

Supplementary information - Material requirements and impacts of the building sector in the Shared Socioeconomic Pathways

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S1 Literature review

S1.1 Dynamic materials modeling of the building sector

In the identified literature, several authors have identified a lack of knowledge of the anthropogenic stock (Schiller et al., 2017). While initial publications have focused on primary materials inflows, the in-use stock has progressively gained attention, for its role a driver of the construction and demolition flows, and the secondary materials providers (Müller, 2006). It enables a better understanding of the dynamics of construction and demolition cycle, and the maintenance and expansion shares of the construction inflow in the building sector. The publications show an increasing share of maintenance in the most developed areas (Wiedenhofer et al., 2015), as well as a grow of the demolition flow (Müller, 2006; Bergsdal et al., 2007). Conversely, the developing regions will experience a slowdown of the annual building expansion in the coming decades, which could have profound industrial consequences for the management of waste and secondary materials.

We further observe an increasing focus on the construction and demolition wastes. It provides estimations of the prospective wastes (Bergsdal et al., 2008), potential secondary resources through a better classification of input materials (Hashimoto et al., 2009), circularity achievements (Arora et al., 2019) or analysis of the impacts of policies, for example in the European Waste Framework Directive (Wiedenhofer et al., 2015). Despite a substantial increase of the building in-use stock in China over the past years, per-capita level could further grow (Wang et al., 2015a; Zhang et al., 2015), before an expected reduction of the building stock expansion in the coming decades in developed areas and China. It could allow a higher reliance on secondary resources to lower primary inputs and environmental impacts, which require the development of an efficient waste management and recycling system (Shi et al., 2012; Wang et al., 2015b).

Finally, the parameters assumptions are discussed in the literature, as they represent a major source of uncertainties (Cao et al., 2018)¹, but could provide tools to better manage materials stocks and flows. Several publications emphasize the significant role of material efficiency to reduce materials demand and their environmental impacts. Material efficiency is a multifaceted concept (Pauliuk and Heeren, 2021), which includes lifetime extension (Hu et al., 2010; Shi et al., 2012; Huang et al., 2018)², a better product design (Shanks et al., 2019)³, or higher recycling and reuse (Shi et al., 2012; Wang et al., 2015b). Pauliuk and Heeren (2021) proposed the implementation of material efficiency strategies in the building and transport sectors in Germany.

S1.2 The role of materials engineering to reduce impacts

Contrary to the common belief that concrete is not recyclable, Japan or the Netherlands manage to achieve close to 100% concrete recycling rate by recycling concrete into new structural applications or in lower grade uses such as road (Tam, 2009; Xicotencatl, 2017). It allows considerable reduction of the environmental impacts of CDW by lowering their amount in landfills. The potential of recycling construction and demolition waste (CDW) is substantial, as they represent a significant part of the society wastes. They reached 25 to 30% of the overall generated European Union wastes in 2017 (European Union (2017)), of which mineral and solidified wastes represented 76% (Reis et al. (2021)). The European Union targeted the CDW in the its Waste Framework Directive⁴, and managed to fulfill the objective to recover or recycle 70% of CDW by 2020 (Villoria Sáez and Osmani (2019)). More than 82% of generated CDW were recycled in 2021, and only 11% were disposed in landfills (European Environment Agency (2021)). However, the European results are not representative of the current global CDW management, and lower recovery rates are observed in other regions (Reis et al. (2021); Tam et al.

¹Liu et al. (2020) provided a household level desegregation study for a more accurate description of appliances life cycle in the building sector.

²Hu et al. (2010) raised questions about the optimal lifetime to reduce resources use and environmental impact through an increase of the share of secondary materials in the input.

³Conversely to light-weighting products, dynamic and increasing materials intensities are sometimes used in MFA studies to model technological change (Shi et al., 2012; Hatayama and Tahara, 2016).

⁴This directive from 2008 aimed at enhancing the recovery and management of CDW in a circular economy perspective.

(2018)).

Concrete is recycled in the form of aggregate by a crushing process, partly replacing natural aggregate content in the concrete production⁵. Two types of recycled aggregates from construction and demolition waste (CDW) are generally considered (Marinković and Carević, 2019). Recycled aggregates (RA) are defined as mixed aggregates. Recycled concrete aggregates (RCA) are mainly made of concrete, and their higher quality would allow a use recycled aggregates concrete (RAC) for structural applications with different standard compositions depending on the countries (Nováková and Mikulica, 2016). Despite potential environmental benefits (Tam et al., 2018), the use of recycled aggregates is often limited to non-structural projects, e.g. in India (Behera et al., 2014), in the USA and in France⁶ because of a lack of confidence of construction actors in RA (Silva et al., 2019; European Environment Agency, 2021). The different applications of RA are described in Reis et al. (2021), and the authors outlined the growing number of encouraging pilot projects using recycled aggregate concrete (RAC). Silva et al. (2019) further provided a review of 31 constructions projects to demonstrate the technical and economic viability of recycled aggregates use. Among these projects, 11 are related to structural use of concrete in buildings. They carry substantial benefits through four reductions : financial, primary resources need, accumulation in landfills and environmental impacts.

