Potentials and Design of a Circular Economy for Autoclaved Aerated Concrete

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Abstract

Limiting anthropogenic climate change and transforming to a more sustainable lifestyle are among the current generation's most vital challenges. The built environment plays a crucial role in this context due to high resource consumption and greenhouse gas (GHG) emissions. Therefore, construction and demolition waste recycling is gaining importance but is difficult to realise for some building materials, including autoclaved aerated concrete (AAC). AAC has a low density and excellent thermal insulation properties due to its porous structure. Hence, AAC is a frequently used building material. However, recycling post-demolition AAC (pd-AAC) from the demolition and deconstruction of buildings is complicated as it has low compressive strength and contains sulphate. Therefore, pd-AAC is mainly landfilled. While there are some new pd-AAC recycling approaches, the quantitative, ecological and economic potential of pd-AAC recycling is unknown. Furthermore, no research compares different recycling approaches or examines recycling network structures to identify a circular economy design for AAC. This dissertation addresses these research gaps and answers the following research question: How can a circular economy for autoclaved aerated concrete be designed, and what quantitative, ecological, and economic potential does it have in Germany and Europe?

Quantification shows that pd-AAC volumes reach 1.2 Mm^3 in Germany in 2020 and are expected to rise significantly to over 4 Mm^3 by 2050 (Study A). At the European level (Study B), ten times the German volumes can be expected. A life cycle assessment is conducted to identify the ecological potential of different pd-AAC recycling options (Study C). Using pd-AAC to partly substitute inputs of the lightweight aggregate concrete, light mortar, shuttering block, and AAC production is most promising. The pd-AAC processing only causes little impact, and significant environmental savings can be achieved due to the avoided production of primary materials, reaching total GHG savings of pd-AAC recycling of around 280,000 t CO_2-Eq/a in Germany and more than 8 Mt CO_2-Eq/a in Europe in the future. Additionally, a new recycling option, the production of recycled belite cement clinker (RC-BCC), proves to be ecologically beneficial despite energy-intensive processing (Study D). RC-BCC can replace emission-intensive primary Portland cement. Moreover, pd-AAC recycling has significant economic potential (Study E). Even smaller recycling plants can process pd-AAC cheaper than the average landfilling costs. However, RC-BCC production is not economically viable with current technologies. Mathematical modelling and optimisation methods are used to determine the best design of a pd-AAC recycling network (Study F). According to the computation results, large recycling plants should be preferred, and landfilling should be avoided. Overall, savings of around 4,600 M€ can be achieved until 2050 compared to the status quo.

This dissertation shows that pd-AAC recycling has a significant quantitative and ecological potential. Establishing high-quality recycling options to deal with the increasing future pd-AAC volumes is urgent. An optimally designed pd-AAC recycling network reaches high economic savings and supports the change towards a circular economy of AAC.

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Abbreviations

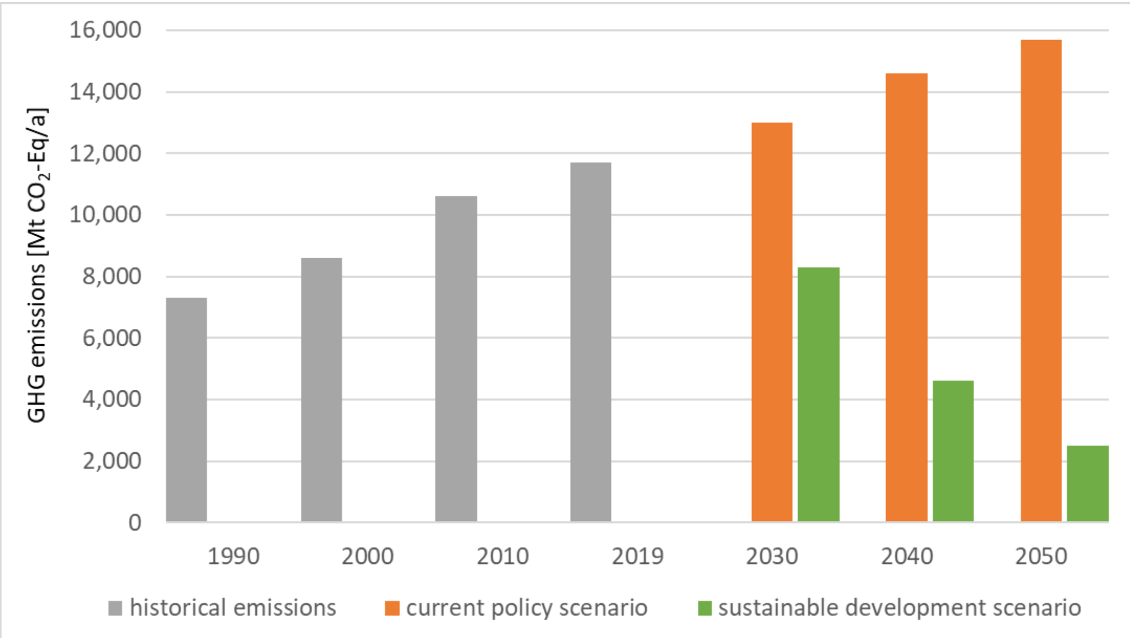
AAC	autoclaved aerated concrete
C&DW	construction and demolition wastes
EPD	environmental product declaration
GHG	greenhouse gas
LCA	life cycle assessment
LWAC	lightweight aggregate concrete
NUTS	Nomenclature des unités territoriales statistiques
pd-AAC	post-demolition autoclaved aerated concrete
RC-BCC	recycled belite cement clinker
TRL	technology readiness level

I Framework, Foundations and Implications

1 Introduction and Motivation

Human-caused greenhouse gas (GHG) emissions and climate change are fundamental global challenges. Habitats on the planet are changing due to higher average temperatures or are being flooded by rising sea levels. In addition, extreme weather events are becoming more frequent (IPCC, 2022) and cause substantial damage to people and nature. Therefore, mitigating climate change is a crucial issue. Central concepts for meeting this issue are the sustainable use of materials and the closing of material loops.

The built environment is essential in the transition to more sustainability as it requires enormous amounts of primary resources, consuming around 50% of the extracted materials in Europe (European Commission, 2020a). Moreover, the building sector accounted for significant GHG emissions of nearly 12 Gt CO_2-Eq[1] worldwide in 2019, corresponding to 21% of total global GHG emissions (IPCC, 2022). In a 2050 projection, these emissions rise to almost 16 Gt CO_2-Eq (+34%), assuming a current policy scenario, while a significant decrease to about 2.5 Gt CO_2-Eq (-79%) would be necessary to meet a sustainable development scenario (IPCC, 2022; Figure 1.1). Therefore, substantial GHG savings in the building sector must be identified and realised to correct the course from the current policy to sustainable development.



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[1] The unit ”t” represents metric tons throughout this dissertation.

The United Nations Sustainable Development Goals, especially "sustainable cities" (No. 11), "responsible consumption and production" (No. 12), and "climate action" (No. 13) (UN, 2023), also aim at a more sustainable building sector. An important issue in achieving these goals is the construction material. Worldwide, the embodied emissions in construction materials caused by the primary production sum up to more than 2 Gt CO_2-Eq annually and thus account for almost 20% of the total GHG emissions of the building sector (IPCC, 2022).

Construction and demolition wastes (C&DW) summed up to more than 3,000 Mt worldwide in 2012 (Akhtar & Sarmah, 2018) and around 220 Mt in Germany in 2020 (Kreislaufwirtschaft Bau, 2023). These enormous quantities lead to significant potential when closing the building material loops and utilising the C&DW. Massive resource and energy consumption savings can be realised, and the embodied emissions can be reduced by decreasing the primary material production. Additionally, the EU has created legal regulations for the sustainable use of construction materials and closing material loops in the construction industry (Directive 2008/98/EC, 2008). Besides, external factors like decreasing landfill capacities and increasing landfill fees (Knappe et al., 2012; Riegler-Floors & Hillebrandt, 2018) create economic incentives for a sustainable C&DW treatment.

Current sustainability approaches in the construction sector focus on C&DW recycling in road construction and earthworks (73% of all C&DW in Germany), as well as asphalt and concrete production (20% of all C&DW in Germany), reaching a recycling rate of more than 90% (Kreislaufwirtschaft Bau, 2023). However, there are no established recycling concepts for some C&DW fractions, leaving a significant potential unused (OECD, 2020). These C&DW fractions without established recycling concepts also include autoclaved aerated concrete (AAC), which will be considered in detail within this dissertation.

AAC is a mineral building material used as masonry units, especially in constructing one- or two-family houses (UBA, 2019), or for mineral insulation boards. It is produced from quartz sand, cement, quicklime, gypsum or anhydrite, some aluminium powder or paste and water (Kreft, 2017; UBA, 2019). Many tiny pores are formed and preserved during AAC's production, leading to a very low density that can reach as low as 305 kg/m³ (DIN 20000-404:2018-04). Consequently, AAC's thermal insulation capability outperforms other monolithic building materials, including concrete, classical clay bricks, and calcium silicate units. Additional insulation layers are unnecessary for houses built with AAC, resulting in high fire resistance and faster construction. These advantages lead to AAC being Germany's second most used construction material for residential buildings in 2021 based on the number of constructed buildings (Destatis, 2022). Total AAC production in Germany in 2021 was 3.3 Mm³ of masonry units and 1.4 Mm² of panels and floorboards (GENESIS, 2023), while current AAC production in Europe exceeds 16 Mm³ (EAACA, 2023). Worldwide AAC production capacity is around 450 Mm³ for non-reinforced AAC blocks (Fouad & Schoch, 2018).

These high AAC production amounts lead to an accumulation in the building stock and, after the demolition of the respective buildings, to significant quantities of post-demolition AAC (pd-AAC) that must be treated. Unfortunately, the recycling approaches for C&DW mentioned above are unsuitable for pd-AAC. Pd-AAC has a low compressive strength compared to other mineral construction materials, which impedes recycling in road construction and using pd-AAC as aggregate in concrete production since strength requirements are not fulfilled. Moreover, pd-AAC recycling in earthworks is impossible as it contains small amounts of sulphate (from the gypsum or anhydrite), which must not contact groundwater (Knappe et al., 2012). This lack of recycling options leads to a landfill of most pd-AAC (UBA, 2019).

Besides, a potential pd-AAC recycling fundamentally differs from the current concrete recycling approach. The basic idea of closing material loops is to use the energy and emissions generated during production for as long as possible or to recover them for new products. Especially with mineral building materials, this recovery also includes primary raw materials used in the production. When recycling concrete, the focus is on recovering the aggregates sand and gravel for further use. However, in contrast to concrete, AAC does not contain any aggregates. It is a homogeneous material. Therefore, no aggregates can be recovered in a potential pd-AAC recycling, and the extensively researched and proven concrete recycling processes are not applicable. However, the binder used in AAC production (cement and lime) has immense potential when closing the material loop. The production of binders is typically associated with very high GHG emissions. Global cement production causes around 1,500 Mt CO_2 Eq emissions annually (Andrew, 2019). Therefore, when recycling focuses on the binder, there is an exceptionally high potential for recovering significant emissions. This kind of recycling is thus auspicious in the particular situation of pd-AAC. However, this approach has hardly been researched. Therefore, assessing this recycling option's ecological potential plays a vital role in this dissertation and is mainly addressed in the context of a life cycle assessment (LCA) of cement clinker production from pd-AAC (Study D).

Hence, achieving sustainable handling of pd-AAC by closing the material loop with circular economy approaches is urgently needed and has enormous potential. The research project "REPOST - Autoclaved aerated concrete recycling cluster: Development of new options for circular economy", funded by the German Federal Ministry of Education and Research (BMBF), was initiated to address this issue. The project aims to find new high-quality recycling options and strategies for pd-AAC to close the material loop while providing ecologically and economically viable business models. The studies and contents of this dissertation were developed within the context of this research project.

Thus, this dissertation tries to answer the research question, "How can a circular economy for autoclaved aerated concrete be designed, and what quantitative, ecological, and economic potential does it have in Germany and Europe?". Researching this question requires consideration of different issues. First, knowledge of current and future pd-AAC amounts is crucial to quantifying the circular economy potential. Recent research has been undertaken to investigate new recycling options specifically for pd-AAC. However, an ecological assessment and comparison of these recycling approaches is necessary to examine if pd-AAC recycling has an ecological benefit and to identify the best recycling options from an environmental point of view. Economic considerations also play an essential role in the overall success of a circular economy for AAC. If sustainable pd-AAC handling is more expensive than landfilling, further actions like taxation of landfilling, a landfill ban, or subsidies for recycling are needed. Finally, an optimised recycling network, including the best possible facility locations, capacities, and logistics, has to be designed to advance the implementation of pd-AAC recycling.

This dissertation includes the following sections. First, the theoretical foundations of the related topics are given (Section 2). The research objectives are formulated in Section 3. Section 4 then includes the summaries of the companion studies. Finally, implications are presented (Section 5), and a conclusion is drawn (Section 6). Afterwards, the companion studies (Study A-F) are attached in Part II. Each study addresses one of the issues mentioned above to answer the overall research question of this dissertation.

2 Theoretical Foundation

This theoretical foundation section describes AAC's history, primary production process, and general characteristics (Section 2.1). Afterwards, a fundamental definition of the circular economy, including its different concepts, is given in Section 2.2. Finally, Section 2.3 discusses current circular economy approaches for AAC.

2.1 The building material AAC

AAC was invented in Sweden in 1923 (patented in 1924) by Axel Eriksson as a reaction to the energy shortage after the 1st World War, which led to stricter requirements for the thermal insulation properties of building materials (BV Porenbetonindustrie, 2018). Fundamental inventions and preparatory work on autoclaving and aerating of lime-sand/cement/gypsum mortars go back as far as the late 19th century (BV Porenbetonindustrie, 2018). Industrial production of AAC started in 1929 in Yxhult, Sweden, under the name “Yxhults Anghärdade Gasbetong”, which was later abbreviated to the brand name “Ytong” (Xella Group, 2021). To this day, “Ytong” is a well-known AAC brand and is sometimes used as a synonym for AAC. In Germany, AAC production and sales started to rise in the early 1950s (UBA, 2019), while the standardisation of AAC masonry blocks and building boards followed in 1959 (DIN 4165 and 4166:1959-10). Since then, AAC's popularity and production volumes have enormously increased.

Today, as statistics show, AAC is one of Germany's most used building materials (Destatis, 2022). In 2021, around 21,500 residential buildings with about 20 Mm³ gross volume and estimated costs of 6,600 M€ were built with ACC as the predominantly used building material. Concerning the number of constructed buildings, AAC has a share of around 21%, the second highest market share behind clay bricks. Based on the gross volume, clay bricks, sand-lime bricks, and reinforced concrete surpass AAC, as AAC is used primarily for one-family houses. 18,000 of the 21,500 residential buildings built with AAC are one-family houses, making up around 84%. AAC's primary production is spread across 31 plants in Germany (see Study F). Production volumes of approximately 3.6 Mm³ consisting of 3.3 Mm³ masonry units and 1.4 Mm² panels and floorboards were reached in Germany in 2021 (GENESIS, 2023). Worldwide, AAC production capacities are expected to equal 450 Mm³ in more than 3,000 AAC plants (Fouad & Schoch, 2018). The largest market is China, where AAC production is predicted to reach 230 Mm³ in 2025 (Aircrete Europe, 2022). Moreover, Russia is one of the largest AAC producers worldwide, with a production volume of 11.6 Mm³ in 2017 (Grinfel'd et al., 2018). The primary production process is illustrated in Figure 2.1.

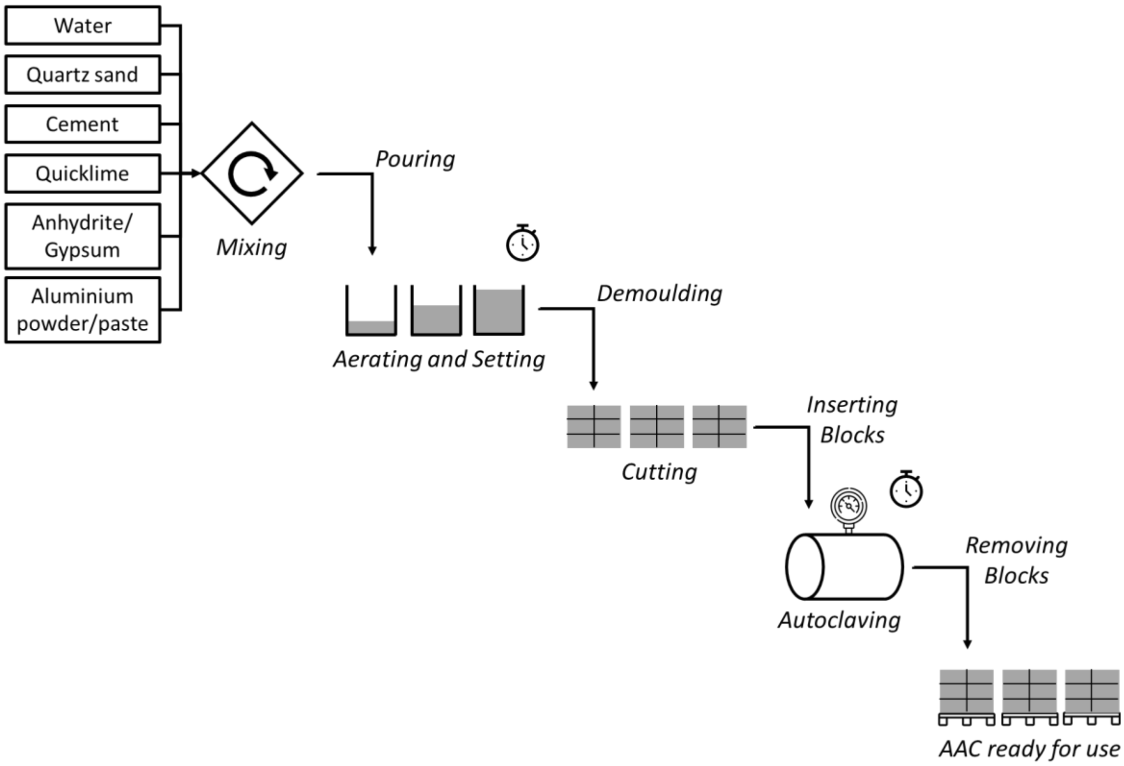

Figure 2.1: AAC's primary production process (based on BV Porenbetonindustrie, 2018; Hamad, 2014).

The process is described in BV Porenbetonindustrie (2018) and Hamad (2014) as follows. AAC is produced from water, quartz sand, cement, quicklime, anhydrite or gypsum, and a small amount of aluminium as a powder or paste. AAC production recipes vary, mainly depending on the density of the AAC that is produced. Exact recipes for three different AAC masonry blocks are presented in Study C. In general, it can be stated that quartz sand is the material with the most considerable input quantity (neglecting water). At the same time, anhydrite/gypsum is only required in smaller proportions and aluminium in tiny amounts.

First, the raw materials are mixed and poured into moulds. Here, two types of chemical reactions take place. First, the aluminium reacts with calcium hydroxide and water to produce calcium aluminate hydrate and hydrogen (Equation 2.1). The hydrogen causes an expansion of up to five times the original volume as numerous tiny pores are formed. Eventually, the hydrogen diffuses, so the pores are filled with air in the final product, which leads to the term "aerated" in AAC's name. The so-called setting reaction of the mixture is the second type of chemical reaction taking place (Equations 2.2-2.4). The reaction of water with calcium oxide (from the quicklime), tricalcium silicate (from the cement), or dicalcium silicate (from the cement) produces calcium hydroxide and tricalcium disilicate hydrate. As a result, the mixture is stable enough to form raw AAC blocks that can be removed from the mould after three to six hours. These blocks are then cut to the desired shape with steel wires. Finally, the blocks reach their ultimate strength in the last process step, the so-called autoclaving, which gives AAC another part of its name. Autoclaving is a steam pressure curing process with pressures of eight to eleven bar and temperatures of 170 to 200 °C, where the raw AAC blocks stay for six to ten hours. The blocks undergo a hardening reaction (Equations 2.5 and 2.6) in which tobermorite is formed from the silicon dioxide in the quartz sand and the products of the setting reaction (calcium hydroxide and tricalcium disilicate hydrate). Afterwards, the AAC is ready to be used as a building material.

$$2\,Al + Ca(OH)_2 + 6\,H_2O \rightarrow CaO \cdot Al_2O_3 \cdot 4\,H_2O + 3\,H_2 \tag{2.1}$$

$$CaO + H_2O \rightarrow Ca(OH)_2 \tag{2.2}$$

$$2\,(3\,CaO \cdot SiO_2) + 6\,H_2O \rightarrow 3\,Ca(OH)_2 + 3\,CaO \cdot 2\,SiO_2 \cdot 3\,H_2O \tag{2.3}$$

$$2\,(2\,CaO \cdot SiO_2) + 4\,H_2O \rightarrow Ca(OH)_2 + 3\,CaO \cdot 2\,SiO_2 \cdot 3\,H_2O \tag{2.4}$$

$$6\,SiO_2 + 5\,Ca(OH)_2 \rightarrow 5\,CaO \cdot 6\,SiO_2 \cdot 5\,H_2O \tag{2.5}$$

$$8\,SiO_2 + 5\,(3\,CaO \cdot 2\,SiO_2 \cdot 3\,H_2O) \rightarrow 3\,(5\,CaO \cdot 6\,SiO_2 \cdot 5\,H_2O) \tag{2.6}$$

AAC reaches compressive strengths of 2.5 to 10 N/mm² (DIN 20000-404:2018-04) while the density is between 305 and 1000 kg/m³ (DIN 20000-404:2018-04). Pores sized millimetres to nanometres comprise 60 to 85 vol.-% of AAC (Anders, 2018). Therefore, AAC is suitable as a building material with excellent thermal insulation properties. The rated value of thermal conductivity for AAC is between 0.11 and 0.29 W/m*K depending on the density (DIN 4108-4:2020-11), well below other monolithic mineral building materials like classical clay brick, calcium silicate units, or concrete. Modern AAC products even reach values as low as 0.07 W/m*K (Xella Deutschland GmbH, 2018). Thus, the operating energy demand for heating or cooling a building built with AAC is comparatively low. Additionally, AAC is a mono-material without any need for extra insulation and with large masonry units that can be handled due to the low density, leading to a faster and less expensive construction process. Finally, AAC is non-flammable and fulfils all fire protection requirements. All these characteristics contribute to AAC's high popularity.

Additionally, there are several standards for determining AAC's physical characteristics. These characteristics include basic physical properties like compressive strength (DIN EN 679:2005-09), bending tensile strength (DIN EN 1351:1997-02), dry bulk density (DIN EN 678:1994-02), moisture content (DIN EN 1353:1997-02), and shrinkage (DIN EN 680:2006-03). Moreover, specific building material characteristics like freeze-thaw resistance (DIN EN 15304:2010-06), creep deformations under compressive stress (DIN EN 1355:1997-02), static modulus of elasticity under compressive stress (DIN EN 1352:1997-02), and shear strength of jointings (DIN EN 1739:2007-07) are considered in the standards. Overall, AAC's characteristics are defined comprehensively.

Besides the standardisation of AAC's physical characteristics, there are standards for ecological assessment. In addition to a general framework and guidelines for LCA (DIN EN ISO 14040:2021-02; DIN EN ISO 14044:2021-02), which is applied in Studies C and D, there are standardised environmental product declarations (EPDs) (DIN EN 15804:2022-03). This EPD standard addresses construction products and provides a basis for their ecological assessment. The assessment is divided into modules A (production and construction phase), B (use phase), C (disposal phase), and D (benefits and burdens outside the system boundary). Moreover, the environmental impact categories to be analysed are specified within the standard to achieve comparability between different EPDs. Despite this helpful standardisation, the framework of DIN EN ISO 14040:2021-02 is used in this dissertation, as the focus is on comparing different end-of-life treatment options and not on preparing an EPD.

2.2 Circular economy

The circular economy is gaining an increasingly important status in times of high and steadily growing global demand for materials. Therefore, there is an extensive discussion, and many initiatives concerning the circular economy have emerged. The most important aspects at the international, European, and German levels are presented below.

At the international level, alongside agreements generally concerned with tackling climate change, such as the Paris Agreement, some global initiatives specifically address the circular economy. These include, for example, the United Nations Sustainable Development Goals (in particular Goal twelve, “responsible production and consumption”) (UN, 2023), and the United Nations Environment Programme resolution on “Innovative pathways to achieve sustainable consumption and production” (United Nations Environment Programme, 2019). Moreover, there are numerous multilateral initiatives concerning the circular economy like the “10 Year Framework of Programmes on Sustainable Consumption and Production Patterns”, the “Partnership for Action on Green Economy”, and the “Platform for Accelerating the Circular Economy” (European Commission, 2020b).

Besides these initiatives, several international standards influence and foster the circular economy. First, the ISO 14000 series deals with environmental management and thus provides essential guidelines for the circular economy. The series includes the requirements for environmental management systems (ISO 14001:2015-09), principles and procedures for environmental labels (ISO 14021:2016-03; ISO 14024:2018-02; ISO 14025:2006-07), a framework and requirements for LCA (DIN EN ISO 14040:2021-02; DIN EN ISO 14044:2021-02), and a standard for integrating environmental aspects into product design and development (ISO 14062:2002-11). Besides, there are a framework and principles for methodologies on climate actions (ISO 14080:2018-06), which also includes the circular economy. Moreover, international standards such as guidelines for integrating sustainability (including circular economy principles) in procurement (ISO 20400:2017-04) influence the circular economy. In the specific area of the circular economy of building materials, the general sustainability principles in buildings and civil engineering works (ISO 15392:2019-12), principles and requirements for a design for disassembly and adaptability of buildings (ISO 20887:2020-01), and core rules for environmental product declarations of construction products (ISO 21930:2017-07) are standardised.

At the European level, the European Green Deal is a fundamental initiative. The aim is the transformation into a resource-efficient economy that produces no net greenhouse gas emissions by 2050 to meet the challenges associated with climate change (European Commission, 2019). A vital element of the European Green Deal is “mobilising [the] industry for a clean and circular economy” (European Commission, 2019). Therefore, the circular economy plays a central role in achieving the objectives of the European Green Deal. Consequently, the EU has introduced various strategies and initiatives intended to specifically contribute to fostering the circular economy. In addition to product-specific initiatives (e.g. on plastics, textiles and chemicals), these include the Circular Economy Action Plan.

The Circular Economy Action Plan described by the European Commission (2020a) aims to use circular economy approaches to achieve a transformation in line with the European Green Deal. Central components are the reduction of waste and the establishment of a European market for high-quality secondary materials. Sustainable business models and products should become the norm. The focus is on critical principles of the circular economy, such as durability, reusability, and reparability of products, remanufacturing, and high-quality recycling. Besides the Circular Economy Action Plan, ambitious circular economy targets were set by amending, among others, the Waste Framework Directive and the Landfill Directive as part of the Circular Economy Package (European Parliamentary Research Service, 2017).

Efforts to foster the circular economy at the European level also cover the specific area of construction products. The construction sector is mentioned in the Circular Economy Action Plan as a key product value chain for which a new strategy is to be launched to enhance circularity (European Commission, 2020a). Part of this new strategy is, among other things, the use of Level(s), an „assessment and reporting framework […] for sustainability performance of buildings” to support lifecycle thinking (Directorate-General for Environment, 2021). Moreover, the revision of the Construction Products Regulation (Regulation (EU) No 305/2011, 2011) is an essential element of the new strategy. In 2022, a proposal was formulated for this desired revision. Besides other aspects, the proposal addresses the circular economy of construction products. According to Article 22 of the proposal, product manufacturers must ensure sufficient durability of the products, enable easy repair or refurbishment, consider requirements for a minimum proportion of recycled material and favour recycled material, and design a product in such a way that reuse, remanufacturing, and recycling are easy to realise (European Commission, 2022).

In recent years, extensive initiatives and legislative proposals to foster the circular economy have been realised at the European level. Additionally, there are similar endeavours at the national level in Germany. The national circular economy strategy is currently being developed based on the German sustainability strategy (Bundesregierung, 2020) and the United Nations Sustainable Development Goals (UN, 2023). This circular economy strategy aims to improve environmental and climate protection, secure the supply of raw materials and prosperity, promote social justice and avoid hazardous materials by fostering the circular economy (BMUV, 2023). Moreover, the Waste Management Act (KrWG/02.03.2023) significantly influences the circular economy in Germany. The Act transposes the EU Waste Framework Directive (Directive 2008/98/EC, 2008) into national law. It intends to foster the circular economy and to protect people and the environment when generating and managing waste.

Furthermore, there is the Standardisation Roadmap Circular Economy. The roadmap analyses the current challenges in the area of the circular economy and identifies the standards required to deal with these challenges. The aim is to define a framework that simplifies launching circular services and products, thus fostering the ongoing transformation into a circular economy (DIN et al., 2023). The roadmap addresses crucial topics similar to the European Circular Economy Action Plan and investigates the construction sector in more detail. Regarding building materials, it is stated that „sustainable solutions not only for reducing the need for resources, but also for closed material cycles, are […] increasingly coming into focus" (DIN et al., 2023). This aspect of closed material cycles is also vital in this dissertation concerning AAC.

In addition to identifying central issues and initiatives concerning the circular economy, a definition of the circular economy is an essential theoretical foundation for this dissertation. There are numerous different approaches. Thus, Kirchherr et al. (2017) have extensively analysed the scientific literature on circular economy definitions to identify a definition that includes as many of the aspects found in the literature as possible. They considered 114 definitions to derive the following one:

> *"A circular economy describes an economic system that is based on business models which replace the 'end-of-life' concept with reducing, alternatively reusing, recycling and recovering materials in production/distribution and consumption processes, thus operating at the micro level (products, companies, consumers), meso level (eco-industrial parks) and macro level (city, region, nation and beyond), with the aim to accomplish sustainable development, which implies creating environmental quality, economic prosperity and social equity, to the benefit of current and future generations."*

Based on this definition, the circular economy can be seen as a strategy focussing on the entire 'economic system'. The aim is to achieve 'sustainable development' for this system. Thus, the circular economy is a general principle considering different levels ('micro level', 'meso level', 'macro level'). This general principle of the circular economy needs to be applied to specific products and related production and processing steps, such as the AAC. This application is achieved by determining the quantitative, ecological, and economic potentials of AAC's circular management and, finally, identifying an optimal design for an AAC circular system in this dissertation. This research improves AAC's circularity and its conformity with the characteristics of a circular economy mentioned in the definition: the concept that the AAC reaches an end of life and is to be disposed of after its use phase is replaced by multiple ways of benefitting from the pd-AAC to achieve economic and environmental advantages. Therefore, this dissertation can make an essential contribution to the circular economy.

to the definition, the crucial point is to replace the end-of-life concept with the alternative approaches of 'reducing', 'reusing', 'recycling' and 'recovering materials'. In this way, the circular economy differentiates itself from the linear economy. Ekins et al. (2019) describe the circular economy's goal similarly. Environmental impacts from raw material extraction, production, use, and end-of-life for all kinds of products should be decreased by reducing the disposal and improving the use of resources. Hirsch and Schempp (2020) extend the alternatives mentioned above to replace the end-of-life concept to a total of nine alternatives, which are well-known as the so-called 9 Rs:

- Refuse (R1): "Make product redundant by abandoning its function or by offering the same function by a radically different (e.g. digital) product or service."
- Rethink (R2): "Make product use more intensive (e.g. through product-as-a-service, re-use and sharing models or by putting multi-functional products on the market)."
- Reduce (R3): "Increase efficiency in product manufacture or use by consuming fewer natural resources and materials."
- Re-use (R4): "Re-use of a product which is still in good condition and fulfils its original function (and is not waste) for the same purpose for which it was conceived."
- Repair (R5): "Repair and maintenance of defective product so it can be used with its original function."
- Refurbish (R6): "Restore an old product and bring it up to date (to specified quality level)."
- Remanufacture (R7): "Use parts of a discarded product in a new product with the same function (and as-new-condition)."
- Repurpose (R8): "Use a redundant product or its parts in a new product with different function."
- Recycle (R9): "Recover materials from waste to be reprocessed into new products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material but does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations."

The 9 Rs can be divided into the categories "smarter product use and manufacture" (R1-R3), "extend lifespan of product and its parts" (R4-R8), and "useful application of materials" (R9) (Potting et al., 2017). Sometimes, recovery (as mentioned in the circular economy definition) is added to this list (Potting et al., 2017). However, recovery is often seen critically as raw material savings are significantly lower than for R1-R9 (Hirsch & Schempp, 2020). Therefore, recovery will not be examined further as an alternative to end-of-life. Specific ways in which the 9 Rs can be applied to improve the circularity of AAC are described in Section 2.3. That section also outlines the focused approaches of this dissertation.

Generally, lower-number strategies should be preferred, as they typically require fewer natural resources and have a lower environmental impact (Potting et al., 2017). This preference is also reflected in the European waste hierarchy, which classifies the following five aspects (Directive 2008/98/EC, 2008):

1. Prevention
2. Preparing for re-use
3. Recycling
4. Other recovery, e.g. energy recovery
5. Disposal

‘Prevention’, which is the most favourable aspect of the waste hierarchy, can be achieved by ‘refuse’ (R1), ‘rethink’ (R2), and ‘reduce’ (R3). The second aspect of ‘preparing for re-use’ comprises the ‘re-use’ (R4) itself and strategies of reusing after some preparational work, namely ‘repair’ (R5), ‘refurbish’ (R6), ‘remanufacture’ (R7), and ‘repurpose’ (R8). Furthermore, the waste hierarchy directly mentions ‘recycling’ (R9). Thus, all 9 R strategies are part of the European waste hierarchy. Besides these circular economy aspects and recovery, disposal is mentioned as the last opportunity to deal with waste. However, disposal reflects the classical end-of-life concept in a linear economy and does not lead to a sustainable use of resources, so all other options should be preferred.

2.3 Circular economy of AAC

This section discusses the status quo of AAC’s circular economy. Figure 2.2 overviews the 9 Rs and the European waste hierarchy aspects in AAC’s life cycle.

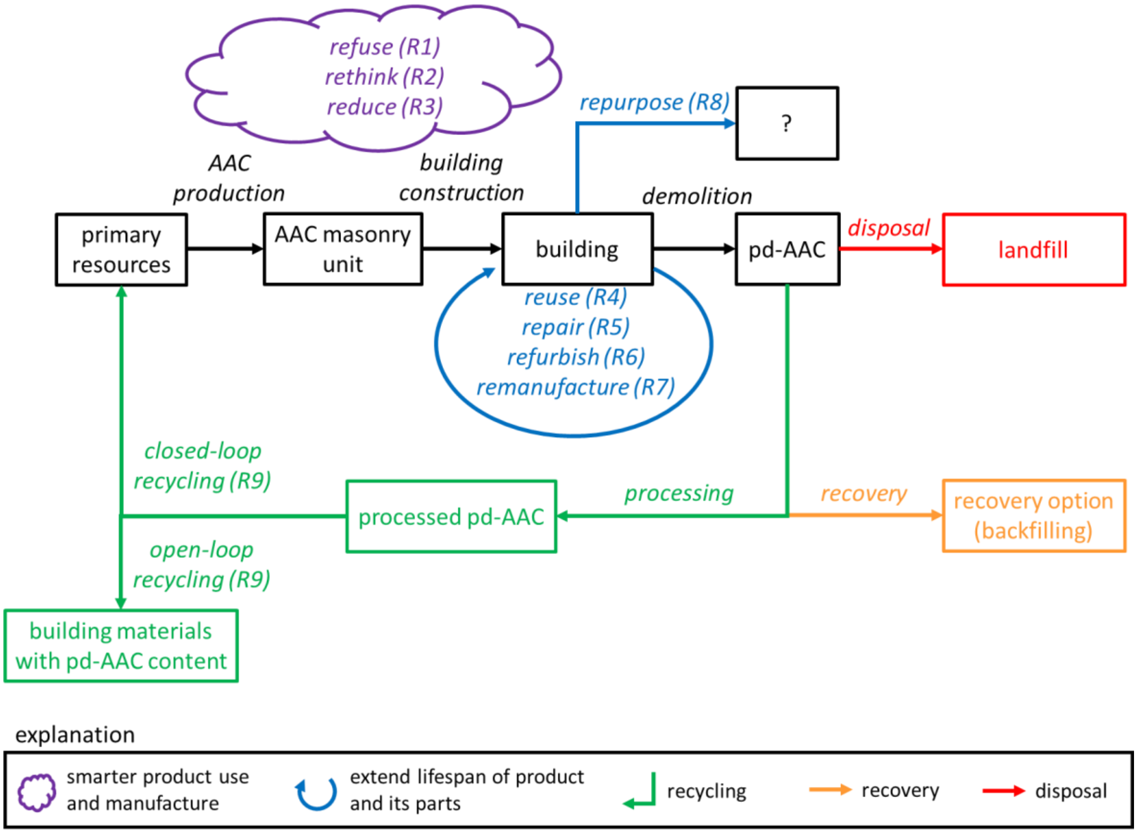

Figure 2.2: Overview of AAC's circular economy strategies, including the categories "smarter product use and manufacture", "extend lifespan of product and its parts", and recycling, as well as the non-circular treatment options recovery and disposal.

2.3.1 Circular economy strategies R1-R8 for AAC

First, the circular economy strategies of the category "smarter product use and manufacture" (R1-R3) are discussed. Smarter product use and manufacture can significantly reduce the environmental impacts of AAC (and the whole construction sector). However, fundamental changes like new living and housing concepts are necessary for this approach. Progress on these aspects requires general research and is not specific to building materials. Therefore, strategies R1-R3 will not be considered further in the context of this dissertation.

Extending the lifespan of a product and its parts (R4-R8) is another option to foster a circular economy. On the one hand, the lifespan of the entire building can be extended. However, since AAC as a building material is the focus of this dissertation, this possibility will not be investigated further. On the other hand, the lifespan of the AAC itself can also be extended, for example, by reusing AAC masonry units. Such a reuse would require separating the individual AAC masonry units from each other during a (partial) deconstruction process of a building. These units could then be used to construct a new building. Thus, no costly and energy-intensive processing of the pd-AAC would be necessary, and the AAC material cycle could be closed. However, prerequisites for reuse are a recovery of the AAC masonry units without destruction during the deconstruction process and a suitable new application. Both prerequisites are challenging to fulfil. First, a meticulous selective dismantling process would be necessary to receive reusable AAC masonry units. Such a deconstruction would significantly increase the costs and is therefore not viable (Gyurkó et al., 2019). In addition, AAC masonry units produced many decades ago no longer meet current regulations, e.g. due to insufficient insulation properties. Therefore, there is no present possibility of extending the lifespan of AAC masonry units (besides extending the lifespan of the whole building) without changing the purpose (R4-R7). The repurpose (R8) of AAC masonry units is still possible as insulation properties may not be necessary for other purposes. However, there are no typical applications for repurposed AAC yet. Further research is needed to identify them.

Therefore, strategies R4-R8 will also not be investigated further. This dissertation focuses on recycling (R9) of pd-AAC, which is examined in more detail in the following sections. Therefore, this dissertation may use recycling synonymously with a circular economy of AAC.

2.3.2 Current use of pd-AAC

Recycling pd-AAC would involve processing and preparing the pd-AAC for various new applications. Clean and homogenous AAC production waste (e.g. offcuts or broken material) is already recycled in AAC production today (BV Porenbetonindustrie, 2018; Dehn et al., 2003). However, the much more difficult-to-process pd-AAC, where various impurities and contaminants can occur (Deilmann et al., 2014), is not yet recycled. For example, mortar, wallpaper or tiles may adhere to the pd-AAC, or there can be a general mixture with other demolition materials. The most common recycling options for building materials include an application in the base course of road construction, in earthworks, and as aggregate in concrete production (Knappe et al., 2012). However, AAC has a comparatively low compressive strength, which impedes recycling as aggregate in concrete production or road construction. Additionally, pd-AAC contains small amounts of sulphate from the gypsum or anhydrite, which does not allow recycling in earthworks due to potential groundwater contamination (Knappe et al., 2012). Overall, the usual recycling options for mineral construction wastes are not applicable for pd-AAC, and new applications must be found.

So far, a circular economy for pd-AAC has not yet been established. Therefore, more than half the pd-AAC is disposed of (i.e. landfilled), while recovery has a one-third share (Figure 2.3). However, recovery mainly implies backfilling the pd-AAC, which does not represent a high-quality use of resources. Recycling currently accounts for only a negligible share.

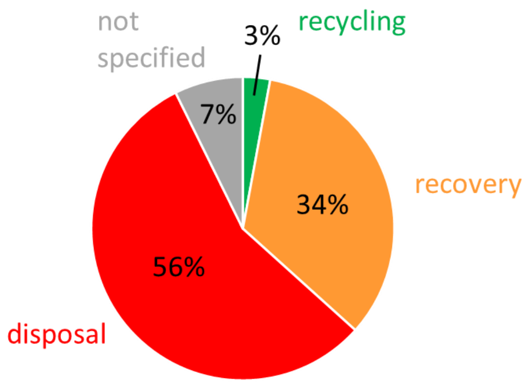

Figure 2.3: Disposal, recovery, and recycling rates for pd-AAC in Germany (Bauhaus University Weimar, 2010; UBA, 2019).

Several research efforts are trying to develop new recycling options for pd-AAC (Table 2.1). It is necessary to clean the pd-AAC before recycling so that as few impurities as possible hinder the recycling process. Furthermore, crushing and grading are required, as the different grain sizes of the pd-AAC strongly influence its properties. In the following, a distinction is made between pd-AAC powder (grain size 0-1 mm) and pd-AAC granulate (grain size > 1 mm). The allocation of pd-AAC powder and pd-AAC granulate during crushing depends on various properties, such as crushing technology (Krampitz et al., 2022) and moisture content of the pd-AAC. In general, it is reasonable to expect pd-AAC powder and pd-AAC granulate to be produced in a ratio of about 3:1 due to the low strength of AAC (Gyurkó et al., 2019). Unlike the pd-AAC powder, the grain size of the pd-AAC granulate is sufficiently large to preserve the porous structure. Consequently, the pd-AAC granulate resembles pumice or expanded clay, whereas the pd-AAC powder is sand-like. Accordingly, possible applications of pd-AAC powder and pd-AAC granulate are different.

2.3.3 Closed-loop recycling of pd-AAC

The closed-loop recycling options for pd-AAC are described in this section. On the one hand, recycling pd-AAC powder in producing new AAC is possible (Kreft, 2017). Fine pd-AAC powder is needed for this recycling option to achieve a good intermixture with the other input materials. Shares of the primary raw materials sand, quicklime, cement, and anhydrite can be substituted by the pd-AAC powder. Primary quicklime and cement production are associated with high efforts and GHG emissions. Thus, the savings potential by using pd-AAC in AAC production is very high. However, the input amount of pd-AAC is limited as normative requirements, for example, the compressive strength of AAC, can no longer be achieved if the proportion of pd-AAC is too high. This recycling process is much simpler but also has less savings potential if only primary sand is replaced and clean AAC production waste without impurities is used, as in the studies of Lam (2021), Rafiza et al. (2019), and Rafiza et al. (2022).

On the other hand, the pd-AAC can also be used as input for cement production. Schoon et al. (2013) investigate the production of Portland cement, which is the most common type of cement, using pd-AAC. However, they find that although substitution of primary inputs with pd-AAC would be possible, no benefits are expected, and recycling is not practical due to additional efforts for drying and grinding the pd-AAC and potential impurities. However, other cement types could be more suitable for pd-AAC recycling. The pd-AAC can also be processed into recycled belite cement clinker (RC-BCC). Belite (dicalcium silicate, $2CaO \cdot SiO_2$) is a well-known cement clinker phase and is usually found alongside alite (tricalcium silicate, $3CaO \cdot SiO_2$) and other minor clinker phases in ordinary Portland cement (DIN EN 197-1:2011-11). However, the proportion of belite (15-30%) in ordinary Portland cement is lower than that of alite (50-70%) (Chatterjee, 1996). In addition, the strength of the belite is mainly formed in the long term and relatively low in the first few days in contrast to alite, so different approaches for activating the belite have already been researched (Chatterjee, 1996; Chen et al., 2017; Ullrich et al., 2021). Therefore, special applications have to be identified for the RC-BCC.

The advantages of producing RC-BCC from pd-AAC are that there is less calcium needed compared to alite, which saves 12% of the process-related CO_2 emissions, and crystallisation takes place faster and at lower temperatures leading to energy savings for providing the process heat (Ullrich et al., 2021). The process temperature can be reduced from 1450 °C to 1000 °C (Ullrich et al., 2021). RC-BCC produced from pd-AAC can generally substitute Portland cement in different applications. One conceivable application of RC-BCC is the production of new AAC, thus achieving closed-loop recycling. Up to 50% of the required primary Portland cement can be substituted by RC-BCC (see Study D). Due to the high energy demand of RC-BCC production, the direct recycling of pd-AAC powder in AAC production could be preferable to the “indirect route” via RC-BCC. However, RC-BCC production is a critical complement to increase the degree of substitution and deal with difficult-to-separate impurities. Study D extensively researches the recycling of pd-AAC in RC-BCC production used for new AAC production. Furthermore, there are other possible applications for RC-BCC. For example, the lower hydration energy of belite has proven to be a significant advantage in constructing massive concrete structures such as the Three Gorges Dam in China (Sui et al., 2015).

2.3.4 Open-loop recycling of pd-AAC in the construction sector

In addition to closed-loop recycling, various options for open-loop recycling of pd-AAC in the construction sector are also being researched. First, there is research on using pd-AAC in floor screed, described by Bergmans et al. (2016). Pd-AAC of grain sizes up to 8 mm is combined with other C&DW and bound with cement. Sulphate from the pd-AAC is bound in ettringite in this recycling option, significantly reducing leaching risk. Therefore, using pd-AAC in floor screed opens up a reasonable recycling option with the possibility of using large quantities of pd-AAC. However, compared to the closed-loop recycling options listed above, only sand from the conventionally produced floor screed is substituted; emission-intensive raw materials such as quicklime or cement are not. Other open-loop recycling options include the production of light mortar, lightweight aggregate concrete (LWAC) and shuttering blocks made from concrete without fine fractions.

The pd-AAC recycling in light mortar is researched by Aycil et al. (2016). Light mortar is a mixture of cement, water, and natural aggregates, which can be replaced by pd-AAC. A mortar with pd-AAC content has worse mechanical properties than other mortars. However, the bulk density and the thermal conductivity are significantly reduced. The intended application areas for the light mortar are mainly in the interior of buildings, where the light mortar could be used as a filling, masonry or plastering mortar. The light mortar production using pd-AAC has already been tested on an industrial scale by Aycil et al. (2016). Therefore, this recycling option could soon process significant quantities of pd-AAC. Zou et al. (2022) also investigate pd-AAC recycling in mortar. However, they study the case where pd-AAC is ground more finely. Still, this recycling option is reasonable as pd-AAC can partially substitute sand in the mortar.

In addition to the production of light mortar, Aycil et al. (2016) describe the production of LWAC using pd-AAC as a lightweight aggregate. LWAC is a commonly used building material, so this recycling option has a high substitution potential. The raw mixture of cement, water, coal fly ash and pd-AAC is first compacted in a vibro-pressing process. The resulting blocks have densities and compressive strengths that tend to be at the lower end of the range for LWAC made from primary materials. Therefore, the current focus of use is on non-load-bearing interior walls. A first run in industrial production has already been carried out for this recycling option. Gyurkó et al. (2019) also propose the production of non-load-bearing and load-bearing LWAC using aggregates from pd-AAC. Input materials are the same as those previously mentioned by Aycil et al. (2016), although, for load-bearing LWAC, the proposed cement dosage is higher than for non-load-bearing. Load-bearing LWAC can be used as masonry units or prefabricated walls. Finally, Murthi et al. (2022) also state that pd-AAC recycling in LWAC production is practical. However, they only investigate 10 to 60% substitution of the primary lightweight aggregates, while the other studies do not use primary lightweight aggregates at all.

Furthermore, pd-AAC granulate can be used as aggregate in no-fines concrete, i.e. concrete without fine fraction (Gyurkó et al., 2019). The critical advantage of no-fines concrete is that only a tiny amount of cement is needed, which allows cost-effective and comparatively low-emission production. Possible applications for no-fines concrete are shuttering blocks and stumped concrete. Stumped concrete can be used for non-load-bearing walls or decorative facades and fences. Shuttering blocks are produced in a vibration press and can be used as non-load-bearing blocks or filled with regular concrete to reach a higher strength. The shuttering blocks made from no-fines concrete with pd-AAC granulate have an advantageous thermal conductivity of only one-sixth of standard concrete.

Finally, Gyurkó et al. (2019) propose using pd-AAC powder as supplementary material in concrete. In this case, pd-AAC powder of minimal grain size (diameter < 0.09 mm) is added to the regular concrete mixture to improve the performance of the concrete. On the one hand, the compressive strength is enhanced as the tiny particles of the pd-AAC powder reduce the porosity of the concrete. On the other hand, the concrete’s durability, i.e. compressive strength reduction after numerous freezing cycles, is also enhanced when pd-AAC powder is added. Overall, using supplementary material in concrete production represents a viable recycling option for pd-AAC powder, which is generally more difficult to recycle than pd-AAC granulate. He et al. (2020) and Qin and Gao (2019) describe using pd-AAC as a supplementary cementitious material. Here, the pd-AAC is ground very finely and mixed with Portland cement after production (not used as an input like in the RC-BCC production). Ecological advantages of the approach result from a reduced use of Portland cement. However, in contrast to using pd-AAC as supplementary material in concrete (Gyurkó et al., 2019), it is impossible to precisely quantify the substitution due to the unspecified use of pd-AAC-supplemented cement in the studies. Thus, this generalised recycling approach is not included in the ecological comparison.

2.3.5 Other recycling options for pd-AAC

Finally, there are other suggested uses for pd-AAC outside the construction industry. First, current recycling options for AAC production waste include its use as an oil and chemical binder and as animal bedding. These recycling options could also be applied to pd-AAC if sufficiently purified. Moreover, pd-AAC can improve soil quality as a fertiliser (Niedersen et al., 2004; J. Volk & Schirmer, 2010). The advantages of pd-AAC for this application are its high water absorption capacity and the significant proportion of alkaline components (Niedersen et al., 2004). Landscaping is also conceivable for similar reasons (Rühle & Maiwald, 2018). Furthermore, pd-AAC can be a filter material for phosphate-containing wastewater (Renman & Renman, 2012). Research into this approach shows that it is possible to filter phosphate from wastewater, but long retention times of 24 h are necessary (Renman & Renman, 2012). Moreover, Bukowski et al. (2015) investigate the possibility of using pd-AAC granulate as a basis for bacteria life. Bukowski et al. (2015) suggest covering landfills with pd-AAC granulate containing methane-oxidising bacteria. Consequently, methane emissions from the landfill are reduced, which is advantageous due to the high greenhouse potential of methane.

However, these recycling options outside the construction industry are mostly cascade recycling cases. The pd-AAC is used for one new application, but further recycling is hardly possible. Moreover, ecological and economic considerations are limited since directly comparable primary products barely exist. Therefore, this dissertation does not consider recycling options outside the construction industry in detail. The focus will be on the closed- and open-loop recycling options in the construction industry. Table 2.1 summarises all presented recycling options.

Table 2.1: Overview of pd-AAC recycling options in the literature, including advantages and disadvantages.

recycling option	reference	advantages	disadvantages	examined in this dissertation
AAC (substituting all inputs)	Kreft (2017)	closed-loop, substituting high-impact primary materials	limited substitution rate	yes
AAC (substituting sand inputs)	Lam (2021), Rafiza et al. (2019), Rafiza et al. (2022)	closed-loop, easier process	not substituting high-impact primary materials	yes
Portland cement	Schoon et al. (2013)	potential closed-loop	not practical	no
RC-BCC	Ullrich et al. (2021), Study D	potential closed-loop, less calcium and lower temperature than Portland cement production	special applications needed (slow strength development)	yes
floor screed	Bergmans et al. (2016)	high substitution potential	not substituting high-impact primary materials	yes
light mortar	Aycil et al. (2016), Zou et al. (2022)	bulk density and the thermal conductivity of the mortar are reduced	application only in the interior of buildings	yes
LWAC	Aycil et al. (2016), Gyurkó et al. (2019), Murthi et al. (2022)	high substitution potential	only non-load-bearing elements	yes
no-fines concrete	Gyurkó et al. (2019)	low amounts of cement needed	only non-load-bearing elements	yes
supplementary material in concrete	Gyurkó et al. (2019)	improved concrete performance	minor pd-AAC inputs	yes
supplementary cementitious material	He et al. (2020), Qin and Gao (2019)	reduced use of Portland cement	unspecified use	no
fertiliser	Niedersen et al. (2004), J. Volk and Schirmer (2010)	helpful characteristics of the pd-AAC	presumably cascade recycling	no
landscaping	Rühle and Maiwald (2018)	helpful characteristics of the pd-AAC	presumably cascade recycling	no
filter (phosphatic wastewater)	Renman and Renman (2012)	helpful characteristics of the pd-AAC	presumably cascade recycling	no
landfill cover	Bukowski et al. (2015)	reduced methane emissions through bacteria	presumably cascade recycling	no

3 Research Fields and Research Objectives

Pd-AAC is currently mainly disposed of, as shown in Section 2. A pd-AAC recycling has not yet been established due to impurities and contaminants, small amounts of sulphate in the pd-AAC, and a comparatively low compressive strength of pd-AAC. However, this is a problem with environmental and economic consequences, as described in the following.

First, disposing of C&DW is not a sustainable use of material. Disposal means that materials are removed from the material cycle, and new primary raw materials must be mined. In 2012 alone, the global demand for sand and gravel was estimated at 25 to 30 Gt (Peduzzi, 2014). However, excessive mining of primary mineral materials has significant environmental consequences. These include land loss, landscape changes, impacts on water supply, climate impacts (especially from transport), loss of biodiversity, and reduced protection from extreme events (Peduzzi, 2014).

Second, primary building material production is associated with considerable GHG emissions. The total embodied emissions in the global building sector sum up to 2,200 Mt CO_2-Eq annually (IPCC, 2022). These embodied emissions account for nearly 4% of the total global GHG emissions of 59,000 Mt CO_2-Eq in 2019 (IPCC, 2022). In Europe, embodied emissions in the building sector reached around 230 Mt CO_2-Eq, while total GHG emissions were approximately 4,800 Mt CO_2-Eq in 2019 (IPCC, 2022). Most embodied emissions are due to cement, whose global production causes about 1,500 Mt CO_2-Eq annually (Andrew, 2019).

Third, the number of available landfills is decreasing in Germany. It has halved in the past 20 years, from 1,999 landfills in 2004 to 999 landfills in 2021 (Destatis, 2019b, 2023). This trend will continue and probably even worsen, as more than half of the active landfills currently have a remaining operating life of fewer than ten years (Destatis, 2023). Landfilled C&DW reached 23 Mt in Germany in 2021, accounting for 58% of the total landfill input (Destatis, 2023) and, thus, contributes decisively to this ongoing shortage. However, such a reduction in landfill capacity inevitably leads to rising disposal prices. Therefore, there are also economic drawbacks of pd-AAC landfilling besides the ecological disadvantages described above.

A circular economy uses existing waste, which can reduce the described problems. The mining and use of primary raw materials can be reduced by substituting them with secondary materials, mitigating the problems caused by the mining. Moreover, the substitution has enormous potential for saving GHG emissions, particularly when primary cement consumption can be reduced. Additionally, a circular economy with avoided disposal of C&DW can conserve the decreasing landfill capacities, save landfilling costs, and leave the landfills available for hazardous waste.

Therefore, this dissertation focuses on the establishment of a circular economy of AAC. The research question of the dissertation is formulated as follows:

> *How can a circular economy for autoclaved aerated concrete be designed, and what quantitative, ecological, and economic potential does it have in Germany and Europe?*

Four research fields are considered to answer this research question comprehensively:

1. The quantification of the regional pd-AAC volumes in Germany and Europe (Section 3.1)
2. The ecological and economic assessment of pd-AAC recycling (Section 3.2)
3. The assessment of the RC-BCC production as a new and innovative recycling option for pd-AAC (Section 3.3)
4. The optimal design of a pd-AAC recycling network (Section 3.4)

A brief overview of the current state of the literature and the targeted research objectives are introduced for every research field. More extensive literature reviews can be found in the respective studies in Part II. Section 3.5 classifies the individual studies' contribution to the research fields. Implications from the research in these fields are presented in Section 5.

3.1 Quantification of the regional pd-AAC volumes in Germany and Europe

The first research field to approach the research question is quantifying the pd-AAC circular economy's potential by determining the pd-AAC quantities that can be expected currently and in the future in Germany and Europe. An assessment of the recycling potential in terms of quantity is of great importance as sufficiently high pd-AAC amounts are the basis to consider pd-AAC recycling worthwhile from an organisational, economic, and ecological point of view. Therefore, analysing this research field is a prerequisite for working on the following research fields.

There are numerous studies on material flow and stock analyses in the built environment. Lanau et al. (2019) and Augiseau and Barles (2017) extensively overview many studies. Furthermore, Deilmann et al. (2014), Ortlepp et al. (2016), Schiller et al. (2010), and Schiller et al. (2015) investigate the German anthropogenic stockpile. However, AAC is not considered in most studies or is only investigated negligibly. There is no study entirely focussing on AAC. Additionally, waste statistics in Germany and Europe specify some building materials like concrete, clay bricks, timber, and glass but do not reveal pd-AAC amounts as there is no unique waste code for pd-AAC (Destatis, 2019a). Thus, calculating current and past pd-AAC amounts is the first research objective of this research field.

Moreover, AAC is still a comparatively young building material that has only been known in Germany since about 1950. Therefore, future volumes may differ significantly from current and past volumes. Deilmann et al. (2014) estimate future pd-AAC volumes in Germany, but they only consider two points in time, 2030 and 2050, and do not disclose a reproducible calculation methodology. Therefore, another research objective is to provide a year-by-year prediction of future pd-AAC volumes.

Since post-demolition building materials are low-value products with a high weight, the material's regional distribution is also vital. However, current literature remains at the national level; only Schiller et al. (2010) attempt to distribute their assessed building material stock regionally. Thus, another research objective is to address this shortcoming by creating a detailed regional subdivision of pd-AAC volumes in Germany.

3.2 Ecological and economic assessment of pd-AAC recycling

In contrast to landfilling or backfilling, recycling processes are associated with some ecological effort for sorting and processing. Therefore, the environmental advantage of different pd-AAC recycling options over landfilling must be verified and quantified as part of the second research field. There are various methodological standards and principles for an ecological assessment in the literature, including standards for EPDs (DIN EN 15804:2022-03), LCAs (DIN EN ISO 14040:2021-02; DIN EN ISO 14044:2021-02), and extensive LCA databases (ecoinvent, 2021). Furthermore, Nakatani (2014) examines approaches for life cycle inventory analysis (as a central part of the LCA) suitable for assessing recycling processes.

Previous applications of these principles in the literature have focused on the LCA of primary building materials (Christoforou et al., 2016; Jonsson et al., 1998; Mitterpach & Štefko, 2016; Zimele et al., 2019) or the LCA of using secondary materials in the production of building materials (Ahmed & Tsavdaridis, 2018; Bories et al., 2016; Colangelo et al., 2018; Knoeri et al., 2013). Rahman et al. (2021) and Nühlen et al. (2020) even focus on improving the sustainability of AAC by reviewing, investigating and assessing the production of AAC with the use of industrial by-products and construction waste. However, these studies do not assess pd-AAC recycling or compare pd-AAC recycling options. In addition, pd-AAC is not examined in these studies since by-products from other industrial processes and general construction waste are considered. Thus, the process does not represent closed-loop recycling. Overall, the literature lacks an ecological assessment of high-quality pd-AAC recycling options, which is an addressed research objective of this dissertation. Moreover, the total environmental savings potential of establishing pd-AAC recycling will be calculated.

The economic perspective is also vital, even if pd-AAC recycling is ecologically advantageous. A financial advantage or incentive is essential for the pd-AAC recycling to be successful in a competitive market environment and to establish itself in the long term. There are already extensive methodological standards in the literature for estimating the costs of industrial plants (Humphreys, 2005; Peters et al., 2003; Smith, 2005). In addition, numerous studies have been carried out to assess the economics of a variety of recycling processes (Athanassiou & Zabaniotou, 2008; Cimpan et al., 2016; Cucchiella et al., 2015; Fivga & Dimitriou, 2018; Granata et al., 2022; Hassanpour, 2021; Larrain et al., 2021). However, an economic assessment of pd-AAC recycling does not yet exist. Thus, determining the costs of pd-AAC recycling and comparing it to the costs of disposal is another crucial research objective. Additionally, the economic assessment allows the calculation of the economic savings potential of pd-AAC recycling or whether there is a need for subsidies.

3.3 Assessment of the RC-BCC production as a new and innovative recycling option for pd-AAC

The recycling of pd-AAC has not yet been established because the usual recycling options for mineral C&DW are unsuitable for pd-AAC. Recent research focuses on developing new recycling options designed explicitly for pd-AAC, which are still relatively rare. In particular, suggestions for new open-loop recycling options for pd-AAC within and outside the construction industry exist in the literature (Aycil et al., 2016; Bergmans et al., 2016; Bukowski et al., 2015; Gyurkó et al., 2019; He et al., 2020; Murthi et al., 2022; Niedersen et al., 2004; Qin & Gao, 2019; Renman & Renman, 2012; Rühle & Maiwald, 2018; J. Volk & Schirmer, 2010; Zou et al., 2022).

However, it can be assumed that only one further use of the pd-AAC takes place before it has to be disposed of (so-called cascade use), as open-loop recycling does not imply a closed material loop. There are already higher-quality and promising approaches in the literature for the closed-loop recycling of pd-AAC in producing new AAC (Kreft, 2017; Lam, 2021; Rafiza et al., 2019; Rafiza et al., 2022). However, the amount of pd-AAC used in these recycling processes is limited. Therefore, finding new and high-quality recycling options for pd-AAC is essential, mainly due to increasing pd-AAC volumes. Additionally, because of the comparatively low strength of pd-AAC, a high proportion of fine pd-AAC powder is produced during crushing and processing, for which there are no sufficient recycling options yet.

Therefore, the design of a circular economy for pd-AAC needs an innovative, high-quality recycling option that can process fine material and handle impurities. The production of RC-BCC from fine pd-AAC is a promising recycling approach currently being researched on a laboratory scale. RC-BCC could be used for primary cement substitution to save the high GHG emissions from primary production. Since cement is also needed as input for AAC production, this recycling option is suitable for closed-loop recycling. Moreover, the RC-BCC production could be scaled to handle high pd-AAC volumes in future. Overall, the RC-BCC production fulfils crucial requirements to become a reasonable pd-AAC recycling option, but its ecological and economic potential is still unknown. Thus, assessing the RC-BCC production from pd-AAC using primary data from laboratory trials is another research objective of this dissertation. The technical development of the recycling option is not within the scope of the dissertation but was carried out by project partners as part of the REPOST research project mentioned in Section 1.

3.4 Optimal design of a pd-AAC recycling network

Pd-AAC, like C&DW in general, has only a low value per weight. Therefore, transports over long distances can significantly influence economic efficiency. Additionally, there is a trade-off between transport costs and economies of scale of the recycling plants. Lower catchment areas lead to lower transport costs but also lower economies of scale due to smaller recycling plants. The opposite is true for larger catchment areas. Therefore, in addition to establishing new and high-quality recycling options, the circular economy design for AAC also requires precise planning and economic optimisation of a possible recycling network.

Consequently, the fourth research field of this dissertation deals with modelling and optimising a pd-AAC recycling network. There are numerous applications of network modelling in the literature, including optimising recycling and secondary raw materials networks (Barros et al., 1998; Ghafourian et al., 2021; Jahangiri et al., 2022; Pan et al., 2020; Rahimi & Ghezavati, 2018; Trochu et al., 2020). However, modelling and optimising a pd-AAC recycling network is a research gap that has not yet been considered in the literature.

The modelling requires a realistic representation of pd-AAC recycling for accurate and conclusive results. Therefore, another research objective of this dissertation is to develop a new model that considers the particular characteristics of pd-AAC recycling in detail. On the one hand, the potential future increase in pd-AAC volumes must be included. Accordingly, the model should cover opening, expanding, closing, or moving recycling plants. On the other hand, there are different new approaches for pd-AAC recycling, including the production of RC-BCC. This characteristic should also be considered by modelling the RC-BCC production as an optional second recycling step, resulting in a multi-stage network. Moreover, capacities must be considered as there is a trade-off between economies of scale in large recycling plants and transport costs.

