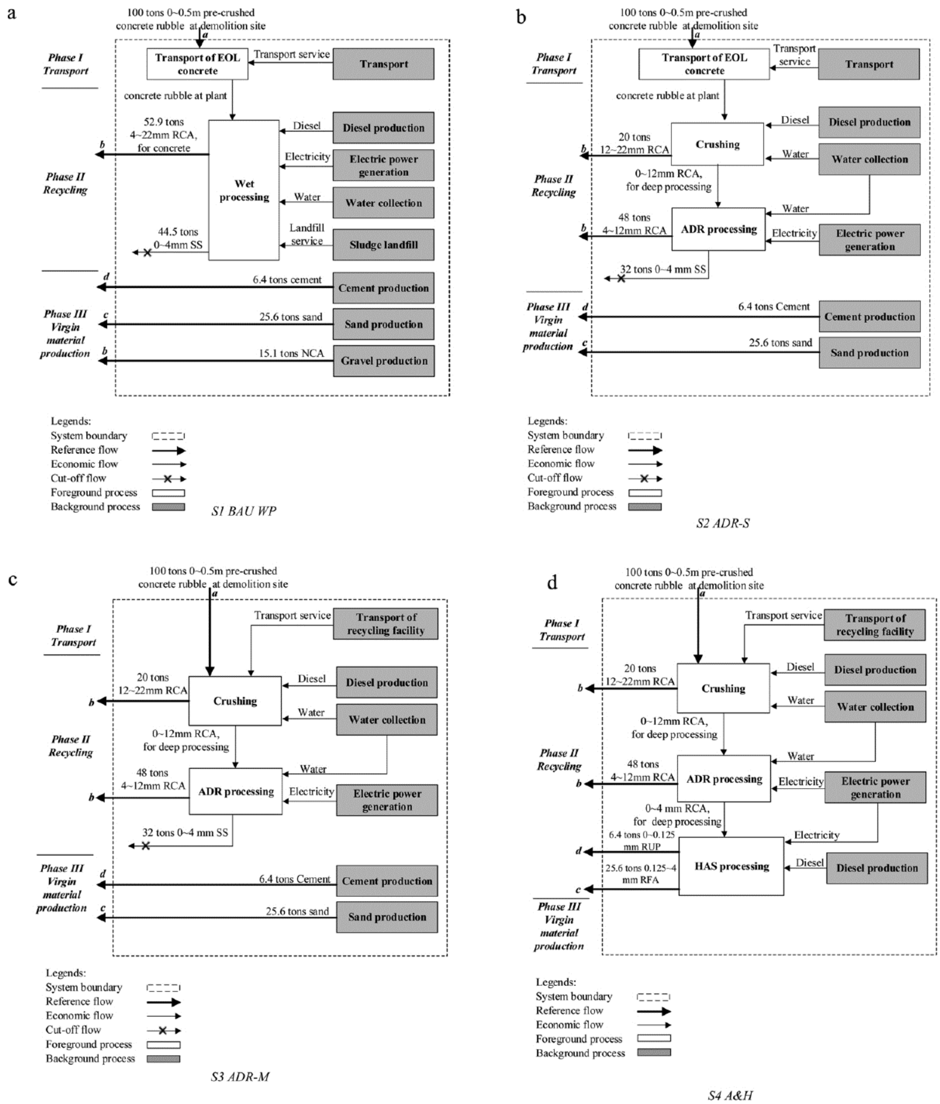

Figure 2.5 Flow diagrams for the four systems: (a). *S1 BAU WP* (wet process) system; (b). *S2 ADR-S* (stationary) system; (c). *S3 ADR-M* (mobile) system; and (d). *S4 A&H* (ADR & HAS) system, those flows which cross through the system boundary are functional flows, those lower-case letters at the end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