The RAC research field has been growing over the past years, and the specific impact of replacing natural aggregates by recycled concrete aggregates (RCA) on the properties of RAC was under focus. RCA is defined as aggregates recovered from concrete in which other materials represent less than few percents (Marinković and Carević, 2019), and it can be used for higher strength concrete than RA (Purnell and Dunster, 2010). However, Behera et al. (2014) showed lower mechanical and durability performance of RAC, and emphasized the need of further assessment of RA quality and its impact on concrete. In reviews on the RCA impacts on RAC production, Silva et al. (2014) and Verian et al. (2018) provided recommendations for an optimal use of recycled concrete aggregates (RCA) in concrete production. The authors showed different challenges depending on the use of fine and coarse aggregates⁷, but a potential was identified for both types. The authors further reported that some researches estimate a 30% limit in the use of coarse RCA in concrete, but state that there could be no limit in its potential share if specific process conditions are respected⁸. Thomas et al. (2013) further outlined the need for more assessment of RAC in different exposure conditions. For moderate conditions, Thomas et al. (2018) estimated that a 25% replacement of NA by RCA displays viable strength results. Guo et al. (2018) and Verian et al. (2018) further identified the potential to enhance recycled aggregate concrete performances, using for example CO₂ treatment and pozzolanic materials (e.g. fly ash). Several studies have conducted further environmental assessment of RCA and natural aggregated for different mixes and geographic perimeters, showing transport distances influence (Marinković et al., 2010) and avoided emissions in steel production and waste disposal (Knoeri et al., 2013). However, no concordance in the potential reduction was observed in the literature, and Wijayasundara et al. (2017) further showed similar embodied energy and impacts of RCA and natural aggregates in a cradle-to-gate analysis of RCA in Australia. Given the lack of consensus, no energy demand and environmental impact reductions were considered.

In parallel of improving concrete recovery and recycling, "green concrete" aims at cutting the environmental impact. It is defined as a low-carbon concrete, which production mainly relies on the use of industries or agriculture by-products, nanoparticles and advanced techniques to further understand concrete structures (Vishwakarma and Uthaman, 2020). Given that most of the energy demand and environmental impacts of concrete production occurs during cement production, the replacement of primary materials by waste materials could achieve lower impacts of concrete production (Reiners and Palm, 2015). The use of wastes in concrete production has already been implemented for decades in the con-

⁵Shares of 12% to 29% of recycled concrete were observed in German, the Netherlands and the United Kingdom in 2012 and 2013 (Reiners and Palm, 2015)

⁶Most of the recycled CDW in France are used in non-structural applications. The RECYBETON project aims at increasing the share of RA in concrete.

⁷Bravo et al. (2015) found that fine aggregates use hampers concrete performance, but that a limited share of coarse aggregates is viable. This higher impact of fine RCA was also highlighted in Guo et al. (2018).

⁸Coarse and fine aggregates are defined by the size of their particles.

crete industry through supplementary cementitious materials (SCM)⁹. The most common SCM are coal fly ash (CFA)¹⁰, blast furnace slags (BFS)¹¹, silica fume, or metakaolin (Habert, 2014; Holland et al., 2016)¹². The study of "green concrete" performances and environmental impacts has showed a growing interest in the literature and several publications recently reviewed the by-products studies (Siddique, 2014; Gu and Ozbakkaloglu, 2016; Paris et al., 2016; Jiang et al., 2018; Sandanayake et al., 2020).

⁹The Lafarge company defines Supplementary Cementitious Materials (SCMs) as "materials that, when used in conjunction with portland cement, portland limestone cement or blended cements, contribute to the properties of hardened concrete through hydraulic and/or pozzolanic activity".

¹⁰CFA is a by-product of the combustion of coal to generate electricity.

¹¹BFS is produced during the iron-making process.

¹²Many other materials can also be used, such as foundry sand, wood ash, plastic, glass and other ashes (Paris et al., 2016; Sandanayake et al., 2020)

S2 Methodology

This section provides further documentation on the methodology in this study. The following items are detailed :

- The general equations of the model in Section S2.1;
- The perimeter of the multinational areas considered in the scenarios are described in Section S2.2;
- The historical regional floor areas are displayed in Section S2.3;
- The regional lifetimes methodology and data are detailed in Section S2.4;
- The regional historical materials intensities and the methodology to set their logistic profiles up to 2100 are explicitated in Section S2.5.