Moreover, the optimisation parameters should reflect the specific pd-AAC recycling situation as precisely as possible. Therefore, cost parameters from the economic assessment (research field 3.2) should be considered, including variable and fixed recycling costs, transport costs, disposal costs and revenues for recycled products. In addition, other parameters, such as demand for recycling products or process efficiencies, should be precisely determined.

3.5 Relation of research fields and companion studies

The four research fields described above are comprehensively explored by the six studies written within the framework of this dissertation. Table 3.1 overviews how the studies contribute to the research fields. The connection of the studies can be explained as follows (Figure

3.1). Study A's pd-AAC volume prediction for Germany forms the basis for the following assessments and the network optimisation and addresses the first research field. Study B is methodologically based on Study A and extends the volume assessment to Europe, contributing to the first research field. Study C carries out an ecological assessment as part of the second research field and assesses the total ecological savings potential of pd-AAC recycling based on the volumes determined in Study A. The assessment of RC-BCC production in Study D is compared to the results from Study C and is directed towards the third research field. Study E contributes to the second research field by performing an economic assessment of pd-AAC recycling for all recycling options from Studies C and D while taking over information on mass and energy balances. Finally, the results of the volume assessment in Study A and the economic assessment in Study E are used as input for Study F, which models and optimises a recycling network for pd-AAC, thereby addressing the fourth research field.

Table 3.1: Overview of the contribution of the studies to the research fields.

Research field	investigated in Study					
	A	B	C	D	E	F
1. Quantification of the regional pd-AAC volumes in Germany and Europe	X	X				
2. Ecological and economic assessment of pd-AAC recycling			X	X	X	
3. Assessment of the RC-BCC production as a new and innovative recycling option for pd-AAC				X		
4. Optimal design of a pd-AAC recycling network						X

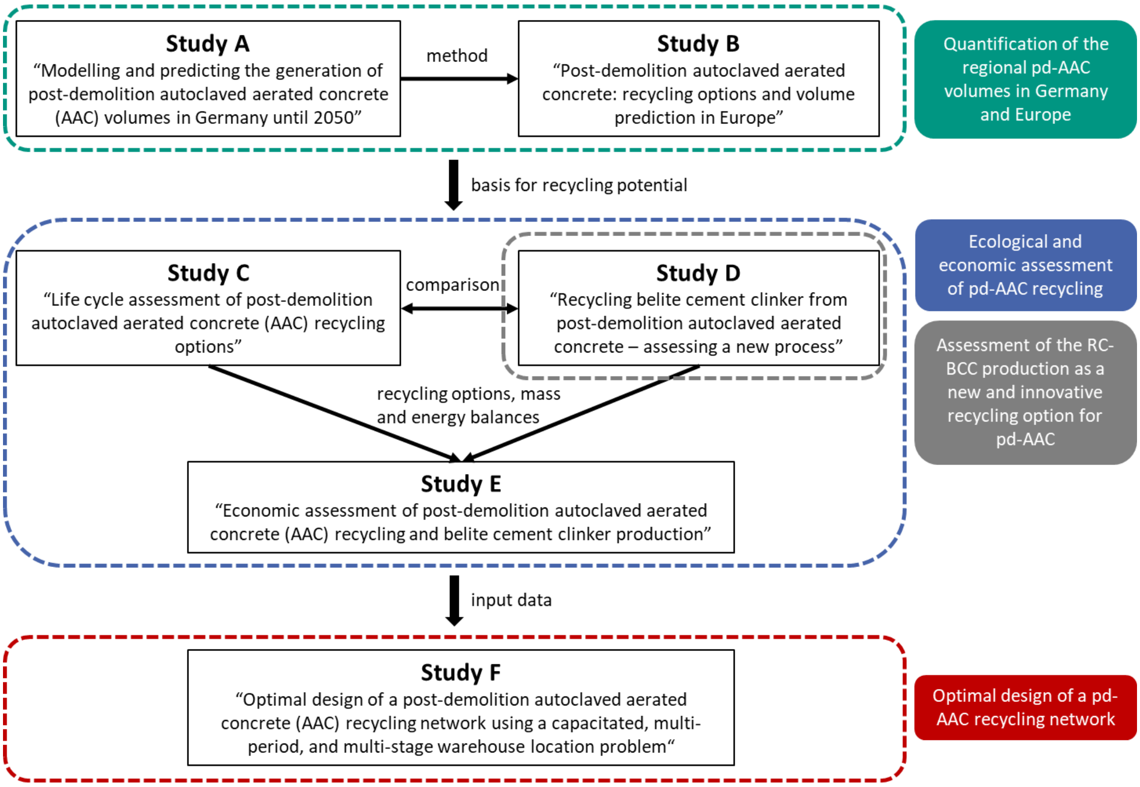

Figure 3.1: Connection of the studies developed within the framework of this dissertation.

4 Summaries of the Companion Studies

This section summarises each of the six companion studies of this dissertation, including the motivation, methodology, results, discussion, and limitations. The full version of the studies can be found in Part II.

4.1 Study A: Prediction of regional pd-AAC generation in Germany until 2050

This section summarises the article “Modelling and predicting the generation of post-demolition autoclaved aerated concrete (AAC) volumes in Germany until 2050” (Steins et al., 2021), developed by Rebekka Volk, Frank Schultmann, and myself. The article was published in the journal “Resources, Conservation & Recycling” in 2021.

4.1.1 Motivation and methodology

C&DW recycling is becoming increasingly important to improve sustainability, foster a circular economy and reduce GHG emissions. There are studies quantifying the current material stock (Deilmann et al., 2014; Hashimoto et al., 2007, 2009; Kapur et al., 2008; Ortlepp et al., 2016; Schiller et al., 2010; Schiller et al., 2015) and statistics on waste generation for various building materials covered by the regulation on the European waste list (AVV/04.03.2016). However, such information is largely lacking for pd-AAC as pd-AAC is not considered at all or only marginally in studies on the building stock. In addition, pd-AAC is not covered separately by the European waste list, so there is no reliable knowledge about respective volumes. However, this lack of knowledge is a decisive obstacle for stakeholders in establishing and promoting a circular economy for AAC. Therefore, Study A develops a quantification model to estimate past, current and future pd-AAC volumes in Germany, including their regional distribution. This distribution is crucial for locating new recycling plants since the pd-AAC transport causes significant ecological and economic efforts.

The quantification model is based on historical AAC production data, historical regional construction activity, regional popularity of AAC, and lifetime distributions of buildings (Figure 4.1). The model can be classified as a dynamic retrospective and prospective flow analysis using a flow-driven model (input flows) following the subdivision of construction material flows and stock studies given by Augiseau and Barles (2017). The quantification model performs the following steps:

1. Research of production data
2. Calculation of annual AAC demand on the federal state level
3. Calculation of a regional popularity index for AAC
4. Calculation of annual AAC demand on the administrative district level (Nomenclature des unités territoriales statistiques (NUTS) level 3)
5. Calculation of the regional past, current and future pd-AAC volumes

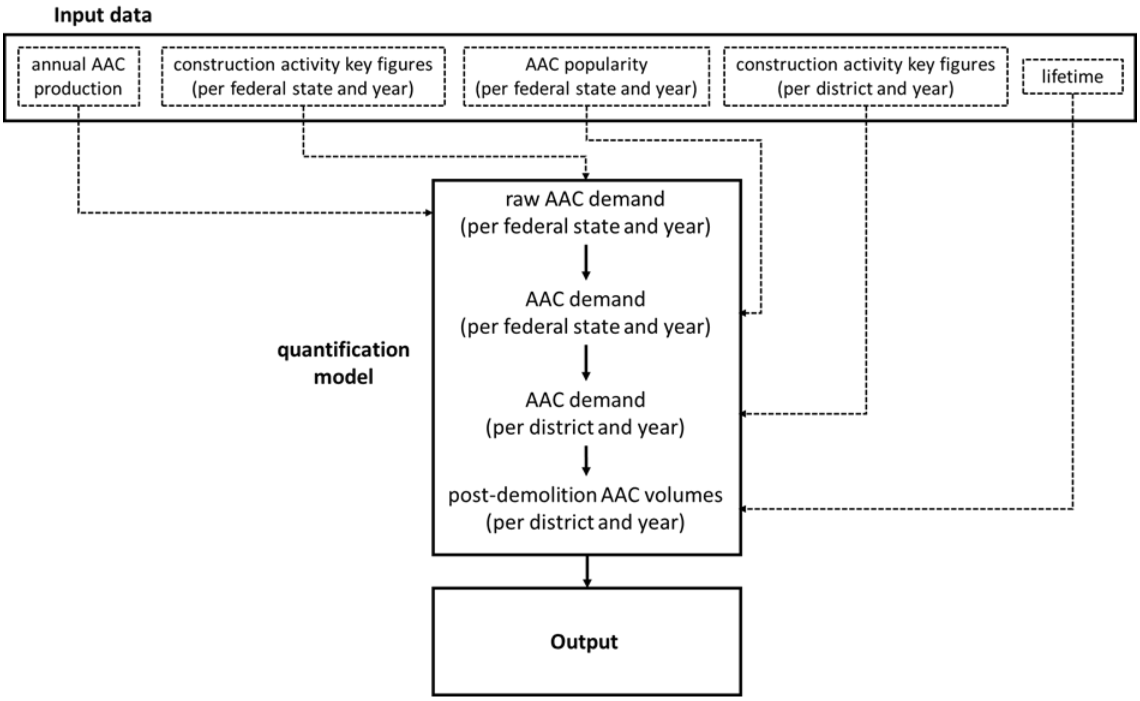

Figure 4.1: Methodology of the pd-AAC quantification model.

First, AAC production statistics are researched. Afterwards, construction activity statistics are consulted to determine a raw AAC demand per federal state in the second step of the model. The correlations and significance levels of different construction activity key figures are calculated to identify the key figure with the highest Pearson correlation and significance level to the AAC production. This key figure can then be used for regional demand calculation. The third step includes an AAC popularity index per federal state (Pestel, 2020) to reflect the regionally varying preferences for different construction materials. Fourth, the demand per federal state is broken down further to the administrative district level (NUTS 3) using the same construction activity key figure. Finally, past, current, and future pd-AAC volumes are calculated from the district-level demand in the fifth step of the quantification model. Triangular building lifetime probability functions are assumed based on literature values (Hossain & Ng, 2018; Rahlwes, 1993; Schmalwasser & Weber, 2012) to calculate the pd-AAC volumes from the demand data (Figure 4.2). Lifetimes for residential and non-residential buildings are distinguished as they differ significantly.

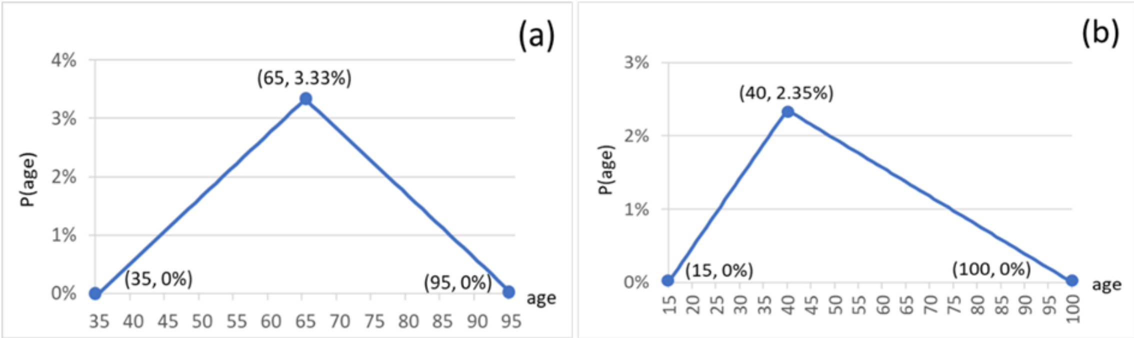

Figure 4.2: Assumed building lifetime probability functions of residential (a) and non-residential (b) buildings.

Validation of the results is performed with a stock-based and a waste-based approach. The stock-based validation approach uses data on the total building stock, the average AAC material intensities in the stock, and demolition statistics to estimate pd-AAC volumes. The waste-based validation approach considers waste statistics (only possible to calculate an upper bound for pd-AAC volumes) and literature estimations on pd-AAC amounts.

4.1.2 Results

The study applies the new pd-AAC quantification model to the use case of Germany. Results for annual German pd-AAC volumes (Figure 4.3a) show that pd-AAC started to emerge around 1990, as the production of AAC in Germany began not earlier than 1950. Since then, however, the increase of pd-AAC volumes has accelerated steadily. The prediction for the future shows that the growth is expected to continue at least until 2050, which is the end of the time horizon of the study. Pd-AAC amounts are expected to reach more than 4 Mm^3 per year by 2050, around 3.5 times the 2020 amount. The AAC stockpile can reach a maximum of approximately 250 Mm^3 in 2040 before the pd-AAC volume outnumbers the AAC production so that the stockpile is reduced (Figure 4.3b). The validation approaches provide several data points above and below the model results. Therefore, the model results seem reasonable.

The results are most sensitive to changes in the assumed building lifetime. Reducing the lifetime by 10% (minimum and maximum lifetime and lifetime peak from Figure 4.2) increases the pd-AAC volumes in Germany by 49% (597,000 m^3) in 2020. A respective lifetime increase of 10% results in a decrease of the pd-AAC volumes by 34% (415,000 m^3) in 2020. However, this sensitivity is based on the shifting of the future increase in pd-AAC volumes through shorter/longer lifetimes. The overall result of rising pd-AAC volumes in the future does not change (Figure 4.3a).

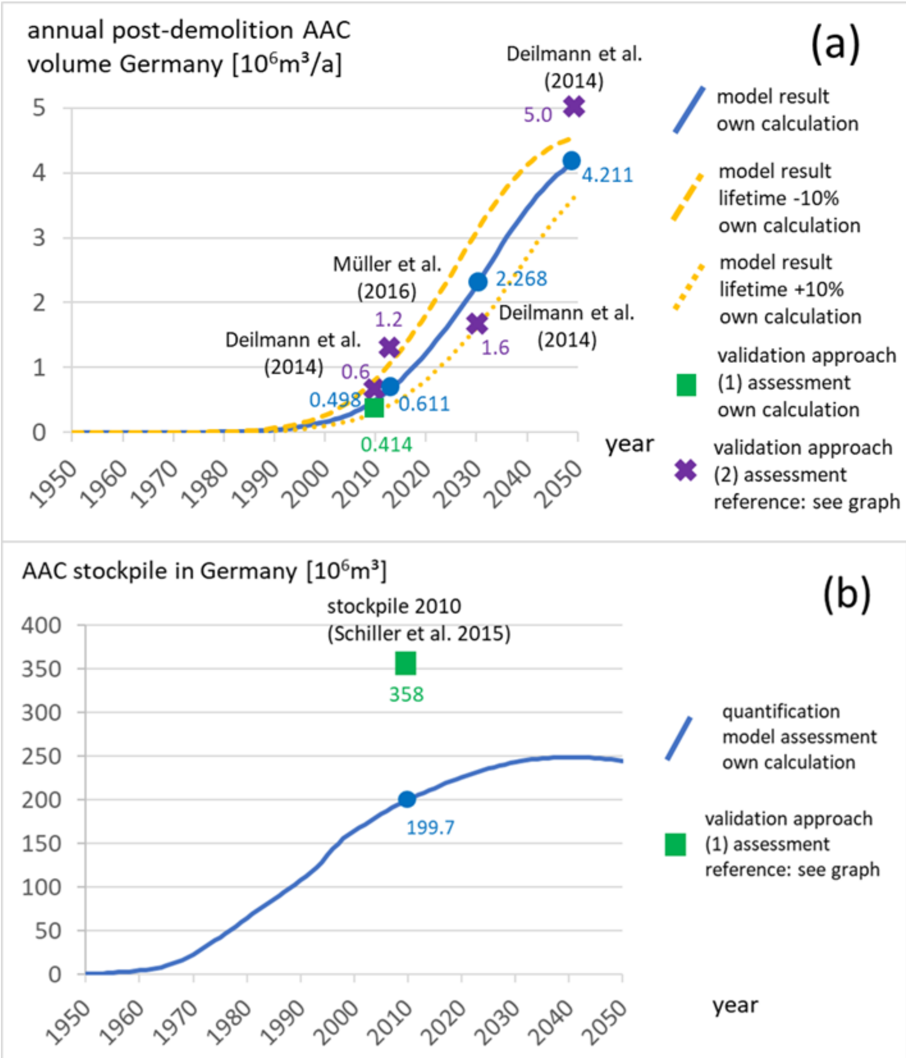

Figure 4.3: Quantification model results for the annual pd-AAC volumes (a) and the AAC stockpile (b) in Germany. Stock-based validation results (approach 1), waste-based validation results (approach 2), and building lifetime sensitivity analysis results are also illustrated.

The expected increase in pd-AAC volumes is also apparent, looking at the results of the pd-AAC quantification per NUTS 3 region (Figure 4.4). Moreover, significant regional differences strike. On the one hand, large German cities such as Berlin, Hamburg, Munich, and Hanover stand out. On the other hand, the regional AAC popularity influences the pd-AAC volumes. The historically developed high AAC popularity in Northern Germany and Baden-Württemberg leads to higher pd-AAC volumes than in Bavaria, where AAC is much less prevalent.

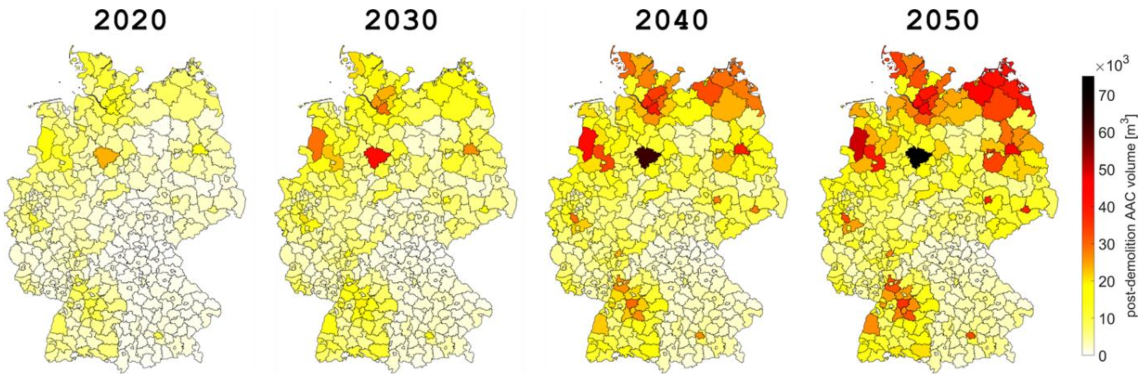

Figure 4.4: Expected pd-AAC volumes at the NUTS 3 level in Germany for 2020, 2030, 2040 and 2050 (fixed scale). The darker the colour, the higher the pd-AAC volumes.

4.1.3 Discussion and limitations

This study quantifies past, current, and future pd-AAC volumes in Germany for the first time. Additionally, a regional subdivision of the volumes is analysed.

The model results are based on the historical German AAC production but ignore imports and exports. Schiller et al. (2015) and BBS (2019) determine that Germany has an export surplus for mineral building materials. Thus, the model results might overestimate the German AAC stockpile and pd-AAC volumes. However, mineral building materials are bulky and generally not transported over large distances, so the overall export does not account for a large proportion of the total production. Besides, building lifetimes are a crucial input to the model. However, the lifetime functions are literature-based assumptions, as no empirical data is available. Furthermore, refurbishments can significantly influence the lifetime.

The AAC density assumption of 500 kg/m³ (according to literature) is used to compare weight-based and volume-based data in the production statistics and literature validation. However, AAC's density decreased in the past to improve thermal insulation properties, reaching 350 kg/m³ and below today (DIN 20000-404:2018-04). Therefore, incorporating a decreasing AAC density over time would improve the quantification model's results.

4.2 Study B: Prediction of pd-AAC generation in Europe until 2030

This section summarises the article "Post-demolition autoclaved aerated concrete: Recycling options and volume prediction in Europe" (Steins et al., 2022), developed by Rebekka Volk, Frank Schultmann, and myself. The article was presented at the "Ecocity World Summit 2021-22" and published in the "Ecocity World Summit 2021-22 Conference Proceedings" in 2022.

4.2.1 Motivation and methodology

Sustainable handling of C&DW is of great importance not only at the German but also at the European level. However, there is no information on current and future pd-AAC volumes for European countries due to the lack of waste statistics disclosing pd-AAC shares and studies on pd-AAC volumes. Although suggestions for pd-AAC recycling exist in the literature, implementing a comprehensive recycling network requires information about current and future mass flows. Therefore, in Study B, the quantification model presented in Study A is consulted, modified and applied to all European countries with more than 100,000 inhabitants to determine national current and future pd-AAC volumes.

The modified quantification model approach differs depending on data availability in the considered countries (Figure 4.5). If data on historic AAC production is available (green in Figure 4.5), pd-AAC volumes are calculated using these production volumes and building lifetime probability functions as presented in Study A. However, data on historic AAC production is only available for Germany and the UK[1]. Therefore, the current AAC market size was researched for twelve countries[2] (yellow in Figure 4.5), including Austria, Belgium, Czech Republic, Denmark, Hungary, Italy, Norway, The Netherlands, Poland, Russia, Slovakia, and Sweden. Based on the detailed calculations for Germany and the UK, a ratio of current and future pd-AAC volumes to the current market size is determined. This ratio can then be used to estimate pd-AAC volumes for the countries where only information on the current market size is available. Finally, there are countries without data on historic AAC production or current AAC market size (orange in Figure 4.5). Thus, the AAC market size of these countries is estimated based on a representative country in the same region. The AAC market size per capita is assumed to be the same as in the representative country. Hence, the total market size can be calculated using population data. Afterwards, the pd-AAC volumes can again be calculated using the ratio of current and future pd-AAC volumes to the current market size. Figure 4.6 discloses all considered European regions and their respective representative countries.

[1] Obtained from a direct inquiry to the UK Department for Business, Energy & Industrial Strategy.

[2] Obtained from an expert interview with Dr. Oliver Kreft, Xella Technologie- und Forschungsgesellschaft mbH.

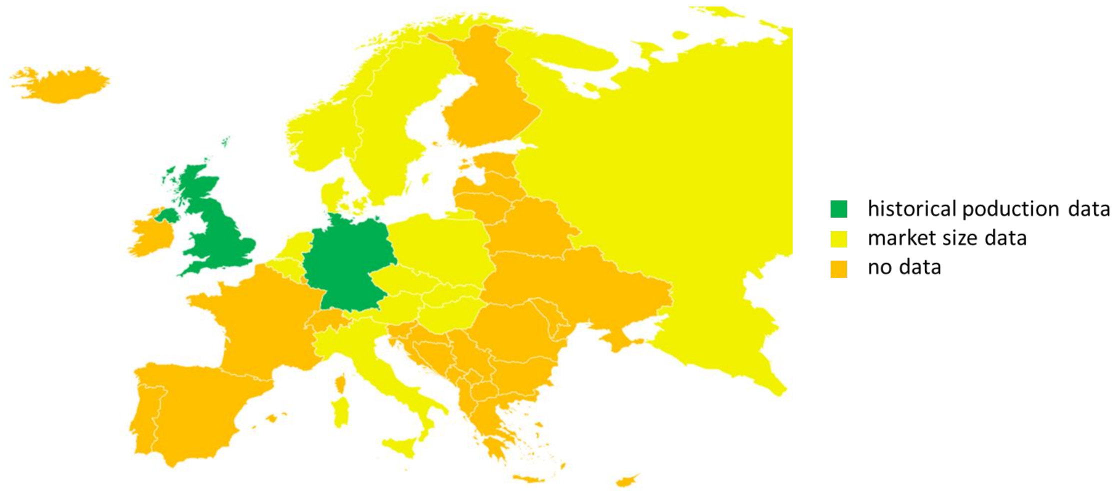

Figure 4.5: Data availability for pd-AAC prediction in the European countries.

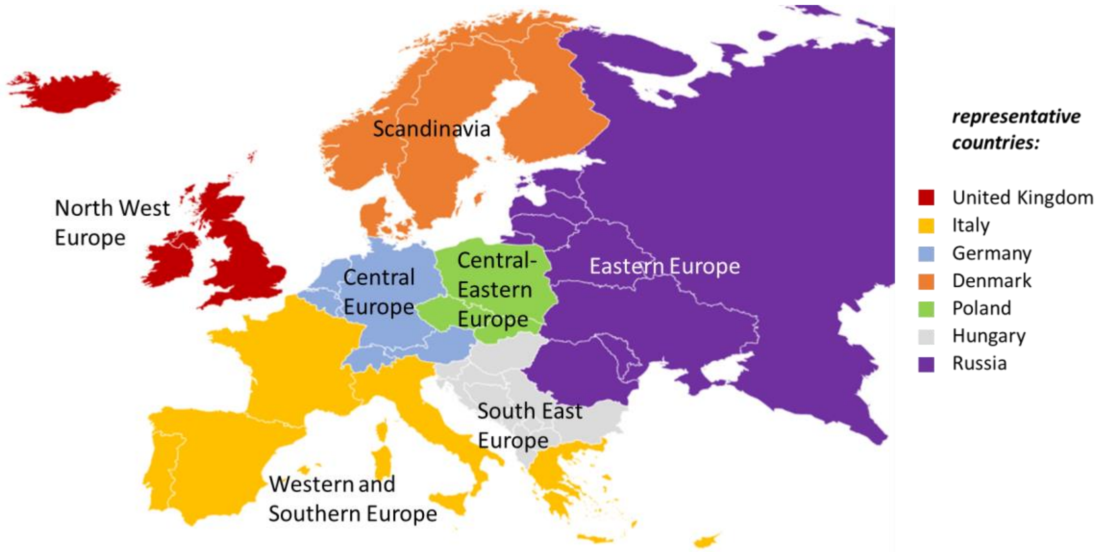

Figure 4.6: Representative countries for all considered European regions.

4.2.2 Results

The calculated pd-AAC volumes in Europe are presented in Figure 4.7 for 2020, 2025, and 2030. In 2020, the highest volumes can be found in Russia (3.9 Mm³), Poland (1.8 Mm³), Germany (1.2 Mm³), Ukraine (1.2 Mm³), the UK (0.7 Mm³), Romania (0.5 Mm³), and the Czech Republic (0.4 Mm³). The highest pd-AAC volumes in the European regions are in Eastern, Central-Eastern, and Central Europe. North West, Western and Southern, Southeast Europe, and Scandinavia have significantly lower volumes. The lower volumes are due to the smaller AAC market sizes in the respective regions. The European pd-AAC volume was around 12.3 Mm³ in 2020, approximately ten times the German amount. The predicted pd-AAC volumes in 2025 and 2030 show a similar regional distribution in Europe. However, total pd-AAC volumes increase significantly, nearly doubling from 2020 to 2030, reaching a sum of around 22 Mm³.

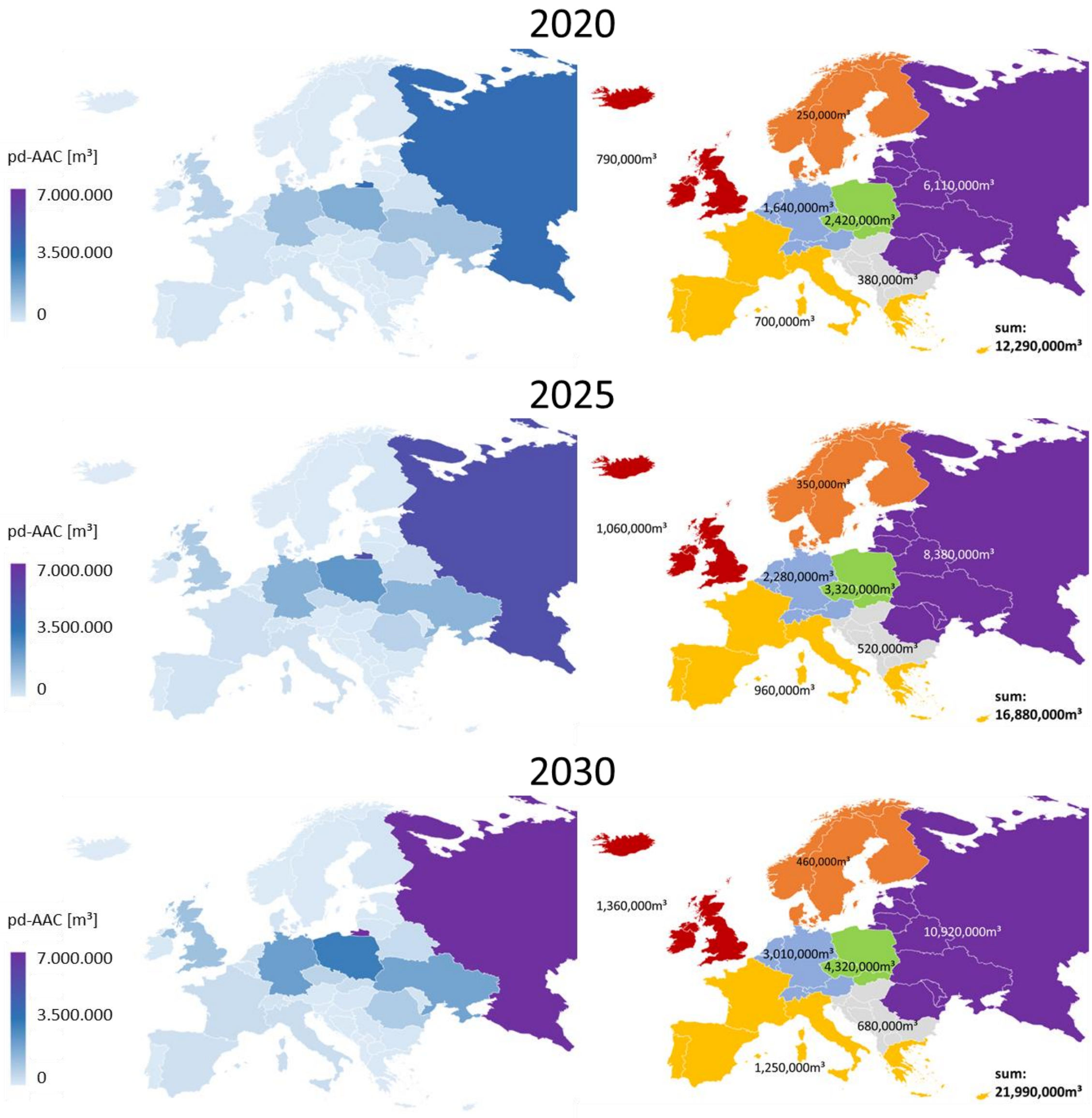

Figure 4.7: Pd-AAC volumes in Europe per country (left) and per region (right) in 2020, 2025 and 2030.

4.2.3 Discussion and limitations

This study extends the quantification of current and prediction of future pd-AAC volumes to Europe on a country-specific level.

The results for Germany and the UK strongly depend on the assumed lifetime probability functions, which are subject to significant uncertainties as they are determined from sparse literature values without empirical validation and neglecting refurbishments (see Study A).

These uncertainties also influence the pd-AAC volume estimation for all other European countries, as the lifetime functions directly impact the ratio of current and future pd-AAC volumes to the current market size.

Moreover, the calculations for all countries except Germany and the UK are based on AAC market size data, as historic AAC production data is unavailable. Therefore, predictions of future pd-AAC volumes in these countries are more imprecise than for Germany and the UK. Countries in Southern and South East Europe are likely to use AAC for a much shorter time than Germany and the UK, making the current and future pd-AAC volumes ratio to the current market size less suitable for pd-AAC prediction.

4.3 Study C: Life cycle assessment of pd-AAC recycling options

This section summarises the article "Life cycle assessment of post-demolition autoclaved aerated concrete (AAC) recycling options" (R. Volk et al., 2023), developed by Rebekka Volk, Oliver Kreft, Frank Schultmann, and myself. The article was published in the journal "Resources, Conservation & Recycling" in 2023.

4.3.1 Motivation and methodology

C&DW recycling essentially contributes to reducing the construction sector's environmental impacts. However, pd-AAC recycling has not been established yet, leaving a significant potential for protecting the environment and reducing emissions unused. Therefore, Study C assesses different pd-AAC recycling options by performing an LCA to identify the most promising options and calculate the ecological savings potential of pd-AAC recycling compared to landfilling.

The assessment comprises closed-loop recycling of pd-AAC in producing new AAC (of density classes AAC-0.35, AAC-0.5, and AAC-0.55) (Kreft, 2017; Lam, 2021; Rafiza et al., 2019; Rafiza et al., 2022) and different open-loop recycling options. These include using pd-AAC as a filler or supplementary material in concrete (Gyurkó et al., 2019), in floor screed to replace sand (Bergmans et al., 2016), in light mortar (Aycil et al., 2016), in LWAC (Aycil et al., 2016; Gyurkó et al., 2019), and in shuttering blocks made from no-fines concrete (Gyurkó et al., 2019). The recycling process consists of a crushing, a grading, and a purifying of the pd-AAC. Crushing and grading are mandatory to reduce the pd-AAC grain size and to separate coarse pd-AAC granulate from fine pd-AAC powder with different applications. The purifying is needed to reduce impurities. Figure 4.8 gives an overview of all assessed processes and recycling options.

The LCA uses the zero burden and avoided burden approach (Nakatani, 2014) to address the specific situation of assessing pd-AAC recycling. Thus, the system boundaries exclude resource extraction, production, and the use phase (Figure 4.8), as these are the same for all considered options (zero burden approach). Moreover, 1 kg pd-AAC for treatment (landfilling or recycling) is the functional unit. The final products from the recycling options are assumed to substitute primary products, giving the recycling process a substitution credit (avoided burden approach). The LCA's goal is to determine the best pd-AAC treatment options from an environmental point of view and to compare pd-AAC recycling with pd-AAC landfilling. The life cycle impact assessment uses the ecoinvent 3.6 dataset and ILCD 2.0 2018 midpoint method (ecoinvent, 2019).

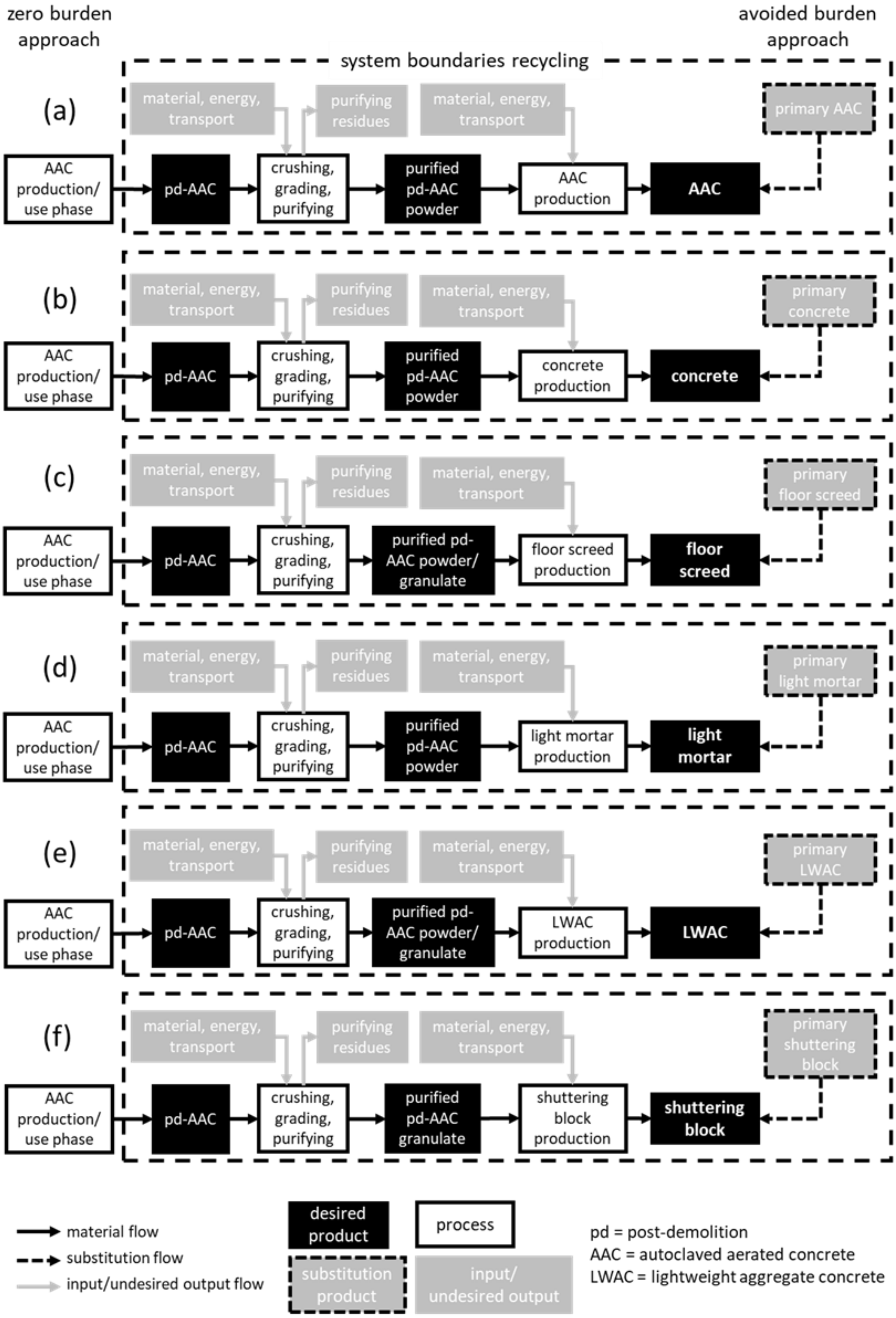

Figure 4.8: System boundaries and assessed pd-AAC recycling options, including AAC (a), concrete (b), floor screed (c), light mortar (d), LWAC (e), and shuttering blocks (f).

4.3.2 Results

Results show that the credits for primary material substitution significantly influence the overall assessment. The highest credits are granted when cement, quicklime, or expanded clay are substituted. Therefore, the recycling options AAC (all density classes, not sand substitution), light mortar, LWAC, and shuttering blocks show the highest environmental savings potential for all considered environmental midpoints, while recycling in AAC (sand substitution), concrete, and floor screed is not associated with considerable savings (Figure 4.9). The substitution credits of these preferred recycling options are much higher than the recycling efforts, resulting in negative efforts (savings). Closed-loop recycling of pd-AAC (AAC-0.35 high substitution) can reach the highest savings for the climate change midpoint of up to 0.50 kg CO_2-Eq/kg pd-AAC compared to landfilling. However, the open-loop recycling options light mortar, LWAC, and shuttering block production outperform the closed-loop recycling for most other midpoints. Usually, the recycling in light mortar production shows slightly lower savings than shuttering block production, while LWAC production performs best for most midpoints. Savings of up to 0.43 kg CO_2-Eq, 7 MJ fossil resources, 0.005 mol H+-Eq (acidification), 0.17 CTU (freshwater ecotoxicity), 0.2 g P-Eq (freshwater eutrophication), $5.2 \cdot 10^{-9}$ CTUh (carcinogenic effects), $4.4 \cdot 10^{-8}$ CTUh (non-carcinogenic effects), $2.5 \cdot 10^{-5}$ g CFC-11-Eq (ozone layer depletion), and 1.6 g NMVOC-Eq (photochemical ozone creation) compared to landfilling can be reached through recycling of 1 kg pd-AAC.

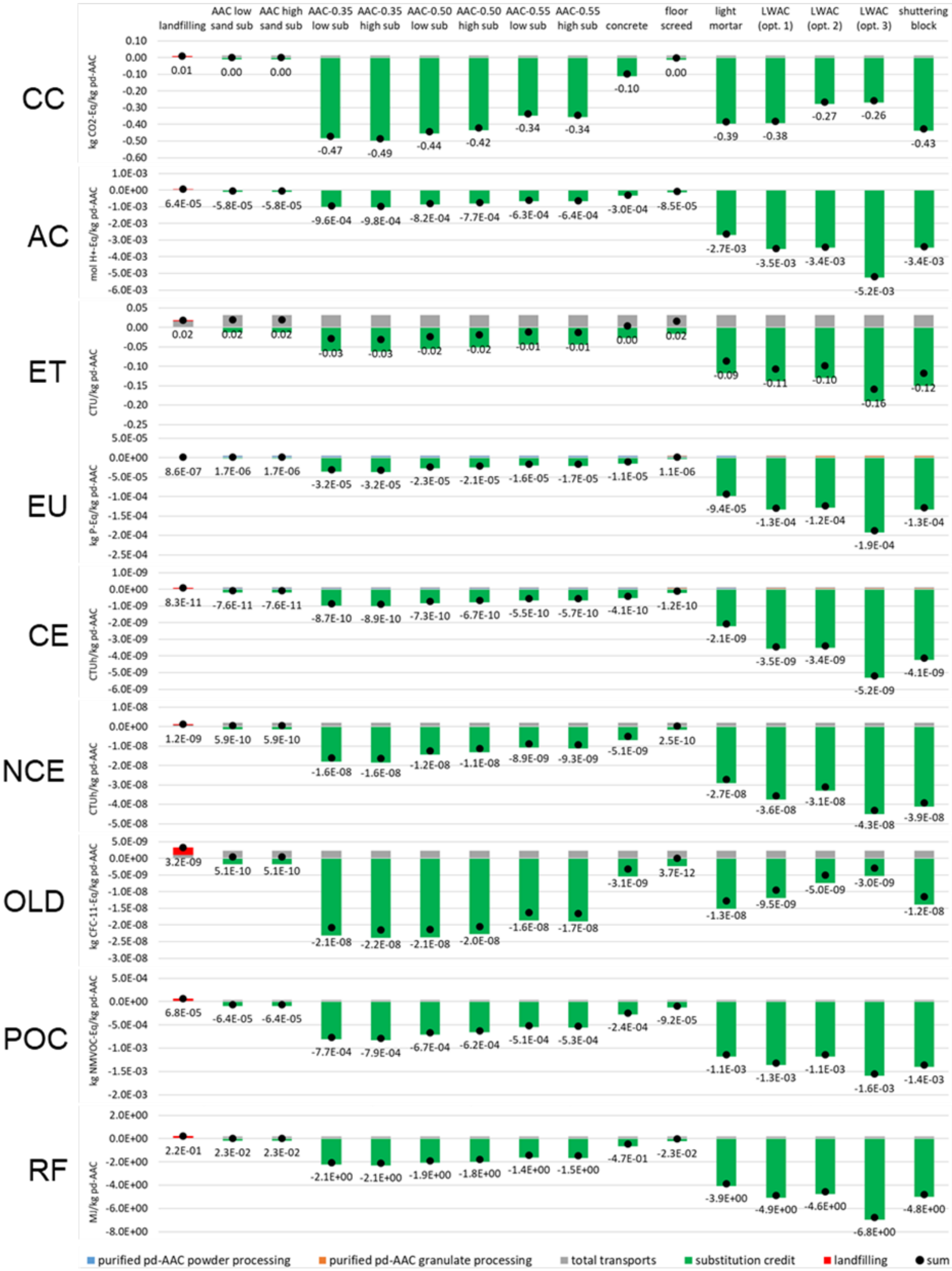

Figure 4.9: Impact assessment of landfilling and various recycling options for 1 kg pd-AAC including the midpoints climate change total (CC), freshwater and terrestrial acidification (AC), freshwater ecotoxicity (ET), freshwater eutrophication (EU), carcinogenic effects (CE), non-carcinogenic effects (NCE), ozone layer depletion (OLD), photochemical ozone creation (POC), and resources – fossils (RF).

The combined German AAC, light mortar, LWAC, and shuttering block production is high enough to recycle the current annual German pd-AAC amount of 700,000 t completely. Overall, pd-AAC recycling in Germany could save more than 280,000 t CO_2-Eq per year in GHG emissions. Moreover, there are significant savings in the other midpoints and the scarce landfill capacity.

A Monte Carlo simulation based on uncertainty functions calculated using the ecoinvent data quality system determines the sensitivity of the LCA results. The median values of all 10,000 simulation runs are near the original LCA value for all recycling options and all midpoints. The variability of the results (25% and 75% percentile of the simulation) is also moderate, reaching changes of a maximum of -20% and +25% for the recycling processes and the substitution credit. Even with these changes, the interpretation of the results stays the same, and the recycling options perform better than landfilling.

4.3.3 Discussion and limitations

This study shows that pd-AAC recycling has clear ecological advantages over landfilling. Therefore, establishing recycling could lead to a significant reduction in GHG emissions and other environmental impacts.

The closed-loop recycling options include different substitution levels, which do not influence the LCA results significantly, as the functional unit is input-related (1 kg pd-AAC) and does not focus on the output. However, higher substitution levels still show significant advantages. First, the environmental impact of the AAC with higher pd-AAC content is lower than that of AAC with lower pd-AAC content (for example, GHG emissions of 0.491 vs. 0.516 kg CO_2-Eq/kg AAC-0.35). Moreover, pd-AAC volumes are expected to increase significantly in the following decades. Thus, recycling options with high substitution levels are necessary to handle these rising volumes. In conclusion, substitution rates should be maximised, even if no direct advantage of high substitution rates is apparent in the LCA results due to the choice of functional unit.

The substitution credits for avoided primary production greatly influence the overall environmental savings of pd-AAC recycling. However, reduced quality of the recycling products compared to primary ones due to impurities in the pd-AAC could lead to lower substitution rates and credits.

The input data for the LCA is from a laboratory scale, and in the case of closed-loop recycling from a large pilot plant scale, as no field data is available yet. Thus, large-scale production systems still need to validate the production recipes and substitution percentages. However, the advantages of recycling options over landfilling are not expected to be eliminated even if the substitution percentage is reduced to ensure that quality requirements can still be met in large-scale production.

4.4 Study D: Assessing recycled belite cement clinker from pd-AAC

This section summarises the article “Recycling belite cement clinker from post-demolition autoclaved aerated concrete – assessing a new process” (Stemmermann et al., 2023), developed by Peter Stemmermann, Rebekka Volk, Günter Beuchle, and myself. The article has been submitted for publication in a scientific journal.

4.4.1 Motivation and methodology

Pd-AAC amounts from building demolition are expected to increase sharply in the following years. However, pd-AAC is still mainly landfilled. The mandatory crushing of pd-AAC produces around 75% of pd-AAC powder, which is particularly hard to recycle and, thus, needs new high-quality recycling options.

Due to suitable chemical conditions, using pd-AAC powder in producing Portland cement is a possible recycling option. However, limestone ($CaCO_3$) must be added to the pd-AAC to achieve an appropriate ratio of calcium to silicon. The added limestone is deacidified in the subsequent clinker production, which requires much energy on the one hand and releases large amounts of CO_2 on the other. A new belite-rich cement clinker, RC-BCC, was developed to reduce these two problems. This clinker can be clinkered at significantly lower temperatures (1000 °C instead of 1450 °C), making electrical heating conceivable. In addition, RC-BCC requires about 10% less emission-intensive limestone than Portland cement. The production of RC-BCC from pd-AAC is also possible for heavily contaminated secondary material because impurities can be safely incorporated into mineral compounds. The RC-BCC production’s technology readiness level (TRL) is 4-5. The scope of the dissertation includes the assessment of this innovative recycling option. The technical development and laboratory trials are not within the scope.

The study involves conducting an LCA for the RC-BCC production and subsequent AAC production using RC-BCC (25% and 50% substitution of primary cement) with zero burden and avoided burden approach (Nakatani, 2014). The impact assessment uses data from ecoinvent 3.8 (ecoinvent, 2021), the ReCiPe 2016 Midpoint (H) impact assessment methodology, and 1 kg pd-AAC as a functional unit. Ecoinvent 3.8 data for Germany, especially the German electricity mix, is used since the LCA results should reflect the current German situation. The RC-BCC production is assessed in four firing scenarios: natural gas, oxyfuel, conventional electricity (German electricity mix), and 100% renewable electricity.

4.4.2 Results

Pd-AAC is first crushed, graded, purified, and dried for RC-BCC production (Figure 4.10). Afterwards, the pd-AAC is mixed with limestone, aiming at a CaO/SiO_2 molar ratio of two. Moreover, flux is added (either Na_2CO_3 or $CaCl_2$) to increase the reaction kinetics and handle contaminations. All input components are milled before calcination and clinkering occur at about 1000 °C.

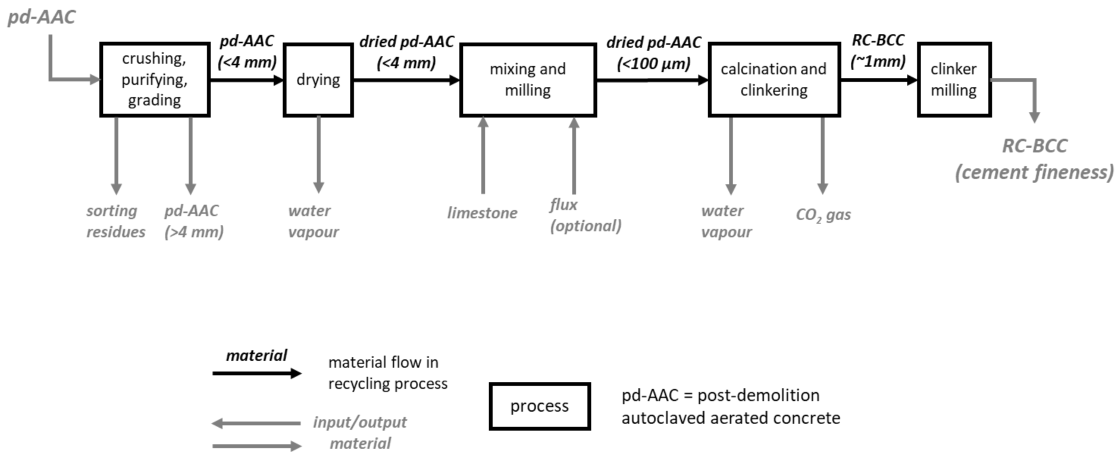

Figure 4.10: Process of RC-BCC production from pd-AAC.

Mass and energy balances (Table 4.1) are determined from thermodynamic equilibrium calculations using Factsage 8.2 (Bale et al., 2016). Measurements from laboratory tests complement the database, as phases such as ellestadite are not available. Generally, the process's lower temperature reduces thermal losses compared to Portland cement production. However, this is offset by the lower TRL of a rotary kiln heated electrically or via oxyfuel. Overall, the thermal efficiency is assumed to equal 60% for natural gas combustion with air, considering the efficiencies of current cement plants and the reduced process temperature. Moreover, the oxyfuel technology is assumed to reach 50% (lower TRL) and the electric heating 60% thermal efficiency (reduced heat losses via off-gas due to lower off-gas amounts). Electricity requirements for raw material grinding and auxiliary devices (e.g. process control, fans, transport) are assumed to be 110 kWh/t clinker (VDZ, 2020).

Table 4.1: Mass and energy balances for RC-BCC production from pd-AAC using electric heating, oxyfuel combustion, or natural gas combustion in air.

Mass balance	Educts RC-BCC [kg/t]		Products RC-BCC	[kg/t]	H_2O, gases, cumulated	CO_2 (process + fuels) [%]
Electric heating	***AAC***	***607***	***RC-BCC***	***1000***		
	Tobermorite	*363*	*Belite*	*852*		
	Quartz	*144*	*Ellestadite*	*138*		
	Water	*31*	*CaO*	*10*		
	Gypsum	*69*				
	$CaCO_3$	*840*	*H_2O*	*107*	*107*	
	$CaCl_2.6H_2O$	*34*	*CO_2*	*369*	*369*	*55%*
			$c(CO_2)$ off-gas: ca. 100%			
	Sum	*1482*		*1476*		
+ Oxyfuel	***+CH_4***	***47***	***+H_2O***	***103***	***210***	
	+O_2	*197*	*+O_2*	*9*	*9*	
			+CO_2	*132*	*502*	*39%*
			$c(CO_2)$ off-gas: ca. 95%			
	Sum	*1725*		*1721*		
+ Gas combustion in air	***+CH_4***	***12***	***+H_2O***	***24***	***235***	
	+O_2	***49***	***+O_2***		***9***	
	+N_2	***800***	***+N_2***	***800***	***800***	
			+CO_2	***38***	***540***	***34%***
			$c(CO_2)$ off gas: ca. 36%			
	Sum	***2586***		***2584***		
Portland cement clinker (for comparison)			***CO_2***	***820***		***100%***

Energy balance	Thermal efficiency*	Heat demand [kJ/kg]	El. heat [kWh/t]	Milling [kWh/t]	Oxygen generation [kWh/t]**	Oxygen [SCM/t]	Sum El. supply [kWh/t]
Electric heating	60%	1937	538	110	-	-	648
+ Oxyfuel	50%	2606	CH_4	110	69	700	179
+ Gas combustion in air	60%	3243	CH_4	110	-	-	110

*: Estimate) **: Assumption 0.5 kWh /SCM)

The LCA results show that the pd-AAC processing and transports do not significantly contribute to the overall RC-BCC environmental impacts (Figure 4.11). For most midpoints, the largest share of the impact is caused by efforts for the required electricity or natural gas supply.

Moreover, direct process emissions from natural gas combustion and limestone calcination significantly influence the global warming midpoint. Overall, electric heating with 100% renewable electricity leads to the lowest impacts of RC-BCC production for all considered midpoints.

The substitution of 25% or 50% primary cement with RC-BCC in AAC production reduces impacts for many midpoints, especially when RC-BCC is produced with renewable electricity (Figure 4.12). Global warming can be reduced from 0.01 kg CO_2-Eq/kg pd-AAC (landfilling) to -0.76 kg CO_2-Eq/kg pd-AAC (AAC with 25%/50% RC-BCC from 100% renewable electricity). The savings are slightly lower for AAC-0.5 and AAC-0.55 because they are produced with less cement. The substitution percentage does not influence the results as the functional unit is 1 kg pd-AAC. However, the absolute impacts of the produced AAC decrease with higher substitution. A Monte Carlo simulation shows that the LCA results are robust. RC-BCC production with 100% renewable energy shows the highest CO_2-Eq saving potential compared to all other pd-AAC recycling options from Study C. However, in the other firing scenarios, RC-BCC production is outperformed by pd-AAC powder recycling in AAC production and some open-loop recycling options, including light mortar, LWAC, and shuttering block production.

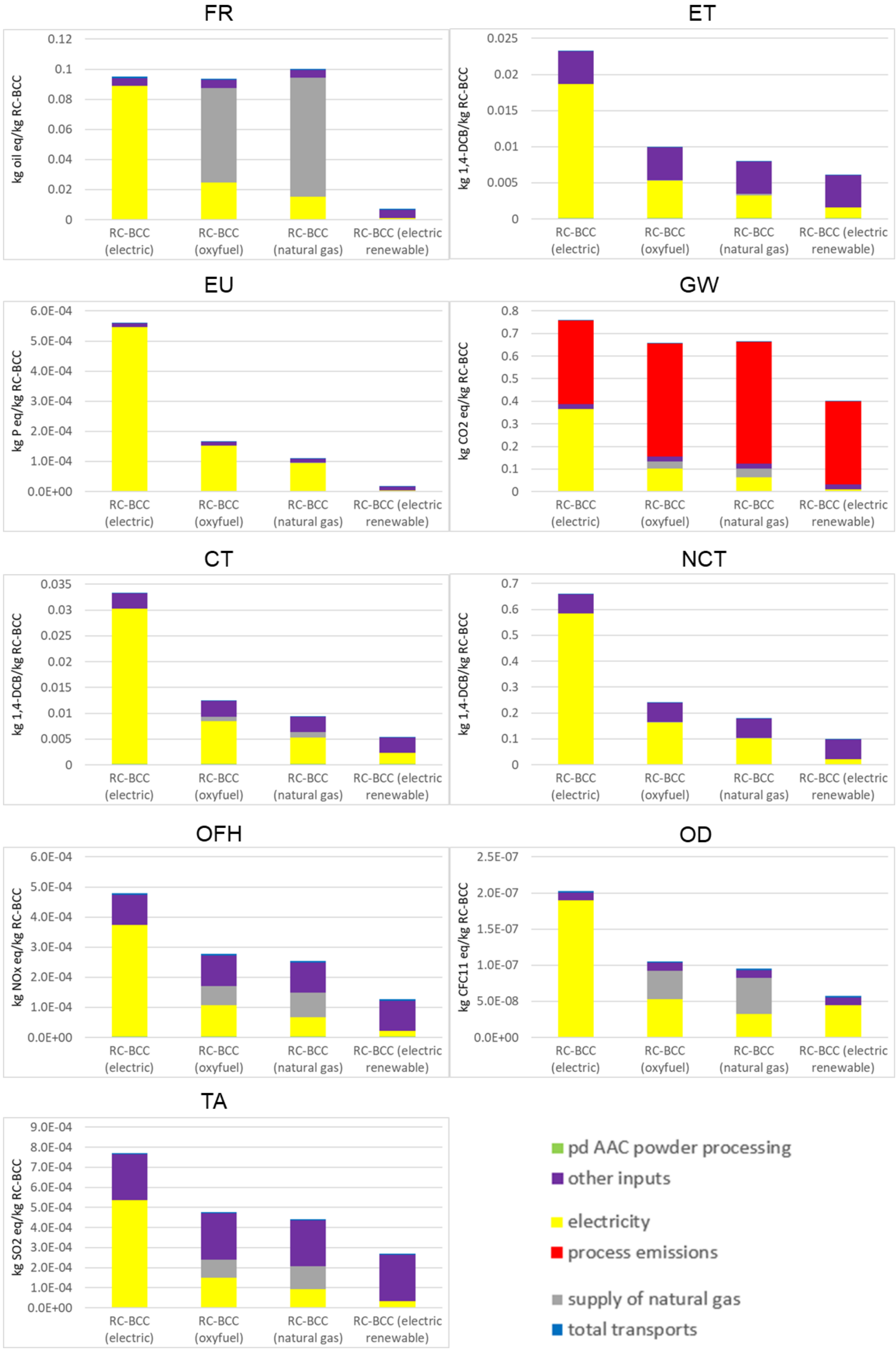

Figure 4.11: LCA results for RC-BCC production with different firing conditions including the midpoints Fossil resource scarcity (FR), Freshwater ecotoxicity (ET), Freshwater eutrophication (EU), Global warming (GW), Human carcinogenic toxicity (CT), Human non-carcinogenic toxicity (NCT), Ozone formation – Human health (OFH), Stratospheric ozone depletion (OD), and Terrestrial acidification (TA).

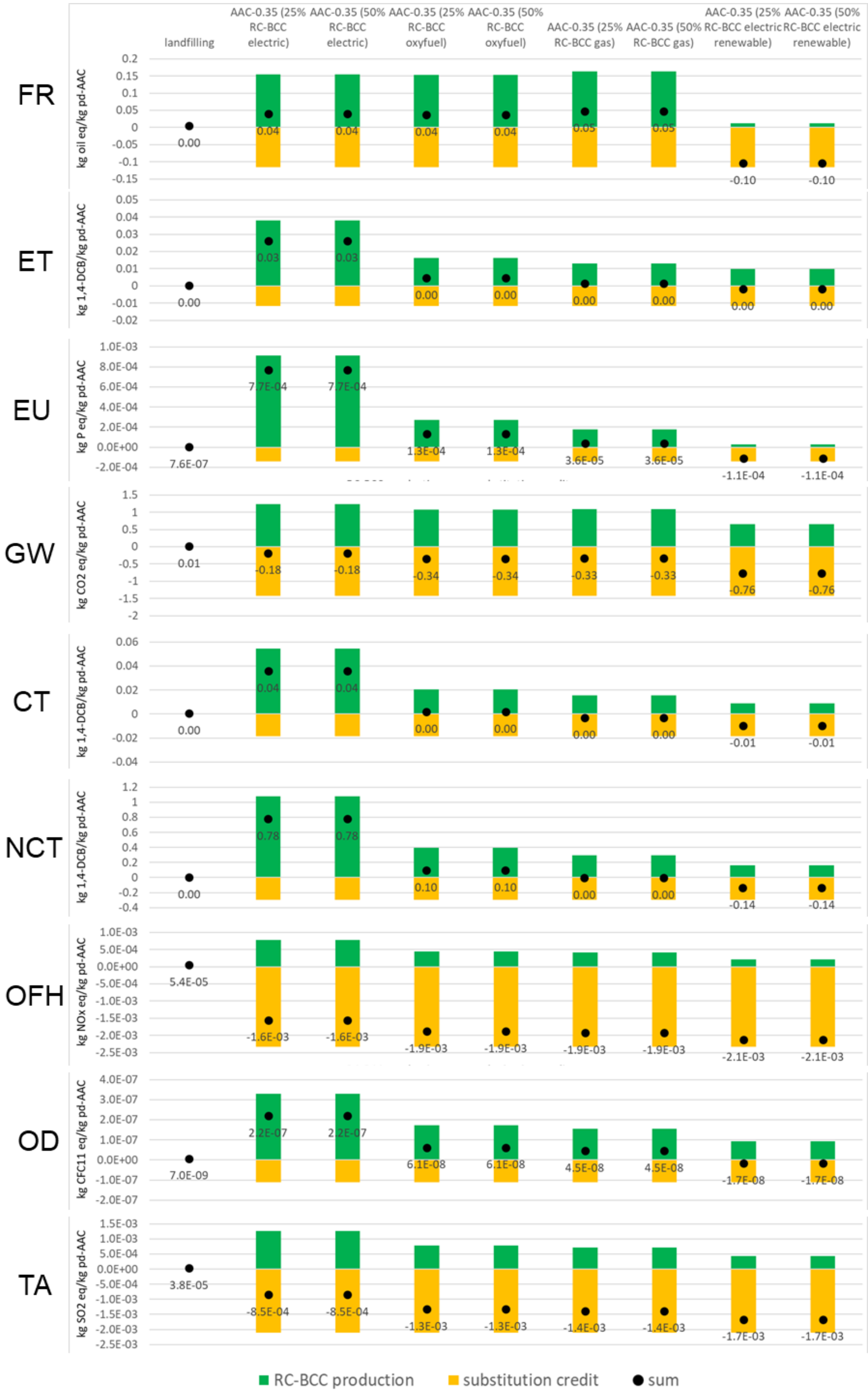

Figure 4.12: LCA results for the AAC-0.35 production with RC-BCC content including the midpoints Fossil resource scarcity (FR), Freshwater ecotoxicity (ET), Freshwater eutrophication (EU), Global warming (GW), Human carcinogenic toxicity (CT), Human non-carcinogenic toxicity (NCT), Ozone formation – Human health (OFH), Stratospheric ozone depletion (OD), and Terrestrial acidification (TA).

4.4.3 Discussion and limitations

This study shows that RC-BCC production from pd-AAC has a significant potential for environmental savings. Due to the high energy consumption in the production, the potential is considerably increased when renewable energies are used.

The RC-BCC production has a TRL of 4-5, and the LCA is based on experimental data. Thus, developing the process to a higher TRL can reduce the assessment's uncertainties. Moreover, the assessment did not consider production plant scaling, which can significantly influence energy efficiency.

The use of secondary materials can raise some issues. First, the constant supply of pd-AAC is significantly more complex than conventional input extracted in a quarry. Pd-AAC quantities are significantly higher in summer than in winter, when the construction industry is generally much less active. Furthermore, it would be desirable if much pd-AAC accrues in a small radius around an RC-BCC plant to operate a large plant with significant economies of scale but without immense transport effort. Therefore, expanding the process's input to include concrete from building demolition is conceivable.

Additionally, the material composition and quality vary, and higher contamination of the pd-AAC leads to a lower proportion of hydraulically effective clinker. It should be noted that belite cement generally has a significantly slower hydraulic reactivity than Portland cement, so process adjustments may be necessary to use the RC-BCC to substitute Portland cement.

4.5 Study E: Economic assessment of pd-AAC recycling and RC-BCC production

This section summarises the article "Economic assessment of post-demolition autoclaved aerated concrete (AAC) recycling and subsequent belite cement clinker production" (Steins, Volk, Beuchle, et al., 2023), developed by Rebekka Volk, Günter Beuchle, Pallavi Reddy, Gourisankar Sandaka, Frank Schultmann, and myself. The article has been submitted for publication in a scientific journal.