2.3.3 Life cycle assessment (LCA)

2.3.3.1 Environment inventory analysis

System boundary and unit processes

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

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

Data collection

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

Multifunctionality

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

2.3.3.2 Environment impact assessment

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

Table 2.4 ILCD impact assessment method

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

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

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

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

2.3.4 Life cycle costing (LCC)

2.3.4.1 Economic inventory analysis

Data collection

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

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

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

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

Multifunctionality

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

2.3.4.2 Economic impact assessment

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

Cost category definition

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

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

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

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

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

Cost impact category selection

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

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

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

Table 2.6 LCC impact categories

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

2.3.5 Integration of LCA and LCC for eco-efficiency indicators

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

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

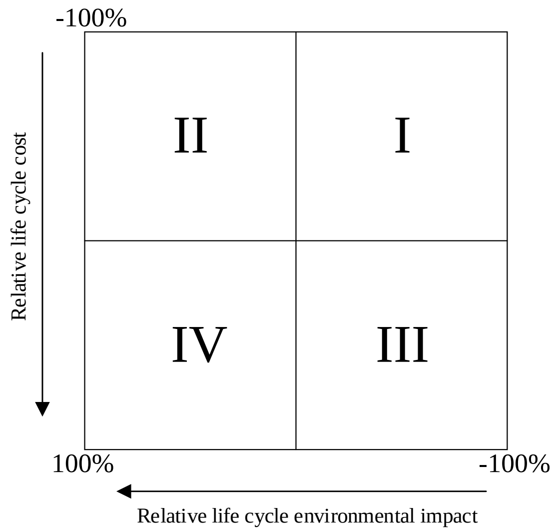

Figure 2.6 Eco-efficiency indicator graph

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

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

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

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

2.4 Results

2.4.1 Results of LCA

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

Table 2.7 Characterized life cycle environmental impact of four systems

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

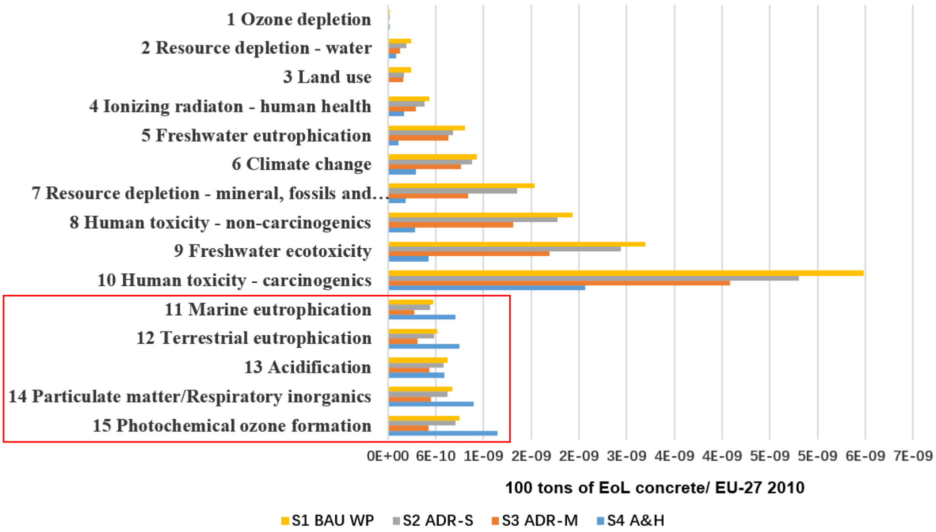

Figure 2.7 Normalized life cycle environmental impact of four systems

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

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

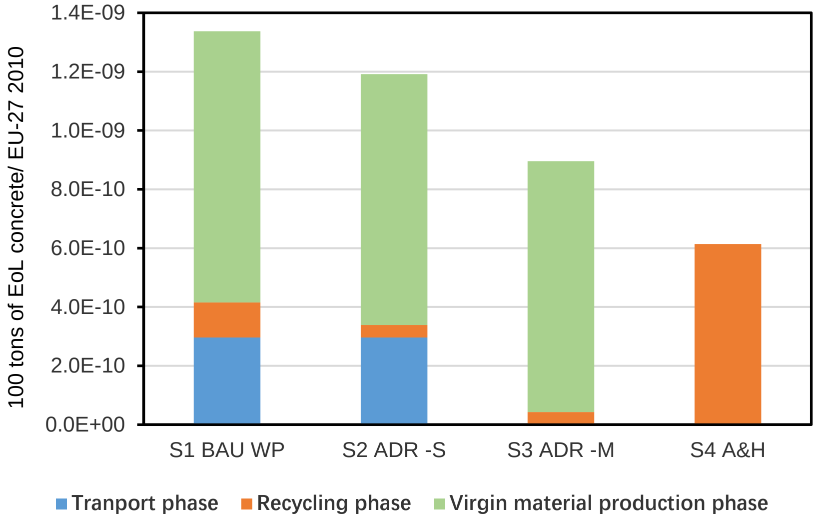

Figure 2.8 Distribution of the weighted environmental impact score in three phases