S2.1 MATER model general equations

In the following section, TU means "Technological Units" (infrastructures such as m^2 of buildings) and RM means "Raw Materials" (such as concrete or steel).

First, the amount of new infrastructures to build each year depends on the evolution of the stock needed and the quantity of old infrastructures to replace :

$$F_{TU_i}^{in} = \frac{dS}{dt} + F_{TU_i}^{out} \quad (S1)$$

$$F_{TU_i}^{out} = \log \text{ normal delay}(F_{TU_i}^{in})$$

with :

- F^{in} : the amount of new infrastructures to build in [TU/y]
- S : the reference stock in [TU]
- F^{out} : the amount of old infrastructures to replace in [TU/y]

Then, the amount of new infrastructures to build is multiplied by their material intensity to calculate the global material consumption :

$$F_{RM_i} = \sum_{TU} (F_{TU_i}^{in} \times i_{TU, RM_i}^{rm}) \quad (S2)$$

with :

- F : the raw material consumption in [ton/y]
- F^{in} : the amount of new infrastructures to build in [TU/y]
- i^{rm} : the material intensity in [ton/TU]

The raw material recycling depends on the old infrastructure flow and the primary raw material production is calculated to fill the gap between the global raw material consumption and the recycling potential :

$$F_{RM_i}^{2nd} = \alpha \times F_{RM_i}^{out^{rm}} \quad (S3)$$

$$F_{RM_i}^{1st} = F_{RM_i} - F_{RM_i}^{2nd}$$

with :

- F^{2nd} : the recycled raw materials in [ton/y]
- α : the recycling rate of the raw materials
- F^{Out^m} : the raw materials contained in the old infrastructures in [ton/y]
- F^{1st} : the primary raw materials in [ton/y]
- F : the raw material consumption in [ton/y]

If, in a given year, recycled production exceeds the total consumption, the primary production is set to zero and any surplus is stored to be used next year.

Finally the energy needed to produce the raw materials is calculated thanks to energy intensities :

$$E_i = \sum_{RM} (i_{RM_i}^{1st} \times F_{RM_i}^{1st} + i_{RM_i}^{2nd} \times F_{RM_i}^{2nd}) \quad (S4)$$

with :

- E : the total energy needed for the raw materials production in [J/y]
- i^{1st} : the energy intensity of the primary raw material production in [J/ton]
- F^{1st} : the primary raw materials in [ton/y]
- i^{2nd} : the energy intensity of the recycled raw material production in [J/ton]
- F^{2nd} : the recycled raw materials in [ton/y]

S2.2 Regional segmentation in the scenarios

Nine multinational areas are considered in this study. They are detailed in the Table S1.

Regions	Countries
Africa	Algeria, Angola, Benin, Botswana, Burkina Faso, Burundi, Cabo Verde, Cameroon, Central African Republic, Chad, Comoros, Congo, Cote d'Ivoire, Democratic Republic of the Congo, Djibouti, Egypt, Equatorial Guinea, Eritrea, Eswatini, Ethiopia, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Kenya, Lesotho, Libya, Madagascar, Malawi, Mali, Mauritania, Mauritius, Morocco, Mozambique, Namibia, Niger, Nigeria, Rwanda, Sao Tome & Principe, Senegal, Seychelles, Sierra Leone, Somalia, South Africa, Sudan, South Sudan, United Republic of Tanzania, Togo, Tunisia, Zambia, Zimbabwe
China	
Europe	Albania, Austria, Belarus, Belgium, Bulgaria, Bosnia and Herzegovina, Cyprus, Czech Republic, Croatia, Denmark, Estonia, Germany, Gibraltar, Greece, Finland, France, Hungary, Ireland, Iceland, Israel, Italy, Kosovo, Latvia, Lithuania, Luxembourg, Malta, Moldova, Montenegro, Netherlands, North Macedonia, Norway, Poland, Portugal, Romania, Serbia, Slovak Republic, Slovenia, Spain, Sweden, Switzerland, Turkey, Ukraine United Kingdom
India	
Latin America	Argentina, Bolivia, Brazil, Chile, Colombia, Costa Rica, Cuba, Curacao, Dominican Republic, Ecuador, El Salvador, Guatemala, Haiti, Honduras, Jamaica, Nicaragua, Panama, Paraguay, Peru, Suriname, Trinidad and Tobago, Uruguay, Venezuela
Middle East	Bahrain, Islamic Republic of Iran, Iraq, Jordan, Kuwait, Lebanon, Oman, Qatar, Saudi Arabia, Syria, United Arab Emirates, Yemen
North America	Canada, Mexico, United States of America
OECD Pacific	Australia, Japan, Korea, New Zealand
Other Asia	Afghanistan, Armenia, Azerbaijan, Bangladesh, Bhutan, Brunei, Brunei Darussalam, Cambodia, Hong Kong, Indonesia, Kazakhstan, Dem. People's Republic of Korea, Kyrgyzstan, Laos, Macao, Malaysia, Maldives, Mongolia, Myanmar, Nepal, Oceania (excluding Australia and New Zealand), Pakistan, Philippines, Singapore, Sri Lanka, Taiwan, Tajikistan, Thailand, Turkmenistan, Uzbekistan, Vietnam

Table S1: Summary of the considered regions.