4.5.1 Motivation and methodology

The pd-AAC recycling shows a significant environmental savings potential, but it is questionable if the recycling is also economically viable. Therefore, an economic assessment of mechanical pd-AAC processing is carried out in this study. This mechanical processing usually produces a large amount of fine pd-AAC powder (75% of the output, Gyurkó et al., 2019), which is particularly difficult to recycle. Therefore, the economic assessment will also include the new RC-BCC production from pd-AAC powder (Study D). RC-BCC is a high-quality recycling product and can substitute primary cement associated with high costs and environmental efforts.

The assessed recycling process is shown in Figure 4.13 and includes crushing, purifying, and grading as part of the mechanical processing (Krampitz et al., 2022; Kreft, 2016) and drying, milling, rotary kiln processing, and cooling in RC-BCC production. The process steps are selected to ensure that pd-AAC powder/granulate and RC-BCC can be produced in high quality. Electricity demands for mechanical processing are determined according to machine specifications. The required energy of the rotary kiln is based on an assumed process temperature of 1000 °C and 9 wt.-% moisture content in the pd-AAC powder, while dryer and ball mill electricity demands are taken from the process simulation software Aspen Plus (Aspen Technology Inc, 2023).

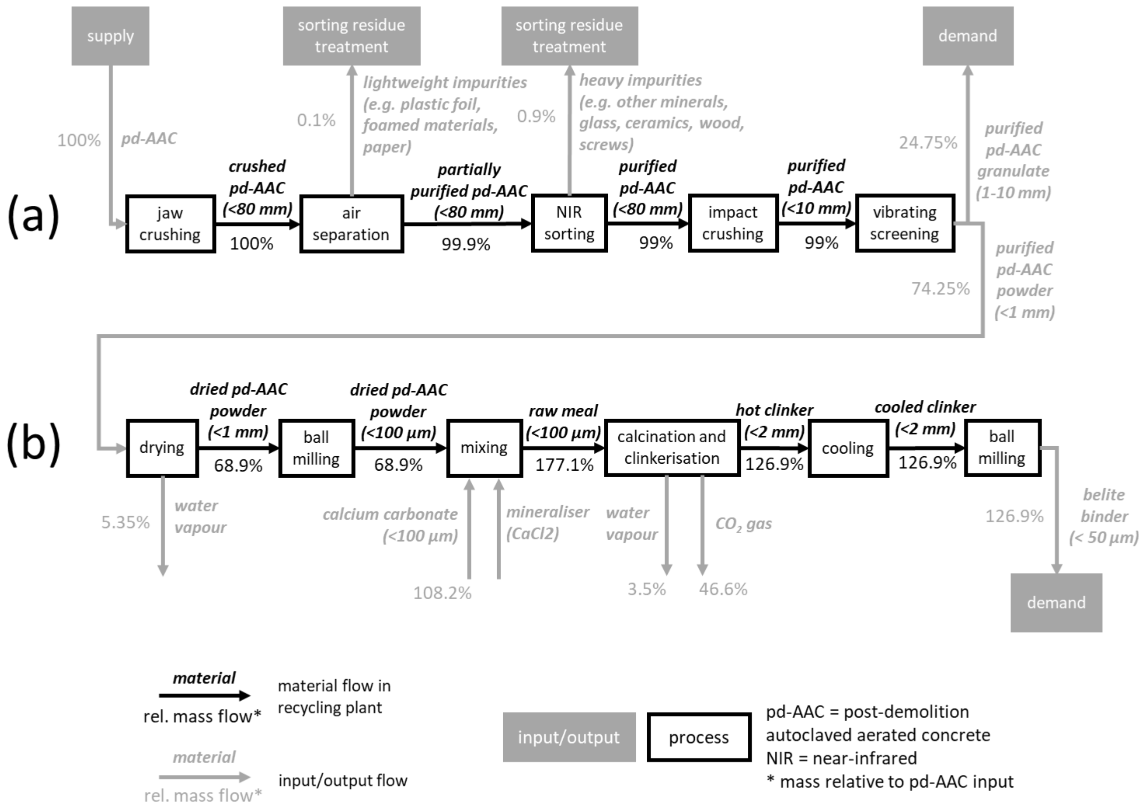

Figure 4.13: Pd-AAC recycling process and relevant mass flows for mechanical processing (a) and RC-BCC production (b).

The cost assessment follows the approach presented by Peters et al. (2003), considering variable costs, fixed costs, overhead costs (for example, costs for safety and protection, packaging costs, and storage facilities), and general expenses (for example, executive salaries, legal costs, and office maintenance). The variable costs include electricity, labour, and maintenance, while the fixed costs are mainly based on the annuity for the investment in the recycling plant. This investment is determined according to the "percentage of delivered-equipment cost" method by Peters et al. (2003), in which the equipment costs are multiplied by given factors to account for different cost aspects. The equipment costs are identified through manufacturer inquiry (for jaw crushing, impact crushing, air separation, NIR sorting, vibrating screening, conveyor belts, and compressed air generation) or calculated using the correlation function from Towler and Sinnot (2012) (for dryer, ball mill, and rotary kiln). Inflation adjustment is considered following the Chemical Engineering Plant Cost Index. Cooling equipment is not included in the cost assessment since data is lacking. The overhead costs and general expenses are calculated as percentages of operating labour costs and total product costs (Peters et al., 2003). The cost assessment includes five scenarios with recycling plant input capacities of 10,000 to 250,000 t/a. Changing equipment costs in the different scenarios are calculated based on the capacity of the recycling plant and a cost-capacity exponent (Humphreys, 2005).

4.5.2 Results

The economic assessment shows that the total costs of the mechanical processing and the RC-BCC production vary significantly depending on the capacity of the recycling plant (Figure 4.14). Smaller plants show higher costs of up to 200 €/t input for the mechanical processing and 1250 €/t input for the RC-BCC production. These costs can go down to 30 €/t input (mechanical processing) and 800 €/t input (RC-BCC production) in the scenario with the largest capacity. As 1.71 t RC-BCC is produced per t pd-AAC powder, RC-BCC production costs sum up to 493 €/t RC-BCC in the latter scenario. The variable costs account for almost 50% of the total cost for mechanical processing and more than 50% (up to 80% in recycling plants with large capacities) for the energy-intensive RC-BCC production. The other cost aspects reach a maximum share of 20% of the total costs and, thus, have a significantly lower influence than the variable costs.

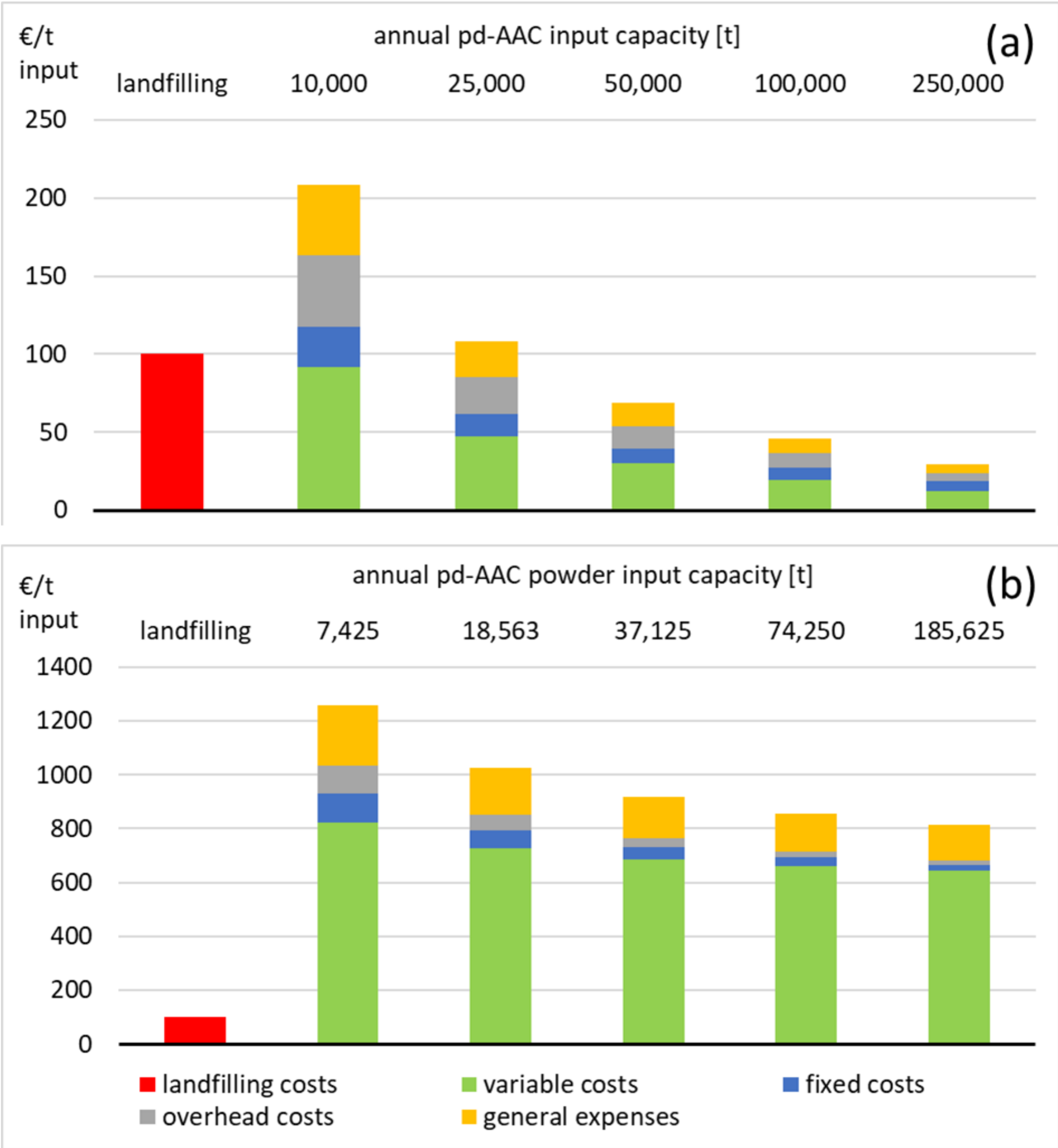

Figure 4.14: Variable costs, fixed costs, overhead costs, and general expenses for mechanical pd-AAC processing (a) and RC-BCC production (b) compared to landfilling costs.

The mechanical processing's total costs are well below the German pd-AAC landfilling costs of an average of 100 €/t in the scenarios with high recycling plant capacities. Therefore, mechanical processing is economically reasonable. This comparison does not even consider sales prices, which could be 5-15 €/t for pd-AAC powder and granulate, similar to other recycled building materials. However, RC-BCC production is significantly more expensive due to high electricity costs. Even in large recycling plants with significant economies of scale, the total costs for mechanical processing and RC-BCC production combined reach about 850 €/t input, of which RC-BCC production accounts for more than 90%. These costs are equal to production costs of 500 €/t RC-BCC. Thus, a 430 €/t RC-BCC selling price would be needed to make the RC-BCC production less expensive than landfilling. In contrast, ordinary Portland cement is only sold for around 150 €/t. Therefore, the RC-BCC production is not economically viable yet.

A sensitivity analysis shows the highest variability of the mechanical pd-AAC processing's total costs when labour or equipment costs change. Concerning the RC-BCC production, the total costs are only considerably sensitive to changing electricity costs.

4.5.3 Discussion and limitations

This study shows that mechanical pd-AAC recycling in sufficiently large recycling plants is significantly less expensive than landfilling. However, subsequent RC-BCC production is associated with very high costs.

The data used for the economic assessment originates from research and experiments, as no field data is available. Moreover, the recycling process design has not yet been tested in practice. A process adjustment might be necessary, especially if the share of impurities is higher than assumed. The limits of scaling the technology for RC-BCC production to significant input amounts are still unknown, particularly concerning the electrically heated rotary kiln. Therefore, switching to natural gas or oxyfuel combustion could be necessary at high capacities. Additionally, the cooling technology was not integrated into the cost assessment as the technology needed for RC-BCC is unclear. The cooling step could increase the equipment costs of RC-BCC production by around 10% (IEA Greenhouse Gas R&D Programme, 2008).

4.6 Study F: Optimal design of a pd-AAC recycling network

This section summarises the article "Optimal design of a post-demolition autoclaved aerated concrete (AAC) recycling network using a capacitated, multi-period, and multi-stage warehouse location problem" (Steins, Volk, & Schultmann, 2023), developed by Manuel Ruck, Rebekka Volk, Frank Schultmann, and myself. The article has been submitted for publication in a scientific journal.

4.6.1 Motivation and methodology

Pd-AAC recycling has not been established yet but has significant ecological (Study C and D) and economic (Study E) savings potential. Thus, Study F develops a new capacitated, multi-stage, and multi-period pd-AAC recycling network model to foster recycling. The model is optimised to identify a German pd-AAC recycling network design with minimal costs. The considered capacity constraints allow a trade-off between transport costs and larger plants' economies of scale. At the same time, the multi-stage formulation enables a trade-off between additional costs and higher revenues of the optional second recycling step (RC-BCC production). Furthermore, pd-AAC volumes are expected to rise sharply, so the model can open, close, expand or relocate plants in every considered period through a multi-period approach.

The mathematical formulation includes sets for supply, recycling plant, and demand locations. Moreover, sets for all commodities, possible capacity levels, and time periods are considered. Decision variables comprise the network flows and binary variables for recycling plant status and time of plant opening. All costs (transport, variable recycling, fixed recycling, plant opening, landfilling) and revenues are part of the model's parameters. Furthermore, the parameters include transport distances, the supply, the demand, efficiencies of the recycling processes, input capacities of the plants, and landfilling limits. The model's objective is to minimise total costs, while the flow conservation at all stages has to be considered. Furthermore, all the pd-AAC has to be treated, and capacities and demand are not allowed to be exceeded.

The model is optimised using specific input data for the German pd-AAC recycling case. The supply and possible recycling plant locations consider the NUTS 2 level, consisting of 38 German regions. Demand locations contain 31 German AAC plants and decentralised demand at construction sites without specific locations. The time horizon includes eleven periods (2023, 2024, 2025, 2026/2027, 2028/2029, 2030/2031, 2032-2034, 2035-2037, 2038-2040, 2041-2045, 2046-2050). Transport costs are based on Persyn et al. (2022). Landfilling costs are calculated using Aycil and Hlawatsch (2020) and online portal data for regionalised costs. All other costs, including capacities and revenues, are based on Study E. Inflation rates of all costs are identified from recent statistics. The supply is taken from Study A, while the demand and the process efficiency are calculated from Studies C and D.

4.6.2 Results

The optimised recycling network prefers large recycling plants at the first recycling stage to use economies of scale (Figure 4.15). There are only three recycling plants in the first period, but additional plants are opened as pd-AAC volumes increase. However, no plants are closed or relocated. The average transport distance from the supply to the first recycling stage is around 185 km in 2023 and will reduce to 117 km as more plants are opened. No RC-BCC production plants are opened due to their high costs.

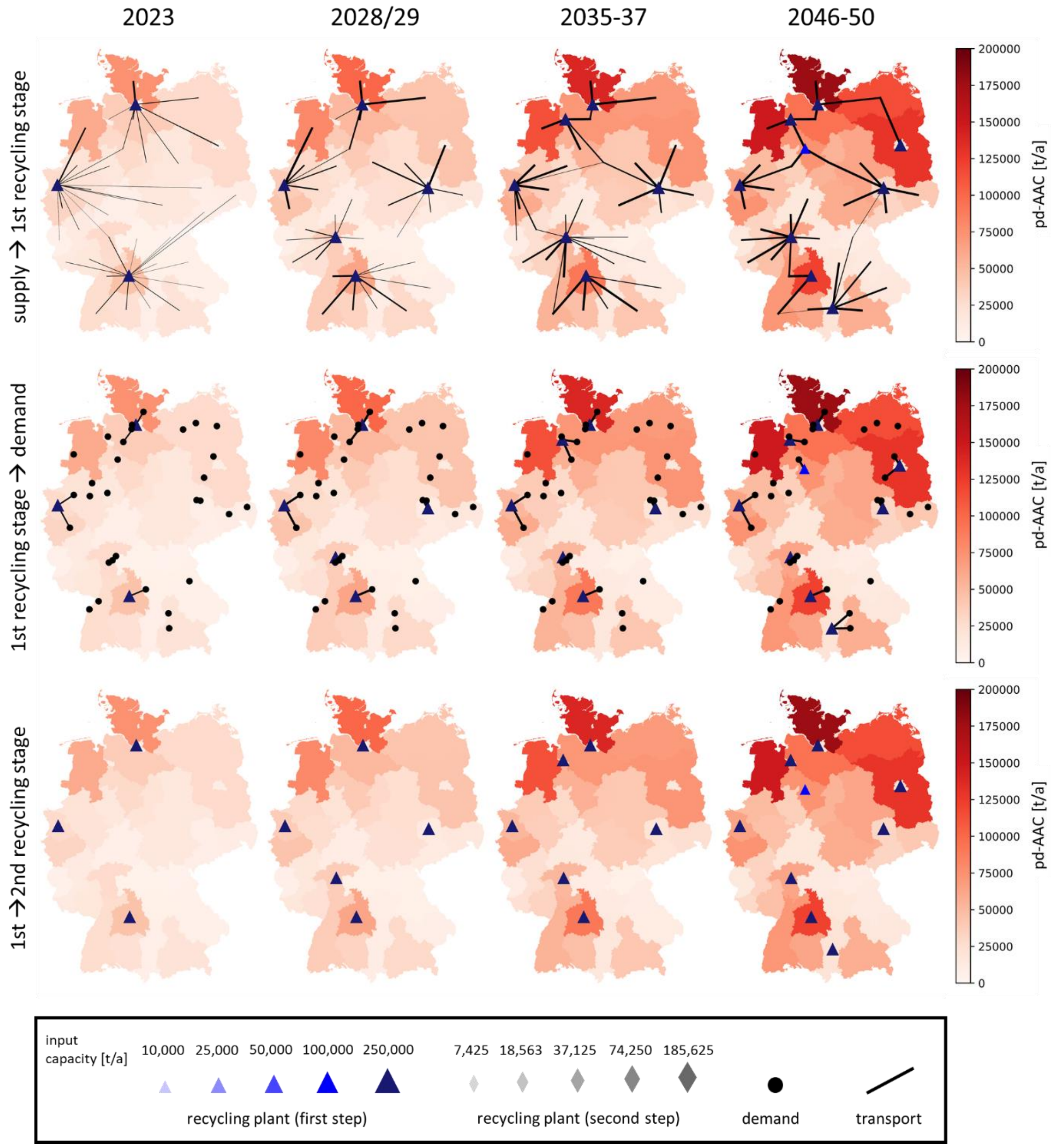

Figure 4.15: Optimal pd-AAC recycling network design for Germany for the periods 2023, 2028/29, 2035-37, and 2046-50.

Transport and fixed costs influence the total network costs the most (Figure 4.16). Variable costs are significantly lower, and opening costs only incur in specific years. Landfilling costs are always near zero, as 99.9% of the pd-AAC is recycled in the optimal network. Revenues offset around 20% of the total costs. All costs, except landfilling and opening costs, continuously increase due to rising pd-AAC volumes and inflation.

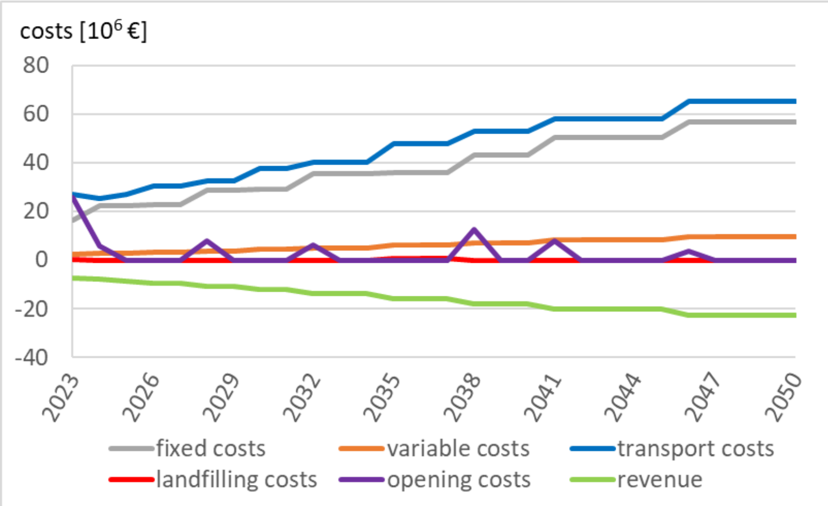

Figure 4.16: Variable, fixed, opening, transport, and landfilling costs as well as revenues in the optimised pd-AAC recycling network until 2050.

The total costs of the optimised recycling network increase from around 50 M€/a in the 2020s to approximately 100 M€/a in the 2040s (Figure 4.17). However, the status quo pd-AAC treatment, which can be supposed to be around ⅔ landfilling and ⅓ recovery (assumed not to cause any costs), is much costlier. It causes costs of just below 100 M€/a in the 2020s, exceeds 200 M€/a in the 2030s and nearly reaches 500 M€/a by 2050 due to increasing pd-AAC volumes and significant landfilling cost inflation. Overall, the status quo pd-AAC treatment would generate costs of 6,800 M€ until 2050, which could be reduced by 68% (4,600 M€) when a cost-minimal recycling network with total costs of only 2,200 M€ is established.

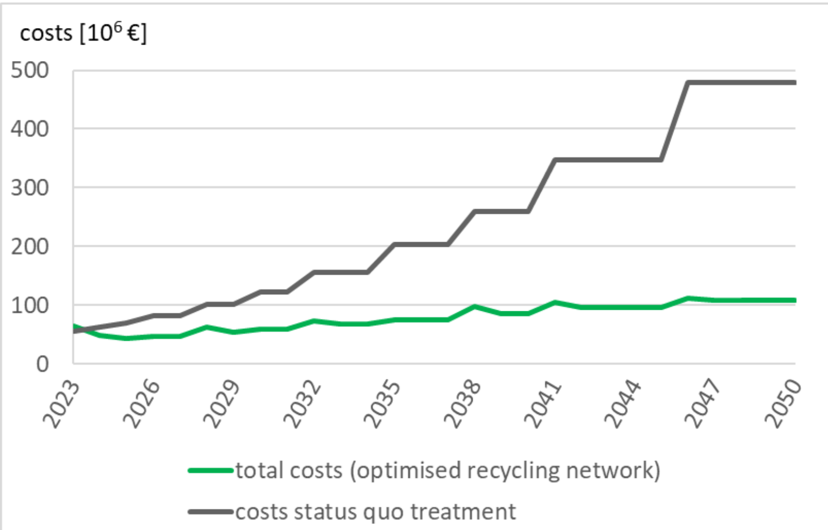

Figure 4.17: Total costs of the optimised pd-AAC recycling network compared with status quo pd-AAC treatment costs.

A sensitivity analysis (±10% variation) shows that the results are most sensitive to changes in the supply (±8%), the transport costs (±7%) and the fixed costs (±5%). Additionally, a scenario analysis discloses that RC-BCC production is used when heavy support is provided (+100% demand, +100% revenue, 6 %/a reduction of variable/fixed/opening costs). However, RC-BCC recycling plants will only be built from 2040 onwards. A lower level of support (halving of the above values) is insufficient to include RC-BCC production in the optimised network. Besides, a recycling stress scenario shows that recycling remains preferred over landfilling even under unfavourable framework conditions (variable/fixed/opening costs +50%, revenues -50%, landfilling costs reduced to 65 €/t). Still, only 3.1% of all pd-AAC is landfilled in this scenario.

4.6.3 Discussion and limitations

This study designs a cost-minimal pd-AAC recycling network in Germany, using large recycling plants and no RC-BCC production due to high costs. Substantial savings can be achieved compared to landfilling.

The input data strongly influences the results. However, the input data is based on calculations and assumptions but not on field data, which is not yet available. Therefore, the results are associated with uncertainties. However, it could be shown in scenarios that recycling is preferable to landfilling even under considerably less favourable conditions. The primary implication that pd-AAC recycling should be fostered is unlikely to change.

Ecological aspects are not considered in the optimisation. However, sustainability gains importance in practical decisions. Thus, a multi-criteria objective function balancing costs and environmental factors like GHG emissions can be a reasonable enhancement. In contrast to the costs, minimising GHG emissions could lead to smaller recycling plants and lower transport distances. Additionally, the ecologically beneficial RC-BCC production could be used more extensively.

5 Implications

This section provides implications for each research field presented in Section 3. The implications are based on the results of Studies A to F.

5.1 Quantification of the regional pd-AAC volumes in Germany and Europe

The results of Studies A and B show that the amount of pd-AAC is increasing sharply in Germany and Europe. As the use of AAC in Germany and Europe only reached significant amounts in the 1960s and early 1970s, there was hardly any pd-AAC until 2000. Since then, a substantial increase in pd-AAC volumes has begun. In 2020, total German pd-AAC volumes reached around 1.2 Mm^3, while the European volumes were approximately ten times higher. This increase is expected to continue over the next few decades. Thus, total annual pd-AAC quantities could reach over three times their current value by 2050. On the one hand, this means that the recycling of pd-AAC has significant potential, as the high volumes in the future can be used to substitute large amounts of primary raw materials in different recycling options. On the other hand, the expected rise also means that pd-AAC recycling urgently needs to be established so that future pd-AAC volumes do not further burden the scarce landfill capacities.

Therefore, manufacturers and recycling companies should continue to technically advance the currently researched recycling options to reach a TRL of 7-9 as soon as possible. Due to the high expected future pd-AAC amounts and limited substitution rates in most recycling options, the focus should be on multiple ecologically and economically promising options (Section 5.2). The technical development and establishment of different recycling options are vital, as alternative circular economy strategies for AAC are currently challenging to implement or can only be realised with fundamental, long-term changes (Section 2.3.1). However, the sharp increase in pd-AAC volumes is imminent. Policymakers should support establishing pd-AAC recycling by simplifying legal requirements. For example, regulations for collection and transport (KrWG/02.03.2023) could be reduced. Moreover, pd-AAC should be legally considered as a secondary raw material (not as waste) at the demolition site, and regional differences in legal requirements for recycling should be minimised.

The regional distribution of pd-AAC volumes shows concentrations in agglomerations and large cities. Therefore, pd-AAC recycling products can be used regionally, allowing short transports, low costs, and avoided GHG emissions. Additionally, considerable regional differences in the waste amounts indicate a high optimisation potential for designing a recycling network. Recycling companies can use the results of the regional distribution and future development of the pd-AAC volumes to achieve greater planning security and improve their business model and investments when establishing pd-AAC recycling.

5.2 Ecological and economic assessment of pd-AAC recycling

Studies C and D identified the environmental benefits of pd-AAC recycling compared to landfilling by conducting LCAs. It was shown that multiple recycling options perform better than landfilling for all considered environmental categories. Closed-loop recycling of pd-AAC in AAC production performs remarkably well. Still, the open-loop recycling options using pd-AAC in producing light mortar, LWAC, and shuttering blocks also show high environmental savings potential. This environmental benefit is mainly the result of significant substitution credits due to avoided primary inputs such as cement, quicklime or expanded clay. Using the example of GHG emissions, the calculated savings reach up to 0.50 kg CO_2-Eq/kg pd-AAC in the case of closed-loop recycling and up to 0.43 kg CO_2-Eq/kg pd-AAC in the case of open-loop recycling. As a result, a total saving of 280,000 t CO_2-Eq per year can already be achieved in Germany today by establishing a pd-AAC recycling system. The ecological savings potential will be even more significant in the future due to increasing volumes and could rise to three times the current value. At the European level, the savings potential could be about ten times higher than at the German level, based on the pd-AAC volume proportion. Thus, the GHG savings through pd-AAC recycling could theoretically grow to more than 8 Mt CO_2-Eq per year in Europe in the future. These savings would reduce the total current 230 Mt CO_2-Eq per year of embodied emissions in the building sector in Europe by around 3.5% (IPCC, 2022).

Political support for market acceptance of secondary materials should be increased to help achieve these significant ecological savings through pd-AAC recycling. For example, only building materials with a significant recycling content could be used in public construction projects. Additionally, the construction process of buildings should be adapted through political guidelines and innovations by construction companies to recover homogeneous products from the dismantling process at the end of the building’s life more easily. Such a dismantling process would minimise the proportion of impurities, reducing sorting and purifying efforts during recycling. Fewer impurities can also improve the quality of recycled products, enabling higher substitution rates. These rates are still relatively low, particularly in the closed-loop recycling of pd-AAC. Moreover, an improved (partial) dismantling process is also a prerequisite to facilitate AAC reuse and other circular economy strategies of the category “extend lifespan of product and its parts” (Section 2.3.1).

The findings on the economic performance of pd-AAC recycling from Study E show that recycling is also worthwhile from an economic point of view. The processing of pd-AAC can be carried out with established technologies so that the costs are reasonable. In contrast, landfilling is associated with high costs of about 100 €/t on average in Germany, which are likely to increase further. Therefore, recycling as an alternative to landfilling can be economically viable even for smaller plants with capacities of 25,000 t/a. The potential economic savings from recycling in larger plants are even more striking due to significant economies of scale. Recycling in plants with capacities of 250,000 t/a can save about 70 €/t compared to landfilling. The operation of such large recycling plants will become more practicable as soon as pd-AAC volumes further increase.

Therefore, recycling companies should offer pd-AAC recycling as soon as it is technically established in larger plants. The companies could charge a fee for accepting the pd-AAC equivalent to the landfill prices and sell the final recycling products. This combined revenue significantly exceeds the recycling costs, ensuring a sufficiently high margin.

5.3 Assessment of the RC-BCC production as a new and innovative recycling option for pd-AAC

The established concrete recycling focuses on recovering the aggregates (sand and gravel). However, the pd-AAC contains no aggregates that can be recovered during recycling. Therefore, the assessed new and innovative recycling approach for pd-AAC focuses on cement. The cement as a binding agent is crucial for AAC's strength and is thus required in significant quantities for primary production. Cement can account for up to a third of the total input, depending on the density class of the AAC (Study C). Conventional cement production is associated with high GHG emissions, making the cement input in AAC production the largest contributor to its total GHG emissions. These significant emissions result in an enormous ecological potential for pd-AAC recycling when recovering the cement content. Due to the difference between concrete and pd-AAC recycling and the novelty of the approach, this new recycling option was extensively assessed in this dissertation, especially in Studies D and E.

Compared to other recycling options, this new option requires additional treatment after the pd-AAC processing. This treatment includes the emission-intensive calcination and the energy-intensive sintering of the cement clinker. Nevertheless, this new recycling option should be further researched. After all, it is a high-quality recycling process that can produce a multipurpose material whose alternative primary production is associated with very high GHG emissions. Additionally, the RC-BCC production can process fine material accrued in large quantities during pd-AAC crushing. Many other recycling options focus exclusively on the use of coarser pd-AAC granulate. Moreover, closed-loop recycling is possible if the RC-BCC is used in AAC production as a substitute for primary Portland cement. The GHG savings of RC-BCC production compared to the landfilling of pd-AAC reach about 0.34 kg CO_2-Eq/kg pd-AAC, which is lower than the previously mentioned maximum savings. However, the process can reach much higher savings when using renewable energies. In this case, the achievable GHG savings increase significantly to 0.77 kg CO_2-Eq/kg pd-AAC and are even higher than with all other recycling options for pd-AAC since very CO_2-intensively produced primary Portland cement can be substituted by the RC-BCC. With pd-AAC volumes rising sharply in the future, RC-BCC production from pd-AAC is a vital addition to the other recycling options to find recycling routes for the large mass flows and reduce landfilling.

Significant technological progress is still required to make the ecologically promising RC-BCC production economically viable. Therefore, policymakers should continue to support research into cement production from secondary inputs. Subsidies for RC-BCC production are also necessary to foster this recycling option. The primary production of cement clinker is currently allocated 50% of the required emission allowances free of charge (UBA, 2022). Thus, the ecological advantage of secondary production is not fully reflected in the economic comparison. Therefore, policymakers should reduce the free allocation of emission allowances for primary cement clinker production.

Furthermore, costs can be reduced by establishing RC-BCC production on a larger scale. Recycling companies should aim to use other secondary input materials in RC-BCC production besides pd-AAC to increase the scale of the RC-BCC production and reduce the catchment area and, thus, transport costs. For instance, including concrete from building demolition in RC-BCC production is conceivable. A larger recycling plant has substantial economies of scale and higher energy efficiency, making RC-BCC production more efficient and affordable. However, process and recipe adaptation to the changed input would also be necessary.

5.4 Optimal design of a pd-AAC recycling network

The results from Study F show that an optimally designed pd-AAC recycling network prefers recycling plants with high capacities over low capacities under an economic objective function. Consequently, the transport costs account for the largest share of the total costs of this recycling network. Moreover, no RC-BCC plants are built due to high expenses (Section 5.3). Further recycling plants will be opened with increasing pd-AAC volumes in the future to reduce transport distances and be able to cope with the volumes. The computational results show that landfilling is hardly used in the optimal network. The avoided landfilling indicates that a cost-minimal recycling network design can be consistent with environmental objectives. The optimal recycling network is considerably less costly than the status quo pd-AAC treatment (assumed ⅔ landfilling and ⅓ recovery). However, the cost difference between the recycling network and the status quo treatment is not uniform over time but significantly increases in the future. The status quo treatment costs are less than 50 M€/a higher than the total costs of the cost-minimal recycling network in the 2020s. However, this difference will increase to more than 300 M€/a at the end of the 2040s. Consequently, recycling companies and building material manufacturers should aim to establish a pd-AAC recycling network as soon as possible to avoid the immense future costs caused by landfilling.

However, pd-AAC recycling has not yet been established, causing uncertainty among recycling companies and building material manufacturers. From the perspective of recycling companies, there is no demand for pd-AAC power or granulate yet. Thus, a pd-AAC processing might not be worthwhile. From the perspective of building material manufacturers, there is no possibility of using homogenous and purified pd-AAC as a secondary input, so they do not adapt their production. Therefore, recycling companies and manufacturers should cooperate to reduce the uncertainties for both sides, for example, through trial supply with incremental realisation of new production recipes. If the collaboration is successful, long-term supply contracts can provide security for both sides. Moreover, policymakers could address the dilemma by providing economic support for building recycling plants or guaranteeing minimal prices for processed pd-AAC to reduce the economic risks for the recycling companies.

The total economic savings potential of establishing pd-AAC recycling in Germany can be estimated at around 4,600 M€ until 2050. Thus, the finding from Study E that pd-AAC recycling is desirable from an economic point of view can be confirmed if the recycling plants' locations, capacities, and transport at all stages are optimally chosen. In that case, recycling is more attractive than landfilling despite additional transport costs. However, policymakers should ensure that all stakeholders act as closely as possible to this social optimum by exchanging information. Independent optimisation by the stakeholders involves the risk of inefficiencies, reducing the economic savings potential.

6 Conclusion

6.1 Summary

This dissertation investigates the quantitative, ecological, and economic potential and the possible design of a circular economy for AAC. Past, current, and future pd-AAC volumes in Germany are estimated using a newly developed methodology since pd-AAC volumes are not currently recorded in waste statistics or explored in existing scientific studies. This methodology uses historical AAC production, construction activity statistics, regional AAC popularity, and buildings' lifetimes to calculate pd-AAC volumes in Germany on a district area level from 1950 to 2050. The estimated volumes have remained very low for a long time. By the turn of the millennium, hardly any pd-AAC volumes had been generated, and estimated pd-AAC volumes only sum up to around 160,000 m³ in 2000. However, the volumes already reached approximately 500,000 m³ in 2010 and around 1.2 Mm³ in 2020. Future projected volumes continue to increase steadily and are expected to reach 2.3 Mm³ (2030), 3.4 Mm³ (2040), and 4 Mm³ (2050), thus exceeding the current production volumes of AAC. The results are validated with two approaches, including a comparison with data from the literature. The regional distribution of the calculated volumes shows that the highest volumes occur in agglomerations such as Berlin, Hamburg, Munich, Bremen, Hanover, Cologne, Frankfurt, and Stuttgart.

An extension of this estimate to Europe results in a country-specific forecast of pd-AAC volumes until 2030. According to this estimate, the highest volumes in 2020 are expected in Russia (3.9 Mm³), Poland (1.8 Mm³), Germany (1.2 Mm³), Ukraine (1.2 Mm³), and the UK (0.7 Mm³). The total European volumes will add up to about 12 Mm³ in 2020 and will almost double to approximately 22 Mm³ by 2030, according to the forecast.

Recycling these pd-AAC volumes can lead to significant environmental improvements compared to the currently predominant landfilling, as shown by an LCA. Based on the functional unit of 1 kg pd-AAC and including the zero burden and avoided burden approach, the environmental impact of potential pd-AAC recycling options in the production of AAC, floor screed, concrete, LWAC, light mortar, and shuttering blocks is assessed. Closed-loop recycling of pd-AAC in AAC production can achieve the highest GHG emission savings of 0.5 kg CO_2-Eq/kg pd-AAC compared to landfilling. Open-loop recycling in producing light mortar, LWAC, and shuttering blocks also shows substantial ecological advantages over landfilling. These advantages apply to all investigated environmental impact categories, not only GHG emissions. The ecological savings are achieved due to the non-energy-intensive mechanical pd-AAC processing and high rewards for substituted primary material. Overall, savings of around 3.5% of the total current embodied emissions in the building sector in Europe can be reached by recycling the high pd-AAC volumes in the future.

From an economic point of view, pd-AAC recycling can also be advantageous. The recycling costs vary greatly depending on the size of the recycling plant. Around a capacity of 25,000 t/a input, total costs sum up to 100 €/t input, equalling the average German landfilling costs. For plants with the largest capacities of up to 250,000 t/a input, costs can drop to about 30 €/t input. Therefore, pd-AAC recycling shows a strong economic incentive. With significantly increasing volumes in the future, such larger plants will be necessary in any case, further supporting the financial advantage of recycling.

The production of RC-BCC from pd-AAC as a substitute for primary Portland cement, e.g. in the production of AAC, is also assessed ecologically and economically. From an ecological perspective, this recycling option is found to be advantageous over landfilling in several environmental impact categories, including global warming. With conventional natural gas firing of the process, savings of 0.34 kg CO_2-Eq/kg pd-AAC can be achieved, so RC-BCC production performs slightly worse than the previously described recycling options. The lower savings are due to significantly higher energy expenditures than mechanical pd-AAC processing alone. However, if RC-BCC production is powered exclusively by electricity from renewable sources, the savings increase to 0.77 kg CO_2-Eq/kg pd-AAC. Thus, the savings would be higher than those of all other recycling options. Overall, the ecological potential of the RC-BCC production as a pd-AAC recycling option is enormous. However, the RC-BCC production is still significantly costlier than primary cement production. The economic assessment results show that costs of mechanical processing and subsequent RC-BCC production sum up to nearly 500 €/t RC-BCC in large-scale recycling plants. Thus, the RC-BCC must reach sales prices of around 430 €/t to match current pd-AAC landfilling costs. However, primary Portland cement is currently only sold for approximately 150 €/t.

A cost-minimised German pd-AAC recycling network confirms the economic advantage of recycling over landfilling. The cost minimisation leads to almost all pd-AAC (99.9%) being recycled, for which primarily large recycling plants should be built to benefit from economies of scale. However, RC-BCC production is not used in the optimised network due to high costs. The total costs are mainly driven by transport and fixed costs, while revenues offset approximately 20% of all expenses. Considering inflation and discounting, total costs are around 2,200 M€ for the entire period up to 2050. However, if the status quo of pd-AAC treatment (⅔ landfilling and ⅓ recovery) is maintained, the costs would amount to around 6,800 M€ by 2050 due to increasing pd-AAC volumes and high inflation of landfilling costs. Therefore, the savings potential by establishing an optimally designed German pd-AAC recycling network is 4,600 M€ or 68%.

6.2 Limitations

The pd-AAC volume assessment (Studies A and B) does not consider imports and exports but is based entirely on domestic AAC production. In Germany (Study A), exports of products such as AAC are significantly higher than imports (BBS, 2019), so domestic AAC production could partly be exported to foreign countries. However, transport over longer distances is usually not economically viable for building materials and thus, exports account for only a small part of domestic production. Additionally, the pd-AAC volume estimation is based on lifetime functions derived from very sparse literature values. Thus, the lifetime is subject to general uncertainties. In particular, extending the building use after its expected end of life due to refurbishments and renovations is not explicitly considered. The presented pd-AAC assessment might be slightly higher than the actual emergence. However, the sensitivity analysis on lifetimes shows that the substantial increase in pd-AAC volumes in the following decades is only shifted temporarily with changed building lifetimes. In any case, pd-AAC volumes will increase in the future, even if building lifetimes are extended. At the European level, comprehensive data on historical AAC production are only available for Germany and the UK. The volumes of all other countries are calculated based on the current market size of AAC and are subject to higher uncertainties, especially regarding their future development.

The environmental assessment of pd-AAC recycling (Study C) considers credits for substituting primary materials. Incorporating substitution credits in the assessment is based on the assumption that the same quality as in primary production can be achieved by using pd-AAC. However, the quality of recycled products is usually lower than that of primary materials since impurities cannot be ruled out. In the case of pd-AAC, for example, impurities from wallpaper, dowels, screws, or tiles are possible. However, when using pd-AAC for producing building materials, an extensive framework of standards and specifications on physical and chemical properties ensures a defined quality. If these standards and specifications can be met, the equal quality assumption can be justified, even if recycled material with potential impurities was used in the production. Thus, considering a full substitution credit for avoided primary production is appropriate.

Furthermore, it should be mentioned that the investigated and compared pd-AAC recycling options are not yet established in practice but are currently under research on a laboratory scale or, at most, on a large-size pilot plant scale. Therefore, data from literature and LCA databases is used while field data is unavailable. Thus, in the case of a large-scale implementation of the recycling options, various aspects, including energy efforts and input quantities, could still change, influencing the ecological assessment. However, efforts for processing pd-AAC tend to be overestimated rather than underestimated by the process and data choices. Thus, the demonstrated environmental benefits of pd-AAC recycling are not expected to be significantly lower once the recycling options are implemented in practice.

In the economic assessment (Study E), the landfilling costs play a significant role as an alternative treatment option to recycling. Compared with average landfilling costs of around 100 €/t, even comparatively smaller recycling plants can operate profitably. However, these costs vary regionally. Therefore, the assessment results should be understood as a general consideration without reference to specific sites. It is demonstrated that pd-AAC recycling is worthwhile under average German conditions. Additionally, the mechanical processing and RC-BCC production processes must verify in practice whether they can achieve the required material qualities with the assumed process steps and machines. So far, no field data on the processes is available. Moreover, vital economic aspects, such as operating labour requirements and the proportion of impurities, are based on theoretical calculations. Therefore, the assumed values must also be verified when implementing pd-AAC recycling. The economic assessment generally uses conservative estimations, so the total costs can be expected to be lower in practice.

The production of RC-BCC from pd-AAC as a new recycling option (Study D) is currently at a TRL of 4-5, as the implementation is at a laboratory scale. Accordingly, the ecological assessment of the process is still based on experimental data. Moreover, RC-BCC requires significantly more time for the hydraulic reaction than Portland cement, which must be considered in the case of substitution. Thus, process adaptations in the form of longer process times or the use of chemical accelerators are necessary for AAC production with RC-BCC content (Equations 2.2-2.4 would be affected and proceed slower). Both can, in principle, lead to higher costs and possibly increase the environmental impact. It should also be considered that the quality of the RC-BCC depends on the input and, thus, the purity of the pd-AAC. While the RC-BCC production can handle impurities in the pd-AAC, the portion of hydraulically effective clinker in the RC-BCC reduces with higher amounts of impurities in the input. Consequently, more RC-BCC would have to be used to achieve the same strength in the final product, which may reduce savings by substituting primary cement.

Uncertainties for the optimal pd-AAC recycling network design (Study F) are considered via sensitivity analysis and scenarios but not directly integrated into the model via stochastics. Such uncertainties are much more prominent in reverse logistics than in primary product networks because waste volumes and material qualities usually vary much more. Therefore, integrating probability distributions for pd-AAC quantities or impurity amounts could be a helpful extension. However, due to the expected significant increase in pd-AAC volumes, the focus is on multi-period modelling to look more closely at the changing network structure. An additional stochastic extension could significantly reduce the solvability and should be modelled and optimised separately. Additionally, there is no well-founded data on possible probability distributions of pd-AAC volumes since the pd-AAC volume assessment determines precise volume estimates and no intervals, apart from a general sensitivity analysis. Moreover, the pd-AAC volumes and the processing plants' capacities are considered annually or even for periods of several years. However, there is a considerable fluctuation of pd-AAC volumes during the year, as the construction (and deconstruction) industry is much less active in winter than in summer. If the recycling plants' capacities are fully utilised, this non-uniform distribution of pd-AAC volumes will have to be stored over several months, incurring additional costs. Alternatively, the plants would need higher capacities to handle the higher volumes during heavier deconstruction activity. However, this also comes at a higher cost.

6.3 Outlook

Future research can transfer this dissertation's methodology for assessing pd-AAC volumes to other building materials, including concrete, clay bricks, sand-lime bricks, lightweight concrete or timber. Moreover, other countries or regions can be investigated. This methodology transfer can lead to a more comprehensive knowledge of future C&DW volumes and, thus, to better planning of C&DW recycling. Moreover, further research on buildings' lifetimes can increase the precision of the assessment. For example, a differentiation of individual building components or a more profound differentiation of buildings (single-family vs. multi-family house, terraced vs. detached house) could be considered. Furthermore, considering the changing density of building materials over time would be a meaningful model extension. In general, the densities of building materials have decreased in recent decades due to increasing thermal insulation requirements.

Future research on the ecological and economic assessment of pd-AAC recycling should use improved data from pilot plants to enhance the precision of the assessment results. It is also essential to increase substitution rates of primary raw materials to deal with rising pd-AAC volumes in the future. This way, significant ecological savings and improved resource efficiency can be achieved. Such an increase in substitution rates requires research on technical feasibility.

This dissertation used the framework and guidelines of DIN EN ISO 14040:2021-02 and DIN EN ISO 14044:2021-02 for the ecological assessment. The focus was on comparing end-of-life treatment options. Impacts from the production, construction and use phases were not included in the assessment for simplification. The ecological assessment should be extended in further research, and a standardised EPD following DIN EN 15804:2022-03 should be prepared. In this way, a more comprehensive ecological assessment would be possible. Moreover, comparability with other EPDs of construction products would be achievable because the standard contains predefined modules for assessment and specified impact categories.

The RC-BCC production has emerged as a promising recycling option for pd-AAC. It enables high ecological savings due to the possibility of substituting primary cement, although it also involves high costs. In future research, this recycling option should be investigated on a larger scale in a pilot plant. This scaling will allow process optimisations and the collection of more precise data. Additionally, the research could further advance the use of RC-BCC as a substitute for Portland cement by investigating the slower hydraulic reaction and exploring options for mitigating it.

In future research, the pd-AAC network modelling and optimisation can be used to optimise recycling networks in other regions or networks of other (construction) materials. The model best fits a waste product with significantly changing volumes in the future (increase or decrease), as in the pd-AAC case, so that the multi-period consideration is of particular added value. Finally, the results of the ecological assessment could also be included in the network optimisation. Either the objective function could be changed to minimise ecological impacts or a multi-criteria approach could be implemented to balance costs and different environmental impacts simultaneously. However, since ecologically inferior landfilling is almost completely avoided in the cost-optimised network, the overall result that a comprehensive pd-AAC recycling network should be established is not expected to change.

The full potential of a circular economy for AAC can only be realised if other strategies are pursued in addition to the recycling considered in this dissertation. New housing concepts should be researched independently of building materials to address the circular economy strategy of “smarter product use and manufacture”. Moreover, the strategy “extending the lifespan of a product and its parts” can be reached by research on improved demolition processes and an adaptation of regulations to enable reusing AAC masonry units. Additionally, approaches and incentives for refurbishing buildings should be pursued, and new applications for repurposed AAC should be identified.

Despite the limitations and the need for further research, this dissertation makes a valuable scientific contribution to fostering the development of a circular economy for AAC. The high quantitative potential shows the importance of establishing pd-AAC recycling. Additionally, the ecological potential of pd-AAC recycling promises high savings in GHG emissions and reductions in other environmental impacts if the most promising recycling options are implemented. Moreover, numerous recycling options are already economically viable today, and high economic savings can be achieved if this dissertation’s result for an optimally designed pd-AAC recycling network is realised.

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II Companion Articles

Overview of Related Publications

Study A

Steins, Justus J.; Volk, Rebekka; Schultmann, Frank (2021): Modelling and predicting the generation of post-demolition autoclaved aerated concrete (AAC) volumes in Germany until 2050. In: *Resources, Conservation and Recycling 171*, S. 105504. DOI: 10.1016/j.resconrec.2021.105504.

Study B

Steins, Justus J.; Volk, Rebekka; Schultmann, Frank (2022): Post-Demolition Autoclaved Aerated Concrete: Recycling Options And Volume Prediction In Europe. In: *Ecocity World Summit 2021-22 Conference Proceedings*. DOI: 10.5445/IR/1000153325.

Study C

Volk, Rebekka; Steins, Justus J.; Kreft, Oliver; Schultmann, Frank (2023): Life cycle assessment of post-demolition autoclaved aerated concrete (AAC) recycling options. In: *Resources, Conservation and Recycling* 188, S. 106716. DOI: 10.1016/j.resconrec.2022.106716.

Study D

Stemmermann, Peter; Volk, Rebekka; Steins, Justus J.; Beuchle, Günter (2023): Recycling belite cement clinker from post-demolition autoclaved aerated concrete – assessing a new process. *Submitted for publication in a scientific journal*.

Study E

Steins, Justus J.; Volk, Rebekka; Beuchle, Günter; Yarka Reddy, Pallavi Reddy; Sandaka, Gourisankar; Schultmann, Frank (2023): Economic assessment of post-demolition autoclaved aerated concrete (AAC) recycling and belite cement clinker production. *Submitted for publication in a scientific journal*.

Study F

Steins, Justus J.; Ruck, Manuel; Volk, Rebekka; Schultmann, Frank (2023): Optimal design of a post-demolition autoclaved aerated concrete (AAC) recycling network using a capacitated, multi-period, and multi-stage warehouse location problem. *Submitted for publication in a scientific journal*.

The information on the publication status of the Studies reflects the situation at the time of submission of this dissertation (05.12.2023).

A Modelling and Predicting the Generation of Post-Demolition Autoclaved Aerated Concrete (AAC) Volumes in Germany Until 2050

Abstract[1]

Autoclaved aerated concrete (AAC) has a porous structure and excellent thermal properties. Therefore, it is a frequently used building material for masonry units, prefabricated reinforced components and lightweight mineral insulation boards with increasing popularity. Post-demolition AAC is currently mainly disposed of in landfills. Decreasing landfill capacities, the legal framework, and protection of primary resources require developing recycling options for AAC. However, so far, no overall recycling of post-demolition AAC has been established yet. Only AAC primary process waste is recirculated or discharged, for example as an absorbent for chemicals or animal bedding. For high-quality post-demolition AAC recycling, only minimal information about recyclable volumes and their regional distribution is available. Therefore, a new dynamic retrospective and prospective AAC quantification model on a national level using AAC production, construction activity, AAC popularity, and buildings' lifetimes is developed to assess geographically distributed current and future post-demolition AAC volumes. This new model is applied to quantify post-demolition AAC volumes in a case study for Germany in the period between 1950 and 2050. For validation, the results are compared with two different approaches and with data from the literature. The AAC quantification allows decision support for the circular management of AAC during its life cycle and along its value/supply chain regarding design, location planning, logistics, production, and recycling.

A.1 Introduction

Growing awareness of climate change and sustainability fosters construction and demolition waste (C&DW) recycling worldwide. Worldwide, C&DW exceeded 3 billion tons in 2012 (Akhtar & Sarmah, 2018) and the global concrete production used between 25.9 to 29.6 billion tons of sand (Peduzzi, 2014). Around 220 million metric tons of C&DW was generated in Germany in

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[1] This section includes the article "Modelling and predicting the generation of post-demolition autoclaved aerated concrete (AAC) volumes in Germany until 2050", developed by Rebekka Volk, Frank Schultmann, and myself. The article was published as Steins et al. (2021) in the journal "Resources, Conservation & Recycling" in 2021. The supplementary material can be found on the journal website.

2017 (Destatis, 2019e) and increased significantly in recent years (Kreislaufwirtschaft Bau, 2018).[2] Besides, landfill capacities, especially in Germany, diminish rapidly.[3] The construction and demolition (C&D) sector plays an essential role in terms of greenhouse gas (GHG) emissions and resource consumption due to its large mass flows and in reaching the UN sustainable development goals 11 (sustainable cities), 12 (responsible consumption and production), and 13 (climate action) (UN, 2020).

Autoclaved aerated concrete (AAC) has been a well-known building material for almost a century. The raw materials for AAC are quartz sand, cement, quicklime, anhydrite or gypsum, the aerating agent aluminium powder/paste and water (DIN 20000-404:2018-04; DIN EN 771-4:2015-11; Kitsch & Rehrmann, 2012). Most AAC products in Germany are installed in terms of masonry units in residential buildings, especially in one- and two-family houses (Destatis, 2019a; UBA, 2019). AAC has a porous structure where pores make up 60 to 85 vol.-% of AAC with pore sizes ranging from millimetres to nanometres (Anders, 2018), while crystalline calcium silicate hydrates are the main constituent by mass (Straube et al., 2008). AAC has a low density and excellent thermal insulation properties that outperform other monolithic mineral building materials like classical clay brick, calcium silicate units, or concrete. A building's envelope made from AAC results in considerably lower operational heating and cooling demand of the building. And, in contrast to layered or composite materials, AAC is a mono-material. In 1950, the production of AAC began in Germany (A. Müller, 2018; UBA, 2019). In 2018, 23% of the annually constructed residential buildings in Germany were built using AAC as masonry (Destatis, 2019b). This share increased by two percentage points[4] over the last five years (Destatis, 2014). AAC is the second most popular building material for residential buildings in Germany after clay bricks (basis: number of constructed buildings). The current annual AAC production sums up to 3.1 million m³ of AAC masonry units and 1.6 million m² of AAC panels and floorboards in Germany (GENESIS, 2019). The annual AAC production (2019) amounts to at least 16 million m³ in Europe (EAACA, 2020) and to 11.6 million m³ (2017) in Russia (Grinfel'd et al., 2018). Worldwide, there is an estimated production capacity in 2018 of 450 million m³ for non-reinforced AAC blocks (Fouad & Schoch, 2018). With average[5] GHG emissions of 0.4361 kg CO_2-eq/kg AAC the GHG emissions from AAC production range between 49 and 98 million tons CO_2-eq worldwide, depending on production capacity utilization.

C&DW recycling in general and AAC recycling in particular is essential to preserve natural deposits of sand, gravel, lime, and other materials necessary for the construction industry and to reduce their GHG emissions. "From an 'urban mining' perspective, the building stock can be

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2 In 2014 (2016), around 202 (215) million tons of C&DW were generated in Germany Kreislaufwirtschaft Bau (2018).

3 Between 2004 and 2017 the number of open landfills in Germany decreased by 45.9% from 1999 to 1082 Destatis (2019).

4 This corresponds to a relative increase of around 9.5%.

5 Ecoinvent 3.6 dataset for "autoclaved aerated concrete block production"; System model: Allocation, cut-off; geography: Rest-of-World; impact assessment method: IPCC 2013, climate change – GWP100; assumed density of 500 kg/m³ (Section A.4.1).

seen as a repository of natural resources" (Ortlepp et al., 2016) and recycling aims at using this repository instead of landfilling it. However, current post-demolition AAC volumes in Germany are neither recorded in statistics nor recycled on a large scale. Therefore, missing knowledge on the accumulated stock and current/future outflows is a crucial issue that hampers an effective circularity of AAC. The material production and end-of-life treatment (besides construction, use phase and demolition) is an important starting point for GHG reduction and therefore climate mitigation in the building sector (Mata et al., 2020).

This study develops a new quantification model (Section A.3) to assess current and to predict future geographically distributed and temporally differentiated post-demolition AAC return flows on national level for the first time. The prediction until 2050 allows precise economic and ecological assessment, capacity planning and the implementation of recycling options and a recycling network for AAC. Temporal differentiation of post-demolition AAC volumes is essential to develop accurate capacity plans for recycling plants and estimate associated transportation and handling. Furthermore, transportation of post-demolition AAC has a significant economic and environmental impact on recycling options, so the geographical distribution of the post-demolition AAC volumes is vital. Since AAC is frequently used for the main structure of buildings, it has a long and non-deterministic lifespan. Consequently, current production volumes do not determine current post-demolition AAC volumes. While the quantification model is production-, construction-, popularity-[6] and lifetime-based, we use a stock-based and a waste-based validation approach. In a case study for Germany (Section A.4), the developed quantification model is demonstrated. Then, the results for the post-demolition AAC volumes in Germany in the period 1950 to 2050 are presented, a sensitivity analysis and validation is conducted, and shortcomings are discussed (Section A.5). Finally, the results are concluded and an outlook is given (Section A.6).

A.2 Literature and state-of-the-art

This section reviews relevant literature on urban mining and material stock quantification as well as material inflow and outflow analysis on the national level. Lanau et al. (2019) and Augiseau and Barles (2017) extensively analyse studies on material flow analyses in the C&D sector and the built environment stock. Augiseau and Barles (2017) provide a structured overview of 31 construction material flows and stock studies including their purpose, time frame, geographic scope, and methodology (Augiseau & Barles, 2017). Besides, they subdivide and classify the investigated studies into six methodological approaches: static bottom-up flow analysis (1), static top-down flow analysis (2), bottom-up stock analysis (3), dynamic retrospective or prospective flow analysis using a flow-driven model (input flows) (4), dynamic retrospective or prospective flow analysis using a stock-driven model (5), top-down retrospective or prospective stock analysis using a flow-driven model (6). Additionally, Lanau et al. (2019) give

[6] Regional AAC popularity is considered in the model. For further information see Section A.3.1.

an overview of 249 built environment stock studies (published between 1985 and 2018) including bibliometrics and analysis of geographic scope, methodology, and considered materials. The studies mainly focus on building materials in general, non-metallic minerals like concrete, brick, and cement or metal ores like steel, copper, and aluminium. Overall, Augiseau and Barles (2017) and Lanau et al. (2019) summarise the state-of-the-art on quantifying both stock and flows of buildings on the regional and national levels. However, in all studies, AAC is neglected or subsumed under concrete or gypsum stock or mass flows.

Due to the scope of this study (Sections A.1 and A.3), approaches of the dynamic retrospective or prospective flow analysis using a flow-driven model (input flows) (approach 4 according to Augiseau & Barles, 2017) are further reviewed. Available studies investigate different case studies of Germany (Deilmann, 2009; Deilmann et al., 2014; Ortlepp et al., 2016; Schiller et al., 2010; Schiller et al., 2015) , Japan (Hashimoto et al., 2007, 2009), the US (Kapur et al., 2008), and France (Orléans) (Serrand et al., 2013). Mostly, methodological approaches/frameworks (Daxbeck et al., 2009; Hashimoto et al., 2009) or a general overview of urban material flows (Deilmann, 2009; Hashimoto et al., 2007; Schiller, 2007; Serrand et al., 2013) are developed. In addition, Xia et al. (2020) and Xiao, Li, et al. (2012)/Xiao et al. (2016)/Xiao, Xie, and Zhang (2012) investigate concrete and general C&DW recycling in China.

Overall, there is no study focusing on quantification of post-demolition AAC volumes on national level. Thus, the quantification of AAC stock and post-demolition AAC flows is developed in this study. The quantification model is applied to the case study Germany and validated using anthropogenic stockpile studies for Germany (Deilmann et al., 2014; Schiller et al., 2015). This is the only currently available literature with certain thematic proximity to post-demolition AAC volumes. However, the publications focus on the total stockpile at one point in time and do not differentiate the temporal and geographical distribution of AAC material flows.

A.3 Methodology

We develop a quantification model that uses the historical production of AAC, the construction activity, regional popularity of AAC and lifetime distribution of buildings as inputs to quantify current and future geographically distributed and temporally differentiated post-demolition AAC volumes (Figure A.1, Section A.3.1). Furthermore, we validate the results with a (1) stock-based and (2) waste-based approach (Section A.3.2). See Table A.1 for a comparison of the model and the validation approaches.

Table A.1: Presentation of the quantification model and the validation approaches used to estimate the post-demolition volume of AAC in Germany for the period 1950 to 2050.

	quantification model	**stock-based validation approach (1)**	**waste-based validation approach (2)**
methodology	dynamic retrospective and prospective flow analysis using a flow-driven model (input flows), own model	top-down (data on building types and stockpile from literature), own calculation based on literature data	top-down (waste statistics, literature estimates), consultation of literature
subject of investigation	production, construction activity, popularity, lifetime	building stock and its changes	demolition wastes
temporal differentiation	annual level for the period 1950 to 2050	annual level for sporadic years	annual level for sporadic years
geographical distribution	NUTS-3[7]	national level (stockpile data at NUTS-3 level)	national level
results	continuous post- demolition AAC volumes	sporadic post-demolition AAC volumes	sporadic post-demolition AAC volumes
use of results	assessment of post-demolition AAC volumes	validation	validation

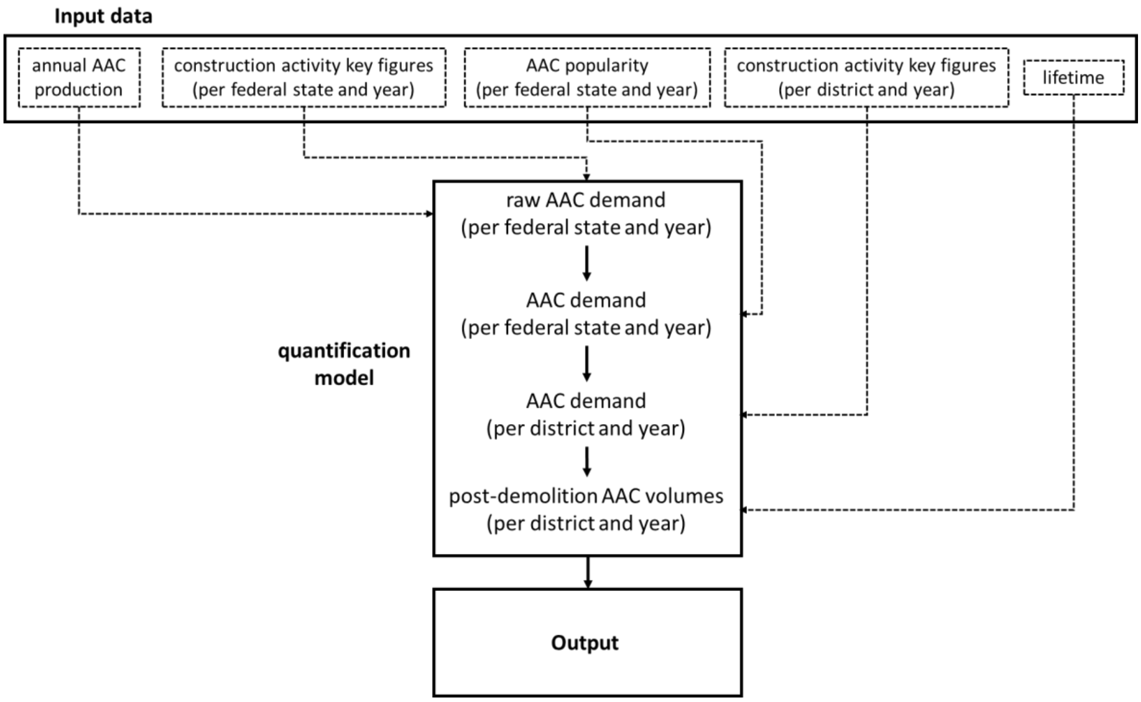

Figure A.1: Graphical overview on the methodology of the quantification model.

[7] NUTS is a hierarchical classification system for identifying and classifying the subdivision of countries for official statistics in the Member States of the European Union. The NUTS-3 level includes 401 administrative districts in Germany.

A.3.1 Quantification model of the post-demolition AAC volumes

Lanau et al. (2019) describe a common problem of outflow estimation as intended in this paper: „Data on inflows of material are usually available […] over [a] long period of time. But outflow data are much more difficult to track, and are thus often calculated through [an] estimated lifetime of [a] product […]“. This statement also applies to the available data concerning post-demolition AAC volumes and the procedure of our quantification model.