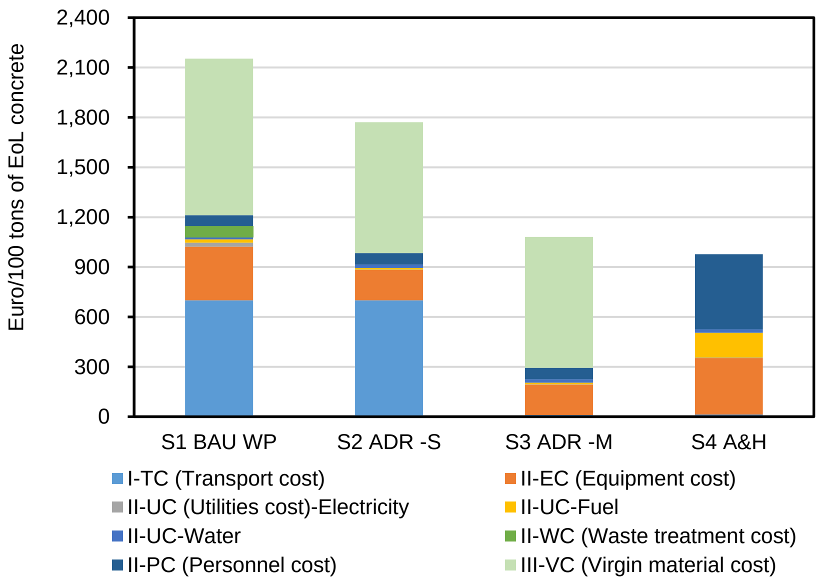

Figure 2.9 Life cycle costs of four systems

2.4.2 Results of LCC

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

2.4.3 Eco-efficiency index

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

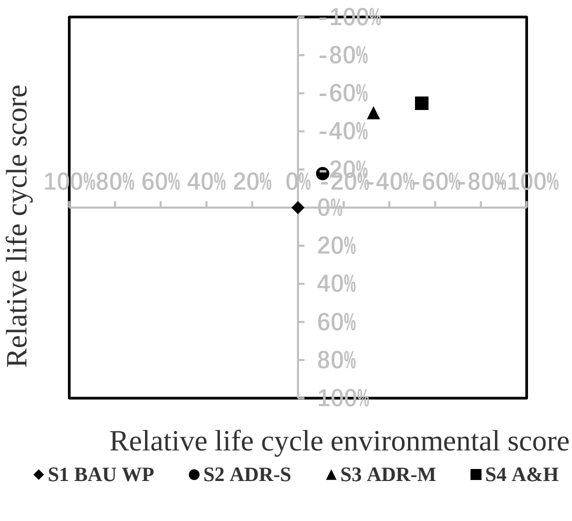

Figure 2.10 Eco-efficiency index diagram

2.4.4 Sensitivity analysis

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

Table 2.8 Factors for sensitivity analysis

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

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

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

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

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

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

2.4.5 Uncertainty analysis

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

Table 2.10 Relevant factors for uncertainty analysis

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

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

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

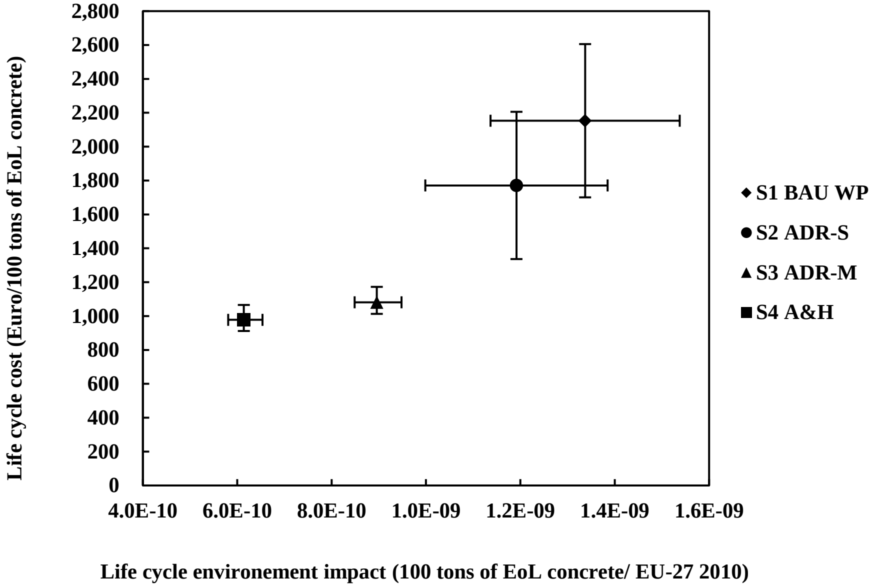

Figure 2.11 The uncertainty of eco-efficiency for four systems

2.5 Discussion

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

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

2.5.1 Current policies related to CDW management

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

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

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

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

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

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

2.5.2 Potential policy options

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

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

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

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

2.6 Conclusions

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

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

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

Acknowledgements

The authors thank the support of EU’s 7th Framework Project C2CA “Advanced Technologies for the Production of Cement and Clean Aggregates from Construction and Demolition Waste” (No. 265189), EU’s Horizon 2020 Project HISER “Holistic Innovative Solutions for an Efficient Recycling and Recovery of Valuable Raw Materials from Complex Construction and Demolition Waste” (No. 642085), and EU’s H2020 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). The first author appreciates the support of the China

Scholarship Council (No. 201706050090). Dr. L. Dong also appreciates the support of the National Science Foundation, China (NSFC, NO. 41701636).

Appendix

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

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

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

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

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

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

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

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