S2.3 Historical regional building stocks

The surfaces obtained from the logistic approach adopted in this article are displayed in the Figure S1 for the residential segment and in the Figure S2 for the non-residential segment.

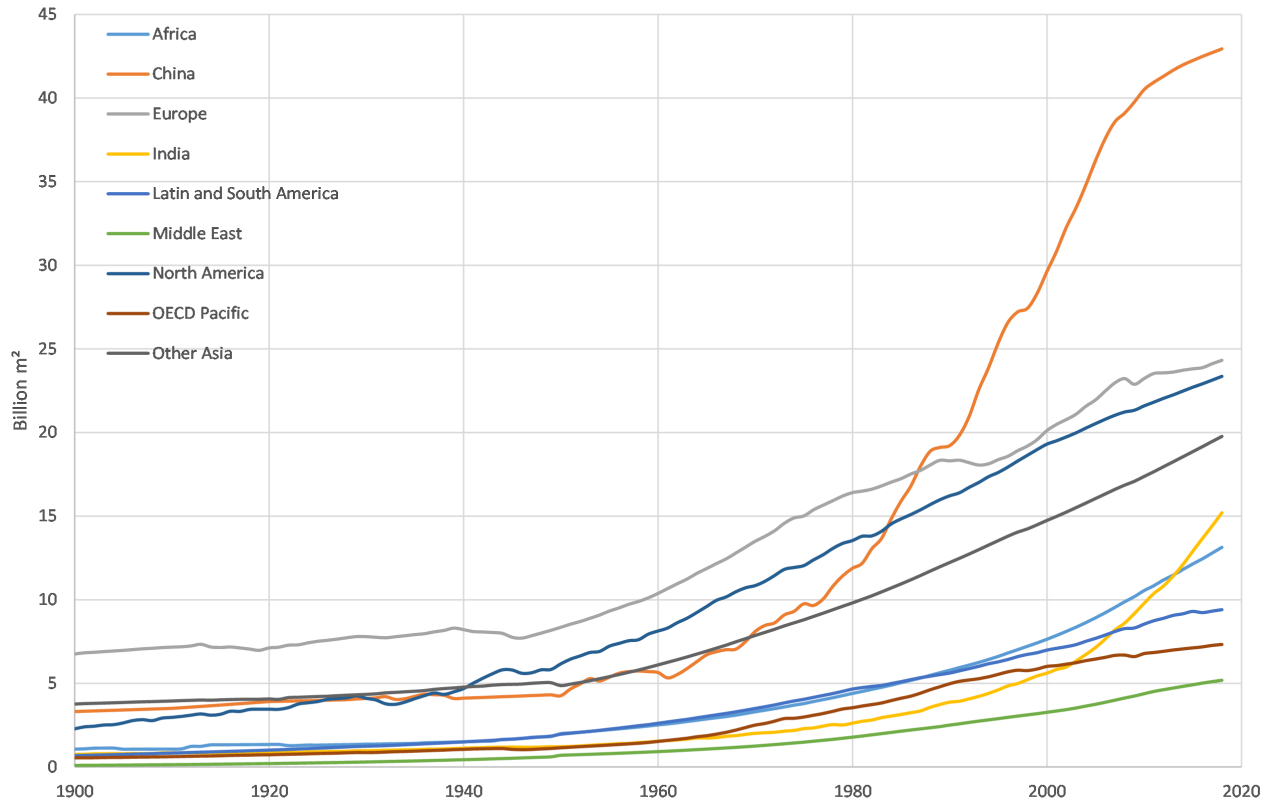


Figure S1: Historical residential regional floor areas.