We assess the geographically distributed and temporally differentiated post-demolition AAC volumes, using historic AAC production volumes and lifetime functions for buildings. D. B. Müller (2006) confirms that this can lead to reliable outflow estimation because precise data on historical input is a significant factor influencing the waste generation. Following Augiseau and Barles (2017) (Section A.2), our model can be classified as a dynamic retrospective and prospective flow analysis using a flow-driven model (input flows), because actual outflows and not the AAC stock is investigated. It is dynamic because “the change in flows over a long period is studied by assuming removal […] of materials […][at the] end of their lifetime” (Augiseau & Barles, 2017) by including lifetime distributions in the model. Finally, the model is based on input flows (annual historical production data). Additionally, we include historic construction activity and the regional AAC popularity as input data.

The quantification of post-demolition AAC is conducted in the following five steps:

1. *Research of production data*

First, we investigate the input of AAC into the stockpile using annual production data on the national level.

2. *Quantification of annual AAC demand on the federal state level*

Second, we determine the annual AAC demand on the federal state level. We choose the federal state level because the AAC popularity index considered in the next step is also calculated on this level. Further breakdown on the administrative district level (NUTS-3) will be conducted in step 4. We consider key figures from population and construction activity statistics to determine the local AAC demand. We calculate the correlations and significance levels and identify the key figure with the highest Pearson correlation to the AAC production and significance level $p < 0.01$. We assume that the produced AAC is demanded in the same year. Thus, the following equation can be stated:

$$d^*_{AAC,f,y} = p_{AAC,y} * \frac{k_{f,y}}{\sum_{f \in F} k_{f,y}} \quad \text{(A.1)}$$

$d^*_{AAC,f,y}$	Raw AAC demand volume in federal state f and year y [m³]
$p_{AAC,y}$	AAC production volume in year y [m³]
$k_{f,y}$	key figure value per federal state f and year y [-]
f	federal state
F	set of all federal states

3. Calculation of a regional popularity index for AAC

In addition to the construction activity, the regional popularity of AAC influences the demand. Therefore, the share of AAC in the construction of buildings per federal state (basis: gross volume) is used to calculate a normalised popularity index:

$$pop_{AAC,f} = \frac{s_{AAC,f}}{s_{AAC}} \tag{A.2}$$

$pop_{AAC,f}$	AAC popularity index per federal state f [-]
$s_{AAC,f}$	share of AAC in the construction of buildings in federal state f [-]
s_{AAC}	average share of AAC in the construction of buildings [-]

$pop_{AAC,f}$ is independent of the overall AAC popularity changes over time by increasing AAC production and use. $pop_{AAC,f}$ is higher than 1, if AAC is above average popularity in the respective federal state and below 1, if not. Then, we couple the raw regional production (Equation A.1) with the regional popularity (Equation A.2) and incorporate it:

$$d_{AAC,f,y} = p_{AAC,y} * \frac{k_{f,y} * pop_{AAC,f}}{\sum_{f \in F}(k_{f,y} * pop_{AAC,f})} \tag{A.3}$$

$d_{AAC,f,y}$	AAC demand volume in federal state f and year y (corrected by popularity) [m³]

4. Quantification of annual AAC demand on the administrative district level (NUTS-3)

Based on the annual AAC demand on the federal state level, we quantify the temporally differentiated AAC demand on the administrative district level (NUTS-3). Detailed data on the construction activity per district is not available for the distant past and the future. Therefore, we calculate the average construction activity share of the districts of the respective federal state over the last few years using the mainly correlating key figure identified in step 2. Then, we allocate the AAC demand per federal state (Equation A.3) among the NUTS-3 districts using these shares:

$$d_{AAC,\delta,y} = d_{AAC,f,y} * s_{\delta,f} \tag{A.4}$$

$d_{AAC,\delta,y}$	AAC demand volume in district δ and year y [m³]
$s_{\delta,f}$	average share of district δ in the construction activity of the respective federal state f [-]

5. *Quantification of the geographically distributed and temporally differentiated post-demolition AAC volume*

Finally, we determine the post-demolition AAC volumes based on the AAC demand (Equation A.4) by considering building lifetime. Schmalwasser and Weber (2012) indicate an average "utilisation period" of 77 (53) years for residential (non-residential) buildings in Germany, ranging from 40 (15) to 95 (113) years, respectively. We assume that the "utilisation period" corresponds to the lifetime of the building. Furthermore, Hossain and Ng (2018) review 155 publications on the life cycle assessment of buildings and find that the majority (65%) assume a building lifetime ranging between 41 and 50 years without differentiating residential and non-residential buildings (Hossain & Ng, 2018) (lower than in Schmalwasser & Weber, 2012). Besides, only 4% of the publications consider lifetimes above 80 years (Hossain & Ng, 2018). In this case, the upper limit in Schmalwasser and Weber (2012) is above the literature average, too. Rahlwes (1993) assessed concrete demolition quantities based on lifetimes of buildings. He assumed that 2% of the buildings exist for 30 years, 40% for 50 years, 30% for 70 years, 20% for 90 years, and 8% for more than 90 years without differentiating between residential and non-residential buildings. This distribution leads to an average lifetime of buildings of 67.6 years[8], which is again below Schmalwasser and Weber (2012) for residential buildings.

Due to the lack of reliable data and varying values in literature, adaptable and straightforward triangular lifetime probability functions are chosen for both residential and non-residential buildings (Figure A.2). For residential buildings, we assume the lifetime limits to be 35 and 95 years. The lower limit is slightly downwardly adjusted compared to Schmalwasser and Weber (2012) because Hossain and Ng (2018) show that 12% of their reviewed publications assume a service life of 40 years or less and Rahlwes (1993) considers a service life of 30 years for 2% of the buildings. The upper limit corresponds to Schmalwasser and Weber (2012). The average lifetime (equals the most probable lifetime in this case) is set to 65 years (according to Schmalwasser & Weber, 2012 and Hossain & Ng, 2018 and close to Rahlwes, 1993). The probability values for all ages in between are calculated using the following function:

$$f(age) = \begin{cases} \frac{2(age-a)}{(b-a)*(c-a)}, & a \le age < c \\ \frac{2}{(b-a)}, & age = c \\ \frac{2(age-a)}{(b-a)*(c-a)}, & c < age \le b \end{cases} \tag{A.5}$$

a lower limit of the lifetime
b upper limit of the lifetime
c most probable lifetime

[8] For the calculation of the average lifetime of buildings 100 years is chosen as the representative service life for the category 'more than 90 years'.

For non-residential buildings, we chose 15 to 100 years as the lifetime boundaries. The lower limit corresponds to Schmalwasser and Weber (2012). The upper limit is slightly downwardly adjusted because only 4% of the publications reviewed by Hossain and Ng (2018) assume a service life of 80 years or more and the assumed maximum lifetime is 100 years (Hossain & Ng, 2018). Rahlwes (1993) does not differentiate the service lives of more than 90 years. We assume an average lifetime of 51.7 years that corresponds to a slightly downwardly adjusted average lifetime indicated by Schmalwasser and Weber (2012), according to the findings of Hossain and Ng (2018). The most probable lifetime shown in Figure A.2b is at lower age (40 years) than the average lifetime (51.7 years) since the probability function is asymmetrical (positively skewed). We include the lifetime distribution of residential and non-residential buildings in the sensitivity analysis (Section A.5.1), due to the lack of reliable literature values and the made assumptions.

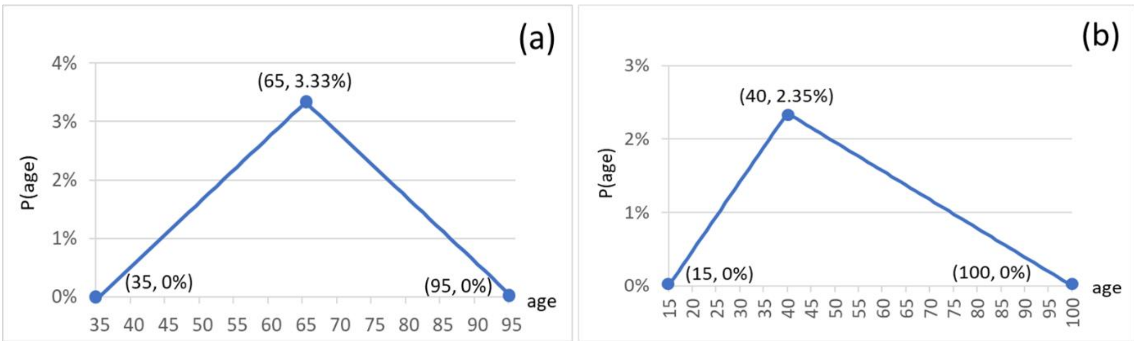

Figure A.2: Assumed lifetime probability functions of residential (a) and non-residential (b) buildings.

We assume that at the end-of-life, the same AAC volume once demanded incurs as post-demolition AAC. Lifetimes of residential and non-residential buildings differ significantly and are considered separately. Thus, the share of AAC used in residential and non-residential buildings is determined using construction activity statistics (basis: gross volume). Finally, we calculate the temporally differentiated post-demolition AAC volume per district with annual granularity based on the annual AAC demand on district level (step 4), the triangle lifetime probability functions (Equation A.5, Figure A.2) and the shares of AAC used in residential respectively non-residential buildings:

$$pd_{AAC,\delta,y} = \sum_{a=1}^{A} \left(d_{AAC,\delta,y-a} * L_r(a) * S^*_{AAC,r} + d_{AAC,\delta,y-a} * L_{nr}(a) * S^*_{AAC,nr} \right) \tag{A.6}$$

$pd_{AAC,\delta,y}$	post-demolition AAC volume in administrative district δ and year y [m³]
A	maximum age [-]
$L_r(a)$	probability of a residential building's lifetime of a years [-]
$L_{nr}(a)$	probability of a non-residential building's lifetime of a years [-]
$S^*_{AAC,r}$	share of AAC used in residential buildings [-]
$S^*_{AAC,nr}$	share of AAC used in non-residential buildings [-]

A.3.2 Validation approaches

For the first (stock-based) validation (1), we used publications focussing on the building stock and consult demolition statistics. We research the average AAC material intensities and the gross floor area demolished per year for residential and non-residential buildings to estimate the total post-demolition AAC volume per year. For this, we performed our own calculation by using the following formula:

$$pd_{AAC,y} = mi_{AAC,r,y} * gfa_{r,y} + mi_{AAC,nr,y} * gfa_{nr,y} \tag{A.7}$$

$pd_{AAC,y}$	post-demolition AAC volume in year y [m³]
$mi_{AAC,r,y}$	average AAC material intensity of residential buildings in year y [m³/m²]
$gfa_{d,r,y}$	gross floor area of residential buildings demolished in year y [m²]
$mi_{AAC,nr,y}$	average AAC material intensity of non-residential buildings in year y [m³/m²]
$gfa_{nr,y}$	gross floor area of non-residential buildings demolished in year y [m²]

Besides, we investigate publications on the geographical distribution of the stockpile to compare it with the post-demolition AAC volumes calculated in the quantification approach. Projections of future post-demolition AAC volumes are not reasonable using validation (1) because of changing material intensities in the buildings over time.[9]

In the second (waste-based) validation (2), we consult waste statistics and literature values on AAC demolition wastes. Official waste statistics for AAC do not exist because of missing unique AAC waste codes. Therefore, waste statistics only provide a very rough upper bound.

A.4 Case study: Post-demolition AAC volumes in Germany

A.4.1 Application of the quantification model

In this section, the methodology (Section A.3.1) is applied to assess the post-demolition AAC volumes in German administrative districts between 1950 when AAC use started (A. Müller, 2018; UBA, 2019) and 2050 including a comprehensive forecast.

[9] In general, changing material intensities occur due to changing building design like more generous floor plans. Concerning AAC, the increasing popularity raises the material intensity additionally. For example, the AAC material intensities of detached houses in Germany are much higher for newly built (97.6 kg/m²) than the average stock (31.5 kg/m²) (source: direct inquiry of the authors to the Federal Environment Agency (Felix Müller)).

1. Research of production data

The annual input of AAC into the German stockpile can be determined using available production data since 1950 from the Federal Statistical Office (Destatis)[10]. If necessary, the production data is converted to m³ using the following thickness and density assumptions.[11] The average density of AAC is assumed to be 500 kg/m³ (Deilmann et al., 2014; A. Müller, 2016; Volk et al., 2019) and the thickness of AAC roof/ceiling panels and wall elements is assumed to be 0.17 m (1950 to 1994). This thickness assumption is based on the official German production statistics for the years 1990 to 1994 (Destatis, 1991, 1992, 1993b, 1994, 1995) which indicate production quantities both in m² and in m³. From 1995, we assume an average thickness of 0.22m (expert interview[12]) because the German Heat Insulation Ordinance became effective (WärmeschutzV/16.08.1994) which led to the construction of thicker and more insulating walls. These assumptions can considerably influence the post-demolition AAC volumes calculated in this case study. Thus, they are included in the sensitivity analysis (Section A.5.1).

AAC production volumes are shown in Figure A.3. All-German production data are available from 1991 onwards. Before 1990, production statistics are only available for Western Germany (Federal Republic of Germany (FRG)). AAC production values in the German Democratic Republic (GDR) are available for the years 1964, 1970, 1975, 1980, 1985, and 1987 (Weise & Kreher, 1988) but missing for the year between. Therefore, we interpolate production values based on operation starts of the four AAC production plants in the GDR (Parchim 1964, Laussig 1970, Calbe 1971, Hennersdorf 1981; Trätner, 2001) to achieve a best possible approximation of the real production.[13] From 2019 on, the production is assumed to be the average of the years 2014 to 2018. This “business as usual” assumption is used because the forecasts are very unreliable in the fluctuating construction sector. Due to the long lifetimes of the buildings, the

[10] Destatis (1952, 1962, 1963, 1964, 1965, 1966, 1967, 1968, 1969, 1970, 1971, 1972, 1973, 1974, 1975, 1976, 1977, 1978, 1979, 1980, 1981, 1982, 1983, 1984, 1985, 1986, 1987, 1988, 1989, 1990, 1991, 1992, 1993b, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008, 2009); GENESIS (2019). Values are missing for Saarland between 1950 and 1959 and for Berlin between 1950 and 1963. From 1964 onwards, AAC production figures for Berlin-West are included. Since the overall production quantity did not reach 1 million m³ (less than 30% of the 2018 production) before the mid-1960s, these missing values do not influence the following results much.

[11] Production statistics on AAC from the Federal Statistical Office usually specify the amount of produced AAC masonry units (in tons or m³) and AAC panels and floorboards (in tons or m²).

[12] Xella Technologie- und Forschungsgesellschaft mbH, Torsten Schoch.

[13] A production leap in the years following the opening of each new plant (Figure A.2a) is assumed. In the remaining years, we consider an annual 5% increase in production (1966-1970), respectively, a linear increase in production (1973-1981, 1983-1989). For 1966-1970, in contrast to later years, no linear increase is assumed as no linearization of the interim period is feasible. This is due to the lack of data for 1965 (the year after the AAC plant Parchim started its operation). However, due to the short period of 1966 to 1970, the assumption of relative increase in production (increase varies between 11,000 and 13,000 m³/year) differs only slightly from a linear increase (constant increase). For 1990 (reunification of Germany), a 30% decreased AAC production is assumed because the GDR production of various types of rocks and soil fell by around 10% to 50% in 1990 Destatis (1993a). This issue is examined in the sensitivity analysis (Section A.5.3) due to the lack of reliable data.

expected future production volumes only have a marginal influence on the post-demolition AAC volume up to 2050 (see Section A.5.3).

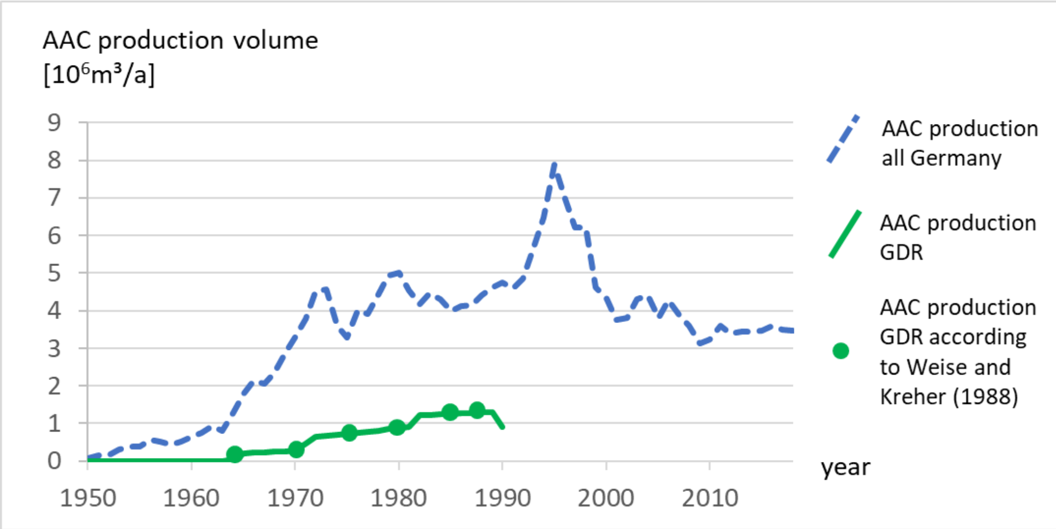

Figure A.3: AAC production volume in the GDR (1950 to 1990) and in all Germany (1950 to 2018).

2. Quantification of annual AAC demand on the federal state level

We consider 22 key figures from population and construction activity statistics to determine AAC demand in Germany. Destatis (2018) provides various data and their regional breakdown on the federal state level. Also, the production index for building construction (GENESIS, 2019) and the population development in Germany (Destatis, 2016b; GENESIS, 2020) is regarded. Then, we calculate the Pearson correlation of the respective key figure and the AAC production in Germany between 1991 to 2018 (annual basis) since detailed data for all key figures is only available for the time after the German reunification in 1990 (Figure A.4).

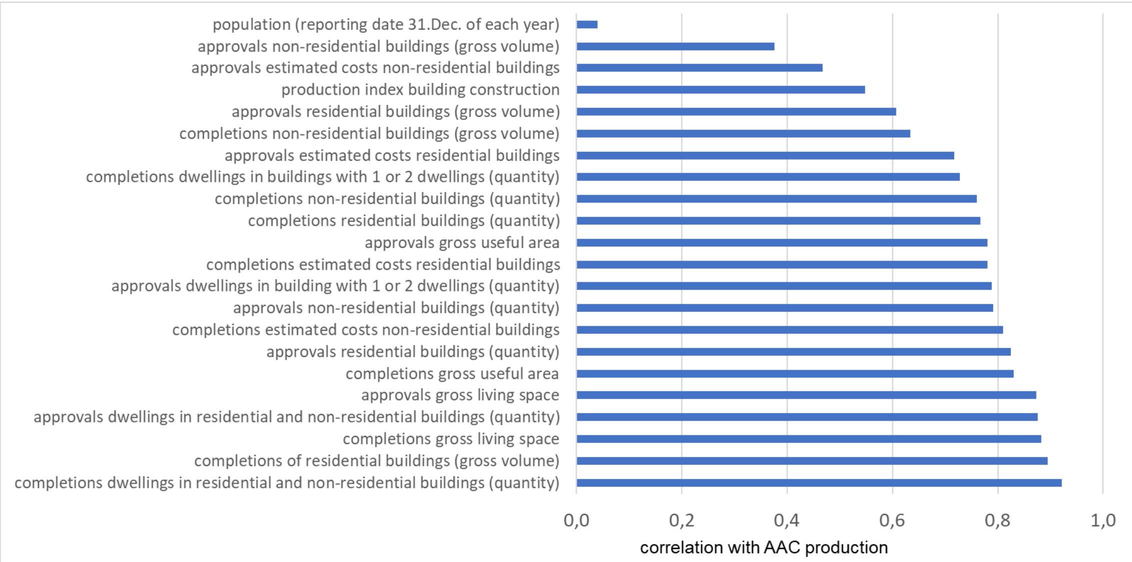

Figure A.4: Correlation of investigated official and publicly available key figures of construction activity and population statistics with AAC production, data sources: Destatis (2016b), Destatis (2018), GENESIS (2019), GENESIS (2020).

For the following calculations, we use the key figure 'completions dwellings in residential and non-residential buildings (quantity)' (Figure A in Supplementary Material) because it has the highest Pearson correlation with AAC production ($r = 92.1\%$).[14] The coefficient of determination ($R^2 \approx 84.9\%$) and a very low p-value ($p \approx 3{,}6 * 10^{-12}$) also indicate excellent suitability. The linear relationship implied by the Pearson correlation is appropriate, as the significance level is $p \ll 0.01$ and the correlation is lower using the Spearman (rank) correlation ($r = 84.8\%$) instead of Pearson correlation ($r = 92.1\%$).

We complement the statistics of this key figure since data for some years are missing (Figure A.5). The "business as usual" assumption for future demand equals that of point 1 in this section. Finally, Equation A.1 is used to calculate the raw annual AAC demand for all 16 German federal states between 1950 and 2050.

[14] The other key figures have a Pearson correlation of less than 90%. The population (correlation of only 4.0%) is far behind the construction activity key figures.

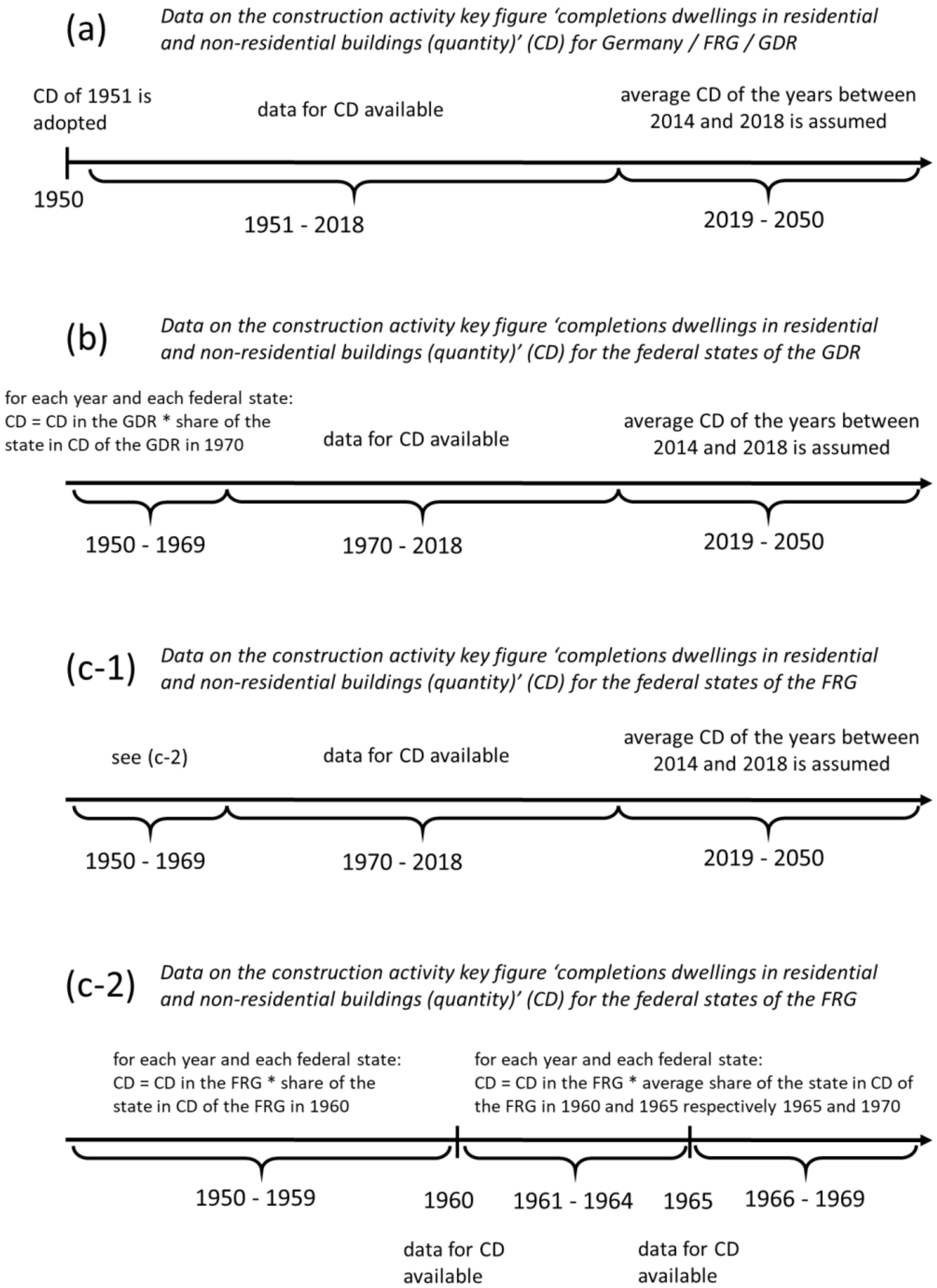

Figure A.5: Correlation of investigated official and publicly available key figures of construction activity and population statistics with AAC production, data sources: Destatis (2016b), Destatis (2018), GENESIS (2019), GENESIS (2020).

3. Calculation of a regional popularity index for AAC

The share of AAC in the construction of dwellings per federal state in 2019 (Pestel, 2020)[15] is used to calculate different regional popularities of AAC (Table A in Supplementary Material). An AAC popularity index is calculated using this data and Equation A.2. This popularity index reflects a normalised share per federal states on the overall AAC demand in Germany. Therefore, a change in construction technology or an increasing demand of AAC in Germany does not influence the popularity indices (but is included in the model via the production volumes, step 1). We assume constant popularity per federal state since 1950 based on an expert interview[16] and due to missing temporally differentiated data on AAC popularity. Furthermore, we assume the same popularity indices for West, East, and reunified Berlin.

Finally, the total annual German AAC production volume is allocated among the different federal states regarding the AAC popularity indices according to Equation A.3 to determine the annual AAC demand on federal state level including AAC popularity. For the period 1950 to 1990, we consider the FRG and the GDR separately regarding both construction activity and AAC production. From 1991 onwards, we use all-German values.

4. Quantification of annual AAC demand on the administrative district level (NUTS-3)

We quantify the annual AAC demand of all 401 independent cities and administrative districts in Germany[17] based on the AAC volumes installed per federal state and year (step 3) and the key figure 'completions dwellings in residential and non-residential buildings (quantity)' on NUTS-3 level (Regionaldatenbank Deutschland, 2020) for the years 1995 to 2018. Based on this data, we calculate the average share of the individual districts in the construction activity of the respective federal states since data for the period of 1950 to 1994 is missing. Then, we distribute the annual AAC demand per federal state among the NUTS-3 districts using these shares and Equation A.4.

5. Quantification of the geographically distributed and temporally differentiated post-demolition AAC volume

Finally, we determine the post-demolition AAC volume in Germany by linking AAC demand (step 4) and buildings' lifetimes (Section A.3.1). We estimate the share of AAC used in residential and non-residential buildings by means of the German construction activity statistics for the years 2011 to 2018 (basis: gross volume) (Destatis, 2012, 2013, 2014, 2015, 2016a, 2017, 2018, 2019b). The share of AAC used in residential buildings ranges between approximately 80% to 90%, with an average percentage of 85.5%. Correspondingly, the average share of AAC used in non-residential buildings is 14.5%. We assume that the share of AAC used in residen-

[15] By courtesy of German Society for Masonry and Housing Construction (Deutsche Gesellschaft für Mauerwerks- und Wohnungsbau e. V., DGfM).

[16] Xella Technologie- und Forschungsgesellschaft mbH, Dr. Oliver Kreft.

[17] A full list of the independent cities and administrative districts is provided by Destatis (2019d). Further data on the districts are available at Opendatasoft (2017).

tial/non-residential buildings corresponds to the share of post-demolition AAC from residential/non-residential buildings over the whole considered period. The high share of AAC used in residential buildings is confirmed by UBA (2019), that also states that AAC is much more likely to be used in residential than non-residential buildings, with exceptionally high shares in one- and two-family houses. Also, we include the share of AAC used in residential and non-residential buildings in the sensitivity analysis (Section A.5.1).

Finally, we calculate the annual post-demolition AAC volume per district in Germany based on the lifetime functions (Figure A.2) and the shares of AAC use in residential and non-residential buildings according to Equation A.6. The results are illustrated in Figure A.6 and Figure A.8 and discussed in Section A.5.

A.4.2 Validation approaches

A.4.2.1 Stock-based validation (1)

Studies on the anthropogenic stockpile usually differentiate residential and non-residential buildings and diverse building materials. Deilmann et al. (2014) and Schiller et al. (2015) estimate the AAC stockpile in Germany to approximately 358 million m³ in 2010 (Figure A.6b). For residential buildings, they use representatives per building type (synthetic buildings) with available material quantities per m² (Schiller et al., 2015), the residential building stock 2010 according to the micro census dataset (Schiller et al., 2015) and the construction activity statistics of 2010. The stock of non-residential buildings is calculated based on gross fixed assets, derived from national accounts (Schiller et al., 2015). Based on this, the material intensity of AAC in different buildings types can be used to calculate the anthropogenic stockpile. According to the Federal Environment Agency[18], the average weighted AAC material intensity in Germany is approximately 31.5 kg/m² (gross floor area) for residential buildings and 21.9 kg/m² (gross floor area) for non-residential buildings. Together with the demolition statistics of Germany (Destatis, 2019c) and Equation A.7 the post-demolition AAC volume for the year 2010 (the base year of Schiller et al., 2015) is calculated. According to this, a gross floor area of approx. 2 million m² for residential buildings and approx. 6.5 million m² for non-residential buildings was demolished in 2010. Thus, the total post-demolition AAC volume in Germany in 2010 amounts to ca. 414,000 m³.

Schiller et al. (2010) provide information on the geographical distribution of the stockpile in Germany but do not disclose the quantity of AAC. However, the average proportion of AAC in the total mineral material stock can be calculated and results in 1.4wt.-% based on Schiller et al. (2015). Thus, we can calculate the total AAC stockpile per district for the year 2005 (the base year of Schiller et al., 2010) as a share of the specified mineral stockpile (Figure A.9). However, this neglects regional preferences for AAC (Section A.3.1, step 3). Results are discussed in Section A.5.2.

[18] Direct inquiry by the authors, contact person: Felix Müller.

A.4.2.2 Waste-based validation (2)

In this validation, the current post-demolition AAC material flow in Germany based on literature on AAC demolition wastes is assessed. Since AAC does not have a unique waste code,[19] no official waste statistics are available for Germany or Europe. Moreover, there are no information on the proportion of AAC in the used waste codes. Therefore, we conclude that waste codes are not appropriate to estimate the post-demolition AAC volume unless a unique waste code for AAC is provided.

A. Müller (2016) indicates that AAC accounts for 1.1wt.-% of the total mass of building rubble (base year 2012). Consequently, the volume of post-demolition AAC in 2012 is 0.6 million tons (A. Müller, 2016), which corresponds to 1.2 million m³. However, a future extrapolation is not reasonable using this approach because of changing material intensities of buildings and the demolition rubble over time. Deilmann et al. (2014) specify the amount of post-demolition AAC to 0.3 million tons in 2010, i.e. about 0.6 million m³. For the future, Deilmann et al. (2014) estimate a post-demolition AAC volume of 0.8 million tons (1.6 million m³) in 2030 and 2.5 million tons (5 million m³) in 2050.

A.5 Results and discussion

A.5.1 Model results for country-wide post-demolition AAC volumes

In this section, the country-wide results of the quantification model are presented, validated and subjected to a sensitivity analysis. An overview of the figures for the post-demolition AAC volume / the AAC stockpile in Germany (Table B) is provided in the Supplementary Material.

Before 1990, there was hardly any annual post-demolition AAC in Germany (Figure A.6a)[20] because AAC was used only since 1950 and production did not increase significantly until the mid-1960ies (Figure A.3). However, around the beginning of the 21st century, post-demolition AAC volumes in Germany started to increase significantly. This increase is projected to continue at least until 2050. Compared to 2020, the amount of post-demolition AAC will increase by a factor of about 3.5 (3 million m³) by 2050 based on the model results. The total AAC stockpile in Germany (cumulated production minus cumulated post-demolition volumes) amounts to around 226 million m³ in 2020 and could nearly reach its maximum of 249 million m³ in 2040 (Figure A.6b, Table B in Supplementary Material).

These model results are compared with the two validation approaches (Sections A.3.2, A.4.2). The quantification model provides the continuous post-demolition AAC volumes of the years

[19] Currently, the waste codes (Verordnung über das Europäische Abfallverzeichnis (Abfallverzeichnis-Verordnung) [Regulation on the European Waste List]) 170101 (concrete), 170107 (mixtures of concrete, bricks, tiles, and ceramics) and 170802 (gypsum-based building materials) are used.

[20] In 1990, the post-demolition AAC volume in Germany amounted to 38,850m³.

1950 to 2050. The stock-based validation (1) results in one data point for the year 2010 (green square in Figure A.6a) that fits well to the quantification of our model. The waste-based validation (2) provides four data points for the years 2010, 2012, 2030, and 2050 (violet crosses in Figure A.6a), showing both higher and lower estimations than the model. Besides, the results of the sensitivity analysis of the lifetime are included as this is the parameter most influencing the results (Figure A.7).

For 2010, the quantification model identifies a post-demolition AAC volume of 498,000 m³ and validation (1) states 414,000 m³ (17% lower), while Deilmann et al. (2014) (validation (2)) specify a volume of 600,000 m³ (20% higher). A. Müller (2016) (validation (2)) indicates the double amount of 1.2 million m³ for 2012 (only two years later) which is 96% higher than calculated in the model (611,000 m³). For 2030, the model quantifies a post-demolition AAC volume of 2.268 million m³ while Deilmann et al. (2014) (validation (2)) estimate 1.6 million m³ (29% lower). Compared to A. Müller (2016), the latter corresponds to a small increase of only 0.4 million m³ in 18 years. For 2050, Deilmann et al. (2014) (validation (2)) predict a post-demolition AAC volume of around 5 million m³ which is 19% higher than the quantification model's estimation of 4.211 million m³. Overall, the model result lies between these five data points and thus seem reasonable.

Besides, the AAC stockpile allows a comparison of the model with the validation (1) (Figure A.6b). An AAC stockpile of approximately 358 million m³ is stated for 2010 (Deilmann et al., 2014; Schiller et al., 2015). This is significantly higher than the stockpile of 199.7 million m³ according to the model. This difference could be explained by the scope and focus of the studies of Schiller et al. (2015) and Deilmann et al. (2014). They examine the entire stockpile in Germany, where AAC only accounts for a tiny proportion compared with other mineral building materials. Uncertainties of only a few per cent result in considerable deviations in the estimated AAC amount.

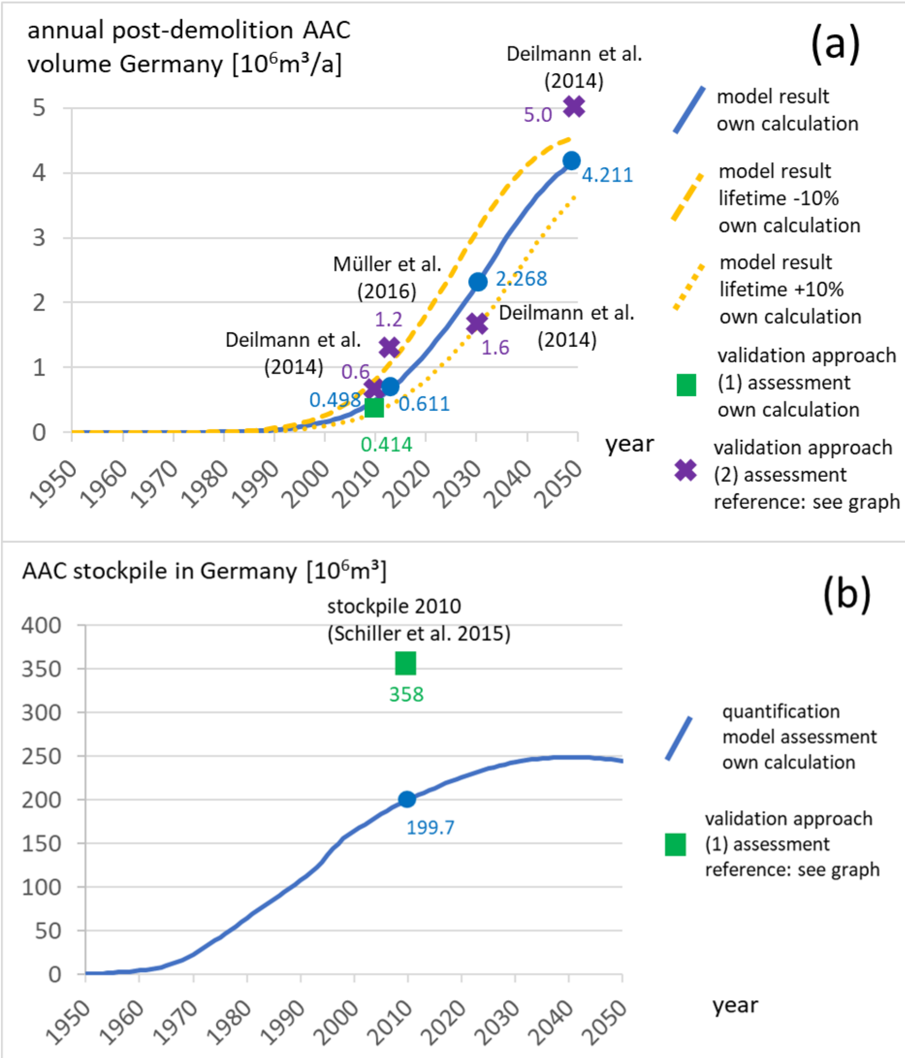

Figure A.6: Comparison of the annual post-demolition AAC volume (a) and the cumulated AAC production (b) calculated with the quantification model, different validations and lifetime sensitivity analysis results.

The following sensitivity analysis of the post-demolition AAC volume in Germany analyses the AAC production in the GDR in 1990, the assumed density of AAC, the assumed thickness of AAC panels (thickness until 1994 and thickness from 1995 on at the same time), the share of AAC used in residential buildings, the lifetime peak (both functions at the same time), and the lifetime (lower/upper bound and peak for both functions at the same time) (Figure A.7).

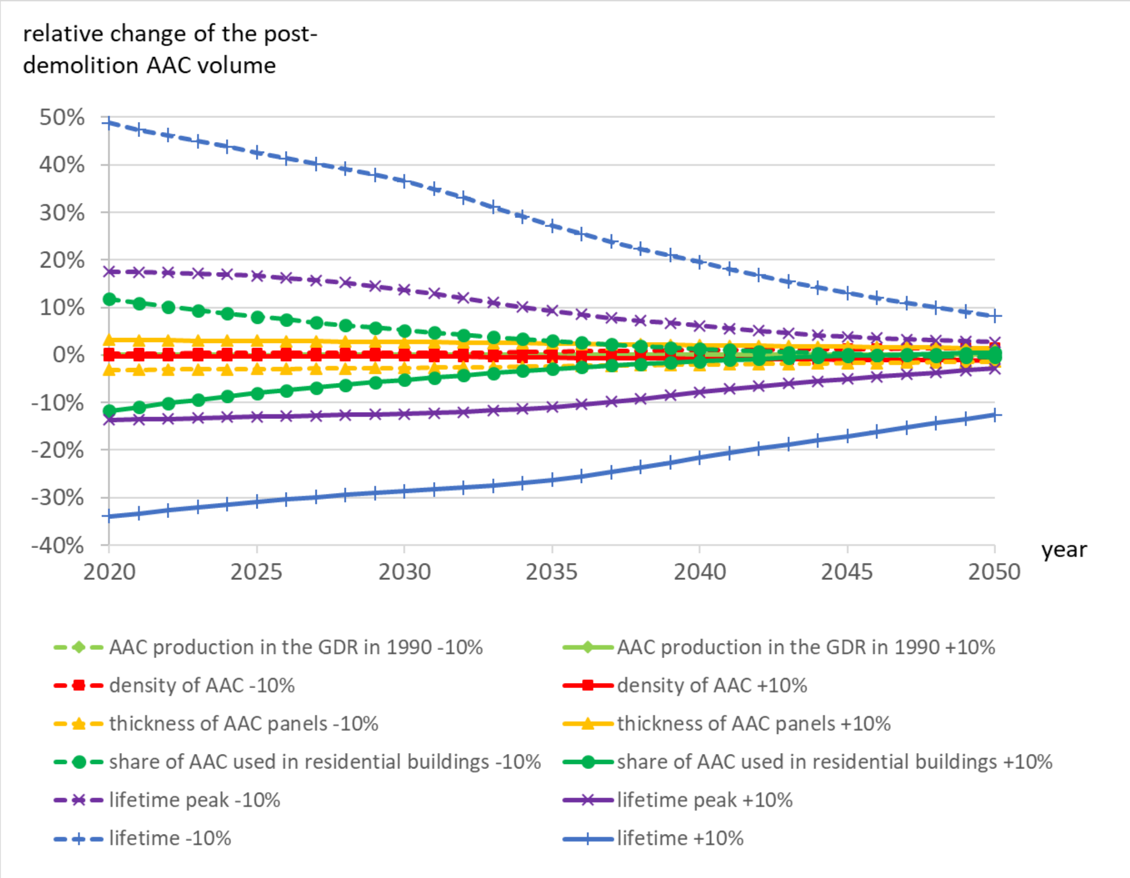

Figure A.7: Results of the sensitivity analysis of the post-demolition AAC volume in Germany.

The production of AAC in the GDR in 1990 (light green curve with diamonds, barely visible in Figure A.7) only has a marginal influence on the post-demolition AAC volume in Germany, as its volume distributes over many years of post-demolition volume due to the long-term lifetime functions. Moreover, the 1990 GDR AAC production only corresponds to about 20% of the 2020 German AAC production (Figure A.3). The density assumption of AAC (red curve with squares) also shows only little influence because most statistical data is given in m³ and does not need to be converted. Only for the years 1950, 1951, and 1995 to 1998, the AAC production is specified in tons and converted to m³ via the AAC density assumption. In contrast, the thickness of AAC panels is used to calculate the production volume of panels (usually specified in m²) for many years. The panel production accounted for 20.1% (on average) of the AAC production volume in Germany (1950 to 2018). A 10% variation of the panel thickness (yellow curve with triangles) has a low (ca. 3%) influence on the post-demolition AAC volume in Germany, but considerably more than the density. A variation in the share of AAC used in residential buildings (green curve with circles) leads to significant changes in the post-demolition AAC volume, that are even higher than 10% in 2020 but converge to low variation (<3%) after 2035. This sensitivity arises from the lifetime that will be discussed next. A higher share of residential buildings extends AAC residence time in the building stock due to longer lifetimes of residential buildings compared with non-residential buildings.

The lifetime has the highest sensitivity of all investigated parameters. Varying only the lifetime peak (violet curve with crosses) shows significant changes in the post-demolition AAC between 12%-18% and converging to < 10% (< 5%) in 2035 (2045). However, a reduction of the lower/upper bound and the peak by 10% at the same time (blue curve with orthogonal lines) increases the post-demolition AAC volume in Germany by 49% (597,000m³) in 2020. A respective increase by 10% results in a decrease of the post-demolition AAC volume by 34% (415,000m³) in 2020. Such high sensitivities can be explained by the steep and asymmetric rise of post-demolition AAC volume in Germany between 2010 and 2050 (Figure A.6a). A shorter lifetime shifts the higher future post-demolition AAC volumes to the present (and vice versa for longer lifetimes). Besides, lifetime sensitivity has less impact approaching the year 2050 because the rise of the post-demolition AAC curve flattens. However, future research on lifetime distributions is needed. The lifetime also has the highest sensitivity regarding the 2020 AAC stockpile in Germany. A 10% reduction (increase) of all lifetime parameters decreases (increases) the baseline of 225.9 million m³ to 218.9 million m³ (230.2 million m³).

A.5.2 Model results for regionalised post-demolition AAC volumes

In this section, the regionalised results of the quantification model are presented and validated. An overview of the figures for the geographical distribution (Table C) is provided in the Supplementary Material. Figure A.8a illustrates the estimated post-demolition AAC volume per NUTS-3 district in Germany in 2020, together with all AAC production plants in Germany. In some cases, a connection of the post-demolition AAC volume with the production sites can be stated. For example, AAC plants in the immediate proximity of the large German cities (Berlin, Hamburg, Munich, Bremen, and Hanover) with estimated high post-demolition AAC volumes stand out. Also, in the Rhine-Main area around Frankfurt, high post-demolition AAC volumes are corresponding to four AAC plants. The same applies to the region around Cologne (three plants), the Emsland (one plant) as well as the northern half of Baden-Württemberg around Stuttgart (four plants at a larger perimeter). The significant correspondence between the locations of the AAC plants and the expected post-demolition AAC volume shows that our model reflects the real situation well. In general, Northern Germany and Baden-Württemberg reveal high post-demolition AAC volumes due to the high AAC popularity. In contrast, buildings in Bavaria are traditionally built with bricks leading to low AAC popularity and low post-demolition AAC volumes.

Comparing 2020 with the expected 2030, 2040 and 2050 projections the strong increase in post-demolition AAC volume is again noticeable (Figure A.8b): Average post-demolition AAC volumes per district rise from 3,050 m³ (2020) to an expected 5,660 m³ (2030), 8,590 m³ (2040) and 10,500 m³ (2050). Especially in Northern Germany and Baden-Württemberg high post-demolition AAC volumes will arise in the future.

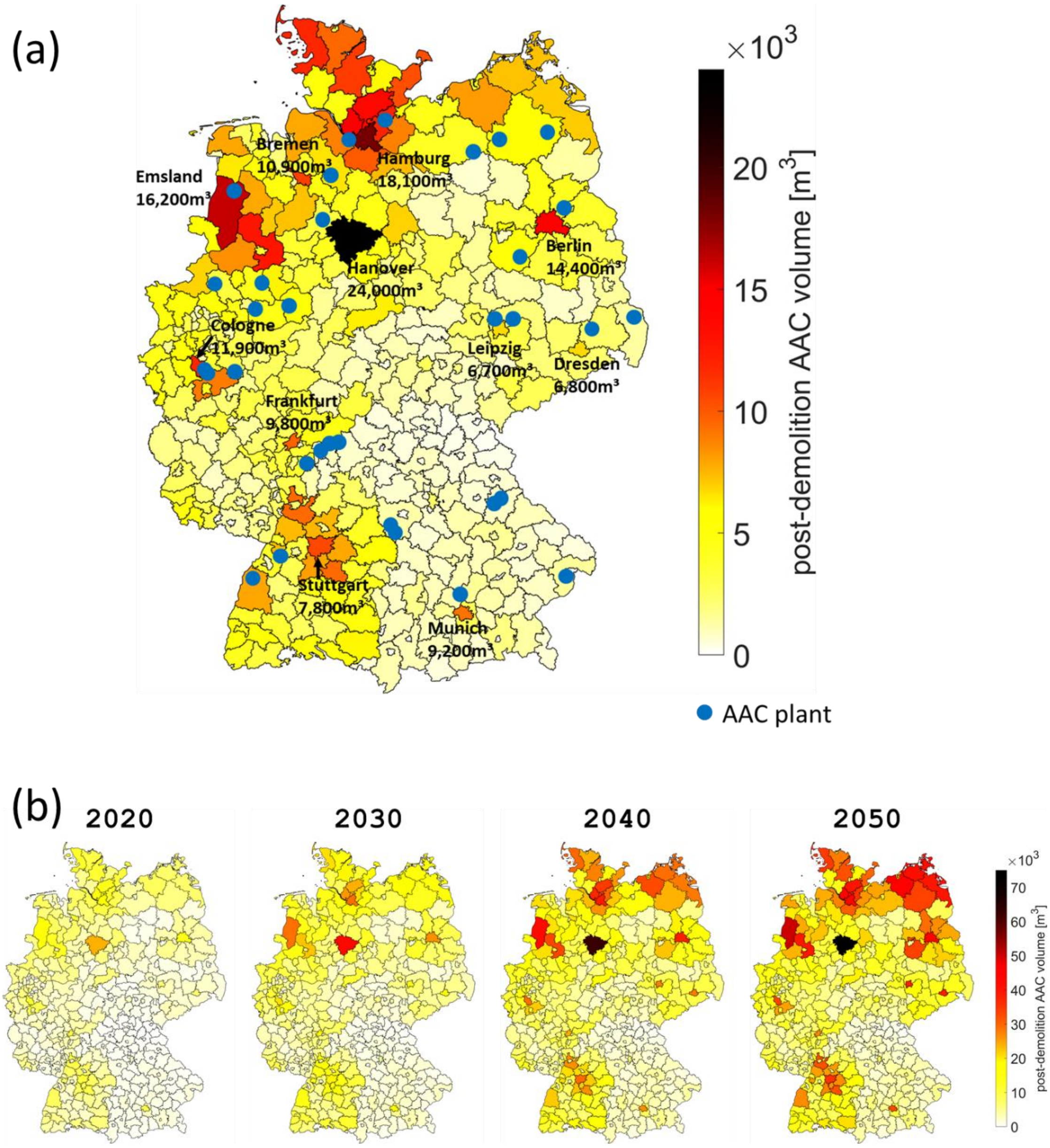

Figure A.8: Post-demolition AAC volume at the NUTS-3 district level in Germany in 2020 and locations of AAC plants (a). Expected post-demolition AAC volume at the NUTS-3 district level in Germany for the individual years 2020, 2030, 2040 and 2050 using a fixed scale (b). The darker the colour, the higher the post-demolition AAC volumes.

Finally, the geographical distribution of post-demolition AAC volumes is compared with the AAC stockpile per NUTS-3 district calculated based on Schiller et al. (2010) (Figure A.9). It confirms high post-demolition and stockpile volumes in the above-mentioned large cities. But due to the influence of the AAC popularity that is considered in Figure A.8, especially Northern Germany and Baden-Württemberg show higher while Berlin shows lower post-demolition AAC volumes compared to the stockpile estimate.

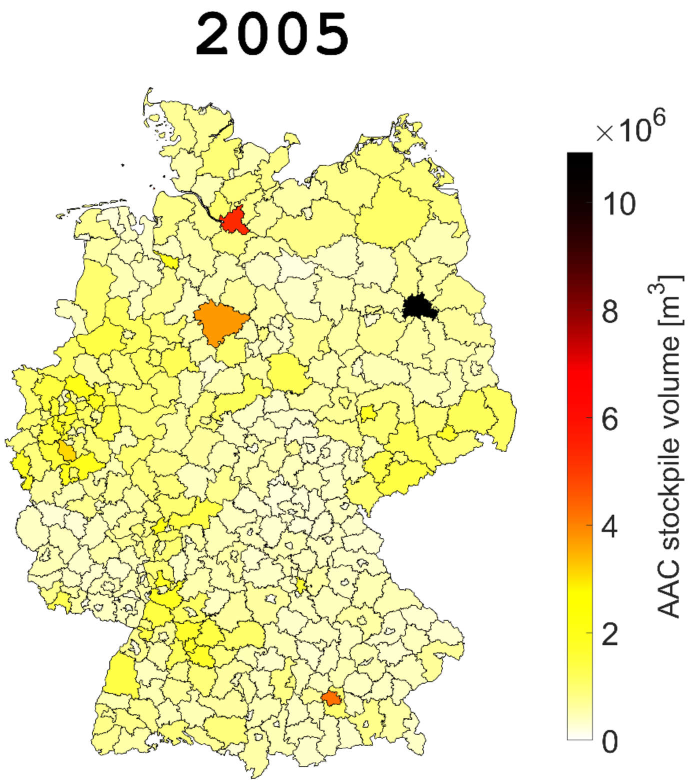

Figure A.9: Total AAC stockpile per district in Germany in 2005 (data source: Schiller et al., 2010).

A.5.3 Shortcomings, limitations and implications

Building materials are massive and bulky products for which long transports are usually avoided. Therefore, and due to missing data, imports and exports are not considered in the model. To reach the AAC stockpile estimation of Schiller et al. (2015) (Figure A.6b), an average import surplus of 56% of the domestic production since 1950 would be required. This seems rather unlikely. Besides, Schiller et al. (2015) show that between 2005 and 2011, imports of mineral building materials remained constant between 20 and 25 million tons annually, while exports fluctuated between 35 and 50 million tons. Moreover, BBS (2019) shows an export surplus for Germany.[21] Therefore, the model rather overestimates the post-demolition AAC volume. Adding import and export aspects to the model could reduce the current overestimation. Hence, these aspects should be subject to future research.

In the model, lifetime functions for buildings are used to determine temporally differentiated post-demolition AAC volumes that can be associated with considerable uncertainties. The used lifetime functions are based on literature (Section A.3.1), because no empirical data is availa-

[21] There is no statistics for AAC foreign trade. But, considering the category ‘production of concrete products, cement products and sand-lime brick products’ (which includes AAC), there is a total import of 1.1 million tons and a total export of 3.2 million tons in 2018.

ble. Besides, many aspects like construction method, building material, floor plan, monument protection, refurbishment or aesthetics can influence the lifetime of a building.

Furthermore, “business as usual” in the production of AAC and the construction activity is assumed to determine future AAC demand because general forecasts are very unreliable in the fluctuating construction sector. However, the demand of 2019 and following years does not significantly influence the prediction until 2050 because of high buildings’ lifetimes. For 2019, only about 12.8% of non-residential and 0% of residential buildings are demolished until 2050 based on the lifetime assumptions of this study. Considering the shares of AAC used in residential and non-residential buildings, only an overall 1.9% of the demand in 2019 will be demolished until 2050. For the following years, this share will constantly decrease and reach 0% from 2035 onwards.

It is also striking that residential buildings account for 63.1% of the total stockpile (Schiller et al., 2015), which is significantly lower than an 85.5% share of AAC used in residential building included in this study. Lanau et al. (2019) generally confirm the share used by Schiller et al. (2015) by stating a share of between 25% and 50% non-residential buildings in the total building stock. Besides, there are no data on the share of AAC in the general construction of buildings, so only the construction of dwellings is included in the calculation of the AAC popularity index.

Increasing insulation demand lead to lighter AAC with better thermal properties and reduced density. According to literature, a constant density of 500 kg/m³ (Section A.4.1) was used. However, the density decreased since 1950 by at least 20% to 30% as modern AAC used for i.e. exterior walls typically has dry densities of 300 to 350 kg/m³, or even less (DIN 20000-404:2018-04; DIN EN 771-4:2015-11; German Institute for Structural Engineering [Deutsches Institut für Bautechnik], 2021). Thus, future work should include a density function over time.

A.6 Conclusion

A model was developed to estimate future annual AAC volumes on a regional level. For validation, a stock-based (1) and a waste-based (2) approach were used. All approaches were applied in a case study to the AAC in Germany to assess post-demolition AAC volumes between 1950 and 2050. Volumes exceeded 100,000 m³/year in 1997 and steeply increased recently from 160,000 m³ (2000) to 498,000 m³ (2010) and 1,224,000 m³ (2020). By 2050, a post-demolition volume of more than 4 million m³ per year can be expected. Therefore, a significant increase of AAC waste to be landfilled can be expected, if AAC recycling is not fostered. In general, the model results can be validated (Section A.5.2) with a maximal deviation of 789,000 m³ for the forecast in 2050. The model results are robust regarding the density of AAC, the thickness of AAC panels, and the AAC production in the GDR in 1990 (Section A.5.1). However, the lifetime of buildings and the share of residential buildings have a significant influence on the results.

The geographical distribution of the projected post-demolition AAC shows significant volumes arising primarily in large German cities/regions like Berlin, Hamburg, Munich, Bremen, Hanover, Cologne, Frankfurt, and Stuttgart. Especially in densely populated areas of industrialized countries, AAC recycling from urban mines could be a promising option to reuse or recycle materials locally. But, current framework conditions need to be adapted to foster post-demolition AAC recycling (e.g. with higher disposal costs or a disposal ban). Demolition and recycling companies could use the model results to get higher planning reliability regarding the regional post-demolition AAC emergence to develop strategies, business models, operations and technologies or to plan necessary investments. Suitable recycling options for post-demolition AAC are investigated in several studies, e.g. Aycil et al. (2016), Bergmans et al. (2016), Bukowski et al. (2015), Gyurkó et al. (2019), Kreft (2017), Rafiza et al. (2019), Renman and Renman (2012).

Future research could specify and extend the model assumptions on AAC density, lifetimes of residential and non-residential buildings, AAC popularity, and the share of AAC used in residential / non-residential buildings. Furthermore, additional aspects like the lifetimes of different AAC products within the different building types, further differentiation of single and multi-family houses, different building cohorts or imports/exports could be included. These extensions could further improve the model results, but will require further data that was not available for the case study in Germany. Moreover, dynamization of the named parameters allowing parameter changes over time could also increase the accuracy of predicting post-demolition AAC volumes in the distant future. Further studies on the field of post-demolition AAC quantification are needed to address these aspects.

The developed model is transferable to other building materials (e.g. bricks, sand-lime bricks, timber or lightweight concrete blocks) and other countries/regions. However, adaption might be required – particularly regarding the available data on building stock, building lifetimes and material or building product production.

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B Post-Demolition Autoclaved Aerated Concrete: Recycling Options and Volume Prediction in Europe

Abstract[1]

Autoclaved aerated concrete (AAC) is an increasingly used building material due to its exceptional thermal properties. Post-demolition AAC is mainly disposed in landfills because of lacking established recycling processes. However, the growing demand for sustainable products, greenhouse gas reduction, decreasing landfill capacities and new legal frameworks require recycling options for post-demolition AAC.

Current research includes using post-demolition AAC recycling in the production of lightweight aggregate concrete, lightweight mortar, no-fines concrete, and floor screed. Even closed-loop recycling could be achieved by adding finely ground post-demolition AAC in the AAC production process or by producing belite cement clinker from post-demolition AAC as a substitution for Portland cement.

Predicting the generation of post-demolition AAC volumes is crucial for a recycling and circular management of AAC. But, post-demolition AAC volumes in Europe are currently neither recorded in statistics nor investigated in comprehensive studies. Therefore, a post-demolition AAC prediction model is presented that quantifies post-demolition AAC on a national and European level. Results show low volumes in South East, Western, and Southern Europe as well as Scandinavia due to small market sizes. In North West and Central Europe, especially the UK (700,000 m³) and Germany (1,200,000 m³) in 2020 drive post-demolition AAC volumes. The most significant post-demolition AAC volumes occur in Eastern Europe, especially in Poland (1,800,000 m³) and Russia (3,900,000 m³) in 2020. While relative volumes between the regions stay similar, the absolute post-demolition AAC volumes in Europe will nearly double in the next decade from 12.3 to 22.0 million m³.

[1] This section includes the article “Post-demolition autoclaved aerated concrete: Recycling options and volume prediction in Europe”, developed by Rebekka Volk, Frank Schultmann, and myself. The article was presented at the “Ecocity World Summit 2021-22” and published as Steins et al. (2022) in the “Ecocity World Summit 2021-22 Conference Proceedings” in 2022.

B.1 Introduction

Worldwide, resource consumption and CO_2 emissions are beyond a sustainable limit. Therefore, the UN aims at sustainable development goals like responsible consumption and production, climate action and sustainable cities (UN, 2023). Circular Economy plays an essential role to reach those goals – especially in the construction and demolition (C&D) sector, where substantial mass flows lead to high CO_2 emissions, energy and resource consumption and significant construction and demolition waste (C&DW) amounts. Worldwide, more than 3 billion tons of C&DW were generated in 2012 (Akhtar & Sarmah, 2018). However, a considerable share of the potential of a circular economy remains unused (OECD, 2020).

Autoclaved aerated concrete (AAC) is produced from quartz sand, cement, quicklime, anhydrite, aluminium powder/paste (as aerating agent), and water. AAC has a porous structure, low density, and exceptional thermal insulation properties among mineral building materials. Therefore, AAC is popular for masonry units and mineral insulation boards, especially in residential buildings. And, construction and deconstruction processes of AAC require less effort than layered insulated materials (e.g. bricks with insulation) because AAC is a mono-material. This leads to time and cost savings in (de)construction processes and contributes to the high popularity of AAC. E.g. in Germany, the production of AAC began in 1950 (UBA, 2019), and in 2018, 23% (trending upwards) of the completed residential buildings were built using AAC (Destatis, 2019).

AAC production waste or breakage is already recirculated. However, post-demolition AAC is not yet recycled because the porous structure, adhering substances, and small quantities of sulphate hamper high-quality recycling. Therefore, post-demolition AAC is mainly landfilled. However, decreasing landfill capacities and legal requirements – especially the European Waste Framework Directive 2008/98/EC – demand for AAC recycling. But, information on post-demolition AAC volumes is crucial to design and manage a circular supply chain for AAC. This study tries to fill the gap because only negligible information about recyclable AAC volumes in Europe is available as official statistics and comprehensive studies are lacking. First, a short overview of post-demolition AAC recycling options is provided. Then, expected post-demolition AAC volumes in Europe are quantified based on historic AAC production data, and building lifetime assumptions. Since comprehensive AAC production data is only available for Germany and the UK, post-demolition AAC volumes in other European countries and regions are predicted based on the current AAC market volume.

B.2 Post-demolition AAC recycling options in literature

Reusing post-demolition AAC blocks is no practical possibility due to the need for an overly careful deconstruction process (Gyurkó et al., 2019, p. 431) and complex transportation and storage. However, crushed post-demolition AAC in fine powder or granulate form could be used in different recycling options. First, post-demolition AAC powder could be used in AAC production (Kreft, 2017; Rafiza et al., 2019) to establish a closed-loop recycling. However, only up to 20% of primary raw materials (Kreft, 2017) or up to 50% of the sand (Rafiza et al., 2019) can be substituted by post-demolition AAC powder. Besides, belite cement clinker production from post-demolition AAC powder (Stemmermann, 2019; Ullrich et al., 2021) could handle significant amounts of post-demolition AAC, because it can be used as primary raw material in many applications such as AAC production (closed-loop recycling) or autoclaved sand-lime brick production (open-loop recycling).

Furthermore, various open-loop recycling options for post-demolition AAC are subject to current research. These options include the application of post-demolition AAC in light mortar (Aycil et al., 2016), lightweight aggregate concrete (Aycil et al., 2016; Gyurkó et al., 2019), floor screed (Bergmans et al., 2016), and no-fines concrete in the form of stumped concrete with decorative function or shuttering blocks (Gyurkó et al., 2019). Besides, there are suggestions for downcycling/utilisation options for post-demolition AAC like the use in phosphorus filters (Renman & Renman, 2012), fertilisers (Niedersen et al., 2004; J. Volk & Schirmer, 2010) and landscaping (Rühle & Maiwald, 2018).

Overall, promising recycling options for post-demolition AAC are presented in the literature. However, implementing a recycling network for these recycling options needs further knowledge on current and future post-demolition AAC volumes that can be expected.

B.3 Post-demolition AAC volume prediction in Europe

B.3.1 Methodology

The European Waste Catalogue specifies different codes to record different types of waste. However, there is no AAC-specific code. In practice, post-demolition AAC is allocated to the codes 170101 (concrete), 170107 (mixtures of concrete, bricks, tiles, and ceramics) and 170802 (gypsum-based construction materials). Thus, no inferences about the post-demolition AAC volume are possible due to large volumes of different types of building rubble recorded in these codes. And, the literature on this topic is limited to stockpile studies, e.g. Schiller et al. (2015) and Ortlepp et al. (2016) for Germany, while comprehensive waste volume studies are missing. Therefore, (Steins et al., 2021) developed a model that predicts regional (NUTS 3 level) post-demolition AAC volumes in Germany based on AAC production, building lifetime assumptions, regional construction activity, and regional AAC popularity. This study uses the same methodology to determine current and future post-demolition AAC volumes in Europe, excluding regionality below the national level. Therefore, data on regional construction activity and regional AAC popularity are unnecessary. Building lifetime assumptions (for residential and non-residential buildings) are adopted from (Steins et al., 2021). Residential and non-residential buildings are considered independently as they have different lifetimes. A triangular lifetime probability function with 35 and 95 years as lifetime boundaries and 65 years as the most probable lifetime is assumed for residential buildings. For non-residential buildings, a similar triangular lifetime probability function with lifetime boundaries of 15 and 100 years and a most probable lifetime of 40 years is assumed. The total post-demolition AAC volume from both building types is added up employing average shares of AAC used in residential and non-residential buildings. The share of AAC used in residential buildings is assumed to be 85.5% and 14.5% in non-residential buildings respectively following (Steins et al., 2021). Overall, the post-demolition AAC volume in a country for a specific year is calculated based on these lifetime assumptions and national historic AAC production using the following equation:

$$pd\ AAC_{c,y} = \sum_{a=1}^{100} \left(AAC\ production_{c,y-a} * P_r(a) * 0.855 + AAC\ production_{c,y-a} * P_{nr}(a) * 0.145 \right) \tag{B.1}$$

$pd\ AAC_{c,y}$	post-demolition (pd) AAC volume in country c and year y [m³]
$AAC\ production_{c,y}$	AAC production volume in country c and year y [m³]
$P_r(a)$	probability of a residential building's (r) lifetime of a years [-]
$P_{nr}(a)$	probability of a non-residential building's (nr) lifetime of a years [-]

However, data availability differs between the European countries (Figure B.1). Thus, there are three different post-demolition AAC prediction approaches based on data availability for every country (Figure B.2).