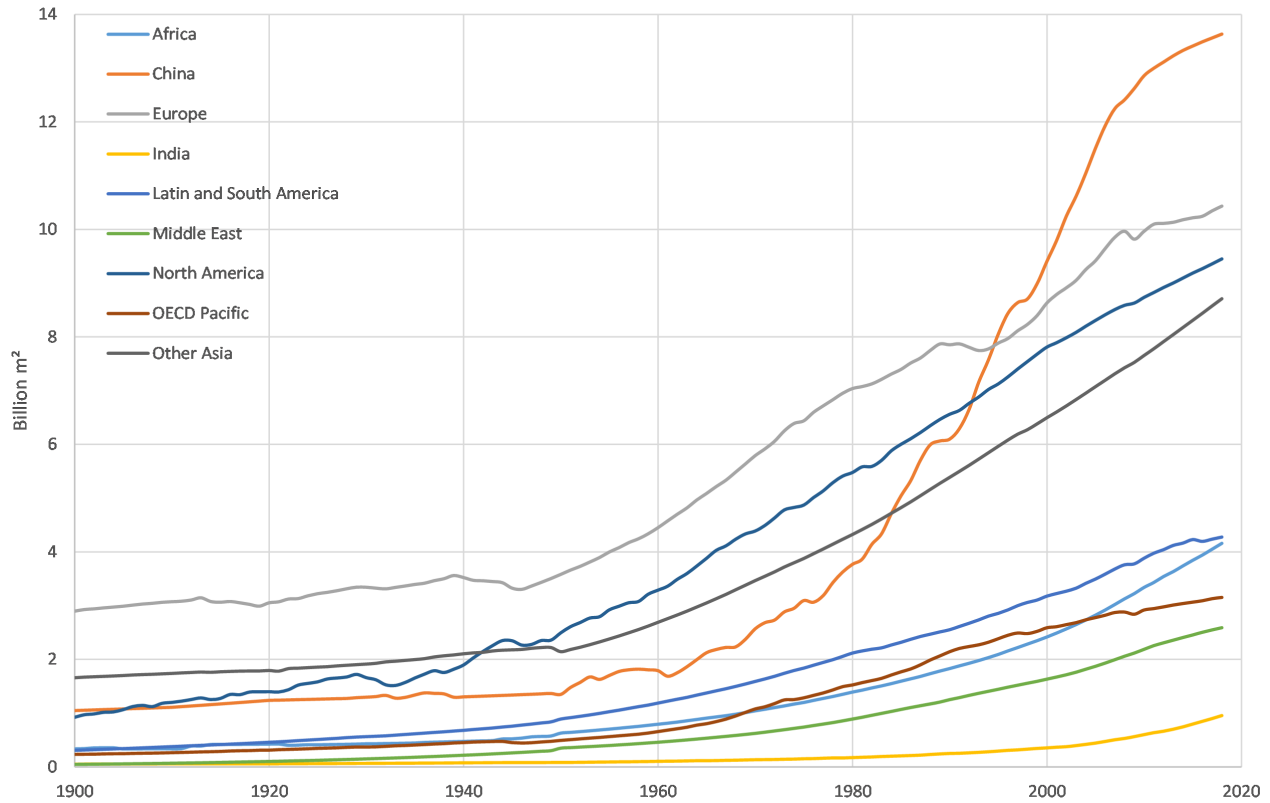


Figure S2: Historical non-residential regional floor areas.

S2.4 Constant and dynamic regional lifetimes

Recent publications analyzed the impact of lifetime in infrastructures modeling (Aktas and Bilec, 2012; Marsh, 2017; Zhou et al., 2019; Cao et al., 2019). The authors outlined two major aspects of lifetime assessment. Firstly, the need for a more systematic use of lifetime distribution. Numerous statistical distributions exist, such as Weibull, normal, log-normal, Gamma or Gompertz-based. They are widely used in the literature (Table S3), but no insights were available about the best suited approach to model building lifetime. Miatto et al. (2017) recently provided such an analysis. The authors tested the sensitivity of the distribution on materials modeling at national and city scales, and concluded on the low effects on stocks results and higher impacts on output flows. They further outlined the need to use several distribution lifetimes when various buildings categories are considered, and the low impact of the distribution choice in an aggregated building stock. Secondly, the inverse relation between lifetime and environmental impacts was depicted in Marsh (2017). They showed that a lifetime value of 100 years reduces the environmental impacts by 38% in comparison with a 50-year level at the Danish scale. A better modeling of building lifetime therefore could therefore lie in a combination of a dynamic mean value with a statistical distribution. This approach was conducted by Hu et al. (2010) for the Chinese residential building stock, assuming a gradual increase of the mean value of a normal lifetime distribution. A constant approach was selected in this study because of the wide scope and the required extensive data collection to compute dynamic regional lifetimes.

The life span of buildings and their typology vary according to geography (AIMCC, 2015). In a first step, current regional lifetimes considered in the literature were studied. They were obtained from a regional analysis and a census of the observed mean values in lifetime distributions. A summary of those latter is presented in Table S3). At the European level, JRC (2011) reported a life span of about 50 years for more than half of the French housing stock, 20% of the buildings reaching 100 years. Wang et al. (2018) identified data from several studies, showing longer building lifetimes for Euro-

pean buildings than Chinese or North American buildings. The values range from 64 to 132.6 years for Germany, Spain, Belgium, the Netherlands, Switzerland, Austria, France, and the United Kingdom (CABR, 2014; Song, 2004). Similarly, AIMCC (2015) presented the results of several European studies predicting lifetimes between 50 and 75 years. Except for Ciacci et al. (2013) and Hatayama et al. (2009), those values are commonly observed in the lifetime distribution summarized in Table S3. Based on these results, an average service life of 70 years was considered for European commercial and residential buildings.

In North America, Aktas and Bilec (2012) reported a slightly increasing life span since the late 1990s. The life span of U.S. residential buildings has evolved from 50 years in 1997 to 61 years in 2009. Wang et al. (2018) estimated close life spans for the United States and Canada buildings, respectively, with 60 and 70 years. There are large differences in service life by construction type. In a study conducted in the United States and Canada, O'Connor (2004) revealed that only one third of North American non-residential concrete buildings are more than 50 years old, compared to 63% of wood buildings and 20% of steel buildings. Wood has a longer life span than concrete, with a majority of buildings destroyed after 75 years of life. Lifetimes ranging from 26 to 50 years are observed for concrete buildings. The practical observations are therefore lower than the theoretical values of 51.6 years for wood buildings and 87.2 for concrete buildings (O'Connor, 2004). It covers a variety of reasons for demolition. In North America, nearly 80% of concrete buildings are demolished for lack of use or for a new building project. Lower levels are observed for wood (50%) and steel (60%). Only 31% of the buildings are demolished at the end of their life in O'Connor (2004), which emphasizes the unreliability of the theoretical life span. An average life of 50 years was therefore considered for the entire North American building stock¹³.

In China, a different trend is observed by Cao et al. (2019). The authors estimated a life span of 39.4 years for the building flows between 1950 and 1969, and 37 years between 1970 and 1979. Lifetime then dropped to 25.3 years for buildings finished between 1980 and 1989, and increases to 26.6 years in the 1990-1999 period and 33.8 years between 2000 and 2015. The life span of Chinese housing thus remains low, explaining the high annual demolition rate. On average, Cai et al. (2015) estimated a current life span of Chinese residential buildings of 23.2 years, while Hu et al. (2010) suggested average life spans of 30/40 years in the urban sector in the 1970s and 1980s, and 15 years in the rural sector. These would increase over time for both rural and urban buildings due to China's economic reform in 1978, allowing for improved construction techniques (Hu et al. (2010)). These values and trend are confirmed by Wang et al. (2018), surveying several studies and announcing a final value of 35 years for the Chinese building stock. Moreover, Hu et al. (2010) depicted a significant increase of life span in the future, finally reaching 75 years. An average value of 35 years was considered in this study.

In Japan, the life span has been decreasing since the end of World War II, reaching 25 to 30 years in recent years (Wuyts et al., 2019; Kobayashi, 2016), although with disparities depending on the construction method¹⁴. This low average life span is explained by the depreciation of buildings observed in Japan, and remains slightly below the observed values in Table S3. An average value of 30 years was finally considered for the Japanese building stock. A low value is also observed in South Korea, with an average of 22.4 years (Seo and Hwang (1999)). A minimum value of 50 years seems to be common in the New Zealand market, and Deetman et al. (2020) modeled the Oceania building stock with a mean value of 83 years. Japan and Korea accounting for 85% of the total ASEAN population in 2020, a 35 year-value is considered for this region.

In the Middle East, a lack of data is observed. The combination of poor quality materials and harsh climatic conditions explains why buildings are renewed faster than in Western countries. Thus, we observe an average building life of about 20 years in the United Arab Emirates¹⁵, 25 to 50 years in Saudi Arabia¹⁶ and 25 to 35 years in Iran¹⁷ (Yousefi and Gholipour, 2018). An average life of 30 years was therefore considered for the Middle-East.

¹³The concrete-made and steel-made building market shares were used to evaluate building lifetime. Concrete market share reached 34% in non-residential and multi-story buildings, while steel market share was 46%.

¹⁴This is in accordance with the reported values of the Chinese Vice-Minister of Housing and Urban-Rural Development.

¹⁵The workshop "Life Expectancy of Buildings in the UAE" aimed at tackling the quality issue and thus increasing the buildings lifetime.

¹⁶Poor quality standards are assumed to be the source of this low life span.

¹⁷This average lifetime is even assumed to decrease as low quality standards actors represent an increasing market share.

The other regions suffer a lack of data. A 60-year value is considered for Latin America, which is lower than theoretical values of [Heeren et al. \(2015\)](#) and [Deetman et al. \(2020\)](#), and a lower value for the entire continent by [Murakami et al. \(2010\)](#). A life span of 50 years was used in [Deetman et al. \(2020\)](#), and is retained in this study for ASEAN and Other Asia. To the author's knowledge, no precise assessments of life span are available for Africa and India, an average value of 50 years was then considered. The final data are displayed in S2.