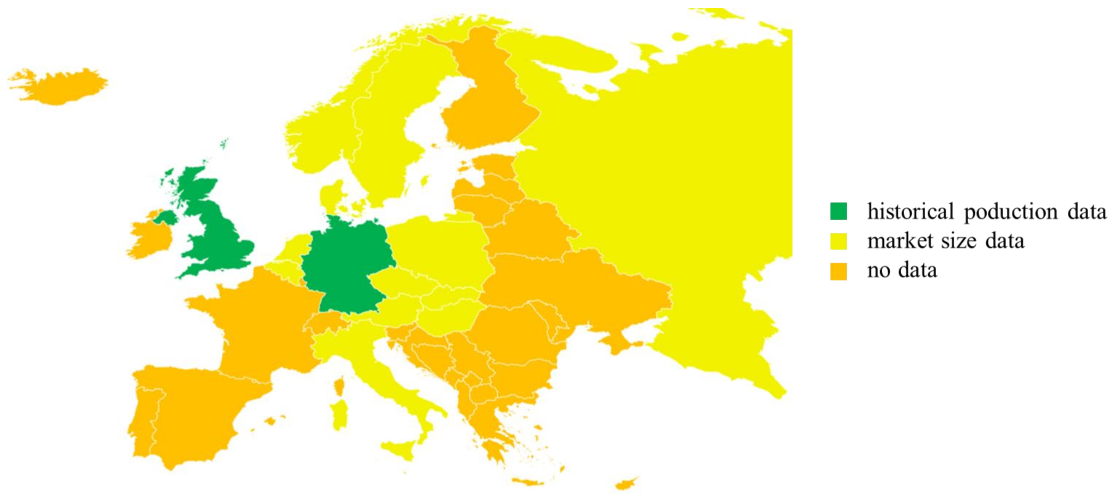

Figure B.1: Data availability for post-demolition AAC prediction in the European countries.

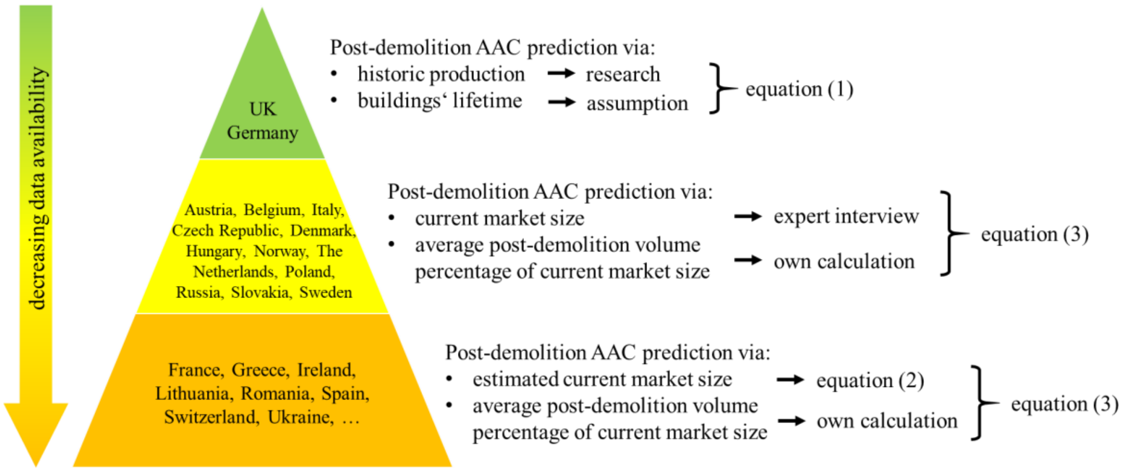

Figure B.2: Different post-demolition AAC prediction approaches depending on national data availability.

tb Comprehensive national AAC production data is only available for Germany (Steins et al., 2021) and the United Kingdom (direct inquiry to the UK Department for Business, Energy & Industrial Strategy). Therefore, for the other European countries another approach is developed: First, current AAC market sizes which reflect current production volumes are researched. Data is available for Austria, Belgium, Czech Republic, Denmark, Hungary, Italy, Norway, The Netherlands, Poland, Russia, Slovakia, and Sweden (expert interview: Dr. Oliver Kreft, Xella Technologie- und Forschungsgesellschaft mbH). Second, Europe is divided into different regions, and a representative country for every region is selected (Figure B.3) to fill the data gap of the missing countries (countries with no data in Figure B.1). All countries with missing market size data in a region are assumed to have the same AAC market size per population size as their representative country. So, the absolute market size is calculated via population data (UN, 2021) using Equation B.2. This calculation allows an extension of the AAC market size estimation to whole Europe.

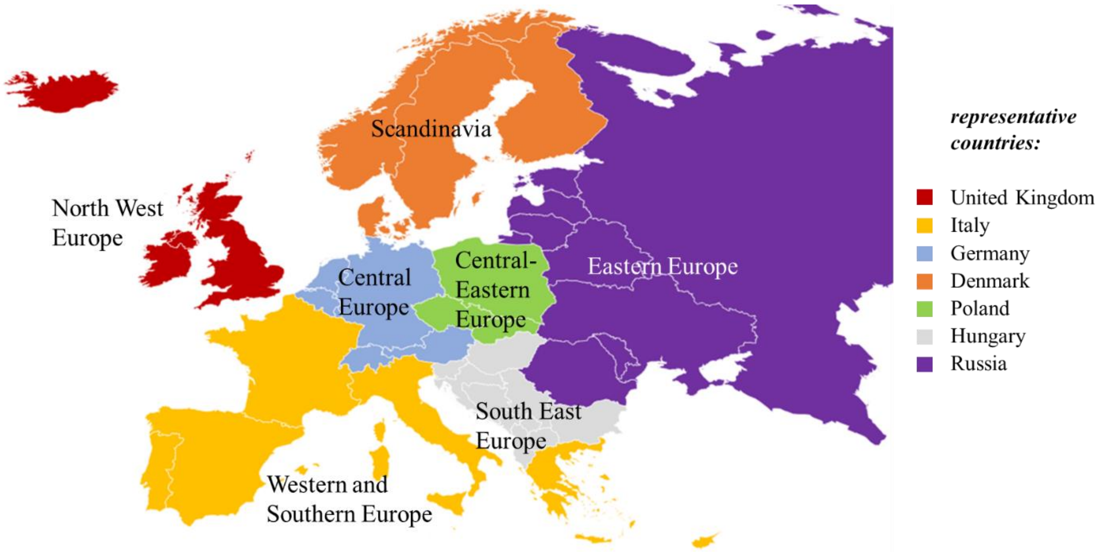

Figure B.3: European regions and their representative countries used for post-demolition AAC prediction.

tb

$$AAC\ market_c = AAC\ market_{rep} * \frac{population_c}{population_{rep}} \tag{B.2}$$

$AAC\ market_c$	current AAC market size in country c [m³]
$AAC\ market_{rep}$	current AAC market size in representative country rep [m³]
$population_c$	population in country c [-]
$population_{rep}$	population in representative country rep [-]

Third, post-demolition AAC volumes have to be calculated for the countries where only current market size data or estimation is available. To do so, the average post-demolition AAC percentages of current production volumes in Germany and the UK are taken which show a similar increase in the following years. Therefore, it is assumed that the post-demolition AAC volume for every country equals a particular percentage of the current market size, depending on the year. Finally, post-demolition AAC volumes for every European country is calculated using the following equation:

$$pd\ AAC_{c,y} = AAC\ market_c * percentage_y \tag{B.3}$$

$pd\ AAC_{c,y}$	post-demolition (pd) AAC volume in country c and year y [m³]
$AAC\ market_c$	current AAC market size in country c [m³]
$percentage_y$	post-demolition AAC percentage of the current market size in year y [-]

B.3.2 Results

Post-demolition AAC volumes can be calculated for Germany and the UK using Equation B.1 because historical production data is available. In contrast to the German dataset (Figure B.4 (a)), UK production data only goes back to 1967. Therefore, a linear increase for the period 1950 to 1966 is assumed (Figure B.4 (b)). In our model, we calculated that significant post-demolition AAC volumes have occurred since the year 2000 in both countries (Figure B.4). In the following years, calculated post-demolition AAC volumes increase sharply and constantly in both countries, exceeding 1 million m³ annually in 2018 (Germany) and in 2026 (UK) and reaching more than 4 million m³ in Germany and more than 2 million m³ in the UK in 2050. This rise would reach/exceed the AAC production in the UK/Germany if production volumes stay on today's level.

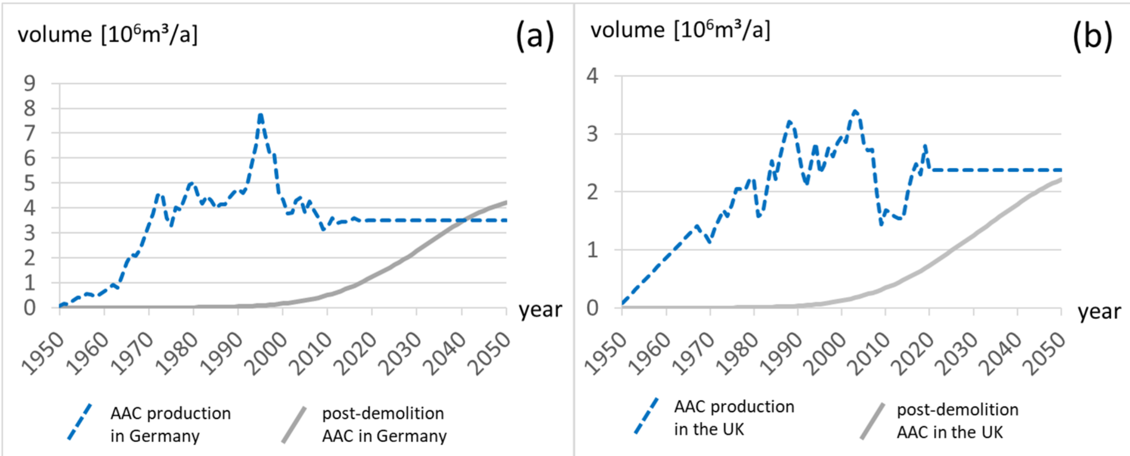

Figure B.4: AAC production (dashed blue line) and post-demolition AAC (grey line) in Germany (a) and the UK (b).

The calculation of post-demolition AAC volumes for all other countries is based on current market sizes or market size estimations and the average post-demolition volume percentage of the current market size. The average percentage is determined for different years using the German and UK calculations calculated e.g. for 2020, 2025, and 2030 (Table B.1), where the German and the UK percentages are relatively close to each other. In the more distant future, they diverge more. The post-demolition AAC calculation (Table B.2, Figure B.5) for 2020 shows relatively low volumes in South East Europe and Scandinavia, which can be explained by their small market sizes. Somewhat larger volumes can be found in the region Western and Southern Europe due to higher population and in North West Europe, where the UK alone accounts for 700,000 m³. In Central Europe, volumes are double the amount of North West Europe with the highest volume in Germany (1,200,000 m³). The most significant post-demolition volumes occur in Central-Eastern and Eastern Europe due to large markets in Poland and Russia. These two countries together (Poland: 1,800,000 m³, Russia: 3,900,000 m³) account for nearly half of the total European post-demolition AAC volume of around 12,290,000 m³ in 2020. A sharp increase of post-demolition AAC volumes throughout Europe is noticeable in the next decade.

Absolute numbers nearly double in Europe from around 12.3 to 22.0 million m³ in only ten years. Increasing AAC production in the 60s, which is expected to reach its end of life around 2030, can explain this rise. However, the relative volumes between the regions stay similar in 2025 and 2030.

Table B.1: Post-demolition AAC volume percentage of the current market size.

year	percentage Germany	percentage UK	average percentage
2020	35.2%	31.9%	33.5%
2025	49.2%	42.9%	46.0%
2030	65.2%	54.6%	59.9%

Table B.2: AAC market sizes and calculated post-demolition AAC volumes for 2020, 2025, and 2030 of all European countries (excluding small countries with less than 100,000 inhabitants).

country	(estimated*) market size [m³]	post-demolition AAC 2020 [m³]	post-demolition AAC 2025 [m³]	post-demolition AAC 2030 [m³]
Albania	79,885*	26,795	36,763	47,871
Austria	122,000	40,921	56,144	73,108
Belarus	751,782*	252,158	345,969	450,502
Belgium	330,000	110,687	151,866	197,751
Bosnia and Herzegovina	92,105*	30,893	42,387	55,193
Bulgaria	195,405*	65,541	89,925	117,095
Croatia	115,159*	38,626	52,996	69,008
Cyprus	12,651*	4,243	5,822	7,581
Czech Republic	1,160,000	389,080	533,831	695,124
Denmark	325,000	109,010	149,565	194,755
Estonia	105,216*	35,291	48,420	63,050
Finland	312,061*	104,670	143,610	187,001
France	696,788*	233,712	320,661	417,547
Germany	3,477,279	1,223,853	1,709,977	2,267,974
Greece	112,809*	37,838	51,915	67,601
Hungary	269,000	90,226	123,794	161,197
Iceland	11,500*	3,857	5,292	6,891
Ireland	164,442*	55,156	75,676	98,541
Italy	650,000	218,019	299,129	389,509
Latvia	153,331*	51,429	70,563	91,883
Lithuania	222,759*	74,717	102,514	133,487
Luxembourg	25,267*	8,475	11,628	15,141
Malta	4,707*	1,579	2,166	2,820
Moldova	322,249*	108,087	148,299	193,106
Montenegro	17,401*	5,837	8,008	10,428

Netherlands	410,000	137,520	188,682	245,690
North Macedonia	57,718*	19,359	26,562	34,587
Norway	17,000	5,702	7,823	10,187
Poland	5,400,000	1,811,235	2,485,074	3,235,923
Portugal	109,958*	36,881	50,602	65,892
Romania	1,551,282*	520,322	713,898	929,598
Russia	11,590,000	3,887,448	5,333,705	6,945,249
Serbia	243,923*	81,815	112,253	146,170
Slovakia	650,000	218,019	299,129	389,509
Slovenia	57,580*	19,313	26,498	34,504
Spain	500,609*	167,911	230,380	299,988
Sweden	106,000	35,554	48,781	63,520
Switzerland	356,663*	119,630	164,136	213,729
Ukraine	3,518,816*	1,180,260	1,619,355	2,108,633
United Kingdom	2,291,135	730,576	982,073	1,251,563

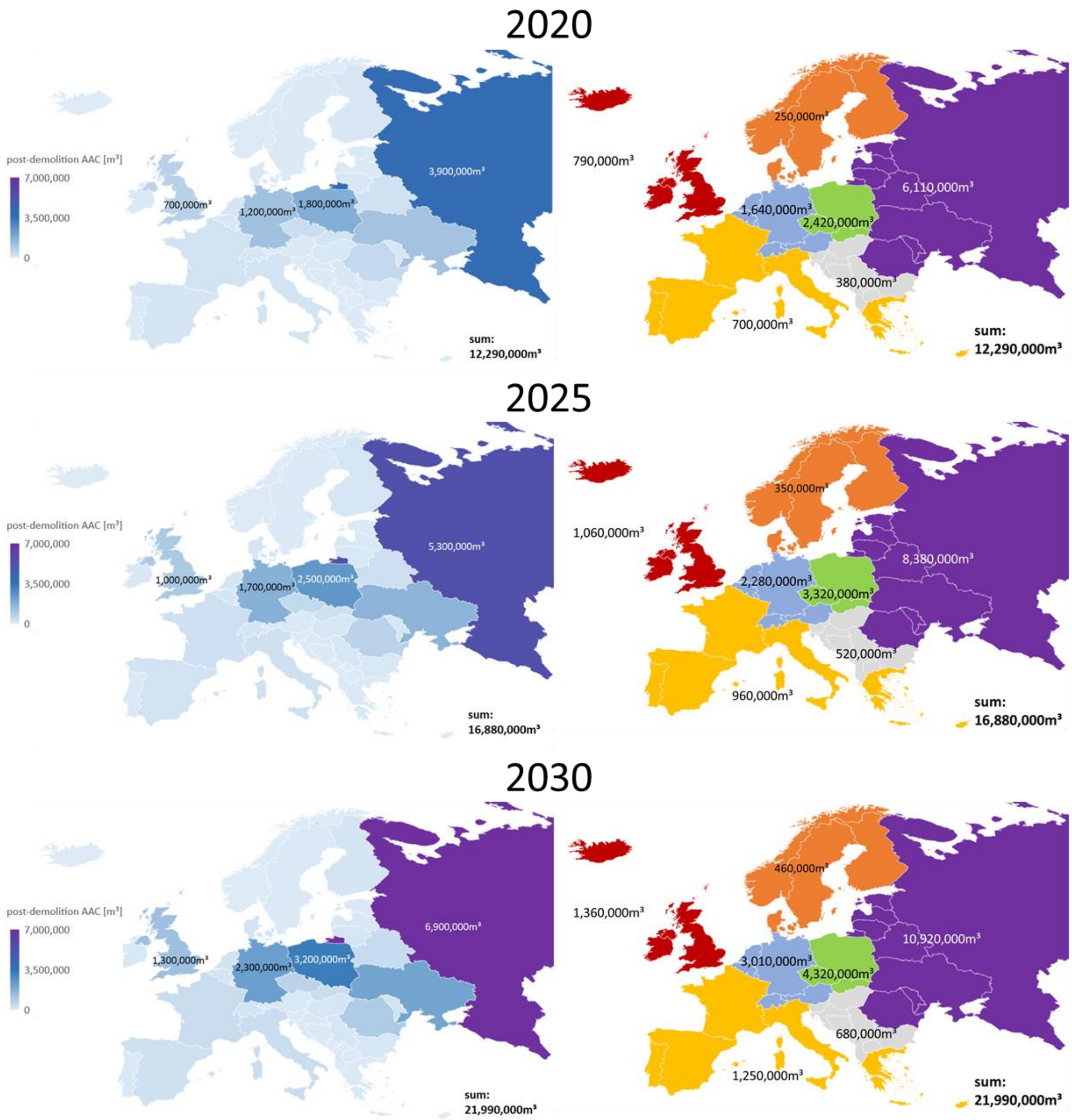

Figure B.5: Total AAC stockpile per district in Germany in 2005 (data source: Schiller et al., 2010).

The average post-demolition volume percentage of the current market size has a great influence on the total European post-demolition AAC volume because it is used in the prediction for every country except for Germany and the UK. Therefore, a sensitivity analysis of European post-demolition AAC volumes with regard to this percentage is carried out (Table B.3). The analysis shows that a +/-10 %-point variation changes the total European post-demolition AAC volume by around 3 million m³. Thus, the post-demolition AAC volume could reach more than 15 million m³ in 2020 and even more than 25 million m³ in 2030.

Table B.3: Sensitivity analysis of European post-demolition (pd) AAC volume with regard to the post-demolition percentage of the current market size.

year	percentage (baseline)	pd AAC volume [m³]	percentage (-10 %-points)	pd AAC volume [m³]	percentage (+10 %-points)	pd AAC volume [m³]
2020	33.5%	12,290,000	23.5%	9,200,000	43.5%	15,360,000
2025	46.0%	16,880,000	36.0%	13,790,000	56.0%	19,950,000
2030	59.9%	21,990,000	49.9%	18,900,000	69.9%	25,070,000

B.3.3 Limitations and shortcomings

An essential component of the post-demolition AAC prediction is the assumption of lifetime functions for residential and non-residential buildings. However, the lifetime functions are based exclusively on literature values. Empirical data is not available. And, various building characteristics (e.g. monument conservation, building material, construction technique, floor plan, renovation) influence its lifetime fundamentally. Overall, the lifetime functions are subject to noticeable uncertainties.

Furthermore, comprehensive production data is scarce, so only post-demolition AAC volumes in Germany and the UK can be calculated using the basic approach. For all other countries, post-demolition AAC volume is calculated using the current market size's average post-demolition volume percentage. This percentage is calculated as the mean value of only two countries (Germany and the UK) and thus also associated with uncertainties. Besides, reliable data on the current market size is only available for some countries. The market size of the remaining countries has to be estimated. Generally, an AAC-specific waste code or AAC waste statistics are missing but could help to quantify the available post-demolition AAC volume for recycling. Furthermore, the used percentage represents the situation in the longer existing AAC markets of Germany and the UK while AAC markets especially in Southern and South East Europe may be much younger and used percentages are not suitable. A change would shift post-demolition AAC volumes into the future. In general, including comprehensive AAC production or market size data of the past would improve the prediction of the model.

B.4 Conclusion

A European prediction model was developed to assess future post-demolition AAC volumes at national level. In 2020, volumes are the largest in Russia (3,900,000 m³) and Poland (1,800,000 m³). In other regions, Germany (1,200,000 m³) and the UK (700,000 m³) account for the largest expected national post-demolition AAC volumes. Furthermore, post-demolition AAC volumes in Europe will increase considerably in the following decade from 12.3 to 22.0 million m³. Therefore, AAC recycling has to be fostered to avoid landfilling these volumes and eventually to substitute primary construction material.

Further research could focus on gathering data on AAC production and AAC market sizes for more European countries to improve the prediction quality. Furthermore, building lifetimes should be further investigated. Especially knowledge on the lifetimes of different AAC products would be interesting. The presented model can be transferred to other regions/continents and to other building materials like clay bricks, timber, sand-lime bricks, or lightweight concrete blocks. To do so, adaptions on production data and possibly building lifetimes are necessary.

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C Life Cycle Assessment of Post-Demolition Autoclaved Aerated Concrete (AAC) Recycling Options

Abstract[1]

Autoclaved aerated concrete (AAC) is a widely used building material for masonry units, prefabricated reinforced components, and lightweight mineral insulation boards. Its low thermal conductivity and good fire resistance increase its popularity in residential buildings. Thus, post-demolition wastes are expected to increase in the future. However, post-demolition AAC (pd-AAC) is mainly disposed in landfills while landfill capacities decrease and legal framework conditions in Europe are tightening. This study performed life cycle assessments (LCA) of different pd-AAC recycling options and compared them to each other and to current landfilling to identify the best end-of-life handling of pd-AAC from an ecological perspective. The functional unit was 1 kg pd-AAC, and the system boundaries included pd-AAC at the demolition site, transports, pd-AAC treatment, and secondary production processes. Final products of the recycling process gained environmental credits/rewards for avoiding primary production using system expansion. Providing primary resources, primary production, and use phase were not in the scope of this study. Results show that especially closed-loop recycling of pd-AAC in AAC production has a high potential of improving environmental impacts. In the best recycling option (high substitution in AAC-0.35), potential savings per kg pd-AAC compared to landfilling reach up to 0.5 kg CO_2-Eq, 7 MJ fossil resources, 0.005 mol H+-Eq (acidification), 0.17 CTU (freshwater ecotoxicity), 0.2 g P-Eq (freshwater eutrophication), 5.2×10^{-9} CTUh (carcinogenic effects), 4.4×10^{-8} CTUh (non-carcinogenic effects), 2.5×10^{-5} g CFC-11-Eq (ozone layer depletion), and 1.6 g NMVOC-Eq (photochemical ozone creation). Despite data uncertainties, recycling of pd-AAC is advantageous for several recycling options, including the production of AAC, light mortar, lightweight aggregate concrete, and shuttering blocks made from concrete without fine fractions (no-fines concrete). In Germany, up to 280,000 t CO_2-Eq could have been saved in 2022 by pd-AAC recycling using different recycling options instead of landfilling.

[1] This section includes the article “Life cycle assessment of post-demolition autoclaved aerated concrete (AAC) recycling options”, developed by Rebekka Volk, Oliver Kreft, Frank Schultmann, and myself. The article was published as R. Volk et al. (2023) in the journal “Resources, Conservation & Recycling” in 2023. The supplementary material can be found on the journal website.

C.1 Introduction

tbd The construction and demolition (C&D) sector is associated with large shares of global greenhouse gas (GHG) emissions and resource consumption. In 2012, construction and demolition waste (C&DW) exceeded 3 billion tons worldwide (Akhtar & Sarmah, 2018), and the global concrete production used between 25.9 to 29.6 billion tons of aggregates (Peduzzi, 2014). In the future, "the pressure on natural resources will increase, while new infrastructure, services, and housing will be needed" due to a rising world population (OECD, 2020). Therefore, reaching the UN sustainable development goals, particularly sustainable cities, responsible consumption and production, and climate action (UN, 2023), will not be possible without the C&D sector. C&DW recycling is a promising approach to preserving natural deposits of sand, gravel, lime, and other construction materials and reducing GHG emissions. However, "the potential of the circular economy to support sustainable cities, regions, and countries still needs to be unlocked" (OECD, 2020) as inappropriate design-for-recycling, ineffective collection/sorting, and immature recycling technologies hamper an effective circularity of building materials. The European waste and recycling regulation (Directive 2008/98/EC, 2008) stipulates C&DW recycling rates of 70% and requires fulfilment of Regulation No 305/2011 (Regulation (EU) No 305/2011, 2011) for products with recycled content.

Autoclaved aerated concrete (AAC) is made of quartz sand, cement, quicklime, anhydrite or gypsum, aluminium powder/paste (as aerating agent), and water (DIN 20000-404:2018-04; Kreft, 2017). During production, a porous structure typical for AAC is formed where the millimetres to nanometres sized pores make up 60 to 85 vol.-% (Anders, 2018). AAC has a low density and excellent thermal insulation properties that outperform other materials like classical clay bricks, calcium silicate units, or concrete. Annually, 16 million m³ AAC is produced in Europe (EAACA, 2023) and 11.6 million m³ (2017) in Russia (Grinfel'd et al., 2018). Globally, a production capacity of 450 million m³ for non-reinforced AAC blocks is prevalent (Fouad & Schoch, 2018). For Germany, an annual post-demolition AAC (pd-AAC) volume of 1.4 million m³ in 2022 and a sharp increase to more than 4 million m³ in 2050 is expected (Steins et al., 2021).

Usually, post-demolition mineral construction materials are downcycled and used in road construction, earthworks, civil engineering, concrete production, and landscaping (Knappe et al., 2012). However, pd-AAC cannot be recycled in the applications mentioned above due to adhering substances (Deilmann et al., 2014), porous structure, and sulphate content. Besides, AAC has a relatively low compressive strength, preventing recycling in load-bearing components. Thus, pd-AAC is mainly backfilled or landfilled. But, landfilling capacities are limited, and landfill fees are expected to rise, especially in densely populated areas (Knappe et al., 2012; Riegler-Floors & Hillebrandt, 2018).

Unlike freshly produced AAC, crushing of pd-AAC (Section C.2) results in more AAC powder (approximately 75%) than granulate (approximately 25%) (practical trials[2]; (Gyurkó et al.,

[2] Expert interview with Xella Technologie- und Forschungsgesellschaft mbH.

2019)). While pd-AAC granulate retains the porous AAC structure, pd-AAC powder does not. If sufficient purity is given, the granulate could be used for different purposes, e.g. oil binder and animal bedding, as it is already done today with granulate from freshly produced AAC. In contrast, only limited applications for pd-AAC powder are available. Therefore, high-quality recycling options, particularly for pd-AAC powder, are needed.

Currently, direct reuse of pd-AAC blocks is not feasible in practice due to high costs resulting from a meticulous demolition process (Gyurkó et al., 2019), separation/cleaning steps, effortful storage, and transportation. Moreover, as historical AAC blocks do not comply with today's requirements for i. e. thermal protection, areas of application would be limited. Therefore, after the demolition process, a crushing and grading of pd-AAC have to be carried out (Kreft, 2016) to separate pd-AAC granulate (grain size > 1 mm) from pd-AAC powder (grain size 0-1 mm). After further sorting, the purified pd-AAC granulate and powder can be used for different recycling options. Figure C.1 shows the pd-AAC landfilling and recycling processes.

First, closed-loop recycling options were investigated. Theoretically, closed-loop recycling could establish a closed material loop. In practice, pd-AAC can substitute primary raw materials in AAC production up to a given threshold, which depends on the density class of the intended AAC product. Kreft (2017) investigates the use of pd-AAC powder in the production of new AAC, substituting the primary resources sand, cement, lime, and anhydrite (Kreft, 2017). Rafiza et al. (2019), Rafiza et al. (2022), and Lam (2021) describe it as well. However, they investigate the substitution of sand with up to 50% (Rafiza et al., 2019) respectively 100% (Lam, 2021) AAC powder, but it is unclear if the used AAC powder stems from post-demolition or production wastes with higher purity. Another way to achieve closed-loop recycling for pd-AAC is to produce intermediate products for AAC production. Stemmermann (2019) and Ullrich et al. (2021) investigate the production of belite cement clinker made from pd-AAC powder that can again be used for producing AAC or other mineral materials. Furthermore, this approach could lead to a reduction in energy consumption, the separation of valuable and associated harmful substances, and the production of a high-quality product. However, data on energy consumption of the belite cement clinker production and recipes for AAC or other material production from belite cement clinker are still missing. Therefore, this new recycling approach cannot be included in this study.

Besides, there are various open-loop recycling options for pd-AAC. First, pd-AAC powder can substitute primary raw materials in cement clinker production. Schoon et al. (2013) conducted a feasibility study including various pd-AAC samples and varying samples with primary clinkers from different sources. They conclude that pd-AAC recycling in cement clinker production is possible but unpractical due to a high energy demand for water evaporation and potential contaminants in the pd-AAC (Schoon et al., 2013). Other research also confirms that only production waste with significantly lower impurities than pd-AAC can be used (Vogel et al., 2011). Also, pd-AAC powder can be used as a filler or supplementary material in the concrete leading to increased strength and durability (Gyurkó et al., 2019). Moreover, pd-AAC powder is used to produce light mortar (Aycil et al., 2016) in laboratory tests. A mixture of pd-AAC powder and granulate can also be recycled in floor screed to replace sand (Bergmans et al., 2016). However, sulphate leaching from pd-AAC is problematic for this application and the recycling of pd-AAC in general (Bergmans et al., 2016). Nevertheless, in floor screed, the sulphate can react with the cement binder forming insoluble ettringite (Bergmans et al., 2016). Similarly, Zou et al. (2022) show that pd-AAC can substitute sand in mortar. Besides, a mixture of pd-AAC powder and granulate can be used to produce lightweight aggregate concrete (LWAC) (Aycil et al., 2016) in laboratory tests. Gyurkó et al. (2019) also investigate an LWAC composition based on a mixture of both pd-AAC powder and pd-AAC granulate and a composition based on pd-AAC granulate only. However, the production of a load-bearing LWAC requires high cement amounts, and the LWAC has relatively low strength and frost resistance, reducing its application potential (Gyurkó et al., 2019). Finally, pd-AAC granulate can be used to produce no-fines concrete, a concrete type without any fine aggregates like sand, with applications as a self-supporting wall, stumped concrete with decorative function (exposed concrete), and shuttering blocks (Gyurkó et al., 2019). In the following, the focus lies on the application as shuttering block.

Additionally, several other open-loop recycling options for pd-AAC granulate outside the construction sector are discussed in the literature: bioactivation for methane emission reduction in landfills (Bukowski et al., 2015), filter material for phosphorus wastewater (Renman & Renman, 2012), soil conditioner (Niedersen et al., 2004), soil materials and fertilisers (Volk & Schirmer, 2010), construction of ponds, canal bases and embankments (Rühle & Maiwald, 2018). However, there are no comparable primary products; thus, an assessment beyond the pd-AAC granulate is impossible. Therefore, these recycling options were excluded from the following life cycle assessment.

Much research focuses on the LCA of building materials (e.g. Christoforou et al., 2016; Jonsson et al., 1998; Mitterpach & Štefko, 2016; Zimele et al., 2019). Additionally, innovative ideas and the use of secondary material in building materials' production are assessed in many studies using LCA (e.g. Ahmed & Tsavdaridis, 2018; Colangelo et al., 2018; Bories et al., 2016; Knoeri *et al.*, 2013). Also, there is research on AAC produced with recycled content (Nühlen et al., 2020). However, pd-AAC recycling options have not been assessed to a large extent, nor are respective LCA data available in the literature. Thus, the central research gap addressed by this study is the environmental assessment of closed-loop and open-loop[3] recycling of pd-AAC in construction materials. Furthermore, the comparison of different pd-AAC recycling options is missing in the literature and will be carried out in this study. Thus, the research objective is to answer whether pd-AAC recycling in construction materials can be environmentally beneficial and which recycling options show the lowest environmental impacts. In the following, the assessment methodology is described (Section C.2). This section is followed by the impact assessment, a sensitivity analysis, and a discussion of shortcomings (Sections C.3 and C.4). Finally, the results are concluded (Section C.5).

[3] Closed-loop recycling means using the pd-AAC after processing steps in the production of new AAC products. In contrast, open-loop recycling options use pd-AAC to produce other (construction) materials than AAC.

C.2 Methods

C.2.1 Materials

In this paper, different end-of-life options for pd-AAC were compared using LCA. On the one hand, landfilling as the state-of-the-art end-of-life option for pd-AAC is included in the comparison. On the other hand, several recycling options are investigated. First, closed-loop recycling of pd-AAC in AAC production is considered. There are studies investigating this recycling option where pd-AAC powder substitutes sand respectively all primary AAC inputs. In contrast to closed-loop recycling, different open-loop recycling options focusing on construction materials are included in the comparison. Pd-AAC powder can be used as supplementary material in concrete production or as a substitute for primary sand and lightweight aggregates in light mortar production. A mixture of pd-AAC powder and granulate can be recycled in the floor screed production by substituting sand or in the LWAC production by substituting lightweight aggregates. Moreover, pd-AAC granulate can be used to produce shuttering blocks made from no-fines concrete, also substituting lightweight aggregates. Production recipes for these considered recycling options and assumptions are explained in detail in the section on the inventory analysis (C.2.3).

C.2.2 Methodological framework (goal and scope)

The goal was to determine whether recycling of pd-AAC is superior to landfilling and which recycling options perform the best. Furthermore, total savings from implementing a beneficial recycling strategy were calculated.

LCA follows the cradle-to-grave approach, which includes all processes from providing the resources to production, use phase, and end-of-life. The final output of the production usually serves as the functional unit. However, for LCA focusing on the end-of-life stage, the so-called zero burden approach can be applied to meet the particular characteristics of end-of-life assessment by adjusting two significant aspects (Nakatani, 2014): the system boundaries and the functional unit. Concerning the system boundaries, the zero burden approach does not consider the processes until the emergence of waste products (providing resources, production, use phase) to focus on disposal or recycling assessment. This simplification is possible since the processes before end-of-life are identical for every option. The second adjusted aspect is a change of the functional unit. When applying the zero burden approach to a waste management system, the input (the waste) serves as the functional unit (Nakatani, 2014). A comparison of different end-of-life options without an input-based functional unit would not be meaningful as different amounts of waste are handled in the scenarios. Here, the pd-AAC end-of-life LCAs followed the zero burden approach to model and analyse the pd-AAC end-of-life processes. Therefore, providing resources, production, and the use phase was not considered, and the functional unit of 1 kg pd-AAC entered the assessed system without any burdens. The system boundaries included the waste product (pd-AAC at the demolition site) and the waste treatment/recycling processes, including their outputs, as shown in Figure C.1.

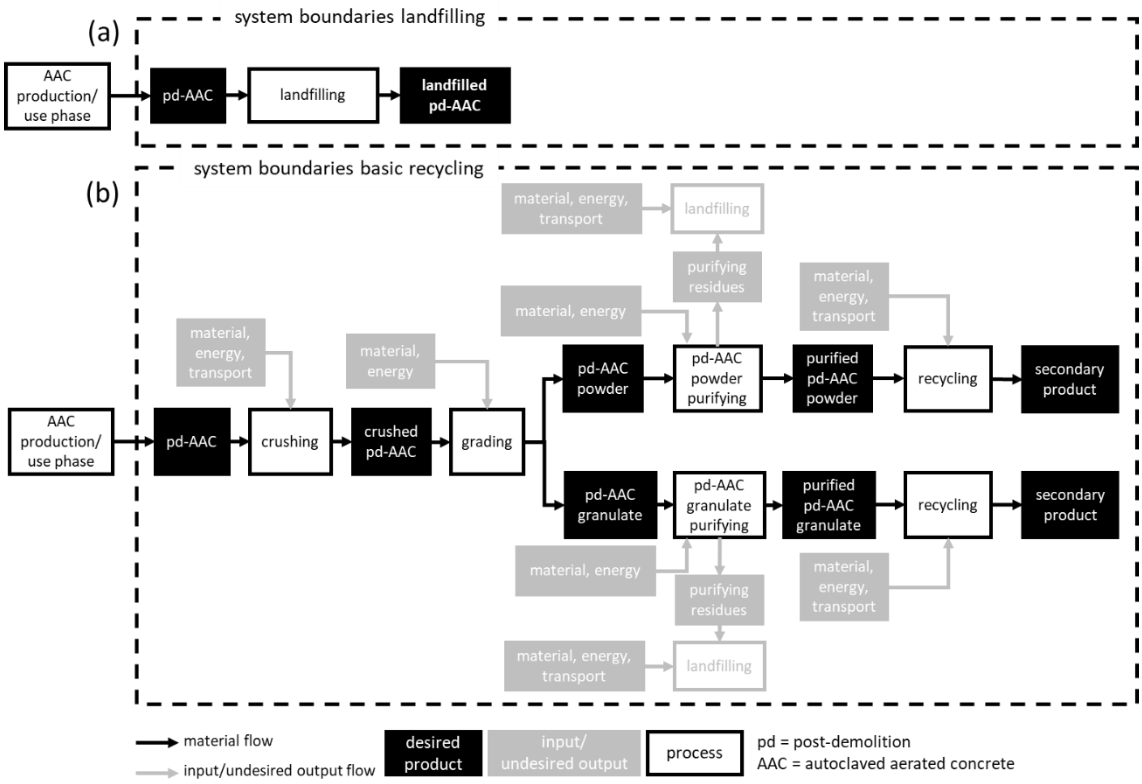

Figure C.1: Overview of (a) the pd-AAC landfilling and (b) the basic pd-AAC recycling process, including crushing, grading, and purifying steps.

In closed-loop and open-loop recycling processes, the desired outputs are valuable products with their LCA data but also come with waste-like sorting residues. Therefore, the ISO standard 14040/14044 encourages a system expansion to include these outputs in the recycling LCA. Nakatani (2014) introduces two different approaches for system expansion. The avoided burden approach assumes that the recycling process' desired output replaces a primary product. Then, the recycling process gains an environmental credit/reward (subtraction) in its LCA because burdens for the primary production of the replaced product are avoided. In the product basket approach, the desired recycling output is rewarded by crediting (addition) the recycling system with the primary product (inverse to the avoided burden approach). These two approaches lead to different absolute LCA values due to the differing credit sign for the replaced primary product. However, the comparative burdens and the overall statement remain the same (Nakatani, 2014). Therefore, this study uses both methodological options without changing the results. In the following, we chose the system expansion using the avoided burden approach. It allows a more comprehensible graphical presentation of the results as subtracted credits/rewards directly oppose the efforts for the recycling process. And the handling of waste (non-valuable undesired outputs) is also considered (Section C.2.3 purifying process).

C.2.3 Process assessment and inventory analysis

This section describes data sources and assumptions for every process under study. Tables that contain the energy and material inputs and outputs per process, their amounts, uncertainty used for Monte Carlo simulation (Section C.3.3), and the references are given in the Supporting Information (SI)-2. Primarily, weight-based amounts of input and output materials were used to achieve the best comparability between the different recycling options and to match the functional unit (1 kg pd-AAC). Conversions from volume- to weight-based amounts are explained in the following where necessary.

The open-source software openLCA was used to model and assess the different end-of-life options. Relevant data, especially recipes for recycling products, are taken from the literature (see below). The ecoinvent 3.6 database was used to assess general processes (crushing, grading, landfilling) and primary production (for substitution credits) and to fill data gaps in the literature. Data from industrial plants is not available for the pd-AAC recycling processes because pd-AAC is mostly backfilled or landfilled today (Section C.1). The ecoinvent data quality system was used for a Monte Carlo simulation to conduct a sensitivity analysis. Figure C.2 shows all assessed recycling options, including preparation steps, inputs, outputs, and substitution products.

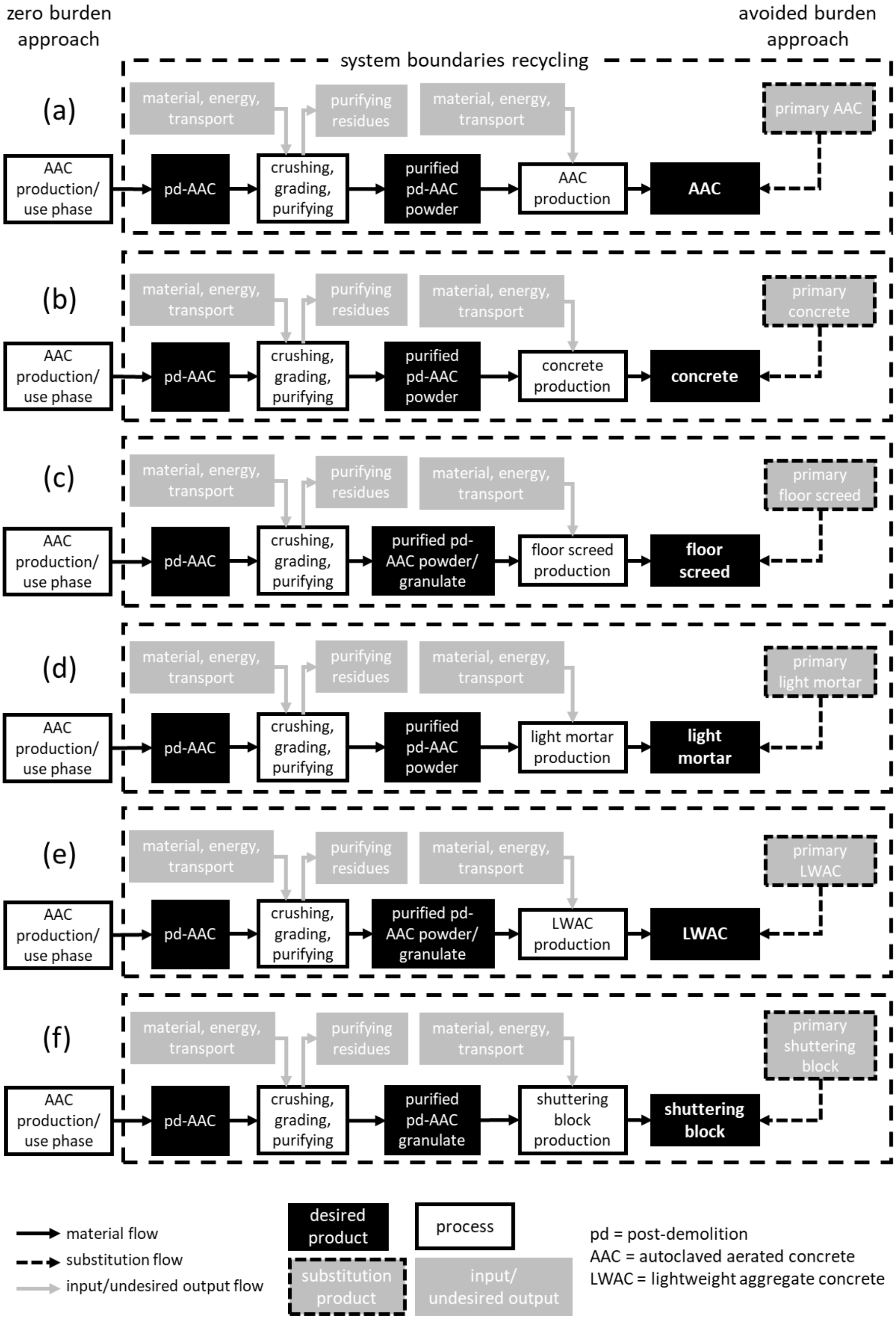

Figure C.2: Assessed pd-AAC recycling options in the construction sector, including (a) AAC production (closed loop), and open-loop options (b) concrete production, (c) floor screed production, (d) light mortar production, (e) LWAC production, and (f) shuttering block production.

The **landfilling** of pd-AAC (reference end-of-life option) and purifying residues was assessed using the ecoinvent 3.6 dataset “treatment of inert waste, inert material landfill”, which discloses electricity/diesel/heat efforts, occupation, and transformation efforts.

The **crushing** of pd-AAC was assessed based on the ecoinvent 3.6 dataset “rock crushing” using one functional unit as input and 1 kg of crushed pd-AAC as output (without material loss).

The **grading** of crushed pd-AAC is mandatory to separate the pd-AAC powder from pd-AAC granulate since they are generally used for different recycling purposes. Additionally, **pd-AAC powder/granulate purifying** is essential to remove as many adhesions and impurities as possible to enable high-quality recycling. However, there is no data from industrial sites for both processes and no grading or purifying process regarding crushed AAC available in the ecoinvent 3.6 dataset. Therefore, the process “treatment of waste brick, sorting plant” was chosen as an approximation for both processes combined because AAC and bricks are masonries. This process was the best fitting available dataset since it is more suitable than those concerning gravel sorting of (reinforced) concrete. Transport efforts were not considered in this recycling step as it is assumed that pd-AAC is crushed, graded, and purified at the same place. However, the outputs are assumed to be transported 50 km for the final recycling step for all considered recycling options. We assumed the purifying of 1.01 kg input results in 1 kg purified pd-AAC powder/granulate and 0.01 kg residue sorted out. The grading was assumed to have no material loss. The residue was supposed to be landfilled using the above assessment of landfilling. The process efforts were allocated physically to the two outputs, purified pd-AAC powder and purified pd-AAC granulate.

The **AAC production with pd-AAC powder** was assessed according to Kreft (2017) with a substitution of sand, quicklime, cement, and anhydrite and according to Rafiza et al. (2019) and Lam (2021) with a substitution of sand only (Section C.1). Therefore, two different LCAs were conducted. The substitution amounts of sand, quicklime, cement, and anhydrite by pd-AAC powder depend on the produced AAC’s density class, which influences the relative shares of the primary inputs. Therefore, three different AAC density classes were considered: AAC-0.35 (class “0,35”, density 305 to 350 kg/m^3), AAC-0.50 (class “0,50”, density 455 to 500 kg/m^3), and AAC-0.55 (class “0,55”, density 505 to 550 kg/m^3) (DIN 20000-404:2018-04; DIN EN 771-4:2015-11). Table C.1 displays typical production recipes for above mentioned AAC density classes. Indicated input share intervals result from the fact that manufacturers have to adapt their production formulations to local raw material qualities (e.g. lime reactivity, sand purity and fineness) and the process technology available on site (various production technologies exist side by side that have evolved historically and were or are partly protected from each other by patents). The recipes provide shares for the main inputs for AAC production, excluding additives but including primary AAC powder from AAC production breakage. The centre of the input share intervals of Table C.1 was chosen for the subsequent assessment. Data on further primary inputs like energy was based on the ecoinvent 3.6 dataset “autoclaved aerated concrete block production”. For substitution, it was assumed that all primary raw materials are replaced according to their input share. The larger the share, the more is substituted by pd-AAC powder. A low and a high substitution was considered for every density class. The high substitution is the maximum substitution realisable in practice without production-related disruptions and without violation of normative specifications or other quality requirements on the final product. To ensure this, prototypes with increased powder content were developed first on "laboratory level" at the small-scale pilot plant of the Xella Technologie- und Forschungsgesellschaft mbH (hereinafter referred to as XTF). In 2021, the new formulations were sucessfully validated by up-scaling to a production-typical casting volume of 5 m^3 at XTF's large-size pilot plant. According to our current knowledge, the increased powder shares do not have negative impacts on product properties (i.e. compressive strength), and the first test productions at Xella AAC plants according to the new formulations are currently implemented.

The low substitution is five percentage points below the high substitution and indicates the assumed minimum degree that can be implemented even in unfavourable framework conditions (again raw material properties and type of production technology).

Thus, assumed weight-based input shares for pd-AAC powder were 2% (low) and 7% (high) for density class AAC-0.35, 2% (low) and 7% (high) for AAC-0.50, and 5% (low) and 10% (high) for AAC-0.55. Overall, AAC powder input (primary and pd-AAC powder) in the high substitution case sums up to 16% (AAC-0.35) and 21% (AAC-0.50 and AAC-0.55).

Table C.1: Primary AAC production recipes for different density classes.

AAC density class	AAC-0.35	AAC-0.50	AAC-0.55
Input share sand	36%-40%	43%-47%	51%-55%
Input share quicklime	13%-15%	16%-18%	13%-15%
Input share cement	29%-33%	18%-20%	15%-17%
Input share anhydrite	4%-6%	2%-4%	2%-4%
Input share primary AAC powder	7%-9%	12%-14%	9%-11%

Rafiza et al. (2019) investigated AAC production recipes with between 15% and 50% of sand substituted by pd-AAC powder. Results for this lower and upper interval limit (low/high substitution) are shown in Section C.3.1. Results for substitution rates in this interval can be directly calculated due to a linear relationship because only one primary input is substituted. Lam (2021) investigated AAC production recipes with up to 100% sand substituted by AAC powder but found that the maximum substitution for meeting crucial requirements is 25%. Therefore, the 50% substitution investigated by Rafiza et al. (2019) stays the upper interval limit for the LCA in this study. The assessment of primary production, recipes after the sand substitution, and rewards for substituting primary AAC were based on the ecoinvent 3.6 dataset "autoclaved aerated concrete block production", which considers AAC of the density class AAC-0.50.

The **concrete production** assessment using pd-AAC powder was based on the ecoinvent 3.6 dataset "concrete production 25-30 MPa", which was also used to calculate the substitution rewards of primary concrete. This concrete was chosen because Gyurkó et al. (2019) state the strength class of the investigated concrete as C25/30. Input amounts of pd-AAC powder, cement, gravel, and sand followed Gyurkó et al. (2019): The cement amount was directly given (270 kg/m³), and the pd-AAC powder was specified as 10 % of this (27 kg/m³). The amount of gravel includes 4/8 mm and 8/16 mm aggregates (1055 kg/m³), while the sand amount corresponds to the 0/4 mm aggregate amount (936 kg/m³). Substituted primary concrete production inputs equal these amounts, so the products are directly comparable, and the pd-AAC powder's environmental impact can be revealed.

Aycil et al. (2016) provide a **light mortar production** recipe using pd-AAC powder. For the assessment, the amounts for pd-AAC powder, aluminium, cement, organic chemicals, and water were given by Aycil et al. (2016). Further efforts like electricity or packing were taken from the ecoinvent 3.6 dataset "light mortar production". This dataset also served as the primary light mortar substitution reward.

In the **floor screed production** using pd-AAC powder and pd-AAC granulate investigated by Bergmans et al. (2016), amounts for pd-AAC, cement, and water are directly given. These were used for the assessment after conversion to a mass-based output using the floor screed density (1.75 t/m³ as the sum of all inputs, Bergmans et al., 2016). However, Bergmans et al. (2016) only provide the total amount of "AAC aggregate" without disclosing pd-AAC powder and granulate shares. As the pd-AAC is crushed before usage in the floor screed (Bergmans et al., 2016), a share of approximately 75% powder and 25% granulate were assumed (Section C.1). However, pd-AAC powder and granulate efforts are the same (see above), so its distribution does not influence the LCA results. Other inputs like primary sand and electricity were taken from the ecoinvent 3.6 dataset "cement cast plaster floor production", which was also used for primary floor screed rewards/assessment. The mortar production using pd-AAC, as described by Zou et al. (2022), was not separately included in the comparison as results would be very similar to those of the floor screed production.

The **LWAC production** using pd-AAC powder and pd-AAC granulate was investigated by Aycil et al. (2016) and Gyurkó et al. (2019). Fundamental inputs and emissions were taken from the ecoinvent 3.6 dataset "lightweight concrete block production, expanded clay". The LWAC recipe (option 1) by Aycil et al. (2016) includes amounts for pd-AAC powder, pd-AAC granulate, cement, water, and hard coal ash, which were used for the assessment. Gyurkó et al. (2019) also provide a recipe for their investigated LWAC (option 2) from pd-AAC powder and granulate, including intervals of amounts for AAC aggregate, cement, and water. In the assessment, the centres of these intervals were considered. Based on a grain size distribution (Gyurkó et al., 2019), a 40% powder (< 1mm) and 60% granulate (> 1mm) allocation of AAC aggregates was assumed. The required water amount was calculated using the water-cement ratio and the cement amount. Gyurkó et al. (2019) also investigated LWAC production (option 3) only using pd-AAC granulate. Again, amounts for AAC aggregate, cement, and water were given. This data was handled the same way as the other recipe (option 2). The assessment of the reference primary LWAC production was entirely based on the ecoinvent 3.6 dataset "lightweight concrete block production, expanded clay".

Besides, Gyurkó et al. (2019) investigated the recycling of pd-AAC granulate in **shuttering blocks** made of no-fines concrete. Again, a recipe disclosed input amounts of pd-AAC granulate, cement[4], and water and was used for the assessment. The water amount is calculated using the water-cement ratio. However, there is no ecoinvent 3.6 dataset on "no-fines concrete" or "shuttering block production". Therefore, the dataset "lightweight concrete block production, expanded clay" was used for this purpose. The main difference between LWAC and no-fines concrete is the existence of fine aggregates in LWAC. Still, the production processes are alike, so it is assumed that this process adequately represents no-fines concrete/shuttering block production. The primary shuttering block production inputs are identical to those described in Gyurkó et al. (2019), but the pd-AAC granulate is replaced by primary expanded clay.

C.3 Results

C.3.1 Life cycle impact assessment

The following results compare all available pd-AAC recycling options in literature and research based on energy and material balances in a life cycle impact assessment (LCIA). The LCIA reflects the specific case of Central Europe/Germany. So, all input providers were chosen to be from the German/European area if possible.[5] The "ILCD 2.0 2018 midpoint" (ecoinvent, 2019) method was chosen for LCIA. Table C.2 shows which midpoints are included in the ILCD 2.0 2018 method. Selected results for each midpoint are presented in the main text. The remainder can be found in SI-1. All numbers used to create the figures are given in the SI-2. First, pd-AAC landfilling and basic processing (crushing, grading, purifying) are compared (Figure C.3).

[4] Gyurkó et al. (2019) specifies the cement input as 300 litres/m³ shuttering block respectively 100 kg/m³. However, this would equal a non-realistic cement density of 0.33 t/m³. Therefore, it is assumed that these values are unintentionally mixed up and the input amount of cement is 100 litres/m³ shuttering block respectively 300 kg/m³ which would equal a realistic density of 3 t/m³.

[5] The priority for the provider selection was: Germany > Europe without Switzerland / Europe > Rest-of-the-World / Global.

Table C.2: Overview of LCIA midpoints in the "ILCD 2.0 2018 midpoint" method and where results are provided (ecoinvent, 2019).

Midpoint	Unit	Results provided in
climate change - climate change biogenic	kg CO_2-Eq	SI
climate change - climate change fossil	kg CO_2-Eq	SI
climate change - climate change land use and land use change	kg CO_2-Eq	SI
climate change - climate change total	kg CO_2-Eq	Section C.3
ecosystem quality - freshwater and terrestrial acidification	mol H+-Eq	Section C.3
ecosystem quality - freshwater ecotoxicity	CTU	Section C.3
ecosystem quality - freshwater eutrophication	kg P-Eq	Section C.3
ecosystem quality - marine eutrophication	kg N-Eq	SI
ecosystem quality - terrestrial eutrophication	mol N-Eq	SI
human health - carcinogenic effects	CTUh	Section C.3
human health - ionizing radiation	kg U235-Eq	SI
human health - non-carcinogenic effects	CTUh	Section C.3
human health - ozone layer depletion	kg CFC-11-Eq	Section C.3
human health - photochemical ozone creation	kg NMVOC-Eq	Section C.3
human health - respiratory effects, inorganics	disease incidence	SI
resources - dissipated water	m^3 water-Eq	SI
resources - fossils	MJ	Section C.3
resources - land use	points	SI
resources - minerals and metals	kg Sb-Eq	SI

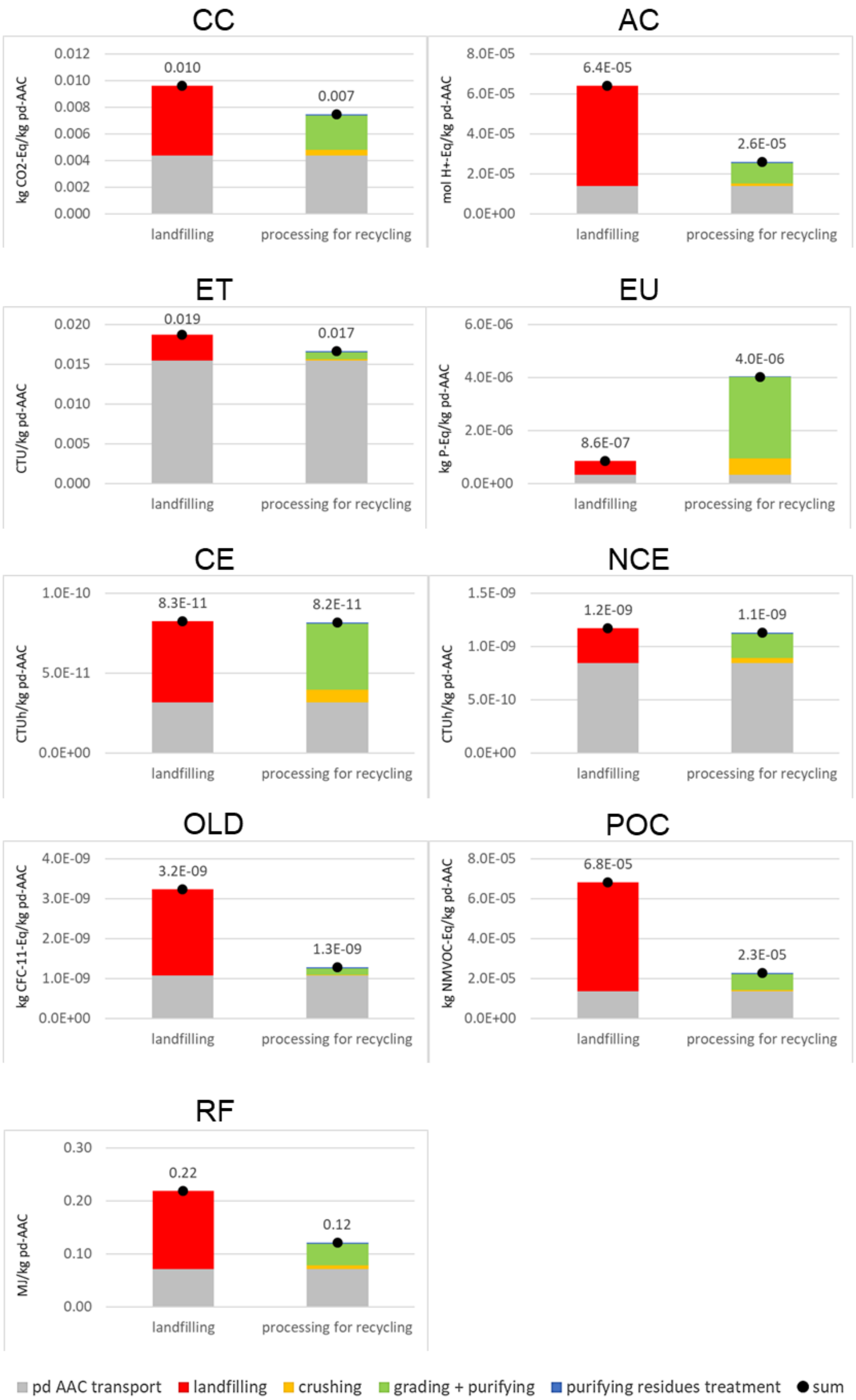

Figure C.3: Impact assessment of landfilling (left vertical-bar in all sub-diagrams) and basic processing for recycling (right vertical-bar in all sub-diagrams) of 1 kg pd-AAC (CC: climate change total, AC: freshwater and terrestrial acidification, ET: freshwater ecotoxicity, EU: freshwater eutrophication, CE: carcinogenic effects, NCE: non-carcinogenic effects, OLD: ozone layer depletion, POC: photochemical ozone creation, RF: resources – fossils).

Transport distances were assumed to be 50 km for both end-of-life options, so these efforts (grey) are identical comparing landfilling and processing for recycling. Further landfilling efforts (red) include the construction of the landfill and energy demand for waste handling and landfill management, especially diesel used in landfill machinery. These landfilling efforts are a bit more impacting than the transport for most midpoints. In contrast, processing for recycling efforts consists of crushing (yellow), grading and purifying (green) and purifying residues treatment (blue). The purifying residues treatment only marginally contributes to the overall impact as only 0.01 kg residues per kg pd-AAC are assumed to be sorted out during the purifying process. Electricity demand is critical for crushing, grading, and purifying efforts. However, grading and purifying contribute significantly more to the overall effort than the crushing, as the total energy demand is around five times higher. Strikingly, overall landfilling impacts exceed the overall processing for recycling impacts for most midpoints. Only concerning freshwater eutrophication, landfilling is environmentally preferable to pd-AAC basic processing. Therefore, the recycling options outperform landfilling if the substitution credits are higher than additional recycling efforts. As a second step, overall results, including substitution credits, were calculated (Figure C.4).

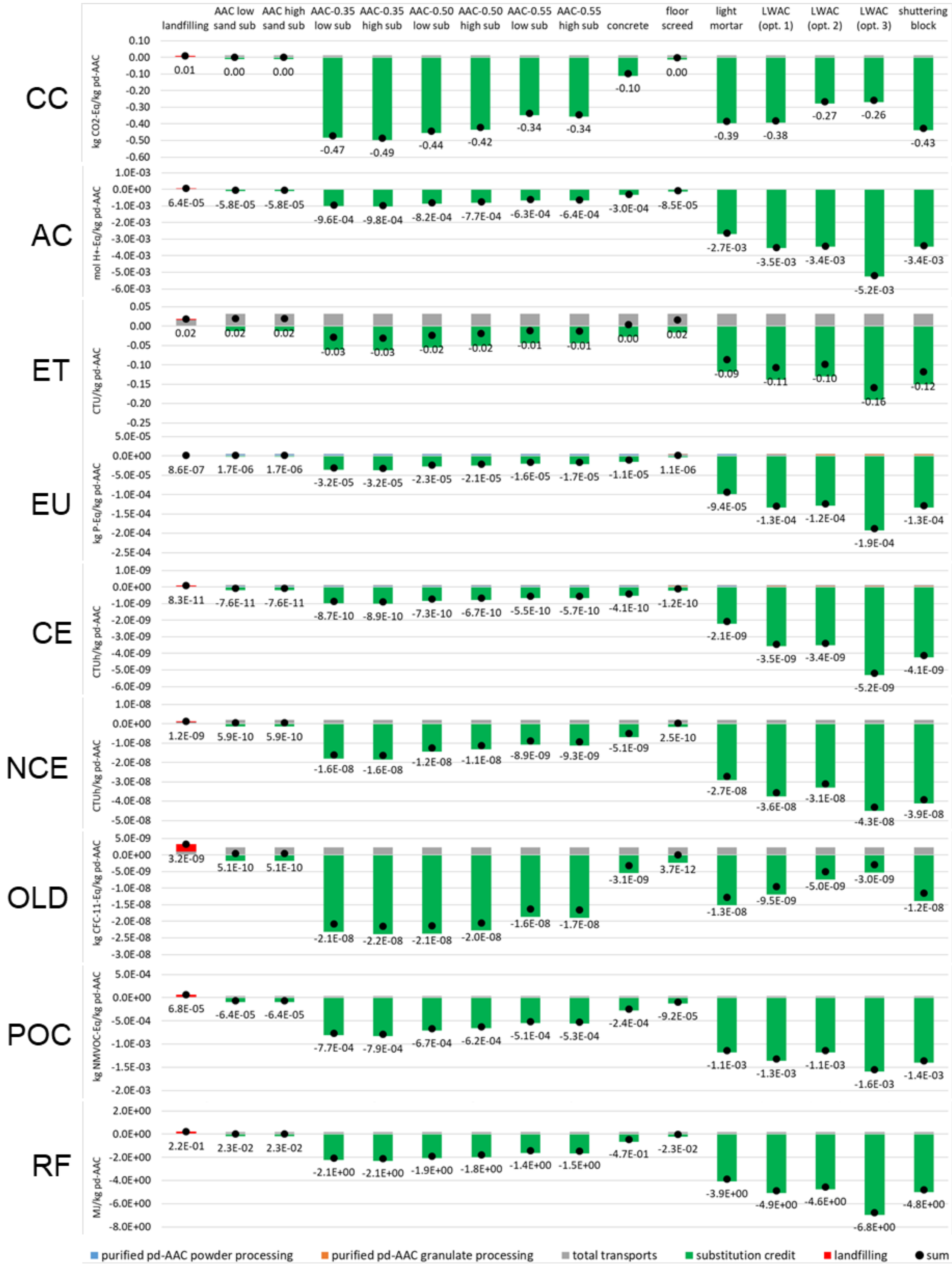

Figure C.4: Impact assessment of landfilling and various recycling options for 1 kg pd-AAC (CC: climate change total, AC: freshwater and terrestrial acidification, ET: freshwater ecotoxicity, EU: freshwater eutrophication, CE: carcinogenic effects, NCE: non-carcinogenic effects, OLD: ozone layer depletion, POC: photochemical ozone creation, RF: resources – fossils).

The recycling options of AAC production (sand substitution) and floor screed production have the most negligible substitution credits because only primary sand, which is not associated with high ecological efforts, is substituted. Therefore, these options hardly outperform landfilling in most midpoint categories. But, all other recycling options included in this study show a significant reduction in ecological impacts compared to landfilling. The pd-AAC recycling in concrete production offers higher substitution credits than the recycling options mentioned before. Still, overall savings are lower than in the AAC, light mortar, LWAC, and shuttering block production. The closed-loop recycling of pd-AAC in the AAC production substituting sand, cement, quicklime, and anhydrite shows high substitution credits and the best overall savings concerning CO_2-Eq emissions. There is hardly any difference between the low and high substitution scenario because the functional unit is 1 kg pd-AAC. However, in the high substitution case, more pd-AAC could be used as input to substitute more primary resources. Thus, the environmental efforts per kg AAC are lower than in the low substitution case (Section C.4). The substitution credit and the overall savings decrease for higher AAC density classes in all midpoints. Higher AAC density is associated with higher sand and a lower cement content (Table C.1), leading to higher sand and lower cement substitution. Thus, substitution credits decrease as cement is associated with significantly higher ecological burdens than sand. However, savings remain significant and higher than in AAC sand substitution and concrete recycling options, even for the high-density AAC-0.55 production. In many midpoint categories except climate change, light mortar, LWAC, and shuttering block production show higher substitution credits and savings than AAC production. Mainly, expanded clay is substituted by pd-AAC in all three recycling options. This substitution leads to notable credits as expanded clay is associated with relatively high impacts, for example, 0.44 kg CO_2-Eq/kg. The light mortar production savings are slightly lower than the LWAC and shuttering block production savings for most midpoints. Regarding the different production recipes for LWAC mentioned in the literature, recycling option 2 shows the lowest substitution credit and savings. Option 1 favours the climate change midpoint, as this production recipe has the lowest cement content. Option 3 performs the best for most other midpoints despite a recipe with a higher cement content since expanded clay is substituted by less pd-AAC. Generally, LWAC production (option 3) shows the highest savings of ecological efforts for most midpoints among all recycling options. The shuttering block production outperforms the LWAC production concerning the midpoint climate change but is slightly behind LWAC production for most other midpoints.

C.3.2 Interpretation

Recycling pd-AAC in the AAC production of different density classes is an excellent option, particularly if cement, quicklime, and anhydrite are substituted. Especially CO_2-Eq emissions could be significantly reduced to 0.49 kg CO_2-Eq/kg pd-AAC compared to landfilling. Further beneficial recycling options include the production of light mortar, LWAC, and shuttering blocks – the latter made from no-fines concrete. These options could also reduce CO_2-Eq emissions and reach savings per kg pd-AAC compared to landfilling of up to 0.43 kg CO_2-Eq, 7 MJ fossil resources, 0.005 mol H+-Eq (acidification), 0.17 CTU (freshwater ecotoxicity), 0.2 g P-Eq (freshwater eutrophication), 5.2×10^{-9} CTUh (carcinogenic effects), 4.4×10^{-8} CTUh (non-carcinogenic effects), 2.5×10^{-5} g CFC-11-Eq (ozone layer depletion), and 1.6 g NMVOC-Eq (photochemical ozone creation). Overall, there are several recycling options for pd-AAC, which can reduce ecological impacts significantly compared to landfilling.

Finally, total potential savings can be estimated using the example of GHG emissions in Germany (Figure C.5). For this estimation, the available pd-AAC was assumed to be used in the described recycling options in the literature in descending order of their GHG efficiency. First, as much as possible pd-AAC was considered to be used for AAC production because its GWP substitution credits are the highest. In Germany, around 0.7 million t of pd-AAC was expected to be generated in 2022 (Steins et al., 2021). This could be recycled in the production of AAC-0.35, AAC-0.50, and AAC-0.55, where it substitutes sand, cement, quicklime, and anhydrite. High pd-AAC substitution percentages were assumed to be 7 %, 7 %, and 10 % for the respective AAC products (Section C.2.3) at shares of 45 % for AAC-0.35, 20 % for AAC-0.50 and 10 %[6] for AAC-0.55 of the overall AAC production of 3.5 million m³. Under these assumptions, around 80,600 t of pd-AAC (11.5 % of the total pd-AAC amount) could be used for the production of AAC in Germany today. However, due to the limited substitution in AAC production, recycling options of light mortar production, LWAC production, and shuttering block production should also be considered for the remaining pd-AAC material.

In Germany in 2022[7], 2.7 million t of masonry mortar and interior plaster are expected to be produced, which are the main application options for light mortar (Aycil et al., 2016). And 900,000 t LWAC is expected to be produced (GENESIS, 2023). Shuttering block production is not disclosed in the official production statistics, so it is assumed that they account for 50% of the category “other concrete building blocks and bricks“ with a total annual production of 390,000 m³ (GENESIS, 2023). Thus, we assumed a shuttering block production of 135,000 t with a density of 0.7 t/m³ (Gyurkó et al., 2019). According to literature, the possible input shares of pd-AAC are much higher for these recycling options than for recycling in AAC, summing up to 61 % (light mortar), 81 % (LWAC option 1), and 36 % (shuttering block) (Aycil et al., 2016; Gyurkó et al., 2019). However, substitution rates might vary between producers because of varying recipes and different product qualities and requirements. After supplying the closed-loop AAC recycling options, the remaining pd-AAC was assumed to be first used for shuttering block production (up to 48,000 t of pd-AAC) due to higher GWP substitution credits than light mortar and LWAC production. The remaining pd-AAC could be equally used for light mortar and LWAC recycling options.

Potential GHG emissions savings were calculated using the difference between landfilling and respective recycling options per assigned pd-AAC mass flow (Figure C.5a). Under the given

[6] The remaining 25% market share is distributed among different AAC products including reinforced AAC wall- and roof-elements.

[7] Production statistics for 2022 are not available yet. Therefore, the production statistics for 2020 and 2021 are used for prediction. The produced amounts did not significantly change over the last two years for all relevant materials. Thus, we assume a constant production amount for 2022.

assumptions, more than 280,000 t CO_2-Eq could be saved via pd-AAC recycling in Germany in 2022 (Figure C.5b). Besides, a landfill capacity of 1.386 million m³ respectively 693,000 t could have been saved in Germany in 2020 if 1% of purifying residues still were landfilled.

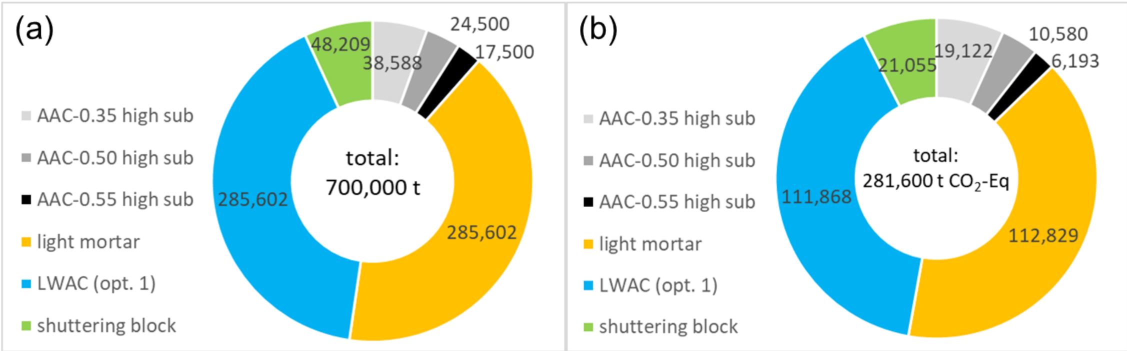

Figure C.5: Recycling strategy for pd-AAC in Germany minimising GHG emission, (a) allocation of the total pd-AAC waste in 2022 [t], (b) allocation of the savings in GHG emissions per chosen recycling option [t CO_2-Eq].

C.3.3 Sensitivity analysis through Monte Carlo simulation

A Monte Carlo Simulation was performed to investigate the sensitivity of the LCA results. This simulation was based on the ecoinvent data quality system that assesses the reliability, completeness, temporal correlation, geographical correlation, and further technological correlation of the data to determine an uncertainty function. Uncertainty values were taken from the ecoinvent 3.6 database as far as possible. The uncertainty of the remaining inputs that were only described in the literature (Section C.2) was determined based on information on data origin (measurements or estimates), the extent of the production sites under consideration, actuality of the study, area of the study, and technological comparability (laboratory data or field data). Finally, a lognormal distribution with standard deviation calculated from the uncertainty values was used for the simulation. All primary production, basic recycling, and final recycling processes were included in the Monte Carlo Simulation, with 10,000 runs for each process. All results of the Monte Carlo Simulation, including mean, median, standard deviation, minimum, maximum, and 25%/75% percentile, are given in the SI-2.

The Monte Carlo simulation results show a median value of all runs near the initially calculated value (Figure C.4) for all processes and all midpoints as expected. Absolute deviations between the originally calculated value and median of the Monte Carlo simulation for the climate change total midpoint are the highest for the AAC production (low sand substitution) and shuttering block production with 0.011 kg CO_2-Eq/kg pd-AAC. The other recycling options usually have deviations around 0.005 kg CO_2-Eq/kg pd-AAC or even less, around 1% of the total savings of pd-AAC recycling in AAC-0.35 production. The most exciting deviation over all midpoints and recycling options may be found at the freshwater ecotoxicity of AAC-0.35 (low substitution), where the simulation median of -0.04 CTU/kg pd-AAC differs around a third from the original result of -0.03 CTU/kg pd-AAC. However, interpretation does not change as other recycling options, including light mortar, shuttering block, and LWAC production, still outperform the AAC production in this midpoint with total savings of up to -0.16 CTU/kg pd-AAC. These findings are also valid for the impacts of landfilling and primary production processes used for substitution credit calculation. The Monte Carlo simulation shows that the overall impacts calculated for all recycling options are reasonable.

The variability of the LCA results is investigated through the 25% and the 75% percentile of the simulation results. However, the variability of the final impacts of the different recycling options is hard to calculate as it is influenced by the variability of the production process and the substitution credit. Therefore, both aspects are considered separately. Figure C.6 and the following interpretation investigate variabilities for the climate change total midpoint and only focus on impacts of the production processes without substitution credits.

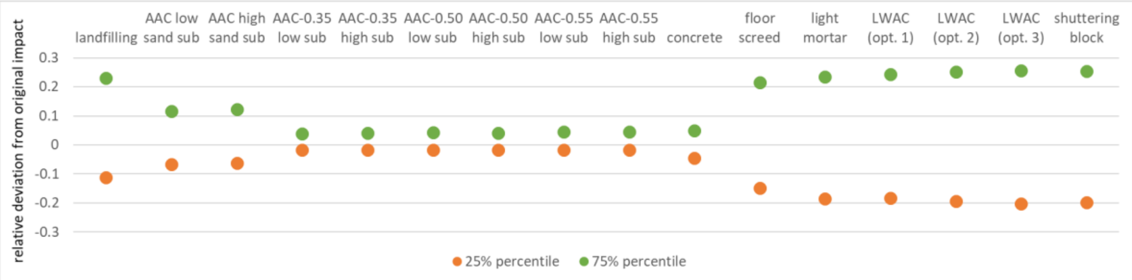

Figure C.6: Variability of the LCA results in a Monte Carlo simulation (25% and 75% percentile) for all recycling options for the midpoint climate change total.

Different variabilities can be observed for the recycling processes considering the 25% and 75% percentile of the Monte Carlo simulation. AAC and concrete production only show very low deviations of up to +/-10% in the AAC production (sand substitution) case. Considerably higher variations of up to -20% and +25% can be observed in the floor screed, light mortar, LWAC, and shuttering block production. The landfilling shows moderate deviations of +23% and -11%. Overall, the interpretation of the results does not change. The AAC-0.35, the preferred recycling option concerning climate change, only shows little variabilities. Other recycling options, including light mortar, LWAC, and shuttering block production, could reach the same level as AAC-0.35 since the impacts could be as much as 20% lower. Either way, the recycling options perform much better than landfilling.

Variabilities of the substitution credit are generally very similar to those of the production process shown above since the processes are the same except for the pd-AAC content. However, variabilities of the substitution credits for light mortar, LWAC, and shuttering block are up to 20 percentage points lower than that of the production processes. Pd-AAC is used in much higher quantities in these three processes, so they differ more significantly from their primary production process. Further information and all data on variabilities for the other midpoints and the substitution credit can be found in the SI-2.

C.4 Discussion

Different end-of-life options of pd-AAC were assessed and compared. In most options, pd-AAC was used in the production of new products, and substitution credits for avoided primary production were granted, assuming that the same quality for all products was achieved. However, the quality of recycling products is often lower due to impurities in recycling materials. Such quality reductions of recycling products could also occur in pd-AAC recycling since wallpaper, plaster, dowels, screws, and ceramics are likely to adhere to the pd-AAC and might reduce substitution rates and credits. However, a wide range of normative and manufacturer-specific requirements on building materials exist. We determined that the AAC produced with pd-AAC fulfils the same building material standards as primary products by laboratory test production (Section C.2.3). Concerning all other recycling options from the literature, we only considered those where secondary products of high quality can be produced. Thus, if the recycling products fulfil the same building material standards as primary products concerning relevant physical and chemical parameters, the same quality assumption and granting full substitution credit is justified.

Literature shows that all investigated recycling options are suitable for replacing primary products. However, some recycling options could be preferred due to technological aspects not being included in this assessment. For example, pd-AAC as aggregate in concrete production performs better than landfilling but worse than some other recycling options concerning the LCA of this study. However, from a technological point of view, pd-AAC improves the strength and durability of the concrete (Gyurkó et al., 2019), which could be a pivotal argument for preferring this recycling option over the others.

Furthermore, two different substitution levels were considered in the closed-loop recycling for three different AAC density classes. The different substitution levels did not show significant differences for any midpoint indicator since the functional unit is 1 kg pd-AAC. However, pd-AAC volumes could soon reach or exceed primary AAC production (Steins et al., 2021). Substitution is currently bound to a few per cent of the AAC production volume, so only a limited share of pd-AAC could be used in closed-loop recycling. Additional savings in environmental efforts between low and high substitution cases reveal when using 1 kg final product as functional unit. Then, primary production of AAC-0.35 is associated with GHG emissions of 0.526 kg CO_2-Eq per kg of AAC, which decreases to 0.516 (-1.9%) in the low (2%) and 0.491 (-6.7%) in the high (7%) substitution scenario. This relationship between primary production, low substitution, and high substitution cases remains the same for the other AAC density classes. Therefore, substitution rates in closed-loop recycling should be maximised to minimise the environmental efforts of the final products. However, pd-AAC shares in AAC production are still quite low and further research and development is required to enhance the shares.

Primary AAC powder emerging from the processing of AAC production leftovers, cutting residues and leftovers returned from job sites is already input for AAC production (Table C.1). This primary powder could be replaced by pd-AAC powder to reach pd-AAC powder input shares of up to 21% (Section C.2.3). Primary AAC powder is generally cleaner than pd-AAC material and could be used in other recycling options demanding exceptionally high quality. Currently used high-quality applications include, e.g. odour-/ammonia-binders in livestock breeding, fertilisers or soil conditioners. In addition, also calcium silicate units (another masonry product) are produced using AAC powder, albeit mostly in small quantities.

So far, no studies have compared the different end-of-life options of AAC yet. Therefore, the results of this study cannot be directly contextualised with the literature. The input data used for the LCA is taken from different studies. These studies focus on technological aspects and the feasibility of recycling. Currently, the investigations are on a laboratory scale, respectively, on a large-size pilot plant scale in the case of closed-loop recycling (Section C.2.3). All proposed production recipes still have to be validated in large-scale production plants. Thus, the input data used for the comparison could still change when the recycling options are implemented more in practice.

C.5 Limitations

The results and interpretation of this study are based solely on the assessment of ecological factors given the technological descriptions in literature and our own experiments. Economic or social aspects were not considered in this study but could significantly influence the decision of selected recycling options in practice (Section C.6).

Additionally, there is no field data for the investigated recycling processes since the performed experiments and data are primarily performed and available on a laboratory scale. Therefore, literature data and the ecoinvent 3.6 dataset were chosen that fit the described processes the best. This approach will likely reflect actual pd-AAC crushing and many primary production processes. However, there is no directly fitting ecoinvent 3.6 or literature dataset for pd-AAC grading and powder/granulate purifying. A dataset for waste brick sorting ("treatment of waste brick, sorting plant – Europe without Switzerland") was chosen for both processes instead, including efforts for a comprehensive treatment process. Hence, pd-AAC grading and purifying effort could have been overestimated.

Moreover, it was assumed that 1 % of residue (impurities) is sorted out based on an expert interview. This percentage heavily depends on actual pd-AAC purity and could reach higher values that would decrease process yield and increase ecological burdens of the respective end-of-life option. Additionally, the purifying efforts depend on the desired quality of pd-AAC powder or granulate and their further usage. Application in AAC production is likely to require a high pd-AAC powder quality, whereas applications such as light mortar or floor screed might be practicable with lower grades. Thus, the chosen recycling process determines the effort of the preceding purifying process. This connection was not explicitly considered in the performed LCAs as profound information on the required quality is not available yet. However, the contribution of purifying to the overall result is low.

The potential national savings by pd-AAC recycling are based on literature values and a rough estimation. Thus, this potential might be further limited by technical or logistical restrictions (e.g. small amounts), material, recyclate and product qualities and specific requirements of LWAC, light mortar and shuttering blocks and market sizes/share of products of different grades. Further research is required to reduce uncertainties in this estimation.

C.6 Conclusion

Life cycle assessment with zero burden approach and avoided burden system extension were performed to assess the environmental impacts of recycling pd-AAC compared to landfilling. Recycling options considered in this study include the production of AAC (with substitution of sand only or of all primary inputs), concrete, floor screed, light mortar, LWAC, and shuttering blocks made from no-fines concrete. Results show that recycling pd-AAC is advantageous over landfilling in all cases for all environmental criteria analysed since processing pd-AAC is not associated with high impacts and rewards for substituted primary material are significant. Especially the closed-loop recycling of pd-AAC in AAC production can considerably reduce environmental impacts, for example, GHG emissions. Light mortar, LWAC, and shuttering block production are the best open-loop recycling options. These options perform best for the midpoints acidification, eutrophication, ecotoxicity, ozone layer depletion, and resource consumption. Additionally, further open-loop options are needed to cope with the increasing amount of pd-AAC that can be expected in the future.

This study shows that pd-AAC recycling should be fostered because potential annual savings by pd-AAC recycling could sum up to 280,000 t CO_2-Eq and 1.386 million m³, respectively, 693,000 t of saved landfill capacity in Germany. The legal framework for the processing and recycling mineral demolition waste should support recycling strategies by reducing regional differences in the legislation and in landfilling prerequisites and cost. Besides, political commitment to secondary building materials with recycling content would increase the acceptance and substantially help to enhance recycling. Public construction projects could, for example, contain fixed rates for secondary building materials.

Future research should focus on improved LCA data of said processes, e.g. from pilot plants instead of laboratory data. Furthermore, an economic assessment of the investigated end-of-life options is mandatory to analyse economic viability, transport and handling, significant impacts, influencing factors, and advantageous framework conditions. Right now, landfilling of pd-AAC becomes more expensive as landfill capacities, especially in Germany, decrease. The regional prices exceed 100 €/t in many districts and can reach up to 200 €/t, but differ from district to district. The recycling options presented in this study will likely remain below these costs if transport distances between the demolition place, the recycling plant, and the final production plant can be kept short. Around 30 €/t can be expected for a 100 km transport of pd-AAC using transport costs given by Wolfermann (2016) (adjusted to 2022). Therefore, adding up the costs of two 100 km transports (demolition site to recycling plant and recycling plant to production plant) and the pd-AAC processing could stay below 100 €/t (= landfilling costs) as the processing uses standard processes and only has moderate electricity consumption. However, this rough estimation does not consider revenues for substituted primary material yet.

Further research and regulation should aim for higher substitution ratios, especially in AAC production. Higher substitution rates can reduce the overall environmental impacts and handle increasing pd-AAC amounts in the future.

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D Recycling Belite Cement Clinker from Post-Demolition Autoclaved Aerated Concrete – Assessing a New Process

Abstract[1]

Autoclaved aerated concrete (AAC) is a popular construction material for residential buildings, however varying between regions. Demolition generates an increasing post-demolition AAC waste stream that is currently mainly landfilled due to its physical properties and lacking recycling processes. In Germany, post-demolition AAC is expected to quadruple until 2050. A promising technology - especially for low-quality wastes is the production of recycled belite cement clinker, which can partially substitute Portland cement clinker. This paper presents experimental data of recycled belite cement clinker production from post-demolition AAC that has been successfully demonstrated on technology readiness level 4-5 and its associated life cycle assessment. Different supply chains for post-demolition AAC and energy are examined. The closed-loop post-demolition AAC recycling via the belite route that aims for Portland cement clinker substitution shows promising results and significant potential savings in environmental impacts. It has lower net environmental impacts than landfilling in all firing scenarios. The savings could reach 0.77 kg CO_2-Eq/kg post-demolition AAC compared to the status quo (landfilling) by using renewable electricity, and 0.34 kg CO_2-Eq/kg post-demolition AAC by using natural gas. The gained reduction of around 13.5% is significant considering that it is the result of substituting only 15.5% of the overall input material. The results indicate that fostering closed-loop recycling for post-demolition AAC could significantly reduce environmental burdens associated with the current landfilling.

D.1 Introduction

Autoclaved aerated concrete (AAC) is a popular mineral lightweight construction material (Steins et al., 2022). AAC is highly porous, with a total pore volume of 65 to 90% (Schober, 2011). The bridges between mm-sized air pores consist of the calcium silicate hydrate tober-

[1] This section includes the article "Recycling belite cement clinker from post-demolition autoclaved aerated concrete – assessing a new process", developed by Peter Stemmermann, Rebekka Volk, Günter Beuchle, and myself. The article has been submitted for publication in a scientific journal as Stemmermann et al. (2023). The supplementary will be found on the journal website after publication.

morite and smaller quantities of other minerals, especially ones containing sulfate and unreacted raw materials, especially quartz. The raw materials include ground quartz, ordinary Portland cement (OPC), quicklime, gypsum, and aluminum powder. After mixing with warm water the resulting slurry is poured in forms. Slow hardening generates a cake that rises by released hydrogen as propellant gas. During the reaction, the volume of the mixture increases 2 to 3-fold, leading to a low density. Finally, the cake is hardened in an autoclave under hydrothermal conditions. Thus, AAC has excellent thermal insulation properties and can be easily assembled on the building site. Typically, in Europe, AAC is produced in plants with a production capacity of between 110,000 and 275,000 m^3/a (Harder, 2009; UBA, 2019). Considerably used since around the 1950s, a rising amount of post-demolition AAC (pd-AAC) is expected to return from the building stock soon (Steins et al., 2021, 2022). Pd-AAC is 'generated' locally as a separate or mixed demolition fraction. In Germany it is jointly collected with other gypsum-containing demolition waste and is usually landfilled (Kreft, 2016; UBA, 2019).

Recent studies address new open-loop recycling options for pd-AAC, e.g. in lightweight aggregate concrete production (Aycil et al.; Gyurkó et al., 2019), light mortar production (Aycil et al.), floor screed production (Bergmans et al., 2016), filler or supplementary material in concrete (Gyurkó et al., 2019), cement clinker production (Schoon et al., 2013), and concrete (deprived of the fine fraction) for specific applications and shuttering blocks (Gyurkó et al., 2019). Volk et al. (2022) and Volk et al. (2023) compare the recycling options in a life cycle assessment (LCA). However, these options are not closed-loop and might be limited with growing mass flows.

Moreover, pd-AAC qualities differ strongly from pure to heavily mixed with wood, glass, metal, screw anchors, gypsum/plastering, wallpaper, colour, or other coatings. Average compositions of variously collected pd-AAC are shown in Table A-1 (in supporting material A). Additionally, pd-AAC has a very heterogeneous fragment size. Therefore, prior to any recycling, preprocessing by various crushing and sorting steps is necessary. Most recycling options focus on granules above 2 mm grain size. However, this material class is a minor fraction of the preprocessing with approximately ¼ share of the input mass. The main fraction is fines (ca. ¾)[2], which is hardly recycled today. Besides, the sulfate content increases significantly in the fines fraction.

Due to the use of raw materials and the reaction with water, AAC is similar in composition to a hardened Portland cement paste but deficient in CaO. Therefore, it is appealing to use pd-AAC fractions in a chemical recycling process to produce Portland cement if there is no better physical option for recycling. However, to use pd-AAC as a raw meal substitute, its calcium oxide content must be increased by adding limestone ($CaCO_3$). The subsequent clinker burning includes calcination of limestone as an essential step, which requires much energy and releases CO_2.

[2] Expert interview within research project REPOST with Xella Technologie- und Forschungsgesellschaft mbH. However, the shares can vary due to water content, the used mill Krampitz et al. (2022) and other factors.

To reduce the specific CO_2 emissions and to broaden the range of suitable raw materials for clinker production, high-belite ($2CaO.SiO_2$ or C_2S) cement can be produced (Kotsay & Jaskulski, 2019). However, production is only economically viable at temperatures above 1200 °C in high-capacity plants. Therefore, a new type of belite cement clinker, recycled belite cement clinker (RC-BCC), has recently been developed at KIT. Its chemistry complies with the European cement standard (DIN EN 197-1:2011-11) for Portland cement clinker (PCC), with a belite content of up to 80 wt%. Special conditions in the kiln allow clinkering at a much lower temperature (RC-BCC 1000 °C vs. PCC 1450 °C) (Beuchle et al., 2013; Ullrich et al., 2021), which lowers energy consumption and enables electric heating with available technologies instead of combustion. Environmentally harmful and cement-damaging impurities are fixed by incorporation into non-soluble minerals, which allows for low-quality secondary raw materials. The slower hydration kinetics of RC-BCC (Chatterjee, 1996; Chen et al., 2017) is compensated by formulating mixed types of cement from alite-rich ($3CaO.SiO_2$ or C_3S in cement notation) PCC, RC-BCC, or other main constituents according to European standards.

The RC-BCC technology reduces the specific consumption of limestone by about 10% and substitutes limestone with CO_2-free secondary materials such as pd-AAC. Both features reduce the specific CO_2 emission and energy consumption of RC-BCC relative to PCC. In addition, natural resources are saved, and waste is reduced. Furthermore, the lower process temperature simplifies the kiln design. Finally, the technology enables easy separation of concentrated CO_2 if process heat comes from oxyfuel combustion or electricity. Both the use and storage of highly concentrated CO_2 are economically favourable.

Therefore, this study aims to describe the technical process of RC-BCC production, including chemical reactions, material balances, energy consumption, and CO_2 emissions (Section D.2). Moreover, a full LCA is conducted to assess the environmental potential of RC-BCC production from pd-AAC and the results are compared to status quo (landfilling) (Section D.3). Finally, the results are discussed (Section D.4) and concluded (Section D.5).

D.2 Recycled belite cement-clinker (RC-BCC): Process description and assessment basics

D.2.1 General process design

Processing of RC-BCC (Figure D.1) starts with the pre-treatment of the raw materials. The main components are crushed, graded, purified, and, if necessary, pre-dried. Each component is ground until 50% of the particles are smaller than 20 microns (d_{50} = 20 µm) with d_{95} < 100 µm. The grinding time depends on the components and is usually 5 to 15 minutes (Cement equipment organisation, 2018). The resulting grain size distribution is checked by static light scattering. The powdery products are mixed into a raw meal with a CaO/SiO_2 molar ratio of 2. The mixture is stored in a silo and analyzed. Depending on its composition, corrective limestone as well as mineralizer is added. The mixture reacts to RC-BCC in a rotary kiln at 1000 °C in a slightly oxidizing CO_2 atmosphere and is cooled in a clinker cooler. Finally, the clinker is ground in a ball mill to achieve a typical cement fineness.

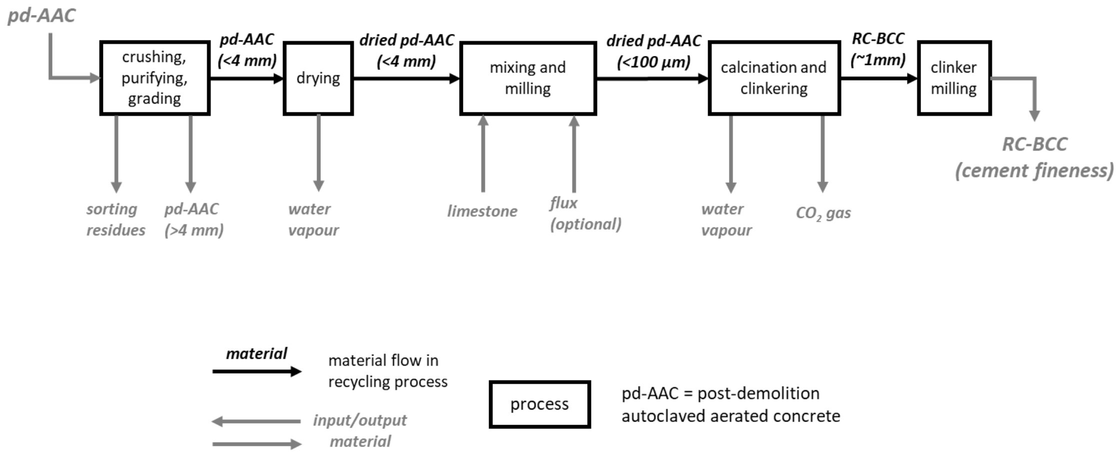

Figure D.1: RC-BCC production flow sheet.

D.2.2 Firing conditions

Three heating technologies have been considered (Table D.1). The simplest variant uses the combustion of natural gas in air, which results in large amounts of off-gas with low CO_2 concentration. The second variant, the oxyfuel technology combusts natural gas in an oxygen(-rich) atmosphere. In the third variant, the rotary kiln is electrically heated. However, heating technologies differ in their level of technological readiness.

Table D.1: Overview of assessed process variants.

	Gas	Oxyfuel	Electric
Firing conditions	Combustion of natural gas	Combustion of natural gas	Electricity
Atmosphere in the rotary kiln	Air	CO_2 + 5% Oxygen	CO_2 + 5% Oxygen
Technology readiness level (TRL)	Established standard (TRL = 9)	Not established for this temperature range (TRL = 5)	Not established for larger plant sizes (TRL = 4)
Addition of oxygen	Yes	No	Yes
Clinker cooling	Established standard (TRL = 9)	Yes, via heat exchange of CO_2 (TRL 7)	Yes, via heat exchange of CO_2 (TRL 5)

D.2.3 Combustion and process gases

Depending on the firing technology, the off-gas stream's composition, quantity, and energy content vary. The CO_2 concentration in the reactor strongly influences the reaction kinetics and, thus, the plant output. The use of secondary raw materials may induce impurities such as CO, SO_2/SO_3, HCl, or trace elements, which affect the quality of CO_2 for later use or storage. Incomplete oxidation is prevented for safety reasons by a slight oversupply of oxygen[3]. The process heat is used as far as possible to preheat the raw meal. In atmospheric combustion, the combustion air is preheated in the clinker cooler. In oxyfuel or electric heating, clinker cooling is done by heat-exchanged concentrated CO_2 of more than 90% purity. The concentrated CO_2 can be used, e.g. in on-site carbonation of waste concrete (Vanderzee & Zeman, 2018), or stored.

D.2.4 RC-BCC from pd-AAC and limestone: Mixing ratio of the starting materials

Three pd-AAC samples (SM(D1), SM(D2), and SM(D3)) and one sample of pure production waste (SM(P)) were provided by a mineral waste processing company and an AAC producer. Details are described in Ullrich et al. (2021). X-ray fluorescence analysis (XRF) is used to measure the chemical composition of the samples, which contain typically 43 to 56 wt% of SiO_2 and 25 to 31 wt% of CaO. Oxide contents are given in Table A-1 in supporting information A.

Bulk quantitative phase analyses of all samples were performed mixed with an internal standard (20 wt% α-Al_2O_3, Alfa Aesar 99.95%) using the Rietveld method following the fundamental parameters approach implemented in TOPAS V6 (Bruker-AXS) to determine crystalline and X-ray amorphous contents. A detailed description of the refinement strategy is given in Ullrich et al. (2022).

RC-BCC samples were synthesized from the four pd-AAC samples with respective limestone additions in a laboratory rotary kiln (Ullrich et al., 2021). Pure $CaCO_3$ was used in the experiments.

Depending on each sample's CaO and SiO_2 content, the addition of $CaCO_3$ is calculated to achieve a raw meal with a molar ratio of CaO to SiO_2 of two – or in cement notation (C=CaO; S=SiO_2): C/S=2 see Table A-2 (supporting information A). Finally, a small addition of a mineralizer, typically 1 to 2 wt% of either Na_2CO_3 or $CaCl_2$ (Garbev et al., 2022; Ullrich et al., 2022), is added to deal with contaminations and increase the kinetics.

Energy and mass balances were calculated based on thermodynamic equilibrium calculations with the database system Factsage 8.2 (Bale et al., 2016). However, the available data sets for the respective systems do not allow for incorporating minor and trace elements in solid solu-

[3] In scenario 1: oversupply of air.

tions. Thus, for stable results, the composition of pd-AAC was approximated with a data set for a simplified calculated average pd-AAC composition (SC-pd-AAC). It was derived from mean values for phase contents according to the ranges given in the safety sheet “YTONG autoclaved aerated concrete” (Xella, 2017) and considering an additional 3% moisture in pd-AAC (Table A-1, A).

D.2.5 Chemical reactions and mineralogy of RC-BCC processing

The clinker raw meal is processed in a rotary kiln. During heating up to about 700 °C, it gradually expels the chemically bound water from mineral phases. After the loss of approx. 3% free moisture, these are in particular ~2 wt% crystal water from gypsum (150 °C) and 3.5 wt% from tobermorite (main weight loss up to 200 °C, then gradually). The total weight loss sums up to approx. 9 wt%.

At about 600 °C, calcium silicates from dehydrated pd-AAC and limestone begin to form and release CO_2. The availability of SiO_2 and CaO determines the reaction rate which can be enormously accelerated by adding mineralizer, e.g. $CaCl_2$ or Na_2CO_3. For combustion in air, the temperature of limestone decomposition is reached at approximately 650-800 °C, depending on the limestone’s crystallinity. Large amounts of free CaO are formed, slowly reacting with the other components and forming the desired clinker phases.

The decomposition temperature of limestone increases by about 100 °C, accompanied by the immediate formation of calcium silicates, if clinkering of RC-BCC is performed in a concentrated CO_2 atmosphere (electric heating, oxyfuel). Clinkering is finished at about 1000 °C. The clinker grains consist predominantly of belite, an x-ray amorphous mixture of highly disordered nanocrystals, and a vitreous fraction. The x-ray amorphous mixture’s chemical composition corresponds to the one of the bulk samples. Sulfate is fixed in a sulfate-rich calcium silicate or sodium sulfate, depending on the used mineralizer. Without the addition of mineralizer, the sulfate-rich calcium silicate ternesite crystallizes. If $CaCl_2$ is used as mineralizer, chloro-ellestadite will form instead. Besides chlorine and sulfate, ellestadite can incorporate other potentially environmentally harmful or cement-damaging impurities in its crystal structure. It is insoluble under normal cementing conditions. When Na_2CO_3 is used as a mineralizer, Na_2SO_4 is formed. Na_2SO_4 is highly soluble in water and can be easily washed out. Table D.2 compares the measured phase contents of RC-BCC with contents calculated with Factsage 8.2 (Bale et al., 2016) for the simplified calculated average pd-AAC composition SC-pd-AAC / 2 wt% $CaCl_2$. Since some of the phases present in the experimental data, e.g. ellestadite, do not appear in the published databases, the data set was expanded based on own measurements.

Table D.2: Measured content of main clinker minerals in RC-BCC prepared from the indicated pd-AAC, limestone, with and without flux. For comparison, the calculated phase content of the standard clinker SC-pd-AAC is given. Data in wt%. XRA: x-ray amorphous; T: Ternesite; Q: Quartz; C: Calcite; L: Lime (CaO); A: Anhydrite (Ca_2SO_4); W: Wollastonite ($CaSiO_3$); E = Ellestadite; NS: Na_2SO_4. Difference from 100 % in measured samples: minor phases (not shown).

pd-AAC / mineralizer	XRA	α'H-C_2S	β-C_2S	T	Q	C	L	A	W	E	NS	Sum
SM(P) / -	-	0	90.0 (6)	0.44 (13)	2.52 (7)7)	0.63 (11)	-	-	0.14 (5)	-	-	93.73 (99)
SM(D1) / -	10.2 (1.7)	4.2 (3)	47.2 (9)	1.17 (11)	8.76 (16)	7.02 (15)	12.07 (17)	-	4.32 (12)	-	-	94.94 (2.52)
SM(D2) / -	4.8 (1.3)	1.08 (1.7)	62.6 (6)	11.36 (17)	5.62 (12)	1.49 (11)	2.52 (5)	4.50 (9)	0.64 (10)	-	-	94.61 (3.70)
SM(D3) / -	-	1.35 (18)	65.4 (6)	8.31 (16)	5.57 (13)	1.23 (12)	1.84 (5)	7.46 (11)	0.99 (11)	-	-	92.15 (92)
SM(P) / 2% $CaCl_2$	15.0 (7)	-	63.8 (4)	0.12 (7)	2.11 (4)	-	0.17 (5)	0.43 (6)	0.6 (1)	9.9 (1)	-	92.13 (1.16)
SC-pd-AAC / 2% $CaCl_2$	-	-	85.2	-	-	-	1	-	-	13.8	-	100.0
SM(P) / 5% Na_2CO_3	22.1 (9)	7.6 (2)	61.3 (4)	-	0.22 (4)	0.07 (4)	-	-	1.0 (2)	-	1.8(4)	94.6 (2.18)

The comparison of the clinker from SM(P)/2% $CaCl_2$ with the simplified calculated average pd-AAC composition SC-pd-AAC / 2 wt% $CaCl_2$ shows a general agreement. Deviating phase contents are due to an incomplete reaction. The experimental sample contains two C_2S modifications (61.4 wt% β-C_2S, 5.4 wt% α-C_2S, total 66.8 wt%). Assuming that the amorphous phase (15.2 %) comprises a proportion of amorphous C_2S that corresponds in mass to the content of C_2S in the crystalline phase (66.8 wt%), the total content of C_2S in the sample is 76.9 wt%. This C_2S content is slightly reduced compared to the calculated composition (85.2 wt%) due to the incomplete turnover of the raw materials quartz, calcite, lime, and anhydrite. The basic agreement of experiment and calculation enables the use of thermodynamic data to calculate mass and energy balances.

D.2.6 Comparing RC-BCC and PCC

The RC-BCC technology is in the stage of basic validation in a laboratory or relevant environment (TRL 4-5). A comparison with PCC is only possible to a limited extent. This section examines whether, and if so, which types of RC-BCC could be suitable as substitutes for PCC, whether they meet the requirements standardized for PCC and cement, and whether adaptation of standards might be necessary.

According to the definition given for PCC in DIN EN 197-1:2011-11, the processing of RC-BCC and PCC does not differ significantly. In particular, the standard does not define a minimum process temperature. As an essential requirement, PCC shall consist of at least two-thirds by mass of calcium silicates (C_3S and C_2S), the remainder consisting of aluminum and iron-containing clinker phases and other compounds. The ratio by mass (CaO)/(SiO_2) shall be not less than 2.0. The content of magnesium oxide (MgO) shall not exceed 5.0 wt%.

Additional chemical requirements apply to certain types of cement that contain PCC, e.g. masonry cement (DIN EN 413-1:2011-07), very low heat special cement (DIN EN 14216:2015-09), Portlandcomposite cement CEM II/C-M, Composite cement CEM VI (DIN EN 197-5:2021-07), supersulfated cement (DIN EN 197-5:2021-07). They mainly concern the loss on ignition (LOI) and insoluble residues (typically < 5 wt%) as well as the sulfate and chloride contents. Contents are analyzed according to DIN EN 196-2:2013-10. While RC-BCC meets the required properties in terms of LOI and insoluble residue, its sulfate and chloride contents vary on the secondary raw materials used, the mineralizer, and the chemical stability of the mineral phases formed. The maximum tolerable sulfate content, typically 3.5 to 4.5 wt% SO_3, can also be maintained by blending with PCC.

On the other hand, the use of (earth) alkali chloride as mineralizer is critical for standardized applications due to chloride-induced corrosion of reinforcements in structural components. Typically, the chloride content is limited to 0.1 wt.%. High chloride contents are only tolerated in special mortars, according to DIN EN 413-1:2011-07.

The typical application for high-belite clinker in China, very low-heat cement, is not standardized within the European framework. Instead, according to DIN EN 14216:2015-09, high amounts of ground granulated blast-furnace slags are combined with PCC for this purpose.

The performance of RC-BCC in AAC processing has been successfully tested on the lab scale (up to 50% substitution of PCC) and on an AAC production line (industrial scale, 25% substitution of PCC, (Stemmermann et al., 2023; Stemmermann et al., 2022). The setup is described in Section D.3.2. Technically, RC-BCC was ground to 90 wt% passing through a sieve of 90 µm. Subsequently, fixed ratios of RC-BCC and OPC were mixed with the other raw materials quicklime, anhydrite, primary sand, aluminium paste or powder, and water to form a thin mortar that was further processed into AAC by the industrial partner Xella (Stemmermann et al., 2022). The used recipes correspond to commercial products. The joint mass of RC-BCC and OPC replaced the cement content of the recipe.

D.2.7 Estimation of material balances, energy consumption, and CO_2 emissions

Based on the simplified formulation for the standard clinker SC-pd-AAC given in Tables A-1 and A-2 (supporting information), mass balances for the production of RC-BCC, including fuel and combustion gases for different firing conditions, were estimated (Table D.3).

Table D.3: Calculated mass and energy balances of the production of 1t RC-BCC from the raw materials SC-pd-AAC / 2 wt% $CaCl_2$ (Table A-2) for an all-electric process, for methane combustion equipped with oxyfuel and for air-operated methane combustion including energy commodities and oxygen/air. The CO_2 emissions from PCC are given for comparison, not differentiating biogenic or fossil fuels. (process emissions + fuels, = direct CO_2 emissions cement / clinker factor (VDZ, 2020, 2022)).

Mass balance

	Educts RC-BCC [kg/t]		Products RC-BCC	[kg/t]	H_2O, gases, cumulated	CO_2 (process + fuels) [%]
Electric heating	***AAC***	***607***	***RC-BCC***	***1000***		
	Tobermorite	*363*	*Belite*	*852*		
	Quartz	*144*	*Ellestadite*	*138*		
	Water	*31*	*CaO*	*10*		
	Gypsum	*69*				
	$CaCO_3$	*840*	*H_2O*	*107*	*107*	
	$CaCl_2.6H_2O$	*34*	*CO_2*	*369*	*369*	*55%*
			$c(CO_2)$ off-gas: ca. 100%			
	Sum	*1482*		*1476*		
+ Oxyfuel	***+CH_4***	***47***	***+H_2O***	***103***	***210***	
	+O_2	*197*	*+O_2*	*9*	*9*	
			+CO_2	*132*	*502*	*39%*
			$c(CO_2)$ off-gas: ca. 95%			
	Sum	*1725*		*1721*		
+ Gas combustion in air	***+CH_4***	***12***	***+H_2O***	***24***	***235***	
	+O_2	***49***	***+O_2***		***9***	
	+N_2	***800***	***+N_2***	***800***	***800***	
			+CO_2	***38***	***540***	***34%***
			$c(CO_2)$ off gas: ca. 36%			
	Sum	***2586***		***2584***		
PCC (for comparison)			***CO_2***	***820***		***100%***

Energy balance

	Thermal efficiency*	Heat demand [kJ/kg]	El. heat [kWh/t]	Milling [kWh/t]	Oxygen generation [kWh/t]**	Oxygen [SCM/t]	Sum El. supply [kWh/t]
Electric heating	60%	1937	538	110	-	-	648
+ Oxyfuel	50%	2606	CH_4	110	69	700	179
+ Gas combustion in air	60%	3243	CH_4	110	-	-	110

*: Estimate)

**: Assumption 0.5 kWh /SCM)

The limestone-pd-AAC mixing ratio, the relative amounts of CO_2 emission, and the clinker quantities rely on the composition of the pd-AAC fines. The following Sankey diagram (Figure D.2) gives a graphical representation of the mass flows and dependencies.

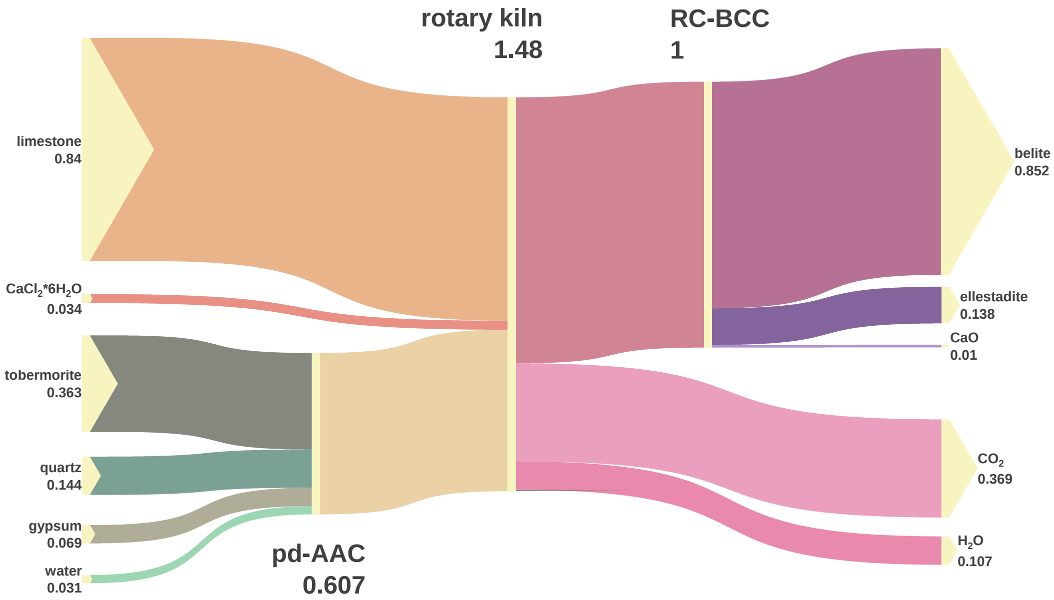

Figure D.2: Sankey diagram of mass-flow for belite clinker production based on pd-AAC input with C/S ratio=2, normalized for RC-BCC.

Pd-AAC consists of tobermorite, quartz, gypsum and water; all amounts of silicon dioxide and calcium oxide are considered reacting. Pure calciumcarbonate is added to adjust belite chemistry: for pd-AAC a typical molar CaO-SiO_2 ratio is 0.7. Pd-AAC samples usually contain up to 5 wt.% of additional oxides of Al, Fe, alkalies and earth alkalies, which are inert, thus not considered here. Hydrated calcium chloride is added as mineralizer.

The resulting RC-BCC leaving the rotary kiln contains belite and ellestadite, the later containing all the sulfate and chloride. The gas phase is composed of carbon dioxide and water vapour.

Table D.3 lists calculated energy balances for all investigated variants. In the case of oxyfuel combustion, additional energy is accounted for oxygen generation. A value of 0.5 kWh per standard cubic meter (SCM) has been taken from a Vacuum-Pressure Swing Adsorption (VPSA) installation with 100 SCM/h and 80% oxygen purity (Banaszkiewicz & Chorowski, 2018). The maximum product temperature is assumed to be 1000 °C.

Thermal efficiencies were estimated based on PCC processing. Thermodynamically, the production of PCC requires approx. 1800 kJ/kg clinker (Locher, 2000). Due to thermal losses by off-gas, wall losses, cooler exhaust air, and clinker, which decrease in this order, the actual thermal energy demand, according to BAT (European Commission - JRC IPTS European IPPC Bureau, 2013) for PCC increases to 3060-3620 kJ/kg clinker if no waste is used as fuel. These values include start-ups and shutdowns and result in a thermal efficiency of 50-60%, which may decrease when waste is used. Recent data from the German cement industry (VDZ, 2022) result in an energy input of 3875 kJ/kg (thermal efficiency 46%), taking into account the current clinker factor (0.73).

Plants for the production of a belite clinker do not yet exist. A hypothetical comparison of the thermal efficiency of the technology with the production of PCC shows great advantages for the losses via off-gas, especially for electric heating, since the quantity and temperature of the off-gas are significantly reduced. This reduction also applies to a limited extent to the oxyfuel technology. The lower firing temperature additionally reduces wall losses, losses from cooler exhaust air, and clinker. However, the lower maturity of the belite technologies has a negative impact. While air-operated gas burners are mature, the oxyfuel technology is in the optimization phase. Large-scale electric kilns at an operating temperature of 1000 °C are a technological challenge. For air-operated gas burners as a mature technology, a thermal efficiency of 60%, equal to the best BAT cement plants, was assumed (also reflecting the reduced temperatures compared to OPC clinkering). Due to the lower technical maturity of the oxyfuel technology, the estimated efficiency was reduced to 50%. For electric heating, 60% efficiency was assumed, as strongly reduced heat losses presumably compensate for the lower technical maturity. Concerning energy efficiency, the scaling of the plants is also essential but was not considered due to a lack of data. The electricity required for grinding the raw materials and products as well as for process control, fans, and transport, was not determined separately. Instead, a standard value of 110 kWh/t clinker was adopted from the production of OPC (VDZ, 2020).

D.3 Life cycle assessment of closed-loop recycling of pd-AAC using RC-BCC

D.3.1 Goal and scope

The LCA of RC-BCC production from pd-AAC and the AAC production using RC-BCC is done with the LCA software openLCA (GreenDelta, 2019) and the database ecoinvent 3.8 cut-off (ecoinvent, 2021). The goal is to determine whether closed-loop recycling of AAC via RC-BCC production is environmentally beneficial compared to landfilling.

Instead of the classical cradle-to-grave approach assessing primary resource extraction, production, use phase, and end-of-life, the following LCA of RC-BCC production from pd-AAC adjusts the system boundaries and the functional unit using the so-called zero burden approach (Nakatani, 2014). It excludes efforts for primary resource extraction, production, and use as well as transports. So, the pd-AAC enters the assessment of the end-of-life stage without any burdens. This assumption is reasonable since we only want to assess and compare waste treatment options at the end of life. The system boundaries include pd-AAC at the demolition site, RC-BCC production, AAC production using RC-BCC, and transports in between (see Figure D.1). This simplification does not influence the comparison of different end-of-life options, as the excluded efforts would be the same for all. Instead of the production output, the system's input (1 kg pd-AAC) is the functional unit. It allows for comparing different end-of-life options with different final products. A system expansion covers the handling of sorting residues following DIN EN ISO 14040:2021-02 and DIN EN ISO 14044:2021-02.

Furthermore, the avoided burden approach is applied to consider the final products' value (Nakatani, 2014). It allows rewards for avoided primary production of the replaced product. DIN EN ISO 14040:2021-02 and DIN EN ISO 14044:2021-02 name the system expansion as the best way to deal with valuable outputs and wastes in an LCA. It's assumed that the AAC produced using RC-BCC replaces primary AAC. Thus, the LCA includes an environmental reward since burdens from primary AAC production are avoided. This LCA reflects the specific case of Central Europe/Germany, so the German electricity mix is chosen for impact assessment.

D.3.2 Life cycle inventory (LCI) analysis

Here, all processes, flows, data sources, and assumptions for the LCA are described. All accompanying inventory data, uncertainties and references are listed in the Supporting Information S2. Data from ecoinvent 3.8 is used for assessing the pre-treatment processes of crushing, grading, and purifying, as well as for landfilling and primary production. The ecoinvent data quality system is used for Monte Carlo simulation to conduct a sensitivity analysis.

Today, landfilling is the main end-of-life option for pd-AAC and is the benchmark. The pd-AAC landfilling is calculated using the ecoinvent 3.8 dataset "treatment of inert waste, inert material landfill | inert waste, for final disposal", considering energy efforts for landfilling and occupation and transformation efforts. Furthermore, purifying residues from the pd-AAC recycling process are assumed to be landfilled and accounted for similarly.

The crushing is assessed with the ecoinvent 3.8 dataset "rock crushing | rock crushing". Material losses in the crushing process are marginal and not considered. Transports from the demolition site to a recycling plant are assumed to be 50 km. Pd-AAC can contain different adhesions or impurities, which should be minimized in the recycling process through purifying. Furthermore, a grading process can separate fine pd-AAC powder from coarse pd-AAC granulate. Only the powder is used for RC-BCC production. The coarse granulate could be used for other recycling options (Section D.1) or recirculated and crushed again to receive pd-AAC powder. The ecoinvent 3.8 dataset "treatment of waste brick, sorting plant | waste brick" is used to assess a combination of both processes. Datasets for crushed AAC do not exist, and primary data from industrial sites are unavailable. Furthermore, 0.01 kg purifying residue per kg purified pd-AAC is assumed to be sorted out while the grading has no material losses. Efforts are allocated by mass to the two final products: pd-AAC powder and granulate.

RC-BCC production is assessed using primary data for energy and mass flows (Section D.2) in four energy supply variants: natural gas, oxyfuel, conventional electricity (German electricity mix), and 100% renewable electricity. AAC production using RC-BCC is calculated for different AAC density classes (AAC-0.35, AAC-0.5, and AAC-0.55). The main inputs for AAC production are sand, cement, quicklime, and anhydrite. Production recipes are taken from Volk et al. (2023). Cement is substituted by 25% or 50% RC-BCC (Table A-3). The German electricity mix is available as an ecoinvent dataset. 100% renewable electricity is represented by the dataset "market for electricity, medium voltage, renewable energy products | electricity, medium voltage, renewable energy products" which is only available for Switzerland. The transport distance between RC-BCC production and the AAC production plant is assumed to be 50 km.

D.3.3 Life cycle impact assessment

The life cycle impact assessment (LCIA) is performed with the "ReCiPe 2016 Midpoint (H)" method. The cultural perspective of Hierarchist (H) is preferred to Individualist (I) and Egalitarian (E) because it reflects common assumptions, e.g., the climate change based on 100 years horizon instead of 20 years (I) and 500 years (E). All included midpoints are presented in Table A-4, and their assessment results are in Figures A-1, A-2, and in supporting information B.

In the RC-BCC production, pd-AAC processing (green) and transport efforts (blue) are relatively low and hardly visible for any midpoint (Figure D.3). In contrast, energy efforts for electricity (yellow) or natural gas (grey) influence the results significantly. Process emissions due to natural gas combustion and direct CO_2 emissions from limestone contribute extensively to global warming. Direct process emissions do not affect other midpoint categories, as only direct CO_2 emissions are considered (Figure D.1). As expected, renewable electricity usage in the rotary kiln has the lowest impact for all considered midpoints. The global warming (GW) of RC-BCC can be reduced from 0.76 kg CO_2-Eq/kg RC-BCC (German electricity mix) / 0.66 (oxyfuel) / 0.67 (natural gas) to 0.40 kg CO_2-Eq/kg RC-BCC (100% renewable electricity). However, natural gas firing (atmospheric or oxyfuel combustion) leads to lower impacts than using the German electricity mix for almost every midpoint, as the German electricity mix still depends significantly on fossil resources.

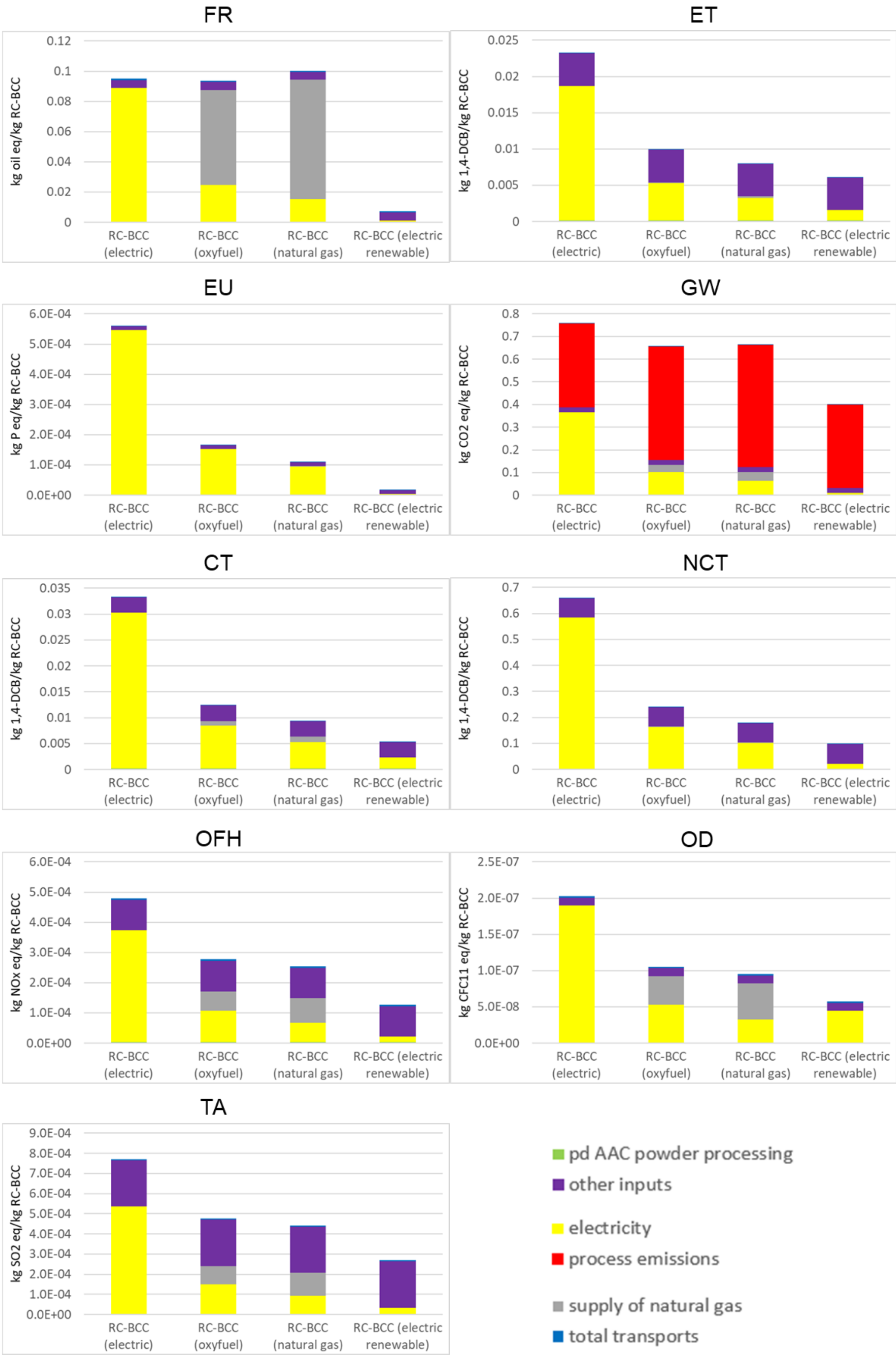

Figure D.3: LCA results for RC-BCC production with different firing conditions (FR: Fossil resource scarcity, ET: Freshwater ecotoxicity, EU: Freshwater eutrophication, GW: Global warming, CT: Human carcinogenic toxicity, NCT: Human non-carcinogenic toxicity, OFH: Ozone formation – Human health, OD: Stratospheric ozone depletion, TA: Terrestrial acidification).

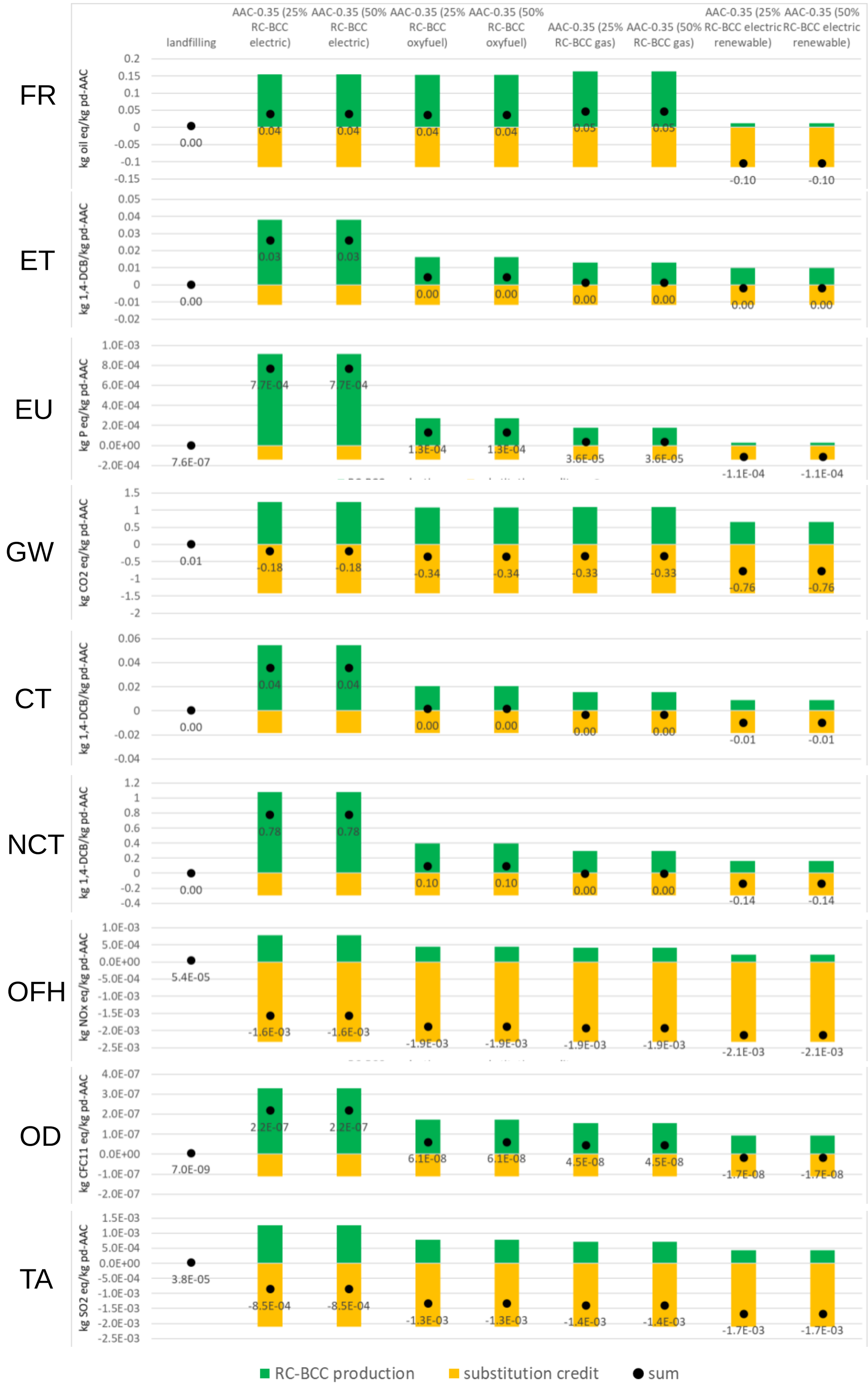

Figure D.4: LCA results for the AAC-0.35 production with RC-BCC content. Savings in relation to the functional unit 1 kg pd-AAC are displayed for different midpoints (FR: Fossil resource scarcity, ET: Freshwater ecotoxicity, EU: Freshwater eutrophication, GW: Global warming, CT: Human carcinogenic toxicity, NCT: Human non-carcinogenic toxicity, OFH: Ozone formation – Human health, OD: Stratospheric ozone depletion, TA: Terrestrial acidification).