Area	Average current life span
Africa	50
ASEAN	50
China	35
Europe	70
India	40
Latin America	60
Middle East	30
North America	50
OECD Pacific	35
Other Asia	40

Table S2: Summary of the current regional building lifetimes considered in this study.

References	Distribution	Area	Building type	Mean lifetime	Standard deviation
Deetman et al. (2021)	Weibull	Brazil	Urban residential	100	
Deetman et al. (2021)	Weibull	Canada	Residential	51	
Hatayama et al. (2009)	Weibull	China	All	32.5	
Zhou et al. (2019)	Weibull	China	Urban residential	34.1	23.5
Cai et al. (2015)	Weibull	China	All	23.2	
Deetman et al. (2020)	Weibull	China	Residential	39	
Wang et al. (2015b)	Weibull	China	Residential	39	
Olaya et al. (2017)	Normal	Colombia	Residential	100	25
Vásquez et al. (2016)	Normal	Czech Republic	Residential	100	25
Deetman et al. (2020)	Weibull	Eastern Europe	Residential	78	
Hatayama et al. (2009)	Weibull	Europe	All	31.5	
Murakami et al. (2010)		Europe	All	75	
Vásquez et al. (2016)	Normal	Germany	Residential	150	40
Bradley and Kohler (2007)	Weibull	Germany	Urban residential		
Ciacci et al. (2013)	Normal	Italy	All	40	14
Hatayama et al. (2009)	Log-normal	Japan	All	38.7	
Tanikawa et al. (2015)	Log-normal	Japan	All		
Hashimoto et al. (2007)	Log-normal and Weibull	Japan	All		
Deetman et al. (2020)	Weibull	Japan	Residential	34	
Murakami et al. (2010)		Japan	All	50	
Fishman et al. (2014)	Normal	Japan and USA	All		
Deetman et al. (2020)	Weibull	Mexico	Residential	56	
Murakami et al. (2010)		North, Middle and South America	All	50	
Brattebø et al. (2009)	Normal	Norway	All	180 down to 95	
Deetman et al. (2020)	Weibull	Oceania	Residential	83	
Deetman et al. (2020)	Weibull	Rest of South America	Residential	60.5	
Deetman et al. (2020)	Weibull	Southeastern Asia	Residential	50	
Heeren et al. (2015)	Log-normal	Rest of South America	Residential and small offices	90	
Müller et al. (2014)	Normal	The Netherlands	Residential	60-90-120	20
Davis et al. (2007)	Weibull	UK	All	60	13.7
Davis et al. (2007)	Log-normal	UK	All	60	12.1
Deetman et al. (2020)	Weibull	USA	Residential	77	
Hatayama et al. (2009)	Weibull	USA	All	75	

S2.5 Regional raw materials and energy intensities

Some material modeling studies consider static materials intensities (Marinova et al., 2020; Deetman et al., 2020; Hong et al., 2016; Schebek et al., 2017; Yang and Kohler, 2008), and other provide useful data about their evolution over time. Hu et al. (2010) assumed a dynamic steel material intensities in rural residential building in China, and dynamic concrete intensities are displayed in Figure S3. An increase of concrete and steel intensities is generally observed over time, except in Sweden (Gontia et al., 2018). Two reasons can explain this upward trend. Firstly, a change in the construction mix occurred in Europe and Japan after the World War II. Concrete increased its domination over material use in a context of rapid and economical reconstruction of destroyed cities (Steele, 2017; Diefendorf, 1989; Chemillier, 1997). The rises represented in Figure S3 may thus provide insight about an evolution toward more concrete in the construction mix rather than a higher use of materials in concrete buildings. A further evolution of the mix (e.g. wood building development) is considered in this study. Higher steel, aluminum and copper intensities over time draw a different trend, such as an increasing use of reinforced concrete, or higher electrification of buildings. This research field is not well studied in the literature, and no quantitative regional trends are available to the author knowledge.