Figure D.4 shows the results for AAC production using RC-BCC to substitute either 25% or 50% of the OPC. Overall, the substitution can reduce the impacts of AAC production. Concerning the GW, savings are highest for substituting OPC with RC-BCC produced using 100% renewable electricity, followed by oxyfuel combustion, natural gas, and conventional electricity. This order corresponds to the LCA results of the RC-BCC production and is the same for all AAC density classes. Global warming, e.g., is reduced from 0.01 kg CO_2-Eq/kg pd-AAC (landfilling, status quo) to -0.76 kg CO_2-Eq/kg pd-AAC (substitution of OPC by RC-BCC with 100% renewable electricity).

The substitution amount (pd-AAC input) hardly influences the total savings as the results are normalized to the functional unit of 1 kg pd-AAC. However, total savings are higher in the 50% than in the 25% substitution scenario if calculated per kg AAC-0.35 (final product). The GW of AAC-0.35 is reduced from 0.52 kg CO_2-Eq/kg (primary production) to 0.45 kg CO_2-Eq/kg (50% substitution, RC-BCC with 100% renewable electricity). Results for AAC-0.5 (reduction from 0.45 to 0.40 kg CO_2-Eq/kg) and AAC-0.55 (reduction from 0.39 to 0.35 kg CO_2-Eq/kg) are comparable to AAC-0.35. Both overall impacts and savings are somewhat lower for AAC-0.5 and AAC-0.55 as their production needs less cement than AAC-0.35.

D.3.4 Interpretation

The LCA results show significant potential for savings in several environmental impact categories, especially considering GW. Savings could reach 0.77 kg CO_2-Eq/kg pd-AAC compared to the status quo (landfilling) when RC-BCC is produced using 100% renewable electricity and OPC is substituted in AAC production. Firing the process with oxyfuel or natural gas still reaches savings of about 0.34 to 0.35 kg CO_2-Eq/kg pd-AAC. Impacts of 1 kg AAC-0.35 can be reduced by 0.07 kg CO_2-Eq/kg. This reduction of 13.5% is significant, considering only 15.5% of the overall input material is substituted. However, many midpoints show higher impacts when the RC-BCC is produced using conventional electricity.

A comparison to other recycling options for pd-AAC investigated by Volk et al. (2023) shows that RC-BCC produced with 100% renewable electricity shows the highest CO_2-Eq saving potential. However, RC-BCC produced with natural gas or oxyfuel combustion would be outperformed by direct usage of pd-AAC powder in AAC production and some open-loop recycling options like light mortar production.

German AAC production is around 3.5 million m³ (GENESIS, 2022), of which AAC-0.35 accounts for about 45%, AAC-0.5 for 20%, and AAC-0.55 for 10% (Volk et al., 2023). More than 82,000 t of pd-AAC could have been recycled in a closed loop, assuming a 50% substitution of OPC with RC-BCC in AAC-0.35, AAC-0.5, and AAC-0.55. This amount equals 12% of Germany's expected total pd-AAC amount of 700,000 t in 2022 (Steins et al., 2021). Total savings in greenhouse gas emissions would sum up to 63,600 t CO_2-Eq annually if the RC-BCC is produced using renewable electricity and 28,200 t CO_2-Eq if the RC-BCC is produced using natural gas.

D.3.5 Sensitivity analysis through Monte Carlo simulation

A Monte Carlo simulation with 10,000 runs for each process is performed for all scenarios (Table D.1-Table D.3) to analyse the sensitivity of the LCA results. The ecoinvent data quality system determines the parameters of a lognormal uncertainty function. It includes data reliability, completeness, temporal correlation, geographical correlation, and further technological correlation. Uncertainty values for these categories are taken from the ecoinvent 3.8 database if possible. Inputs for the RC-BCC production are rated 1 (data reliability), 5 (completeness), 1 (temporal correlation), 1 (geographical correlation), and 5 (further technological correlation) to reflect the presented situation. The simulation shows that the initially calculated values are robust (Figure D.5 and supporting information B). Median values for all scenarios and all midpoints are close to the original value. The RC-BCC production shows sensitivities of around -10% to -20% and +20 to +30%, depending on the firing variant. The final AAC production with RC-BCC shows even lower sensitivities of less than -10% and up to +10%. The highest sensitivity for all scenarios and midpoints is around -20% to +40% in the 25%/75% percentiles (supporting information B). Overall, calculation results seem robust even though the RC-BCC production at the current TRL is associated with some data uncertainty, especially for completeness and technological correlation.

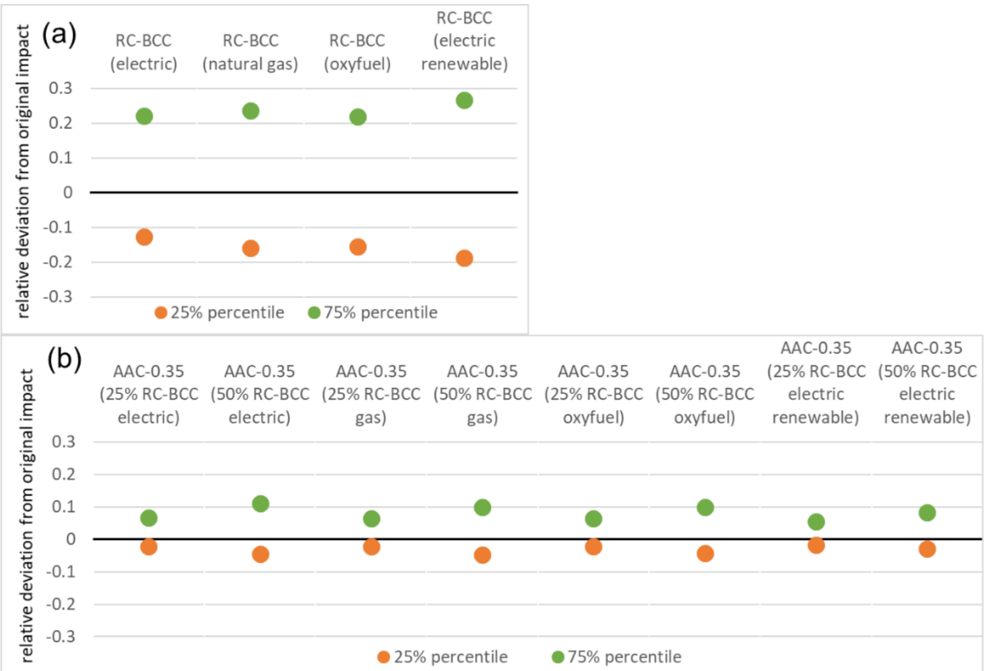

Figure D.5: Sensitivity of the LCA results in a Monte Carlo simulation (25% and 75% percentile) for all considered scenarios, including RC-BCC production (a) and final AAC production (b), for the midpoint GW.

D.4 Discussion

Processing RC-BCC in general and in particular from pd-AAC is new; details are presented here and in (Stemmermann et al., 2022). Significant amounts of RC-BCC have been produced by a service provider and used in a semi-industrial test in AAC production, where RC-BCC successfully replaced 25% of OPC. The LCA is based on our experimental data. However, the process has a TRL of 4-5, only optimized on the laboratory scale.

RC-BCC shows clear advantages concerning energy consumption and CO_2 emissions compared to PCC. However, the technology also has disadvantages and risks; e.g. at least to date, hydraulic reactivity is significantly slower, which may result in longer processing times for products or the need for accelerators, thus higher costs.

Other critical issues are homogeneity and quality of the product if variably composed and contaminated secondary raw materials are processed. High levels of impurities result in a reduced proportion of hydraulically effective clinker minerals. Another difficulty is the permanent supply of distributed residual materials of defined quality for clinker production and its transport logistics compared to quarrying raw materials. Work on technical applications of the RC-BCC and the generated CO_2 is in progress but remains largely unresolved. Legal framework conditions are lacking.

For future assessments, two cases are relevant: (1) Exclusive processing of pd-AAC and (2) Blending of pd-AAC with other materials (fiber cement (cellulose-based), waste concrete). The latter might increase plant utilization and help to increase recycling material flows but would require adjustments of raw meals and process parameters.

The assessment results are associated with considerable uncertainty. The process shall be further developed to a higher TRL with more precise process parameters to reduce these uncertainties. However, the LCA data's uncertainty is somewhat included via Monte Carlo simulation (Section D.3.5).

Moreover, the scaling and placement of plants have to be discussed. Large plants are cost-efficient due to economies of scale (Norman, 1979) and energy-efficient due to reduced heat losses. Comparing the energy demand in BAT (3,000 t/d, 3,340 kJ/kg, mean) and average German plants (2,675 t/d, 3,875 kJ/kg; energy average of all kilns, capacity for rotary kilns only, 98.8% of the total capacity, VDZ (2022) suggests that other factors play a significant role. One factor is the increasing energy demand when using secondary wastes as fuel. If waste fuels are combined with carbon capture usage and storage, indirect heating options would be required not to contaminate the concentrated CO_2. Moreover, transport and reverse supply chain problems might arise in a large central plant. Given the high processing temperature (1450 °C), small-scale PCC processing seems complicated.

RC-BCC production on a small scale (125 t/d) seems technologically feasible. However, data is lacking, especially for various heat input options. Appropriate approaches are currently being developed. Small-scale RC-BCC production would require many more plants with smaller sourcing areas, stocks and transport distances. Moreover, they could face material shortages if sourcing regions become too small. Large and small plants must adapt to the volatile and seasonal demolition, e.g. via stockholding.

Besides, the assumption of 110 kWh/t for milling of pd-AAC might be too conservative since the material is quite soft. Moreover, the LCA results might change due to efficiency gains, significantly higher pd-AAC volumes in the future, higher belite amounts in recipes, or changing recycling strategies. It is striking how much the choice of electricity mix (German electricity mix or 100% renewables) influences the results. In the case of oxyfuel/natural gas, the German electricity mix was used to assess electricity efforts. When changing this to 100% renewable electricity, the impact of the oxyfuel process would be reduced more than of the normal combustion since additional electricity is needed for oxygen production. Economic aspects, market barriers, or further process integration are not assessed.

AAC production using RC-BCC is considered in different scenarios, combining two substitution levels and three AAC density classes. The density class influences the composition of the input materials and, therefore, affects the results. However, the substitution levels do not influence the results since the functional unit is 1 kg pd-AAC. Total savings per kg produced AAC are higher in the 50% substitution scenario. Since pd-AAC volumes are expected to rise significantly in Germany and Europe (Steins et al., 2021, 2022), AAC recycling should aim at high substitution rates to minimize environmental impacts.

D.5 Conclusion

Using a small semi-industrial kiln RC-BCC from waste concrete has been successfully processed. The suitability of RC-BCC as a partial cement substitute in aerated concrete production has been demonstrated in technical trials (Stemmermann et al., 2022). The processing of RC-BCC from pd-AAC was established on a laboratory scale using samples from real collected waste with different levels of contamination. Both the specific energy requirement and the specific emissions of CO_2 can be minimised if electrical heating based on renewable electricity is used in the production process. For economic reasons, however, plants with an annual production of at least 50 kt are necessary. An electrically heated rotary kiln intended for upscaling the technology on this scale is uncharted technological territory. Further difficulties of upscaling concern the control of a homogeneous raw meal composition, heat recovery for high energy efficiency and the constant supply of suitable primary and secondary raw materials.

Improvements in various environmental aspects through pd-AAC recycling are possible, especially concerning CO_2-Eq savings and landfill capacity. Closed-loop pd-AAC recycling via the belite route and substituting OPC shows promising results and significant potential savings of 0.77 kg CO_2-Eq/kg pd-AAC compared to the status quo (landfilling) by using renewable electricity. Savings reach 0.35 kg CO_2-Eq/kg pd-AAC when using oxyfuel combustion or natural gas firing. The gained reduction of 13.5% is significant, considering that only 15.5% of the overall input material is substituted. The results indicate that closed-loop recycling of pd-AAC could significantly reduce environmental burdens associated with the current landfilling. However, the assessment also shows some worsening of environmental impact categories, e.g., freshwater ecotoxicity or eutrophication, human carcinogenic toxicity, and stratospheric ozone depletion when using the German electricity mix. Other recycling options are more promising regarding these environmental impact categories (Volk et al., 2023).

Future research should focus on assessing a pilot plant instead of laboratory data to enhance decision-making. Furthermore, a system analysis and network design for a full AAC circularity could help to identify the optimal recycling routes, plant capacities, and placements under given or future conditions, e.g. spatial and temporal availability of rising pd-AAC, quality aspects and current regulation.

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E Economic Assessment of Post-Demolition Autoclaved Aerated Concrete (AAC) Recycling and Subsequent Belite Cement Clinker Production

Abstract[1]

Autoclaved aerated concrete (AAC) is a building material with high thermal insulation properties used as masonry units in the construction of residential buildings. Post-demolition AAC (pd-AAC) wastes are expected to rise in the following decades as AAC's popularity significantly increased in the 1960s and 1970s. However, pd-AAC is mainly landfilled today while landfill fees rise, legal framework conditions in Europe are tightening, and climate protection needs extensive efforts in the area of recycling. This study presents an economic assessment of pd-AAC recycling, consisting of mechanical processing (crushing, grading, purifying) and subsequent belite cement clinker production from the fine pd-AAC fraction.

The processes are modelled in detail to determine needed equipment, material flows, and energy demands for five different plant capacity scenarios. Calculated total costs of pd-AAC recycling, consisting of variable costs, fixed costs, overhead costs, and general expenses, vary significantly between the different scenarios. Mechanical processing of pd-AAC has total costs between 30 €/t input (plant capacity: 250,000 t/a) and more than 200 €/t input (plant capacity: 10,000 t/a). The mechanical processing is economically viable compared to average pd-AAC landfilling costs of 100 €/t for recycling plants with capacities of at least 25,000 t/a. Additional costs for subsequent belite cement clinker production from pd-AAC sum up to 800 €/t input (plant capacity: 250,000 t/a), respectively 1250 €/t input (plant capacity: 10,000 t/a). Thus, the minimum sales price for the resulting belite cement clinker would need to be around 430 €/t to compete with current landfilling costs.

[1] This section includes the article "Economic assessment of post-demolition autoclaved aerated concrete (AAC) recycling and belite cement clinker production", developed by Rebekka Volk, Günter Beuchle, Pallavi Reddy, Gourisankar Sandaka, Frank Schultmann, and myself. The article has been submitted for publication in a scientific journal as Steins et al. (2023). The supplementary will be found on the journal website after publication.

E.1 Introduction

The building sector has a high and increasing resource consumption and causes vast greenhouse gas (GHG) emissions during construction, operation, and end-of-life. Therefore, considerable savings in the building sector have to be implemented to reach the UN sustainable development goals, particularly “sustainable cities”, “responsible consumption and production”, and “climate action” (UN, 2023). Recycling construction and demolition wastes (C&DW) is a promising approach for reducing GHG emissions and primary resource consumption. The savings potential is enormous as C&DW exceed 3 billion tons worldwide annually (Akhtar & Sarmah, 2018). Furthermore, legal requirements for recycling are getting stricter. For example, the European waste and recycling regulation (Directive 2008/98/EC, 2008) demands recycling rates of at least 70% for C&DW. But, until now, “the potential of the circular economy to support sustainable cities, regions, and countries still needs to be unlocked” (OECD, 2020).

Autoclaved aerated concrete (AAC) is produced from quartz sand, cement, quicklime, anhydrite/gypsum, aluminium powder/paste, and water (DIN 20000-404:2018-04; Kreft, 2017). The aluminium powder/paste acts as an aerating agent that forms numerous pores in the AAC during production. The porous structure leads to a low density and excellent thermal insulation properties of AAC, which is the main reason for its high popularity. The current European AAC production exceeds 16 million m³ annually (EAACA, 2023), while approximately 11.6 million m³ AAC were produced in Russia in 2017 (Grinfel'd et al., 2018). The global production capacity is expected to be around 450 million m³ for non-reinforced AAC blocks (Fouad & Schoch, 2018). Post-demolition AAC (pd-AAC) volumes are currently increasing. In Germany, an annual pd-AAC volume of 1.4 million m³ in 2022 and a sharp increase to more than 4 million m³ in 2050 is expected (Steins et al., 2021). Therefore, the recycling potential is enormous.

Unfortunately, the usual recycling of mineral C&DW in road construction, earthworks, and aggregate in concrete production is impossible for pd-AAC due to the porous structure, relatively low compressive strength, and sulphate content. Besides, adherences and impurities impede recycling (Deilmann et al., 2014). Thus, recycling of pd-AAC is not established yet, and the majority of pd-AAC is backfilled or landfilled, even though landfill fees are expected to rise, and landfilling capacities are limited (Knappe et al., 2012; Riegler-Floors & Hillebrandt, 2018). Additionally, reusing pd-AAC blocks is impractical due to the immense costs of an extremely careful demolition process (Gyurkó et al., 2019) and the incompatibility of historical AAC blocks with up-to-date thermal protection requirements.

Current research investigates new possibilities for pd-AAC recycling in the construction sector. Proposed options include the production of new AAC (Kreft, 2017; Lam, 2021; Rafiza et al., 2019; Rafiza et al., 2022), floor screed (Bergmans et al., 2016), light mortar (Aycil et al., 2016), lightweight aggregate concrete (Aycil et al., 2016; Gyurkó et al., 2019), and shuttering block made from concrete without fine fraction (Gyurkó et al., 2019). These recycling options need the pd-AAC to be crushed, purified and graded. However, they primarily use the pd-AAC granulate (grain size > 1 mm) to replace natural aggregates. Any mechanical treatment also generates a relatively large pd-AAC powder fraction (grain size 0-1 mm; up to 75 wt.%) that is difficult to recycle. As pd-AAC volumes are expected to rise (Steins et al., 2021), recycling options for both fractions are needed.

Pd-AAC powder can be used as a raw meal component for producing a recycled belite cement clinker (RC-BCC) in a low-temperature process at 1000°C (Ullrich et al., 2021). The main clinker reaction takes place in an indirectly heated, electric rotary kiln under a CO_2 atmosphere. RC-BCC can substitute parts of the ordinary Portland cement needed for new AAC blocks (Stemmermann et al., 2022) or other applications.

This study conducts an economic assessment of pd-AAC recycling, focusing on the mechanical processing (crushing, purifying, and grading) required for different recycling options and the RC-BCC production. Data for the latter is from lab and pilot plant tests. The current technology readiness level of the RC-BCC production from demolition wastes is 4-5. Therefore, the exact technology and equipment required to set up a future production plant are not well established, resulting in inaccuracies in the cost assessment of RC-BCC production. Much literature performs (techno-) economic assessments for numerous processes and products. Assessments for recycling processes are also available in the literature, for example, concerning the recycling of plastics (Fivga & Dimitriou, 2018; Larrain et al., 2021; Volk et al., 2021), lightweight packaging (Cimpan et al., 2016), e-waste (Cucchiella et al., 2015), solar photovoltaic panels (Granata et al., 2022), lithium-ion batteries (Thompson et al., 2021), agricultural waste (Hassanpour, 2021), and municipal solid waste in general (Athanassiou & Zabaniotou, 2008). This study addresses the economic assessment of pd-AAC recycling and subsequent RC-BCC production, which is not available in the literature yet. Therefore, the research question to be answered is: Under which circumstances can pd-AAC recycling and RC-BCC production from pd-AAC be economically beneficial? The following sections describe the methodology (Section E.2) and the results (Section E.3). Then, the results are discussed, and limitations are presented (Section E.4). Finally, a conclusion is drawn (Section E.5).

E.2 Methods

This section describes how the economic assessment is conducted and which input data is used. First, the supply of pd-AAC is investigated, and the revenue that can be made from the final products is determined (Section E.2.1). Furthermore, the recycling process is described and illustrated in detail, including information on the mass flows. The mechanical processing is examined in Section E.2.2, while the RC-BCC production is described in Section E.2.3. Additionally, the economic assessment methodology is disclosed (Section E.2.4), and scenarios are discussed (Section E.2.5).

E.2.1 Pd-AAC supply and revenue for the final product

In contrast to other countries and regions, detailed information on pd-AAC volumes is available for Germany (Steins et al., 2021). Thus, the German case is investigated in this study. 2022, around 1.4 million m^3 of pd-AAC can be expected (Steins et al., 2021). This volume equals 0.7 million t of pd-AAC, assuming a density of 0.5 t/m^3 (Deilmann et al., 2014; Müller, 2016; Steins et al., 2021; Volk et al., 2019). Most popular modern AAC has a density of around 0.35 t/m^3, but, historically, thermal insulation requirements were not as high as today, and AAC's density used to be higher (Schlegel & Hums, 2002). Most pd-AAC is landfilled today, demanding disposal costs, which is, thus, the comparative value for the recycling process costs. Pd-AAC landfill fees in Germany vary between 65 to 180 €/t (Aycil & Hlawatsch, 2020). Enquiries in online portals[2] and an expert interview[3] validate this variability, while the average disposal costs for landfilling pd-AAC are around 100 €/t.

The final products of the mechanical processing and purifying of pd-AAC are pd-AAC powder and pd-AAC granulate. These two products can substitute different primary resources depending on their final application. The pd-AAC powder usually replaces sand and, in the case of closed-loop recycling, also partly cement, quicklime, and anhydrite, which are needed for AAC production (Volk et al., 2023). The pd-AAC granulate can be used in several open-loop recycling options serving as lightweight aggregate substituting, for example, primary expanded clay (Volk et al., 2023). Overall, pd-AAC powder/granulate as the final product of the mechanical processing is assumed to reach a sales price of 10 €/t. Actual market prices for pd-AAC powder or granulate do not exist yet. Research on recycling sand/split/broken rocks showed sales prices between 5 €/t and 15 €/t (initial interactive gmbh, 2023). Additionally, pd-AAC powder can be used in RC-BCC production. The costs of the RC-BCC production are compared with the average price of ordinary Portland cement prices of 150 €/t (cemex, 2022; Dyckerhoff, 2022), as a direct substitution is possible.

E.2.2 Mechanical pd-AAC processing

The mechanical processing of pd-AAC consists of crushing, purifying, and grading steps (Krampitz et al., 2022; Kreft, 2016). Krampitz et al. (2022) show that pd-AAC can be treated with established demolition waste processing machinery, especially regarding crushing. Figure E.1 (a) illustrates the detailed process considered adequate for pd-AAC mechanical processing in this study. First, the pd-AAC is crushed to grain sizes < 80 mm with a jaw crusher. The crushed pd-AAC is then purified using air separation to separate lightweight impurities like plastic foils, foamed materials, and paper. A second purifying step is near-infrared (NIR) sorting. This step can sort out heavy impurities like other minerals, glass, ceramics, wood, and screws. Afterwards, the purified pd-AAC is crushed a second time using an impact crusher to

[2] The portals abfallscout.de and clearago.de were used.

[3] Xella Technologie- und Forschungsgesellschaft mbH, Dr. Oliver Kreft.

reach the desired grain size of < 10 mm. Finally, a vibrating screen separates the pd-AAC powder (< 1 mm) from the pd-AAC granulate (1-10 mm).

The relative mass flows in the pd-AAC recycling process are also given in Figure E.1 (a). In total, 1% of the input mass is assumed to be an impurity that is sorted out (Volk et al., 2023). The air separation is supposed to sort out 0.1%, while the NIR sorting is assumed to sort out 0.9% of the total input mass. The final impact crushing is considered to produce pd-AAC powder and granulate in a proportion of 3:1 (Gyurkó et al., 2019; Volk et al., 2023), leading to an overall output of the mechanical processing of 74.25% pd-AAC powder and 24.75% pd-AAC granulate. Electricity for the machines is the only energy needed for mechanical processing. Electricity demands were researched in machine specification sheets and are given in Section E.3 (Table E.2). The electricity demand per ton is calculated from the maximal power input and, thus, should be considered as a conservative electricity cost assessment.

E.2.3 RC-BCC production from pd-AAC powder

The resulting purified pd-AAC powder (< 1mm) from the mechanical processing is processed in a multistage process to produce a RC-BCC. This process involves drying, milling, rotary kiln processing, and cooling, as illustrated in Figure E.1 (b).

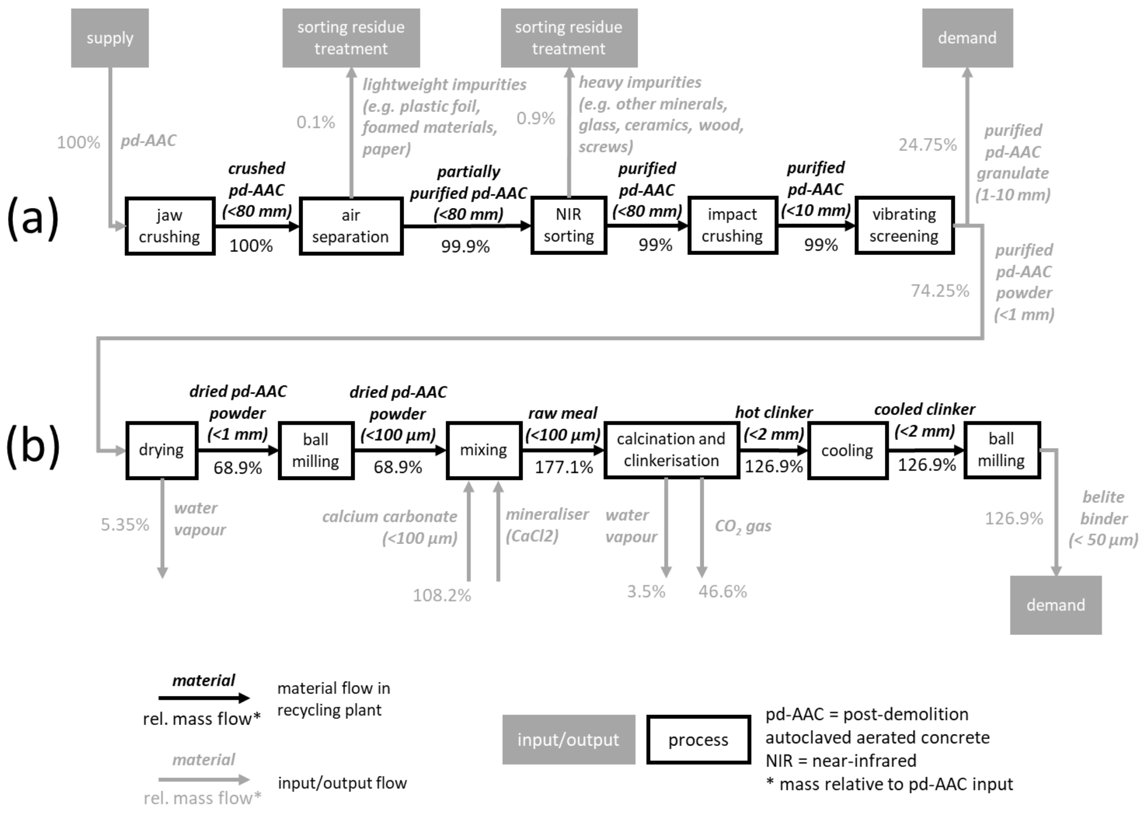

Figure E.1: Schematic representation of (a) the mechanical pd-AAC processing and (b) RC-BCC production from pd-AAC powder, including relative mass flows.

First, the pd-AAC powder is dried to reduce the moisture content. Pd-AAC samples considered in an experimental study by Ullrich et al. (2021) have a moisture content ranging from approximately 9 to 15 wt.%. All mass and energy balance calculations in this work are based on 9 wt.% moisture, according to Stemmermann et al. (2023). The second step is milling pd-AAC to d_{80}=100µm size. Typically, the pd-AAC has an average molar ratio of CaO/SiO_2 = 0.5, while the formation of belite requires a CaO/SiO_2 molar ratio of 2. Therefore, calcium carbonate is added. Furthermore, a mineraliser ($CaCl_2$) of 2 wt.% is added to improve the reaction kinetics. Then, the raw material is fed into an electrical rotary kiln heated to 1000°C, in which calcination and clinkerisation reactions lead to the formation of belite clinker. The hot clinker is finally fed into a cooler and ground to a size of d_{80} = 50µm.

E.2.4 Methodology of the economic assessment

The calculation of the total costs for pd-AAC recycling and RC-BCC production is based on the methodology of Peters et al. (2003). Total costs are made up of variable costs, fixed costs, overhead costs and general expenses. Variable costs comprise several components, including operating labour, electricity, and maintenance.

The fixed costs are determined by the fixed-capital investment for building a recycling plant, the required working capital, and land costs. The land costs are calculated from the assumed required area for the plant (1 ha in the baseline scenario, scaled with an exponent of 0.9 for the other scenarios) multiplied by the average costs per m² (Table E.1). The fixed-capital investment and the working capital are calculated using the “percentage of delivered-equipment cost” approach by Peters et al. (2003). The costs for the required equipment are the basis of this method. Further cost aspects, total fixed-capital investment, and working capital are estimated by multiplying percentages with equipment costs. Included cost aspects and their respective cost percentages of the equipment costs are given in Section E.3. The required equipment is derived from Figure E.1. Besides the machines directly shown, a compressed air generation for the NIR sorting and nine conveyor belts (one for each transport, i.e. arrow in Figure E.1 (a)) for general product transport through the facility are needed for mechanical processing. The RC-BCC production uses an additional eight conveyor belts. Equipment costs were researched by direct inquiry to manufacturers for jaw crushing, impact crushing, air separation, NIR sorting, vibrating screening, conveyor belts, and compressed air generation. The dryer, the ball mill, and the rotary kiln costs were calculated using the correlation function introduced by Towler and Sinnot (2012) and given in Equation E.1.

$$C_e = a + b * S^n \tag{E.1}$$

C_e = cost of purchased equipment

a, b = cost constants for the equipment

S = size parameter

n = exponent for that type of equipment

The size parameter of each piece of equipment relates to different scenarios described in Section E.2.5. The size of the dryer is measured by area (m^2), the ball mill by capacity (t/h), and the rotary kiln by power (MW). The constants a, b, n are taken from Towler and Sinnot (2012) for the respective equipment. The area of the dryer is calculated using dimensions given by Michaud (2016) for different capacities. However, inaccuracies might occur in these calculations due to the ambiguity of the provided data on the dryer area. Ball mill capacities (t/h) for pd-AAC feedstock and clinker are taken from mass balances (Figure E.1) for different plant sizes. The electrically heated kiln is assumed to be a cylindrical furnace with a theoretical energy demand calculated based on the enthalpy of reactions, calcination and clinkerisation and the assumption that 60% of the specific heat of both clinker and hot CO_2 can be recovered. The resulting energy demand is multiplied by an assumed factor of 1.5 to account for heat losses throughout the process. This factor implies a process efficiency of 67%. Compared to this factor, Locher (2000) calculates an efficiency of 83% for a modern ordinary Portland cement production by considering a factor of 1.2.

Clinker cooling equipment cannot be considered in this study due to a lack of data. RC-BCC for lab kilns shows an agglomerate size of up to 2mm, which is different from ordinary Portland cement clinker. Thus, specific cooling technology and setups have to be investigated in the future.

Finally, the Chemical Engineering Plant Cost Index (CEPCI) is used to adjust the equipment costs for inflation over time. Aspen Plus (Aspen Technology Inc, 2023) is used to estimate the electricity requirements for the ball mill and dryer across different capacities.

Annual fixed costs resulting from the fixed-capital investment are calculated using an annuity factor that is calculated using Equation E.2 (Smith, 2005). Annual fixed costs from the working capital and land costs are calculated using the interest rate per year, not the annuity, as no amortisation is needed in these categories.

$$annuity\ factor = \frac{i*(1+i)^n}{(1+i)^n-1} \tag{E.2}$$

i: interest rate per year

n: service life of the plant in years

Overhead costs and general expenses are calculated as percentages of operating labour costs and total product costs. Transports of input material to the recycling plant and final products to the point of demand are not included in the economic consideration since the focus is on assessing the recycling plant itself. A brownfield investment is assumed since it could be added to existing infrastructure for construction and demolition waste treatment or AAC/cement production plants. A recycling plant at a greenfield location could double the brownfield investment (Peters et al., 2003). Since the pd-AAC presumably comes from different demolition sites and due to uncertainty in the availability of raw materials, 275 working days per year are assumed. All relevant primary data and assumptions for the economic assessment are given in Table E.1.

Table E.1: Relevant primary data and assumptions for the economic assessment of pd-AAC recycling.

parameter	value	reference
operational time mechanical processing [h/a]	3,300	assumption (12 h/d, 275 d/a)
operational time RC-BCC production [h/a]	6,600	assumption (24 h/d, 275 d/a)
interest rate [-]	0.07	assumption
share of borrowed capital [-]	1	assumption
service life of the plant [a]	15	assumption
annuity factor [-]	0.11	own calculation using Equation E.1
labour costs [€/working hour]	41.90	labour costs in the manufacturing sector in Germany in 2021 (Destatis, 2022)
sorting residue treating [€/t]	100	assumption
limestone [€/t]	40	expert interview[4]
$CaCl_2$ costs [€/t]	92	assumption based on own research in online portals (chemieshop24.de, german.alibaba.com)
CO_2 certificate costs [€/t]	85	European Energy Exchange AG (2023)
electricity costs [€/kWh]	0.265	BDEW Bundesverband der Energie- und Wasserwirtschaft e.V. (2023), as of: July 2023
land costs (economically used building land) [€/m²]	63.48	GENESIS (2022), average 2021 value
CO_2 certificate costs [€/t]	85	European Energy Exchange AG (2023)

E.2.5 Scenario definition

The size of a plant usually significantly impacts the total product costs. Therefore, different plant sizes are considered in scenarios to disclose the range of the total costs. The recycling

[4] Dr.-Ing. Jesko Gerlach, Holcim GmbH.

plant has an input capacity of 50,000 t/a in the baseline scenario. It would need 14 of these recycling plants to handle the current German pd-AAC amount of around 700,000 t/a and 40 to handle the expected increase of up to 2,000,000 t/a until 2050 (Section E.2.1). Moreover, other recycling plants with lower input capacity (10,000/25,000 t/a) are investigated to reflect more decentralised recycling possibilities where transport can be minimised. Additionally, recycling plant scenarios with higher input capacity (100,000/250,000 t/a) are included in calculating total costs when significant economies of scale become effective. The varying capacity in the different scenarios influences the required capacity of the machines. Generally, the change in equipment costs due to increased or reduced capacity can be calculated using Equation E.3 (Humphreys, 2005).

$$C_2 = C_1 * \left(\frac{Q_2}{Q_1}\right)^x \tag{E.3}$$

C_2: cost of capacity Q_2

C_1: cost of capacity Q_1

x: cost-capacity factor

Equation E.3 is also used for scaling the electricity inputs of the machines since the electricity inputs are assumed to follow the same sublinear relationship with the capacity as the costs. Factor x is calculated separately for costs and electricity input (electricity-capacity factor). There are various suggestions for cost-capacity factors for different plants, machines, and machine parts in the literature. However, calculating the factor for every machine directly from specific cost and electricity input data is the most precise approach. Thus, machines' prices and electricity inputs were researched for two to six capacities. Rearranging Equation E.3 allows a calculation of machine-specific cost-capacity/electricity-capacity factors resulting in 0.37/0.82 for the crushers, 0.42/0.68 for the air separator, 0.5/0.55 for the NIR sorting machine, 0.2/0.64 for the compressed air generator, and 0.61/0.65 for the vibrating screen. It's assumed that all machines reach the highest capacity of 250,000 t/a (around 70 t/h), except for the NIR sorting machine. A direct manufacturer enquiry disclosed a maximum capacity of available NIR sorting machines of about 15 t/h for pd-AAC with an assumed density of 0.5 t/m³ after primary crushing (< 80mm). Higher throughputs are supposed to be handled by parallel sorting on multiple NIR machines. However, one NIR sorting machine can take the pd-AAC input of up to 50,000 t/a. The parallel use of several machines is only relevant for the plant size scenarios of 100,000 t/a (two NIR sorting machines required) and 250,000 t/a (five NIR sorting machines required). Equation E.3 is used for cost calculation of jaw crushing, impact crushing, air separation, NIR sorting, vibrating screen, and compressed air generation. The cost calculation of the dryer for the highest capacities, i.e., 100,000 and 250,000 t/a, is also performed using Equation E.3, as the dimension data was not given in the reference. The ball mill and rotary kiln costs in all scenarios are calculated equally to the baseline scenario.

E.3 Results

This section shows the results of the economic assessment. First, the costs of mechanical pd-AAC processing and RC-BCC production are calculated for the considered scenarios (Section E.3.1). Then, a sensitivity analysis of the results is presented (Section E.3.2).

E.3.1 Costs of mechanical pd-AAC processing and RC-BCC production

The recycling process, including mechanical processing and RC-BCC production, uses equipment described in Section E.2.2 and Figure E.1. Table E.2 discloses the costs and the electricity demand for the entire equipment of both recycling steps. The values for the conveyor belts reflect nine belts for the mechanical pd-AAC processing and eight belts for the RC-BCC production, one between all components of the recycling plant and all materials sorted out (one belt per material flow/arrow in Figure E.1). The conveyor belts are always needed for material transport. Therefore, no scaling of costs or electricity demand is performed for recycling plants of different capacities. The total costs of all equipment are then used as input for the total capital investment assessment (Table E.3) following the method described in Section E.2.4. The total capital investment is around 4 M€ for the mechanical pd-AAC processing and an additional 13.6 M€ for the RC-BCC production in a plant with a pd-AAC input capacity of 50,000 t/a.

Table E.2: Equipment costs and electricity demand for mechanical pd-AAC processing and RC-BCC production in the baseline scenario (50,000 t pd-AAC/a).

equipment	equipment costs [€]	electricity demand [kWh/t input]
jaw crusher	59,422	1.4
impact crusher	43,110	2.7
air separator	52,035	1.0
NIR sorting machine	388,755	1.0
compressed air generation	9,981	0.3
vibrating screen	26,571	0.1
conveyor belts	131,400	0.1
total for mechanical processing	711,274	6.7
dryer	374,154	476.1
ball mill 1	615,709	14.6
rotary kiln	894,353	932.2
ball mill 2	794,756	24.5
conveyor belts	102,200	0.3
total for RC-BCC production	2,781,171	1447.7

Table E.3: Capital investment calculation for the mechanical pd-AAC processing and RC-BCC production in the baseline scenario (50,000 t pd-AAC/a) based on the percentage of delivered equipment cost method by Peters et al. (2003).

cost category	percentage of delivered equipment	costs mechanical processing [€]	costs RC-BCC production [€]
direct costs			
purchased equipment delivered	n.a.	711,274	2,781,171
purchased-equipment installation	0.45	320,074	1,251,527
instrumentation and controls (installed)	0.18	128,029	500,611
piping (installed)	0.16	113,804	444,987
electrical systems (installed)	0.1	71,127	278,117
buildings (including services)	0.25	177,819	695,293
yard improvements	0.15	106,691	417,176
service facilities (installed)	0.4	284,510	1,112,469
total direct plant costs	2.69	1,913,328	7,481,351
indirect costs			
engineering and supervision	0.33	234,721	917,787
construction expenses	0.39	277,397	1,084,657
legal expenses	0.04	28,451	111,247
contractor’s fee	0.17	120,917	472,799
contingency	0.35	248,946	973,410
total indirect plant costs	1.28	910,431	3,559,899
total costs			
fixed-capital investment	3.97	2,823,760	11,041,251
working capital	0.70	497,892	1,946,820
land costs	n.a.	634,800	634,800
total capital investment	n.a.	3,956,452	13,622,871

The results of the total product cost calculation for the baseline scenario are presented in Table E.4. The variable costs do not consider all aspects mentioned by Peters et al. (2003) since the following are not associated with any charges in the case of mechanical pd-AAC processing and RC-BCC production: costs for fuel (as only electricity is used), refrigeration, steam, process water, cooling water, and royalties. Additionally, there are no raw material costs for the mechanical pd-AAC processing. Potential acceptance fees for the pd-AAC treatment are considered in comparing recycling and landfilling costs. However, raw material costs for limestone for the RC-BCC production are included (Table E.1). The assessment of operating labour costs is calculated from data on operating labour requirements (Peters et al., 2003), assuming a highly automated process with two process steps (sorting, crushing/grading) for mechanical pd-AAC processing and three steps (drying, clinkerisation, milling) for RC-BCC production. The operating labour requirement is then multiplied by average labour costs (Table E.1) to get total operating labour costs. Electricity costs are determined from the electricity demand (Table E.2) and costs per kWh (Table E.1). Waste treatment and disposal costs are calculated from treatment costs (Table E.1) multiplied by the assumed 1% of all inputs to be sorted out. Additionally, CO_2 certificate costs must be considered for the RC-BCC production as the cement industry has to buy certificates for the direct emissions. The CO_2 emissions of the process (Stemmermann et al., 2023) are multiplied by current certificate prices (Table E.1). Costs for operating supervision, maintenance and repairs, operating supplies, and laboratory charges are calculated from operating labour costs and fixed-capital investment using percentages given by Peters et al. (2003). Finally, costs for catalysts and solvents include the $CaCl_2$ needed as mineraliser in the RC-BCC production and are calculated using the input mass (Figure E.1) and $CaCl_2$ costs (Table E.1).

The annuity is the central influencing aspect of fixed costs. It is determined by the annuity factor (Table E.1) multiplied by fixed-capital investment (Table E.3). The interest for working capital (Table E.3) is considered separately. The interest rate (Table E.1), not the annuity factor, is used for calculation because the working capital is not amortised. Additionally, taxes and insurance are considered as a percentage of the fixed-capital investment given by Peters et al. (2003). Similarly, overhead costs and all general expenses are calculated from fixed factors of a base value specified in Table E.4. The overhead costs include various aspects, for example, medical, safety and protection, packaging, and storage facilities. The general administrative expenses include executive salaries, legal costs, office maintenance, and communications. Adding up the variable, fixed, overhead, and general expenses leads to the total product costs. These amount to around 69 €/t input for the mechanical processing and an additional 920 €/t input for the RC-BCC production in the baseline scenario.

Table E.4: Product cost calculation for a pd-AAC recycling plant in the baseline scenario (50,000 t pd-AAC/a).

cost category	costs mechanical processing [€/t input pd-AAC]	costs RC-BCC production [€/t input pd-AAC]	reference
variable costs			
raw materials	0.00	55.35	own modelling
operating labour	17.51	32.59	own calculation (based on Peters et al., 2003)
operating supervision	2.63	4.89	0.15*operating labour (Peters et al., 2003)
electricity	1.78	507.15	own modelling
waste treatment and disposal	1.00	50.72	own modelling
maintenance and repairs	3.95	20.82	0.07*fixed-capital investment (Peters et al., 2003)
operating supplies	0.59	3.12	0.15*maintenance and repairs

			(Peters et al., 2003)
laboratory charges	2.63	4.89	0.15*operating labour (Peters et al., 2003)
catalysts and solvents	0.00	5.15	own modelling
total variable costs	30.10	684.69	
fixed costs			
annuity	6.20	32.65	0.11*fixed-capital investment (own calculation)
interest for working capital and land costs	1.59	4.87	0.07*(working capital+land costs) (own calculation)
taxes (property)	1.13	5.95	0.02*fixed-capital investment (Peters et al., 2003)
insurance	0.56	2.97	0.01*fixed-capital investment (Peters et al., 2003)
total fixed costs	9.48	46.44	
overhead costs	14.46	34.98	0.6*(operating labour + supervision + maintenance) (Peters et al., 2003)
general expenses			
administrative expenses	3.50	6.52	0.2*operating labour (Peters et al., 2003)
distribution and marketing expenses	7.53	101.18	0.11*total product costs (Peters et al., 2003)
research and development	3.42	45.99	0.05*total product costs (Peters et al., 2003)
total general expenses	14.46	153.69	
total costs	68.50	919.80	

The product costs are calculated similarly for all other scenarios. These scenarios include mechanical processing with a 10,000 t/a to 250,000 t/a capacity. The RC-BCC production capacities correspond to around 74% of the mechanical processing input (Figure E.1), leading to scenarios with 7,425 t/a to 185,625 t/a pd-AAC powder input. Figure E.2 shows all scenarios' total mechanical processing and RC-BCC production costs. The total product costs highly depend on the capacity of the recycling plant. The smallest plant of 10,000 t/a treats the pd-AAC with costs of more than 200 €/t input for mechanical processing and around 1250 €/t input for RC-BCC production. The larger the plant, the lower the total costs, reaching approximately 30 €/t input for mechanical processing and an additional 800 €/t input for RC-BCC production in the scenario with the largest capacity. Due to the limestone input, the RC-BCC production can produce 1.71 t RC-BCC per t pd-AAC input. Thus, the total costs of the combined mechanical processing and RC-BCC production would be 493 €/t RC-BCC in the scenario with the highest capacity. Detailed results for all scenarios are given in Supporting Information S1.

The variable costs account for the largest share of the total costs, just below 50% of the total costs for mechanical processing and more than 50% for the energy-intensive RC-BCC production, reaching nearly 80% in large-capacity scenarios. Overhead costs and general expenses contribute around 20% each to the total costs of the mechanical processing. In comparison, general expenses (just below 20%) are higher than overhead costs (10% for small capacities, < 5% for large capacities) for the RC-BCC production. Moreover, the fixed costs account for only about 10% (in scenarios with lower capacity) to 20% (in scenarios with higher capacity) of the total costs for mechanical processing. The fixed costs nearly equal the overhead costs for the RC-BCC production (10% for small capacities, < 5% for large capacities).

These total pd-AAC recycling costs can be compared to pd-AAC landfilling costs of approximately 100 €/t (Section E.2.1). The mechanical processing's total costs are nearly equal to the average landfilling costs for a recycling plant with 25,000 t/a input capacity. Total costs in higher-capacity scenarios are well below the landfilling costs. Thus, mechanical pd-AAC recycling is economically desirable, even without considering sale prices of around 5-15 €/t (Section E.2.1). However, the additional RC-BCC production is costly. Thus, total costs are significantly higher than landfilling costs. Cement sales prices of around 150 €/t (Section E.2.1) are not sufficient to reach an economic break-even. A pd-AAC recycling plant of 250,000 t/a input capacity would have total costs for mechanical processing and subsequent RC-BCC production of nearly 850 €/t pd-AAC powder (nearly 500 €/t RC-BCC). Thus, the RC-BCC would need to generate a sales price of around 430€/t so the recycling process's costs would not exceed landfilling costs.

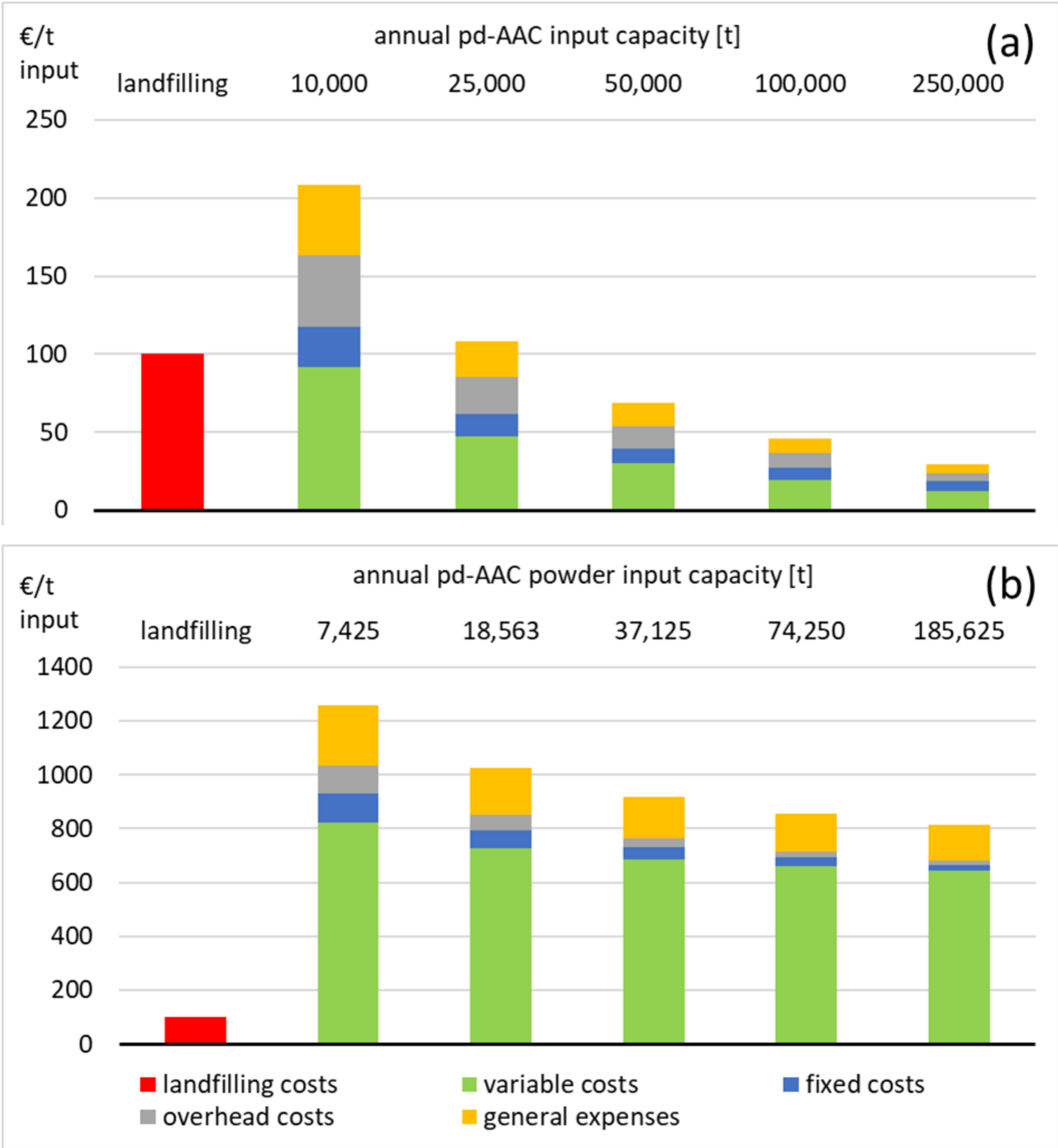

Figure E.2: Total costs of mechanical pd-AAC processing (a) and RC-BCC production (b) and their composition for all considered recycling plant capacities.

E.3.2 Sensitivity analysis

A sensitivity analysis was performed to determine the variation in total costs when input parameters change. The sensitivity analysis for both recycling steps includes the labour costs, equipment costs, electricity costs, interest rate, and recycling plant service life. Additionally, the limestone costs, $CaCl_2$ costs, and CO_2 certificate costs are varied for the RC-BCC production. All parameters are changed by ±10%, and the resulting changes in total costs are compared (Figure E.3).

The mechanical pd-AAC processing's total costs show the highest variability when changing labour costs (±6.7%) or equipment costs (±2.7%). In contrast, varying the electricity costs does not significantly influence the total costs (< 1% change) as the mechanical processing is not very energy-intensive. The interest rate and the service life of the recycling plant also only show a minor influence on the total costs (< 1% change).

In contrast, RC-BCC production is very energy-intensive, and electricity costs account for the largest share of the total costs. Therefore, varying the electricity costs leads to a significant change in total costs (±6.6%). All other parameters show much lower effects on the total costs of around ±1% for labour and equipment costs and even < 1% for interest rate, service life, limestone costs, $CaCl_2$ costs, and CO_2 certificate costs.

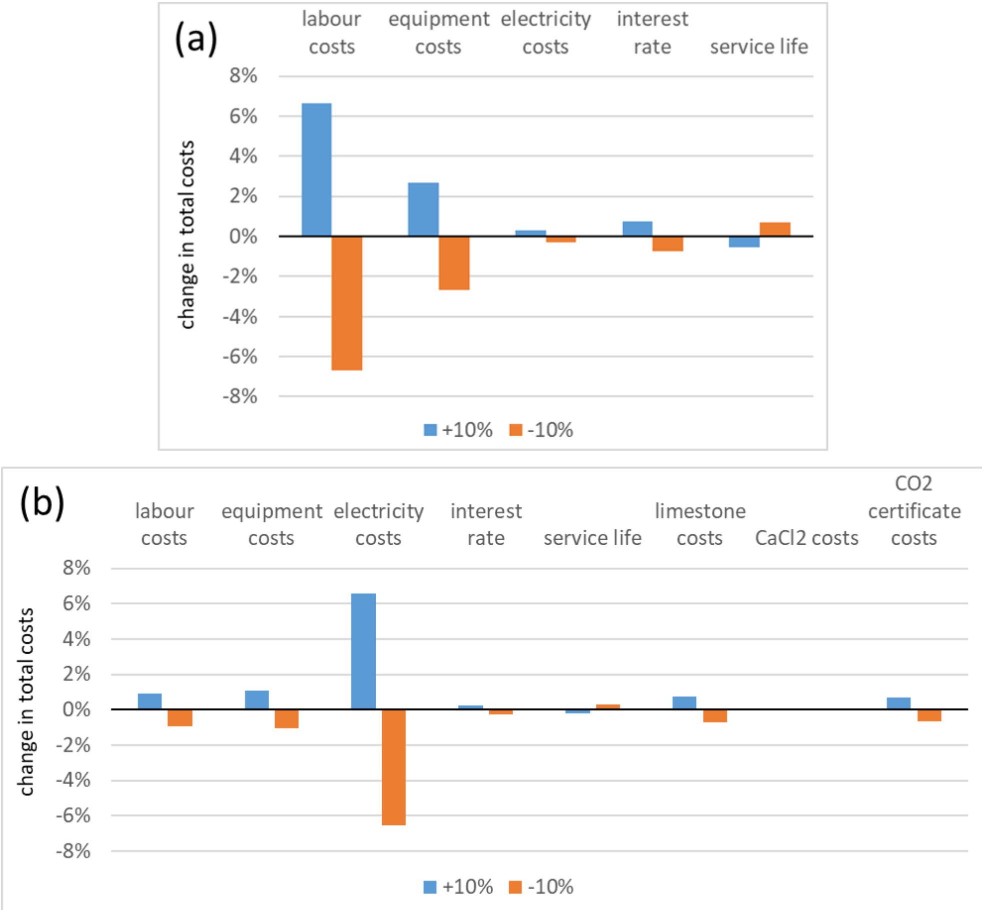

Figure E.3: Sensitivity analysis of total costs of (a) mechanical pd-AAC processing and (b) RC-BCC production.

E.4 Discussion

First, pd-AAC landfilling costs of assumed 100 €/t impact the overall comparison and conclusion of whether pd-AAC recycling is economically viable or not. However, the landfilling costs differ significantly between countries and regions. Therefore, landfilling pd-AAC could be economically more attractive than recycling in areas with low pd-AAC landfilling costs. On the other hand, pd-AAC recycling in small plants, potentially even including RC-BCC production, can be economically viable in regions with very high landfilling costs.

The equipment electricity demand was identified through product data sheets and calculations but not measured from an actual recycling plant. Generally, a conservative estimate was performed. The analysis does not consider, for example, the lower strength of AAC compared to other mineral building materials. Thus, in practice, the electricity demand will probably be well below the calculated values in this study, especially for crushing and milling steps. Krampitz et al. (2022) confirm this impression by giving specific jaw-crushing energy for AAC below the assumed electricity demand in this study. This observation is probably also valid for other processing steps. However, the dryer and the rotary kiln dominate the total electricity demand, while the electricity demand for mechanical processing is low.

Limitations of this study mainly include data availability and quality, as pd-AAC recycling is not yet established. Thus, pd-AAC recycling plants do not exist yet, and there is no field data on the different cost aspects. The study uses newly researched costs and electricity demands of recycling equipment and combines them with literature data to assess all relevant cost aspects. For example, the operating labour requirement strongly influences the variable costs and, thus, the total costs, especially for mechanical processing (Section E.3.2). However, the requirement is determined by an estimation based on Peters et al. (2003), not a measurement in recycling plants for pd-AAC or similar products.

Furthermore, the assumed processing steps and respective equipment must be tested on this scale in practical trials to determine their suitability for pd-AAC recycling and RC-BCC production. It has to be verified if the processes and equipment for input material purifying are sufficient to reach the desired final product quality. Moreover, it might influence the shares of pd-AAC granulate and powder produced. Furthermore, the impurities can vary substantially from the 1% assumed in this study. Higher percentages would increase the costs of waste treatment and disposal (part of the variable costs) and reduce potential sale revenues for the final product per ton input as a lower amount of the final product is produced. However, the influence of these aspects on the total costs is limited. One additional per cent of impurities in the input would increase the waste treatment and disposal costs by 1 €/t input. Overall, this study is the first approach to assess pd-AAC recycling economically. The results could be subject to noticeable changes when pd-AAC recycling is implemented in practice.

The existing knowledge about the technology used is also limited concerning the RC-BCC production. In particular, scaling the process to large input streams might lead to a change in technology. The limit of the currently proposed technology for the RC-BCC production, especially the electrically heated rotary kiln, has not yet been determined in practical trials. The rotary kiln might need to be fired by other technologies in large-capacity scenarios. Oxyfuel technology (natural gas combustion in pure oxygen) could be an option but would slightly increase the energy demand as additional energy is needed for oxygen generation. Overall, there is a need for further technology development.

Moreover, the clinker cooling as part of the RC-BCC production was not integrated into the cost calculation. As mentioned above, the exact cooling technology remains uncertain. An estimated 10% of additional equipment costs would emerge based on the cooler cost in an ordinary Portland cement plant (IEA Greenhouse Gas R&D Programme, 2008). However, the sensitivity analysis (Section E.3.2) shows that a 10% increase in equipment costs would increase the total product costs of the RC-BCC production by only around 1%.

E.5 Conclusion

This study modelled a pd-AAC recycling plant, including mechanical processing and RC-BCC production from pd-AAC for different input capacity scenarios to calculate total costs for pd-AAC recycling. Results show significant economies of scale for the recycling plant. Total costs for mechanical processing vary between 200 €/t input for the smallest plant of 10,000 t/a and 30 €/t input for the largest plant of 250,000 t/a. A subsequent RC-BCC production would incur additional costs between 800 €/t input (largest plant) and 1250 €/t input (smallest plant). Overall, production costs for RC-BCC are around 500 €/t, so minimum sales prices of about 430 €/t would be necessary to reach average pd-AAC landfilling costs. However, the recycling products from mechanical pd-AAC processing can also be used for different recycling purposes. The minimum capacity of the recycling plant needs to be around 25,000 t/a input for the total recycling costs to equal the average pd-AAC landfilling costs of 100 €/t. Plants with higher capacities could even mechanically treat the pd-AAC for costs well below the landfilling. In conclusion, this study indicates that pd-AAC recycling can be economically beneficial compared to landfilling and, thus, should be fostered.

Increasing pd-AAC volumes in the following decades will further extend the economic advantage of pd-AAC recycling as high recycling plants' capacities lead to significant reductions in total costs. Moreover, landfilling costs are most likely to increase significantly in future. However, the legal framework for pd-AAC recycling can still be improved. For example, regional modifications in the legislation complicate recycling.

Future research should enhance the data availability to assess pd-AAC recycling, for example, providing data from pilot plants to improve the quality of this economic assessment further. Additionally, location and logistics planning will be essential to advance pd-AAC recycling. Regional differences in pd-AAC volumes, demand, and landfilling fees can significantly influence the establishment of an AAC recycling network. Moreover, transport costs presumably impact the total recycling costs, making logistics planning vital.

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F Optimal Design of a Post-Demolition Autoclaved Aerated Concrete (AAC) Recycling Network Using a Capacitated, Multi-Period, and Multi-Stage Warehouse Location Problem

Abstract[1]

Autoclaved aerated concrete (AAC) is a popular building material in constructing one- and two-family houses because of its low thermal conductivity and fire resistance. Since AAC production rose significantly in the 1960s and 1970s, increasing post-demolition AAC volumes can be expected in the following decades. However, post-demolition AAC is currently landfilled as high-quality recycling options are still to be established.

This study develops a new capacitated, multi-period, and multi-stage network model for optimising a Germany AAC recycling network. The multi-period character of the model enables the precise consideration of increasing post-demolition AAC volumes by constantly allowing the move of recycling plants or opening new ones throughout the planning horizon. Additionally, the multi-stage formulation facilitates incorporating an optional second recycling step, which involves additional effort and higher revenues. The model aims to find a cost-minimised recycling network and identify optimal network transformations until 2050. Results show that recycling is preferred over landfilling. The optimised recycling network uses large recycling plants for economies of scale and opens new plants in the future to handle the expected increase in post-demolition AAC. Transport costs account for the largest share of total costs (50%), while fixed costs reach around 40%, and revenues offset approximately 20% of all costs. The total costs of the network reach about 2,200 M€ until 2050, which is 4,600 M€ (68%) less than without establishing recycling. The results offer new insights into cost-minimal network structures and their future development to encourage decision-makers to promote AAC recycling.

1 This section includes the article "Optimal design of a post-demolition autoclaved aerated concrete (AAC) recycling network using a capacitated, multi-period, and multi-stage warehouse location problem", developed by Manuel Ruck, Rebekka Volk, Frank Schultmann, and myself. The article has been submitted for publication in a scientific journal as Steins et al. (2023).

F.1 Introduction

Recycling of construction and demolition wastes (C&DW) enjoys increasing popularity as vast amounts of primary resources can be saved (Akhtar & Sarmah, 2018). Comprehensive recycling can support reaching the UN sustainable development goals of "sustainable cities", "responsible consumption and production", and "climate action" (UN, 2023). Moreover, legal requirements for C&DW are tightening, and landfill capacities are decreasing. Thus, there are recycling or downcycling options for most building materials today. In Germany, 79% (95% if all recovery options are included) of C&DW waste is recycled (Kreislaufwirtschaft Bau, 2023). This rate satisfies the 70% recycling rate requirement by the European waste and recycling regulation (Directive 2008/98/EC, 2008). However, among other waste fractions, autoclaved aerated concrete (AAC) is still disposed of nowadays.

AAC is a popular building material produced from quartz sand, cement, quicklime, anhydrite/gypsum, aluminium powder/paste, and water (DIN 20000-404:2018-04; Kreft, 2017). A chemical reaction started by the aluminium produces hydrogen gas that causes the material's porous structure, leading to AAC's very low density. Therefore, AAC shows excellent thermal insulation properties without needing additional insulation. This monolithic construction has different advantages, including a fast and low-cost construction process and a high fire resistance. The current AAC production in Europe amounts to more than 16 million m³ annually (EAACA, 2023), of which 3.5 million m³ are produced in Germany (GENESIS, 2023b). AAC's global production capacity is expected to be around 450 million m³ (Fouad & Schoch, 2018). Post-demolition AAC (pd-AAC) volumes are estimated to be approximately 1.4 million m³ in Germany in 2022, while they are expected to reach more than 4 million m³ by 2050 (Steins et al., 2021). Thus, the OECD (2020) statement "the potential of the circular economy to support sustainable cities, regions, and countries still needs to be unlocked" is especially true for AAC. Today, the main reasons hindering AAC recycling are the low compressive strength compared to other mineral building materials and the small sulphate contents in AAC from the anhydrite/gypsum. Thus, the most widespread recycling options for mineral building materials (road construction, earthworks, and aggregate in concrete production (Knappe et al., 2012)) are not practicable for pd-AAC. Furthermore, reusing AAC masonry blocks is impractical as older AAC does not comply with current standards, and the deconstruction process would have to be overly careful and, thus, expensive (Gyurkó et al., 2019).

However, new recycling options for pd-AAC are investigated: Open-loop recycling options that are examined include the production of lightweight aggregate concrete (Aycil et al., 2016; Gyurkó et al., 2019), light mortar (Aycil et al., 2016), floor screed (Bergmans et al., 2016), and shuttering blocks from no-fines concrete (Gyurkó et al., 2019) made with pd-AAC input. These open-loop recycling options have in common that other products than AAC are produced. In contrast, in closed-loop recycling, pd-AAC is used to produce new AAC. This approach is also studied in the literature. On the one hand, fine pd-AAC powder can be used in AAC production to substitute sand, cement, quicklime, and anhydrite proportional to their input amount (Kreft, 2017; Lam, 2021; Rafiza et al., 2019; Rafiza et al., 2022). On the other hand, the closed-loop can be reached through a second recycling step where recycled belite cement clinker (RC-BCC) is produced (Stemmermann et al., 2023; Ullrich et al., 2021). RC-BCC can partly substitute Portland cement in the AAC production to reach a closed-loop. Additionally, it has been shown that AAC recycling can be superior to landfilling regarding environmental (Volk et al., 2023; Volk et al., 2022) and economic aspects (Steins, Volk, Beuchle, et al., 2023) for many recycling options. However, the design and optimisation of a pd-AAC recycling network for the following decades is missing in the literature[2] and will be the focus of this study.

There are numerous studies on (recycling) network models available. Reverse logistic networks with multi-period consideration and possible capacity adjustments are studied by Alumur et al. (2012), Jahangiri et al. (2022), Pan et al. (2020), Rahimi and Ghezavati (2018), and Rosenberg et al. (2023). Multi-stage networks are also investigated by Figueiredo and Mayerle (2008), Jahangiri et al. (2022), Mansour and Zarei (2008), and Tuzkaya et al. (2011), while multi-product formulations are given by Ene and Öztürk (2015), Gomes et al. (2011), Listeş and Dekker (2005), and Pati et al. (2008). Reverse logistic networks that encompass stochastic factors are investigated by Lieckens and Vandaele (2007), Listeş and Dekker (2005), Roghanian and Pazhoheshfar (2014), Ene and Öztürk (2015), and Trochu et al. (2020). Moreover, some studies focus on modelling the recycling of (mineral) construction waste (Barros et al., 1998; Listeş & Dekker, 2005; Rahimi & Ghezavati, 2018; Trochu et al., 2020).

However, none of these studies includes a capacitated, multi-stage, multi-period approach considering different products and investigating scenarios for uncertainties, which is required for pd-AAC recycling network modelling. Thus, this paper aims to develop a new recycling network model that extensively extends the warehouse location problem (WLP) to consider the specific characteristics of pd-AAC recycling. These are (1) the dynamics of significantly increasing supplies, (2) different recycling plant capacities with economies of scale, and (3) two recycling stages, with the second one being optional.

The model's multi-period approach allows it to react precisely to increasing volumes. It is possible to open, close, expand or relocate plants at anytime. Moreover, the model considers the capacity limitations of recycling plants and economies of scale in larger plants. As the trade-off between economies of scale and transport costs will be a central element of the optimisation, exact modelling of realistic transport distances and product-dependent transport costs is ensured. The optional second recycling step produces a new, higher-quality product from the pd-AAC. This production needs additional effort but can also achieve a higher revenue. Deciding whether or not to use this optional second recycling step is another critical element of the optimisation. Therefore, the model includes this characteristic using a multi-stage formulation with independent locations at the second recycling stage and precisely calculated recycling costs.

[2] So far, the authors have only examined a rough estimate of an optimal European network design with substantial simplifications compared to this study (Steins, Volk, and Schultmann (2023).

An optimal solution is to be calculated for this model to answer the following research questions: How does a cost-optimal recycling network for pd-AAC look like, and how does it need to be adapted in the future when waste volumes increase significantly?

This study is structured as follows. First, Section F.2 describes the methods, including the mathematical formulation of a capacitated, multi-period, and multi-stage pd-AAC network model, as well as the input data. Section F.3 presents the results of the model optimisation and their interpretation. Section F.4 includes the discussion and states limitations. Finally, a conclusion is drawn in Section F.5.

F.2 Methods

F.2.1 Mathematical formulation

The newly developed capacitated, multi-stage, and multi-period model (Section F.1) is used for pd-AAC recycling network modelling and optimisation. Future expected pd-AAC amounts, given by parameter S, are assumed to emerge at different supply locations specified in the set $\mathcal{S}$. These amounts can be transported to recycling plants or landfilled. Variable x determines the recycling network product flows, while variable y indicates opened recycling plants, and variable z specifies the time when a recycling plant is opened. Pd-AAC amounts can be landfilled (variable l), but landfilling is limited to a maximum share (L) of the supply to reach a specified recycling rate.

The pd-AAC recycling process consists of a mandatory first step that includes crushing and purifying to produce pd-AAC powder and pd-AAC granulate and an optional second step to produce RC-BCC. Efficiencies of the two recycling steps are reflected by the parameter E. The model's set $\mathcal{M}$ contains all the commodities. Both recycling steps do not necessarily have to be executed at the same location. Thus, possible recycling plant locations are given separately for the first (set $\mathcal{R}_1$) and the second recycling step (set $\mathcal{R}_2$).

The recycling plants can differ in input capacity, influencing recycling costs. The sets $\mathcal{K}_1$ and $\mathcal{K}_2$ contain all possible recycling plant capacity levels (for indexing: 0, 1, 2, …) for both recycling steps, while parameter K specifies concrete input capacities of the different levels. Finally, the recycling end products are delivered to demand locations (set $\mathcal{D}$). These are limited to a maximum demand D. Also, the model includes different time periods specified by set $\mathcal{T}$, as pd-AAC amounts are expected to rise in the future, and the optimal network design could, thus, significantly change over time.

The overall objective is to minimise the total costs of the pd-AAC recycling network. Costs are divided into the categories variable recycling costs (C^{var}), fixed recycling plant costs (C^{fixed}), recycling plant opening costs (C^{open}), landfilling costs ($C^{landfilling}$), and transport costs ($C^{transport}$). Including opening and fixed costs enables realistic modelling of a recycling plant's cost structure. The parameter for the transport costs only gives costs per distance. So, the actual distance of the transport ($Dist$) has to be specified. Additionally, revenues for the final products of the recycling process are included ($C^{revenue}$).

The model reflects a capacitated, multi-period, and multi-stage recycling network. Further characteristics of the model include determinism (no stochastics for supply amounts or costs are considered), multi-sourcing (different regions can supply a recycling plant), and direct delivery without interactions between plants of the same stage. The model's sets (calligraphic upper-case characters), decision variables (lower-case characters), and parameters (upper-case characters) are presented in Table F.1, along with all indices to precisely determine the values for all locations, capacities, commodities, and periods. The cost-minimising model is formulated in Equations F.1 to F.16. The model can be classified as a mixed-integer problem.

Table F.1: Sets, decision variables, and parameters used for the pd-AAC recycling network modelling.

Sets	
$\mathcal{D}$	set of demand locations
$\mathcal{K}_1$	set of possible recycling plant capacity levels (first recycling step)
$\mathcal{K}_2$	set of possible recycling plant capacity levels (second recycling step)
$\mathcal{M}$	set of commodities
$\mathcal{R}_1$	set of possible recycling plant locations (first recycling step)
$\mathcal{R}_2$	set of possible recycling plant locations (second recycling step)
$\mathcal{S}$	set of supply locations
$\mathcal{T} = \{0, \dots, T\}$	set of time periods
Decision variables	
l_{smt}	quantity of commodity $m \in \mathcal{M}$ landfilled at supply location $s \in \mathcal{S}$ in time period $t \in \mathcal{T}$
x_{sr_1mt}	quantity of commodity $m \in \mathcal{M}$ transported from supply location $s \in \mathcal{S}$ to recycling plant $r_1 \in \mathcal{R}_1$ in time period $t \in \mathcal{T}$
$x_{r_1r_2mt}$	quantity of commodity $m \in \mathcal{M}$ transported from recycling plant $r_1 \in \mathcal{R}_1$ to recycling plant $r_2 \in \mathcal{R}_2$ in time period $t \in \mathcal{T}$
x_{r_1dmt}	quantity of commodity $m \in \mathcal{M}$ transported from recycling plant $r_1 \in \mathcal{R}_1$ to demand location $d \in \mathcal{D}$ in time period $t \in \mathcal{T}$
x_{r_2dmt}	quantity of commodity $m \in \mathcal{M}$ transported from recycling plant $r_2 \in \mathcal{R}_2$ to demand location $d \in \mathcal{D}$ in time period $t \in \mathcal{T}$
$y_{k_1r_1t}$	indicator variable for the status of a recycling plant of capacity level $k_1 \in \mathcal{K}_1$ at location $r_1 \in \mathcal{R}_1$ in time period $t \in \mathcal{T}$
$y_{k_2r_2t}$	indicator variable for the status of a recycling plant of capacity level $k_2 \in \mathcal{K}_2$ at location $r_2 \in \mathcal{R}_2$ in time period $t \in \mathcal{T}$
$z_{k_1r_1t}$	indicator variable for the opening of a recycling plant of capacity level $k_1 \in \mathcal{K}_1$ at location $r_1 \in \mathcal{R}_1$ in time period $t \in \mathcal{T}$
$z_{k_2r_2t}$	indicator variable for the opening of a recycling plant of capacity level $k_2 \in \mathcal{K}_2$ at location $r_2 \in \mathcal{R}_2$ in time period $t \in \mathcal{T}$
Parameters	

$C_{sr_1mt}^{transport}$	transport costs of commodity $m \epsilon \mathcal{M}$ from supply location $s \epsilon \mathcal{S}$ to recycling location $r_1 \epsilon \mathcal{R}_1$ in time period $t \epsilon \mathcal{T}$
$C_{r_1r_2mt}^{transport}$	transport costs of commodity $m \epsilon \mathcal{M}$ from recycling location $r_1 \epsilon \mathcal{R}_1$ to recycling location $r_2 \epsilon \mathcal{R}_2$ in time period $t \epsilon \mathcal{T}$
$C_{r_1dmt}^{transport}$	transport costs of commodity $m \epsilon \mathcal{M}$ from recycling location $r_1 \epsilon \mathcal{R}_1$ to demand location $d \epsilon \mathcal{D}$ in time period $t \epsilon \mathcal{T}$
$C_{r_2dmt}^{transport}$	transport costs of commodity $m \epsilon \mathcal{M}$ from recycling location $r_2 \epsilon \mathcal{R}_2$ to demand location $d \epsilon \mathcal{D}$ in time period $t \epsilon \mathcal{T}$
$C_{mt}^{var,first\ step}$	variable recycling costs of commodity $m \epsilon \mathcal{M}$ in time period $t \epsilon \mathcal{T}$ in the first recycling step
$C_{k_1t}^{fixed,first\ step}$	fixed costs for operating a recycling plant of capacity level $k_1 \epsilon \mathcal{K}_1$ in time period $t \epsilon \mathcal{T}$ in the first recycling step
$C_{k_1r_1t}^{open,first\ step}$	opening costs for a recycling plant of capacity level $k_1 \epsilon \mathcal{K}_1$ at recycling location $r_1 \epsilon \mathcal{R}_1$ in time period $t \epsilon \mathcal{T}$ in the first recycling step
$C_{mt}^{var,second\ step}$	variable recycling costs of commodity $m \epsilon \mathcal{M}$ in time period $t \epsilon \mathcal{T}$ in the second recycling step
$C_{k_2t}^{fixed,second\ step}$	fixed costs for operating a recycling plant of capacity level $k_2 \epsilon \mathcal{K}_2$ in time period $t \epsilon \mathcal{T}$ in the second recycling step
$C_{k_2r_2t}^{open,second\ step}$	opening costs for a recycling plant of capacity level $k_2 \epsilon \mathcal{K}_2$ at recycling location $r_2 \epsilon \mathcal{R}_2$ in time period $t \epsilon \mathcal{T}$ in the second recycling step
$C_{smt}^{landfilling}$	landfilling costs of commodity $m \epsilon \mathcal{M}$ at supply location $s \epsilon \mathcal{S}$ in time period $t \epsilon \mathcal{T}$
$C_{dmt}^{revenue}$	revenue for selling the commodity $m \epsilon \mathcal{M}$ at demand location $d \epsilon \mathcal{D}$ in time period $t \epsilon \mathcal{T}$
D_{dmt}	demand of commodity $m \epsilon \mathcal{M}$ at demand location $d \epsilon \mathcal{D}$ in time period $t \epsilon \mathcal{T}$
$Dist_{sr_1}$	distance between supply location $s \epsilon \mathcal{S}$ and recycling plant $r_1 \epsilon \mathcal{R}_1$
$Dist_{r_1r_2}$	distance between recycling plant $r_1 \epsilon \mathcal{R}_1$ and recycling plant $r_2 \epsilon \mathcal{R}_2$
$Dist_{r_1d}$	distance between recycling plant $r_1 \epsilon \mathcal{R}_1$ and demand location $d \epsilon \mathcal{D}$
$Dist_{r_2d}$	distance between recycling plant $r_2 \epsilon \mathcal{R}_2$ and demand location $d \epsilon \mathcal{D}$
E_{nmt}	efficiency of the production of commodity $m \epsilon \mathcal{M}$ from commodity $n \epsilon \mathcal{M}$ in time period $t \epsilon \mathcal{T}$
K_{k_1mt}	recycling plant input capacity for commodity $m \epsilon \mathcal{M}$ at capacity level $k_1 \epsilon \mathcal{K}_1$ in time period $t \epsilon \mathcal{T}$
K_{k_2mt}	recycling plant input capacity for commodity $m \epsilon \mathcal{M}$ at capacity level $k_2 \epsilon \mathcal{K}_2$ in time period $t \epsilon \mathcal{T}$
L_{mt}	maximum share of the supply allowed to be landfilled for commodity $m \epsilon \mathcal{M}$ in time period $t \epsilon \mathcal{T}$
S_{smt}	supply of commodity $m \epsilon \mathcal{M}$ at supply location $s \epsilon \mathcal{S}$ in time period $t \epsilon \mathcal{T}$

$$
\begin{aligned}
min \quad & \sum_{t \in \mathcal{T}} \Bigg[\sum_{m \in \mathcal{M}} \Bigg(\sum_{s \in \mathcal{S}} \sum_{r_1 \in \mathcal{R}_1} x_{sr_1mt} \cdot Dist_{sr_1} \cdot C_{sr_1mt}^{transport} \\
& \qquad + \sum_{r_1 \in \mathcal{R}_1} \sum_{r_2 \in \mathcal{R}_2} x_{r_1r_2mt} \cdot Dist_{r_1r_2} \cdot C_{r_1r_2mt}^{transport} \\
& \qquad + \sum_{r_1 \in \mathcal{R}_1} \sum_{d \in \mathcal{D}} x_{r_1dmt} \cdot Dist_{r_1d} \cdot C_{r_1dmt}^{transport} \\
& \qquad + \sum_{r_2 \in \mathcal{R}_2} \sum_{d \in \mathcal{D}} x_{r_2dmt} \cdot Dist_{r_2d} \cdot C_{r_2dmt}^{transport} \\
& \qquad + \sum_{s \in \mathcal{S}} \sum_{r_1 \in \mathcal{R}_1} C_{mt}^{var, first\ step} \cdot x_{sr_1mt} \\
& \qquad + \sum_{r_1 \in \mathcal{R}_1} \sum_{r_2 \in \mathcal{R}_2} C_{mt}^{var, second\ step} \cdot x_{r_1r_2mt} \Bigg) \\
& \qquad + \sum_{r_1 \in \mathcal{R}_1} \sum_{k_1 \in \mathcal{K}_1} C_{k_1t}^{fixed, first\ step} \cdot y_{k_1r_1t} \\
& \qquad + \sum_{r_2 \in \mathcal{R}_2} \sum_{k_2 \in \mathcal{K}_2} C_{k_2t}^{fixed, second\ step} \cdot y_{k_2r_2t} \\
& \qquad + \sum_{r_1 \in \mathcal{R}_1} \sum_{k_1 \in \mathcal{K}_1} C_{k_1r_1t}^{open, first\ step} \cdot z_{k_1r_1t} \\
& \qquad + \sum_{r_2 \in \mathcal{R}_2} \sum_{k_2 \in \mathcal{K}_2} C_{k_2r_2t}^{open, second\ step} \cdot z_{k_2r_2t} \\
& \qquad + \sum_{s \in \mathcal{S}} \sum_{m \in \mathcal{M}} l_{smt} \cdot C_{smt}^{landfilling} \\
& \qquad - \sum_{m \in \mathcal{M}} \Bigg(\sum_{r_1 \in \mathcal{R}_1} \sum_{d \in \mathcal{D}} x_{r_1dmt} \cdot C_{dmt}^{revenue} \\
& \qquad + \sum_{r_2 \in \mathcal{R}_2} \sum_{d \in \mathcal{D}} x_{r_2dmt} \cdot C_{dmt}^{revnue} \Bigg) \Bigg]
\end{aligned}
\tag{F.1}
$$

$$
s.t. \quad \sum_{r_1 \in \mathcal{R}_1} x_{sr_1mt} + l_{smt} = S_{smt} \qquad \forall s \in \mathcal{S}, \forall m \in \mathcal{M}, \forall t \in \mathcal{T} \tag{F.2}
$$

$$
\sum_{s \in \mathcal{S}} l_{smt} \leq L_{mt} \cdot \sum_{s \in \mathcal{S}} S_{smt} \qquad \forall m \in \mathcal{M}, \forall t \in \mathcal{T} \tag{F.3}
$$

$$
\sum_{r_2 \in \mathcal{R}_2} x_{r_1r_2mt} + \sum_{d \in \mathcal{D}} x_{r_1dmt} = \sum_{n \in \mathcal{M}} \left(E_{nmt} \sum_{s \in \mathcal{S}} x_{sr_1nt} \right) \qquad \forall r_1 \in \mathcal{R}_1, \forall m \in \mathcal{M}, \forall t \in \mathcal{T} \tag{F.4}
$$

$$
\sum_{d \in \mathcal{D}} x_{r_2dmt} = \sum_{n \in M} \left(E_{nmt} \sum_{r_1 \in \mathcal{R}_1} x_{r_1r_2nt} \right) \qquad \forall r_2 \in \mathcal{R}_2, \forall m \in \mathcal{M}, \forall t \in \mathcal{T} \tag{F.5}
$$

$$
\sum_{r_1 \in \mathcal{R}_1} x_{r_1dmt} + \sum_{r_2 \in \mathcal{R}_2} x_{r_2dmt} \leq D_{dmt} \qquad \forall d \in \mathcal{D}, \forall m \in \mathcal{M}, \forall t \in \mathcal{T} \tag{F.6}
$$

$$
\sum_{s \in \mathcal{S}} x_{sr_1mt} \leq \sum_{k_1 \in \mathcal{K}_1} K_{k_1mt} \cdot y_{k_1r_1t} \qquad \forall r_1 \in \mathcal{R}_1, \forall m \in \mathcal{M}, \forall t \in \mathcal{T} \tag{F.7}
$$

$$\sum_{r_1 \in \mathcal{R}_1} x_{r_1 r_2 mt} \leq \sum_{k_2 \in \mathcal{K}_2} K_{k_2 mt} \cdot y_{k_2 r_2 t} \qquad \forall r_2 \in \mathcal{R}_2, \forall m \in \mathcal{M}, \forall t \in \mathcal{T} \qquad \text{(F.8)}$$

$$\sum_{k_1 \in \mathcal{K}_1} y_{k_1 r_1 t} \leq 1 \qquad \forall r_1 \in \mathcal{R}_1, \forall t \in \mathcal{T} \qquad \text{(F.9)}$$

$$\sum_{k_2 \in \mathcal{K}_2} y_{k_2 r_2 t} \leq 1 \qquad \forall r_2 \in \mathcal{R}_2, \forall t \in \mathcal{T} \qquad \text{(F.10)}$$

$$y_{k_1 r_1 t} - y_{k_1 r_1 (t-1)} \leq z_{k_1 r_1 t} \qquad \forall k_1 \in \mathcal{K}_1, \forall r_1 \in \mathcal{R}_1, \forall t \in \mathcal{T} \backslash \{0\} \qquad \text{(F.11)}$$

$$y_{k_2 r_2 t} - y_{k_2 r_2 (t-1)} \leq z_{k_2 r_2 t} \qquad \forall k_2 \in \mathcal{K}_2, \forall r_2 \in \mathcal{R}_2, \forall t \in \mathcal{T} \backslash \{0\} \qquad \text{(F.12)}$$

$$y_{k_1 r_1 0} = z_{k_1 r_1 0} \qquad \forall k_1 \in \mathcal{K}_1, \forall r_1 \in \mathcal{R}_1 \qquad \text{(F.13)}$$

$$y_{k_2 r_2 0} = z_{k_2 r_2 0} \qquad \forall k_2 \in \mathcal{K}_2, \forall r_2 \in \mathcal{R}_2 \qquad \text{(F.14)}$$

$$x_{s r_1 mt}, x_{r_1 r_2 mt}, x_{r_1 dmt}, x_{r_2 dmt}, l_{smt} \geq 0 \qquad \forall s \in \mathcal{S}, \forall r_1 \in \mathcal{R}_1, \forall r_2 \in \mathcal{R}_2, \forall d \in \mathcal{D}, \forall m \in \mathcal{M}, \forall t \in \mathcal{T} \qquad \text{(F.15)}$$

$$y_{k_1 r_1 t}, y_{k_2 r_2 t}, z_{k_1 r_1 t}, z_{k_2 r_2 t} \in \{0,1\} \qquad \forall r_1 \in \mathcal{R}_1, \forall r_2 \in \mathcal{R}_2, \forall k_1 \in \mathcal{K}_1, \forall k_2 \in \mathcal{K}_2, \forall t \in \mathcal{T} \qquad \text{(F.16)}$$

The model's objective function (Equation F.1) is set to minimise total costs consisting of transport, variable recycling, fixed recycling, opening, and landfilling costs. Revenues for the final products are subtracted from these costs. The model has fifteen constraints specified by Equations F.2-F.16. Equation F.2 ensures that all supply has to be treated in the same period as it emerges. It is either transported to a recycling location or landfilled. However, the total amount that can be landfilled is restricted in Equation F.3 to a specified share of the total quantity supplied. This restriction reflects legal regulations or self-imposed recycling goals. Equation F.4 defines the flow conservation constraint for the first recycling step. The amount transported to a recycling location and treated there must be further transported to the second recycling step or demand locations in the same period. Storage between different periods is not allowed in the model. Moreover, the efficiency of the recycling process is considered. Similarly, the flow conservation of the second recycling step is described in Equation F.5. Equation F.6 ensures that the amounts transported to the demand locations do not exceed the demands for the different commodities in every period. Undersupply of the demand is allowed in the model as the focus is on the waste to be recycled instead of the entire demand to be met. Equations F.7 and F.8 determine the capacity limitations of the recycling plants, including the fact that material can only be transported to opened recycling locations. Moreover, Equations F.9 and F.10 specify for both recycling steps that at most one recycling plant can be opened in one region at the same time. Equations F.11 and F.12 ensure that opening costs are considered when a recycling plant is opened. The first period is handled separately in Equations F.13 and F.14 as negative period indices are not defined ($t - 1 = -1$ for the first period). Finally, Equation F.15 defines the non-negativity of the decision variables for transport and landfilling. Moreover, Equation F.16 ensures that recycling plant opening and recycling plant status are binary.

F.2.2 Use case and input data

SI-1 in the supplementary information discloses all input data, including information that could not be described entirely in this section, for example, all elements of the considered sets and values of comprehensive parameters like the supply.

The use case investigated in this study focuses on pd-AAC in Germany since it is a crucial ACC market with all the required data available. The geographic subdivision uses the NUTS[3] 2 level, where 38 German regions are considered. Every region resembles a supply location ($\mathcal{S}$) and is a possible recycling plant location for both recycling steps ($\mathcal{R}_1$ and $\mathcal{R}_2$). The demand locations ($\mathcal{D}$) include all AAC plants in Germany where pd-AAC powder or RC-BCC can be used in a closed-loop recycling process. Overall, 31 AAC plants in Germany are identified in our own research. Other recycling options described in the literature do not necessarily depend on production plants. Instead, the pd-AAC recycling products are needed at decentralised locations, for example, directly at the construction site. The recycling plants can be constructed in five different capacity levels at both recycling stages ($\mathcal{K}_1$ and $\mathcal{K}_2$). Commodities ($\mathcal{M}$) include pd-AAC as the initial product, pd-AAC powder and pd-AAC granulate as the outputs of the first recycling step, and RC-BCC as the output of the second recycling step. Finally, all years from 2023 to 2050 are considered, but several years are merged into one period to reduce the number of decision variables and the associated complexity. The further the period is in the future, the more years are combined. Thus, the following eleven periods ($\mathcal{T}$) are considered: 2023, 2024, 2025, 2026/2027, 2028/2029, 2030/2031, 2032-2034, 2035-2037, 2038-2040, 2041-2045, 2046-2050.

The cost parameters are determined as follows. Variable costs (C^{var}) for both recycling steps are disclosed in Table F.2. They include costs which directly depend on the treated amount but are independent of the capacity of the recycling plant (raw materials, electricity, waste treatment/disposal, operating supplies, catalysts, CO_2 certificates). Thus, a multiplication of decision variables in the mathematical model's objective function is avoided. All capacity-depending costs are considered in the fixed costs or opening costs.

Fixed costs (C^{fixed}) include operating labour, operating supervision, maintenance and repairs, laboratory charges, interest for working capital, taxes, insurance, overhead costs, and general expenses. Precise costs for all capacity levels and both recycling steps are given in Table F.2.

Opening costs (C^{open}) for both recycling steps and all capacity levels reflect the fixed-capital investment of a recycling plant. Additionally, land costs have to be included. These depend on the recycling plant location and, therefore, are calculated separately based on regional prices per m² from Destatis (2021) and AK OGA (2021). Opening costs and the required land for all capacity levels and both recycling steps are given in Table F.2, while regionally differing land costs are listed in SI-1.

[3] Nomenclature des unités territoriales statistiques is a system for regional division of countries.

Table F.2: Capacities, variable costs, fixed costs, opening costs, and required area for all capacity levels and both recycling steps (based on Steins, Volk, Beuchle, et al., 2023).

First recycling step					
capacity level	1	2	3	4	5
capacity [t input/a]	10,000	25,000	50,000	100,000	250,000
variable costs [€/t input]	3.38	3.38	3.38	3.38	3.38
fixed costs [€/a]	1,843,912	2,333,815	2,901,608	3,720,276	5,511,086
opening costs [€]	1,609,503	2,185,915	2,823,760	4,607,165	9,678,336
required area [m²]	2,349	5,359	10,000	18,661	42,567
Second recycling step					
capacity level	1	2	3	4	5
capacity [t input/a]	7,425	18,563	37,125	74,250	185,625
variable costs [€/t input]	621.51	621.51	621.51	621.51	621.51
fixed costs [€/a]	17,616,962	27,560,537	38,883,341	57,007,255	100,901,908
opening costs [€]	5,133,004	7,982,309	11,041,251	15,691,789	25,836,904
required area [m²]	2,349	5,359	10,000	18,661	42,567

Transport costs ($C^{transport}$) are based on Persyn et al. (2022). They identified the costs for a 40 t truck on routes between all possible combinations of NUTS 2 regions in Germany. Thus, route-specific transport costs can be used for the optimisation. These costs vary between 1.37 €/km for long distances (up to 1000 km) and more than 3 €/km for very short transports (less than 30 km). The costs are adjusted by the rise in transport prices between January 2020 and July 2023 (BGL, 2023). However, the truck's maximum payload depends on the transported commodity, leading to different costs per ton and kilometre. Pd-AAC powder and granulate, as well as belite cement clinker, can be transported in an articulated truck with an assumed maximum payload of 25 t and a maximum volume of 90 m³. Bulk densities of pd-AAC powder (0.6 t/m³; Gyurkó et al., 2019) and cement clinker (> 0.9 t/m³; VDZ, 2017) are high enough to use the whole 25 t payload of the truck. Pd-AAC granulate has a bulk density of 0.255 t/m³ (Gyurkó et al., 2019) and, thus, only reaches a payload of 22.95 t when the maximum volume of 90 m³ is used. Pd-AAC is assumed to be transported in a tipper truck with 25 t maximum payload and 60 m³ maximum volume. The bulk density of pd-AAC is supposed to be around 0.2 t/m³ (Gyurkó et al., 2019)[4]. This low density leads to a maximum of 12 t pd-AAC payload within the given 60 m³ truck volume and to comparably high costs.

Landfilling costs ($C^{landfilling}$) vary significantly between regions, ranging from 65 to 180 €/t pd-AAC (Aycil & Hlawatsch, 2020). Research in the online portals clearago.de and abfall-

[4] Gyurkó et al. (2019) only specify bulk densities for grain sizes up to 8-16 mm which is less than usual pd-AAC from the demolition site. However, the assumption of 0.2 t/m³ bulk density for pd-AAC is used as the density rises with grain size and shows convergence to around 0.2 t/m³.

scout.de discloses regional pd-AAC landfilling costs for nearly every postal code area in Germany. Landfilling costs per NUTS 2 region are calculated as an average of all included postal code areas within each NUTS 2 region (SI-1). It is assumed that transport costs to the landfills are already included in the landfilling fees.

The revenues ($C^{revenue}$) are based on market prices for similar products since pd-AAC recycling products are not established yet. The pd-AAC powder and pd-AAC granulate are assumed to generate 10 €/t revenues as other mineral recycling products have market prices of 5-15 €/t in Germany (initial interactive gmbh, 2023). The RC-BCC can substitute Portland cement in some applications. Thus, average German cement prices of 150 €/t (cemex, 2022; Dyckerhoff, 2022) are assumed as revenue.

Besides the costs, further parameters include distances ($Dist$), demand (D), efficiencies (E), capacities (K), maximum landfill share (L), and supply (S). Persyn et al. (2022) provide average road distances for transports within a NUTS 2 region and between every pair of German NUTS 2 regions. These distances are relevant for all transports from supply to recycling locations and between the recycling stages. Only transports to the demand are calculated separately as demand arises at decentralised locations (construction sites) or concrete AAC plants. A lump-sum distance of 100 km is assumed for transports of pd-AAC powder/granulate to decentralised demand locations. Moreover, distances from recycling plants to AAC plants are calculated using the geodesic distance multiplied by a factor of 1.33 to reflect the average proportion of road distance to geodesic distance in Germany (Persyn et al., 2022).

The demand of the AAC plants is calculated by multiplying the total AAC production of the respective plant with a substitution percentage of primary raw materials with pd-AAC powder or RC-BCC, respectively. All 31 German AAC plants are assumed to produce 50,000 t AAC/a due to missing specific information. Currently, pd-AAC powder can substitute 7-10% of the overall input depending on the AAC type (Volk et al., 2023), while RC-BCC can substitute a maximum of 50% of the cement input, i.e. 8-16% of the overall input depending on the AAC type (Stemmermann et al., 2023). Thus, the pd-AAC powder demand per plant is assumed to be 4,000 t/a (8%), while RC-BCC demand is 6,000 t/a (12%). Furthermore, a decentralised demand at construction sites is assumed for pd-AAC powder and granulate that is used in other recycling options, especially in the ecologically promising recycling options of light mortar, lightweight aggregate concrete, and shuttering block production (Volk et al., 2023). It is assumed that the maximum possible amount of pd-AAC powder and pd-AAC granulate is used for the entire production of light mortar (61% of all input material substituted by pd-AAC powder), lightweight aggregate concrete (46%-81% of all input material substituted by pd-AAC granulate and smaller amounts of pd-AAC powder, depending on the production recipe), and shuttering blocks (36% of all input material substituted by pd-AAC granulate) (Volk et al., 2023). Thus, the decentralised demand is approximately 1,800,000 t/a pd-AAC powder, primarily for light mortar production, and 600,000 t/a pd-AAC granulate for lightweight aggregate concrete and shuttering block production. This decentralised demand would be high enough to ensure a full pd-AAC recycling even for high predicted volumes in the future.

Additionally, efficiencies (E) are considered independently from the plants' capacities. The mechanical processing produces powder and granulate from pd-AAC with an overall efficiency of 99%. Only 1% of impurities are sorted out (Volk et al., 2023). The shares of produced pd-AAC powder and granulate can vary, depending on the crushing technology and humidity of the pd-AAC. Generally, the production of ¾ powder and ¼ granulate is a reasonable assumption (Gyurkó et al., 2019; Volk et al., 2023), reaching an overall efficiency of 74.25% (powder) and 24.75% (granulate). The RC-BCC production needs pd-AAC powder and additional limestone to reach the target C/S ratio, resulting in an efficiency (related to pd-AAC powder) of 171% for the second recycling step (Steins, Volk, Beuchle, et al., 2023). Efficiencies for other commodity combinations (for example, RC-BCC to pd-AAC) are zero since production is impossible.

A recycling plant of the first step is assumed to have five different capacities (K): 10,000, 25,000, 50,000, 100,000, or 250,000 t input/a. The second recycling step's plant has capacities of 7,425, 18,563, 37,125, 74,250, or 185,625 t input/a (Table F.2). The latter capacities correspond to the capacities of the first recycling step multiplied by the efficiency for pd-AAC powder production, allowing the entire output of a first-stage plant to be further processed in a second-stage plant of the same capacity level.

The maximum pd-AAC amount allowed to be landfilled (L) is 30% to reach a recycling rate of at least 70%, according to the European waste and recycling regulation (Directive 2008/98/EC, 2008). The pd-AAC supply (S) until 2050 is based on Steins et al. (2021) and disclosed in SI-1. The model includes a time horizon of almost 30 years. Therein, costs will not remain constant. Therefore, inflation and discounting are considered to compare current and future costs (Table F.3).

Table F.3: Annual inflation rates for relevant products/categories and the resulting inflation for all cost categories in the model.

Product/category	***Inflation p.a. [-]***	***Reference***	***Explanation***
C&DW disposal	7.8%	Interzero Circular Solutions Germany GmbH (2022)	average inflation 2018-2021
electricity for industrial application	3.0%	GENESIS (2023a)	average inflation 2013-2022
building land	6.4%	Federal Statistical Office (2022)	average inflation 2012-2021
salaries	3.9%	Federal Statistical Office (2023)	average inflation 2013-2022
transport	3.8%	Noerpel-Schneider and Stölzle (2019)	average inflation based on EU sport market prices 2016-2019
machines for sorting and grading of minerals	2.8%	GENESIS (2023a)	average inflation 2013-2022
cement	3.1%	GENESIS (2023a)	average inflation 2013-2022

limestone	5.9%	GENESIS (2023a)	average inflation 2013-2022
chlorides	3.3%	GENESIS (2023a)	average inflation 2013-2022
CO_2 certificates	5.0%	assumption	past inflation rates have been subject to strong fluctuations, so an assumption is necessary
goods made from concrete or cement	3.5%	GENESIS (2023a)	average inflation 2013-2022
cost category in the model			
landfilling	7.8%		equals C&DW disposal inflation
variable costs first recycling step	4.4%	own calculation	weighted inflation of electricity, disposal, and machines
variable costs second recycling step	3.4%	own calculation	weighted inflation of electricity, disposal, limestone, chlorides, CO_2 certificates, and machines
fixed costs first recycling step	3.7%	own calculation	weighted inflation of salaries and machines
fixed costs second recycling step	3.0%	own calculation	weighted inflation of salaries and machines
opening costs first/second recycling step	2.8%		equals machine inflation
land costs first/second recycling step	6.4%		equals building land inflation
transport costs	3.8%		equals transport inflation
revenue pd-AAC powder/granulate	3.5%		equals goods made from concrete or cement inflation
revenue RC-BCC	3.1%		equals cement inflation
discount rate	3%	assumption	

F.2.3 Scenario definition

The case described in Section F.2.2 is considered the baseline scenario (scenario 0). Besides, this study investigates scenarios to show how the optimal solution, the network structure, and the network costs change under different circumstances.

First, RC-BCC production has a substantial potential to reduce environmental impacts (Stemmermann et al., 2023) but is very costly (Table F.2). Thus, scenario 1 considers support for the RC-BCC production. First, the RC-BCC demand is assumed to be 50% higher than in the baseline scenario due to, for example, political objectives for increased secondary inputs or sustainable products. Moreover, 50% higher revenues for RC-BCC are assumed in this scenario due to the product's sustainable image or potential subsidies. Finally, the variable, fixed, and opening costs (without land costs) are assumed to decrease by 3 %/a due to technological progress. However, cost inflation from the baseline scenario is still considered and offset against this cost decrease.

Given the extremely high costs for RC-BCC production, scenario 2 assumes an even higher support for the RC-BCC production. The demand is increased by 100%, the revenues are increased by 100%, and the costs are reduced by 6 %/a. Such support could only be realised with great effort. Nevertheless, possible network adaptations and saving potentials are investigated in this study.

Finally, scenario 3 assumes a significant increase in recycling costs and a reduction in landfilling costs. It is investigated whether recycling remains the preferred pd-AAC treatment option even under considerably worse conditions. Therefore, the variable, fixed, and opening costs (without land costs) for both recycling steps are increased by 50%. Additionally, the revenues are decreased by 50%, and the landfilling costs are reduced to 65 €/t in all regions, corresponding to the minimal pd-AAC landfilling costs given by Aycil and Hlawatsch (2020). Cost inflations are considered like in the baseline scenario.

F.2.4 Implementation

The mathematical model is optimised using the CPLEX solver in the 22.1.1 version, implemented in Python 3.10 via the docplex library. Computation time was approximately 900 s (scenario 0), 750 s (scenario 1), 14 h (scenario 2), and 200 s (scenario 3) on the used machine (AMD Ryzen 9 3900X at 4.00 GHz, 128 GB RAM). The results for all scenarios reached optimality.

F.3 Results

F.3.1 Baseline scenario

The results of the optimised network design in the baseline scenario (Figure F.1) show the pd-AAC volumes with a lighter/orange colouring for lower volumes and a darker/red colouring for higher volumes for selected years and periods. Blue triangles specify the optimum recycling plant locations for the first recycling step, and grey diamonds for the second step. Larger symbols and a darker shade correspond to higher input capacities of the plants. Black dots illustrate demand locations (AAC plants), while decentralised demand is not shown since it does not have a specific location. Transports are indicated by black connection lines, which are thicker when larger amounts are transported.

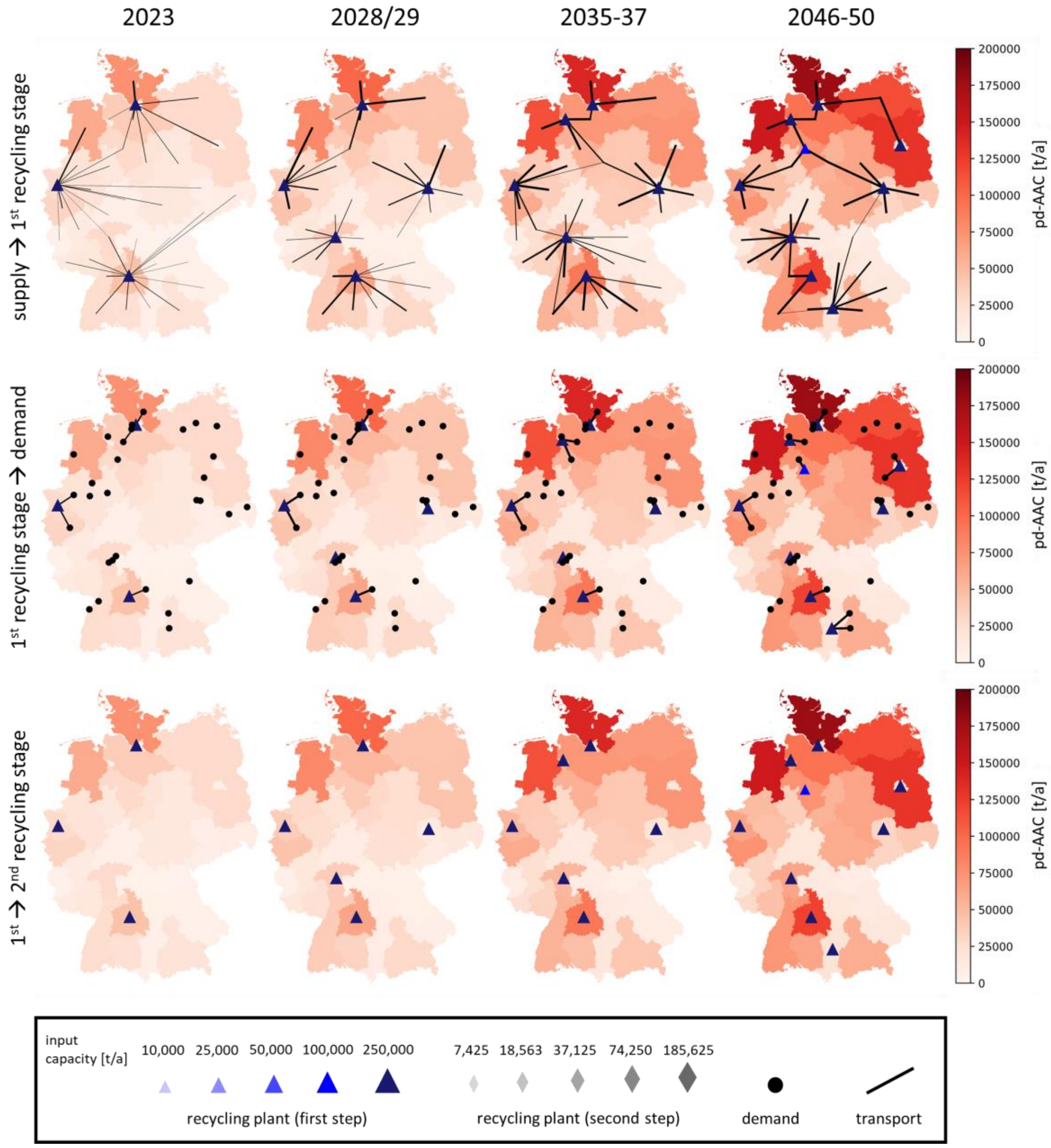

Figure F.1: Optimal pd-AAC recycling network design for Germany in the baseline scenario for the periods 2023, 2028/29, 2035-37, and 2046-50.

First, it is striking that the pd-AAC amount increases significantly in the considered time frame. Accordingly, the number of first-stage recycling plants also rises. Only three plants are opened in 2023, but five are already active in 2028/29. In 2035-37, the number of recycling plants increases to six, and even nine plants are operational in 2046-2050. The optimised network prefers large recycling plants to benefit from economies of scale in fixed and opening costs. Almost all opened plants have the highest possible input capacity of 250,000 t/a. Only one plant in the last period has a lower but still high capacity of 100,000 t/a (second highest capacity level). Additionally, the network completely avoids closing or relocating plants. Consequently, the first plants are placed so that new plants can reasonably complement them over time.

For example, there is a large area without recycling plants in central and eastern Germany in 2023, where new plants are built in the subsequent periods.

The increasing volume and number of recycling plants also leads to an increased pd-AAC transport per region, a recycling plant sourcing from fewer regions, and a general reduction in transport distances (supply → 1st recycling stage). Overall, the average pd-AAC transport distance to a recycling plant decreases from 185 km (2023) to 117 km (2046-50). The transport distances from the first recycling step to the demand locations are also relatively low (1st recycling stage → demand). The decentralised demand is supplied when the demand of the AAC plants within a radius of 100 km around the recycling plants is fully served because a lump-sum transport distance of 100 km is assumed. The second recycling step (RC-BCC production) is not used in the optimised network due to its high costs (1st → 2nd recycling stage).

The overall costs of the network and the different cost categories are presented in Figure F.2. Transport costs (blue) and fixed costs (grey) influence total costs the most. They increase steadily over time as expected pd-AAC amounts rise. The share of transport costs in total costs is consistently around 50%, while the share of fixed costs is about 40%. The variable costs (orange) also rise continuously. However, they are significantly lower than fixed costs and transport costs since the relatively high labour costs are attributed to fixed costs, not variable costs. Opening costs (purple) do not incur in most years and are only significantly high initially, as three large recycling plants are opened in 2023. Finally, pd-AAC is hardly landfilled in the optimised network, leading to almost no landfilling costs (red). The revenues (green) offset around 20% of all costs.

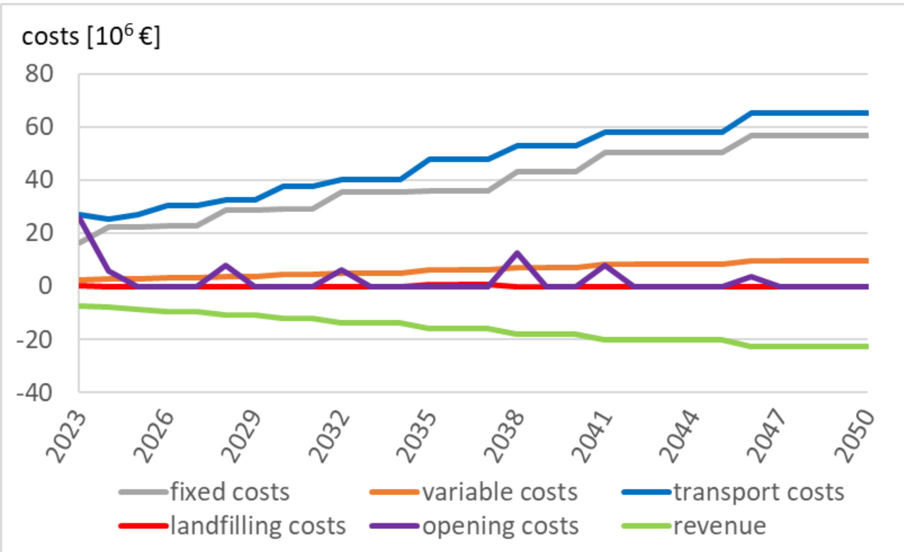

Figure F.2: Development of variable, fixed, opening, transport, and landfilling costs as well as revenues in the baseline scenario in the optimised recycling network until 2050.

Total costs of the pd-AAC recycling network are around 50 M€/a in the first years, ignoring the non-recurring opening costs in 2023 (around 20 M€). The total costs increase significantly in the future, primarily due to strongly rising pd-AAC volumes. Additionally, the inflation considered in the model increases all costs. Thus, the total costs are expected to rise to around 70 M€/a in the early 2030s, potentially reaching more than 100 M€/a in the 2040s (Figure F.3). However, the total costs of the cost-minimised recycling network are significantly lower than the costs of the status quo of pd-AAC treatment, which can be estimated at ⅔ landfilling and ⅓ recovery (e.g. backfilling) in Germany (Bauhaus University Weimar, 2010). Even assuming that recovery does not cause any costs, the landfilling costs are around 55-70 M€/a in the first years. These costs are expected to increase immensely due to the increasing pd-AAC volumes and the exceptionally high inflation rate of landfilling costs (Figure F.3). Thus, status quo pd-AAC treatment without recycling is expected to cause costs of 120 M€/a in 2030, 260 M€/a in 2040 and 480 M€/a in 2050, summing up to around 6,800 M€ until 2050. In contrast, the cost-optimised recycling network only causes costs of approximately 2,200 M€. Thus, the savings potential when establishing pd-AAC recycling in an optimally designed recycling network sum up to 4,600 M€ (68%) until 2050.

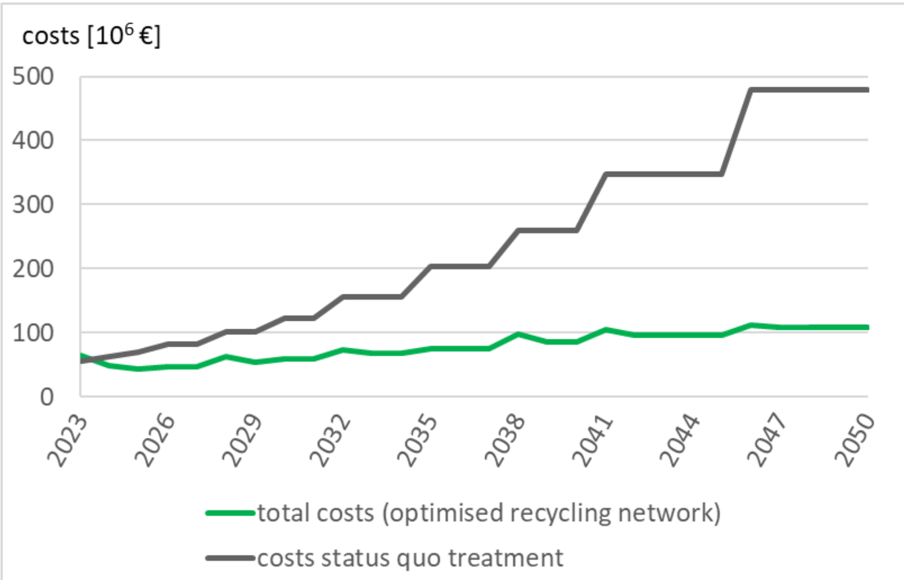

Figure F.3: Comparison of the total costs until 2050 of an optimised recycling network with the status quo (⅔ landfilling, ⅓ recovery).

F.3.2 Sensitivity analysis

The results are subject to uncertainties, mainly of the input data. These are based on cost calculations, research, and assumptions, not field data. Therefore, the relative influence of different input variables on the results is determined in a sensitivity analysis.

Variable, fixed, opening, transport, and landfilling costs, revenues, supply, demand, inflation, and discount rates are considered. These parameters are increased and decreased by 10% compared to their baseline scenario value. The associated change in total costs shows the model's sensitivity to this parameter. The variation of variable, fixed, and opening costs is simultaneously applied to the costs at both recycling steps. Additionally, investigating the sensitivity of the inflation rate means all inflation rates are changed concurrently. Changing inflation rates of individual cost categories would have a similar effect as changing the costs themselves.

Results of the sensitivity analysis show that the total costs are most sensitive to changes in the supply (Figure F.4), as a changed supply directly affects transport, variable, fixed, and opening costs. Moreover, the largest recycling plants have already been built in the baseline scenario, so only small additional economies of scale occur with increased supply. Transport and fixed costs show high sensitivities according to their high share in total costs.

Variations of inflation and discount rates change the total costs by ±2.5-3-5%. These rates affect the total costs differently than the other examined parameters. The impacts of a change only become noticeable over time but are much more significant towards the end of the time horizon as inflation and discount rates have an exponential influence on total costs. Thus, the overall sensitivity considering all periods is significant but not as high as for fixed costs and supply. Revenues show an expectable sensitivity equal to their contribution to total costs. A variation of variable costs, opening costs, landfilling costs, or demand does not change the total costs significantly. Besides, a 10% change in demand does not influence the optimal solution at all.

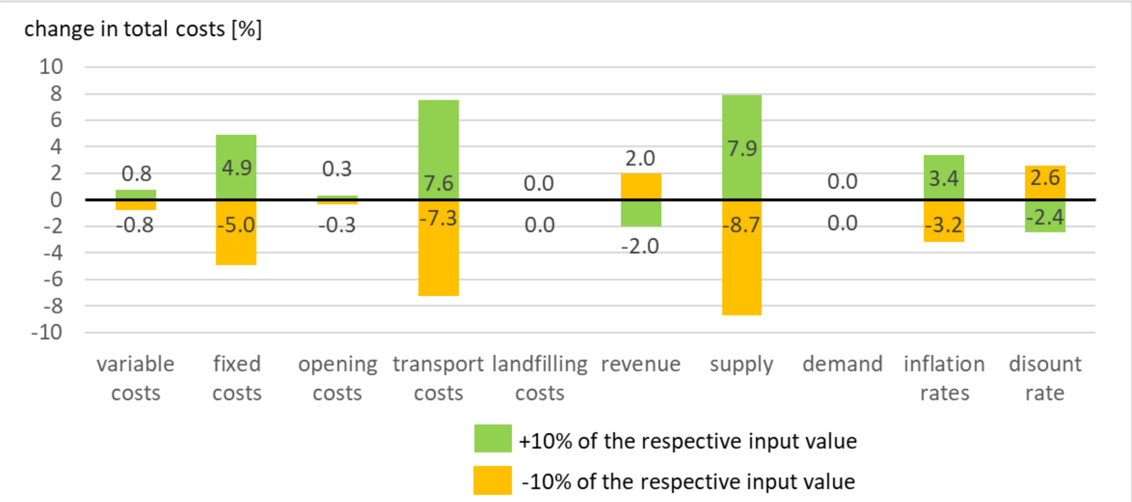

Figure F.4: Results of the sensitivity analysis of total costs, including all cost parameters, revenue, supply, demand, and inflation/discount rate.

Some model parameters are not included in the sensitivity analysis. First, changing the distance would have the same impact as changing the transport costs since the distance is only used in multiplications with transport costs. The efficiency is also not included in the sensitivity analysis. An adjusted efficiency changes variable costs since more or less material is sorted out. Additionally, the revenues change as the amount of valuable output varies. However, variable costs and revenues are already considered in the sensitivity analysis. Moreover, the exact impact of a change in efficiency on variable costs can only be determined in a detailed cost analysis and not within the recycling network modelling. Additionally, the sensitivity of the plant capacity was not investigated because of different capacity levels in the model and higher opening and fixed costs with higher capacities, which are already part of the sensitivity analysis. Finally, the maximum landfill amount is irrelevant for the optimised network as landfilling is widely avoided. Therefore, a 10% change in this parameter would not lead to any change in the result.

F.3.3 Scenario analysis

Three scenarios were considered to investigate how the optimal network structure behaves under changed framework conditions. In scenario 1, the RC-BCC production is supported (+50% demand, +50% revenue, -3 %/a variable/fixed/opening costs). However, the optimal solution of this scenario equals that of the baseline scenario. No RC-BCC plants were opened, and the total costs of the network remain unchanged.

Therefore, the support is doubled in scenario 2 (+100% demand, +100% revenue, -6 %/a variable/fixed/opening costs). In this scenario, RC-BCC plants are opened, but only from 2040 onwards due to the decreasing costs over time (Figure F.5). In the period 2046-2050, even two RC-BCC plants are used, one with an input capacity of 37,125 t pd-AAC powder/a and one with 74,250 t pd-AAC/a. The network structure in the first stage remains unchanged compared to the baseline scenario. The additional RC-BCC plants increase variable, fixed, and opening costs. However, the revenues also increase significantly since the RC-BCC is sold. Thus, the total costs are reduced in the entire time horizon from 2.221 M€ to 2.141 M€ (-3.6%). Overall, the RC-BCC production is only used when extensive support and high technological progress are available.

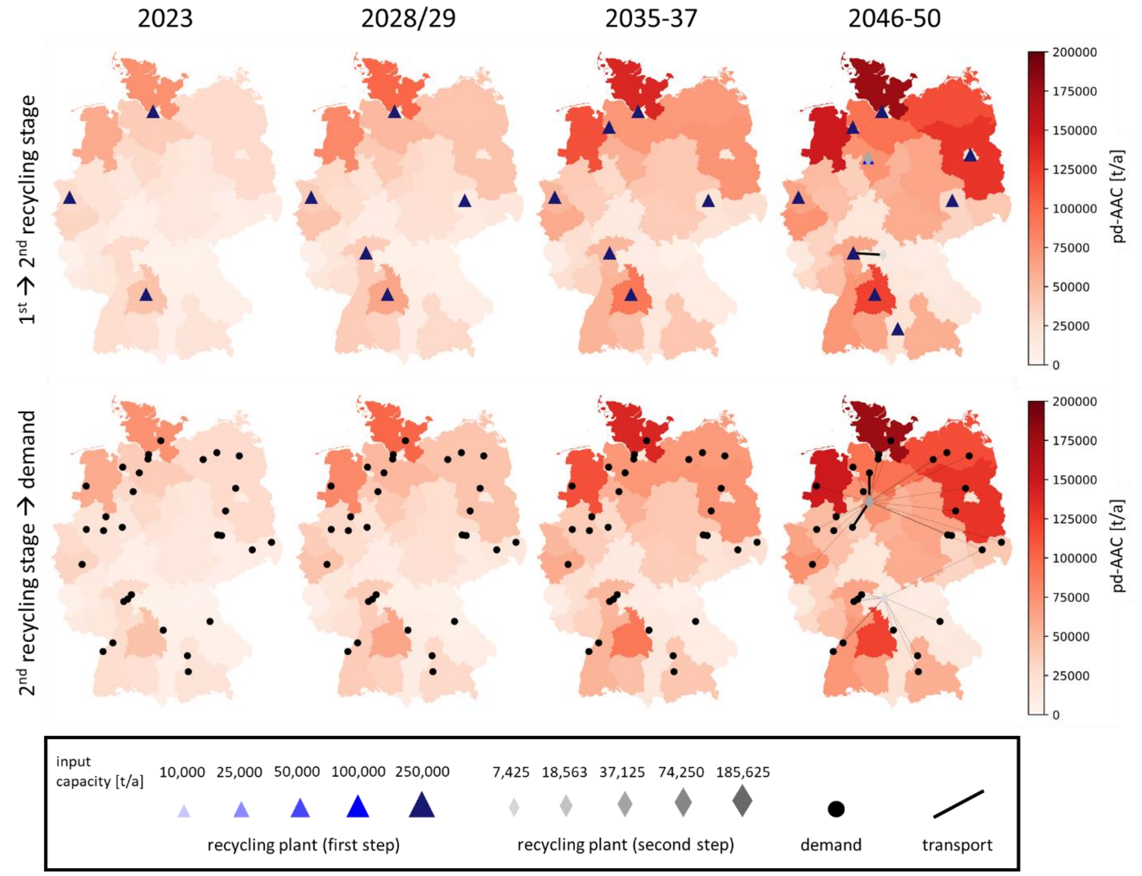

Figure F.5: Optimal German pd-AAC recycling network in scenario 2 (heavy support for RC-BCC production) for the periods 2023, 2028/29, 2035-37, and 2046-50.

Scenario 3 represents a stress scenario for recycling (variable/fixed/opening costs +50%, revenues -50%, landfilling costs reduced to 65 €/t). In this scenario, landfilling is used more than in the baseline scenario, but the limit of 30% is still not reached in any period. The accumulated recycling volume decreases slightly from 41.7 Mt in the baseline scenario to 40.4 Mt over the entire time horizon. However, the recycling rate remains high at 96.9% (99.9% in the baseline scenario). The total costs increase considerably from around 2,200 M€ to approximately 3,100 M€ due to the significantly higher recycling costs (Figure F.6). In addition, revenues are lower and total landfilling costs are higher as more pd-AAC is landfilled. However, the recycling network structure remains mostly unchanged compared to the baseline scenario. Opening a new plant is sometimes postponed to save the fixed costs of one period. Additionally, the location of a few plants changes slightly. However, the largest capacities are still preferred, and up to nine plants are built to recycle most of the pd-AAC. Overall, recycling remains the preferred option for pd-AAC treatment even when facing unfavourable framework conditions.

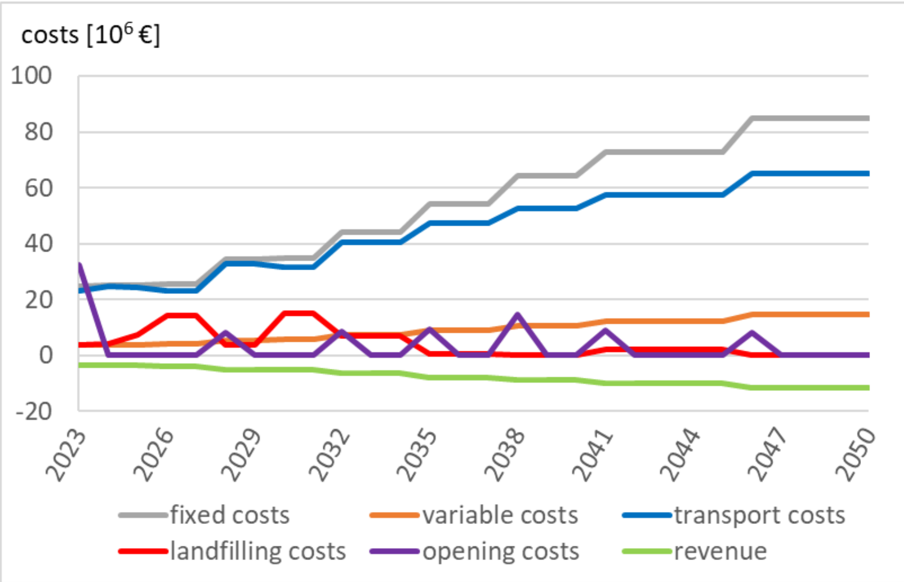

Figure F.6: Development of variable, fixed, opening, transport, and landfilling costs as well as revenues in the optimised recycling network until 2050 in scenario 3.

F.4 Discussion and limitations

The model results depend decisively on the used input data. The input data are primarily based on calculations from previous studies (Persyn et al., 2022; Steins, Volk, Beuchle, et al., 2023; Steins et al., 2021) and are supplemented by assumptions. Field data is not yet available for pd-AAC recycling. Furthermore, it should be considered that inflation rates can fluctuate strongly over such a long horizon and that a change has an exponential effect on the costs of subsequent periods. All in all, the input data are associated with relevant uncertainties that can impact the result. Therefore, the actuality of the input data, including recycling costs, transport distance, and demands, should be reviewed and updated as the establishment of pd-AAC recycling progresses. However, the sensitivity and scenario analyses show that no fundamental change in the result is expected even with significantly changed framework conditions. In particular, recycling will still be preferred to landfilling, even with considerably higher costs. Thus, the current result and the associated implications can be considered relatively robust.

This study's modelling focuses on cost minimisation. Ecological criteria are not considered but are of great importance, especially in the case of modelling and optimising a recycling process. If, for example, greenhouse gas (GHG) emissions are minimised instead of costs, the optimal solution could change significantly. Transports cause significant GHG emissions, while many aspects of the fixed costs (operating labour, interest payments, insurance, overhead costs, general expenses) are not associated with GHG emissions. Thus, GHG minimisation of a pd-AAC recycling network could lead to the opening of more and smaller plants to reduce transport impacts without considerably higher fixed impacts. Additionally, RC-BCC production is more favourable than in terms of costs. If renewable energy is used to produce the RC-BCC, no other recycling option can achieve similarly high GHG savings of up to 0.77 kg CO_2-Eq/kg pd-

AAC (Stemmermann et al., 2023). Therefore, it can be expected that RC-BCC production will be used much more in the GHG-minimal recycling network than in the cost-minimal one. A multi-criteria objective function may be appropriate since practical decisions often consider costs and sustainability aspects, including GHG emissions. This way, the different optimisation aspects can be balanced, and individual weights can be used depending on the decision maker's preferences.

The decentralised demand is calculated from maximum input material substitution with pd-AAC in the entire production of the light mortar, lightweight aggregate concrete, and shuttering block production. This strong assumption leads to a demand of 1,800,000 t/a pd-AAC powder and 600,000 t/a pd-AAC granulate, which might be considerably less in practice. However, there are further alternatives besides these three recycling options, including using the pd-AAC as supplementary material in concrete or for producing floor screed. While these alternatives are ecologically less attractive than the previously mentioned recycling options (Volk et al., 2023), they would nevertheless be available for a cost-minimising recycling network. Therefore, the assumption of a high decentralised demand is reasonable.

The pd-AAC supply must always be treated in the same period it arises, as storage between periods is not allowed. In practice, storage is only conceivable in the short term due to the large quantities to be treated. It is also not practicable to leave the pd-AAC at the demolition site. Only temporary storage at a landfill and processing in a subsequent period would be conceivable. However, this alternative is not modelled due to additional landfill management efforts and transport costs.

F.5 Conclusion

This paper developed a new capacitated, multi-period, and multi-stage model for pd-AAC recycling network optimisation in Germany. The cost-minimised recycling network prefers large recycling plants to use economies of scale in the pd-AAC treatment. However, RC-BCC production plants (second recycling stage) are not opened due to high costs. Instead, pd-AAC powder and granulate are directly used for different recycling purposes. With increasing pd-AAC volumes in the future, the network opens new recycling plants to treat all pd-AAC. Landfilling is mainly avoided. An increasing number of recycling plants leads to reduced transport distances in the future, almost reaching 100 km on average. The transport costs account for around 50% of the total costs, while fixed costs sum up to about 40%, and revenues offset nearly 20% of the total costs. Variable, opening, and landfilling costs are pretty low. Pd-AAC recycling costs for the whole period until 2050 sum up to 2,200 M€ and, thus, have a significant savings potential compared to the status quo, which would cause costs of 6,800 M€.

Future research can use field data to optimise the model when pd-AAC recycling is established, and robust data is available. Furthermore, the model can be expanded regionally to optimise, for example, a European pd-AAC recycling network. Additionally, the model can be transferred to other use cases. Generally, all similar recycling processes are suitable for model transfer, especially those involving construction materials. With its multi-period formulation, the model can deliver the highest added value in situations with increasing (or decreasing) future supply and changing costs. Moreover, future research could investigate stochastic modelling of similar settings to consider uncertainties directly in the model.

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