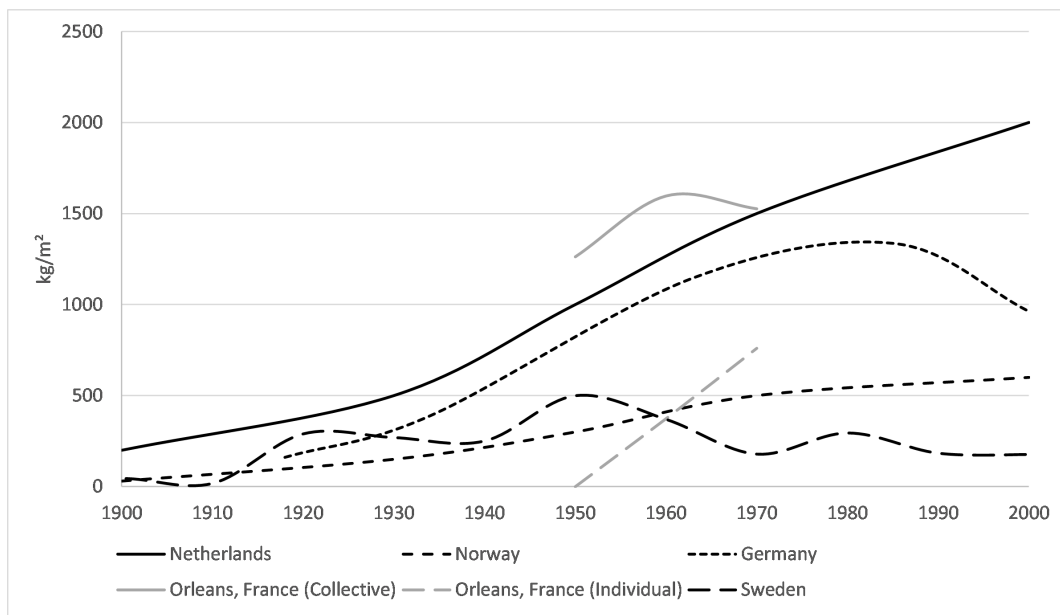


Figure S3: Dynamic residential concrete intensity values observed in the literature. Netherlands data from Müller (2006), Norway data from Bergsdal et al. (2007), Germany data from Ortlepp et al. (2018), French data from Serrand et al. (2013) and Sweden data from Gontia et al. (2018). The decrease of intensity observed in Ortlepp et al. (2018) could be explained by a recent rise of masonry use.

In this study, dynamic material intensities were considered. IEA (2018) distinguished five building structures in nine major regions: composite, wood, steel, masonry and concrete. While wood structures accounted for nearly 80% of the residential stock in North America, Oceania and Africa, concrete was chosen in the rest of the world. The share of concrete structures reaches nearly 60% in Europe, China, India, ASEAN and most Asian countries residential buildings, as well as in Latin America, while wood and masonry are the secondary structures. The situation is more homogeneous in the non-residential sector. A majority of the stock relies on concrete structures - ahead of steel and masonry - except in North America. Along with those market shares, IEA (2018) specify regional material intensities for concrete and steel. The concrete material intensities provided by the IEA (2018) exceed more than 1000 kg/m² for residential in regions with concrete as the main building material. This is in agreement with Liu (2010) for China, Kleemann et al. (2016) for the Austrian capital, Müller (2006) for the Netherlands, Ortlepp et al. (2016) for Germany or Serrand et al. (2013) for the city of Orleans. However, the values observed in the literature vary and are mainly focused on European countries, which

prevents the comparison of data from the majority of regions. Tertiary material intensities of concrete are higher than residential in all regions in IEA (2018). Moreover, while concrete does not exceed 20% of the residential market in North America, Oceania and Africa, its share of the tertiary market in these regions is increasing significantly. It reaches 40% in North America, 60% in Oceania, and nearly 90% in Africa. Conversely, the market shares of steel show little differences in material intensities between regions.

IEA (2018) did not specify aluminum and copper consumption of the global building stock. Less data are available for those two materials than for concrete and steel in the literature. Dong et al. (2019) estimated dynamic residential and non-residential copper intensities from 1980 to 2050 for China. Koutamanis et al. (2018) compiled copper intensities in kg/capita from Drakonakis et al. (2007); van Beers and Graedel (2007); Wittmer et al. (2007); Huang et al. (2013); Cochran et al. (2007); Mália et al. (2013) and Li et al. (2013) for Florida, the USA, China, Korea, Norway, Switzerland, Australia, New Heaven, and Cape Town. The authors showed significant differences between buildings, even within the same perimeter and emphasize the difficulty to estimate copper intensity over large areas. Schipper et al. (2018) reported material intensities of copper in residential building range from 0.92 to 3.18 kg/m², and from 0.46 to 1.58 kg/m² for non-residential buildings. Copper is consumed in the building sector for wiring, air conditioning, plumbing and roof building (Koutamanis et al., 2018). This large range of uses and their regional disparities explain the significant heterogeneity observed in the copper intensities. Residential aluminum intensity varies between 0.37 kg/m² and 0.73 kg/m² in Vienna, Austria (Kleemann et al., 2016), and between 0.6 kg/m² and 3.6 kg/m² in France depending on the type of building (ADEME et al., 2018). Hong et al. (2016) estimated dynamic residential and commercial aluminum intensity from 2010 to 2025 in China, while Marinova et al. (2020) and Deetman et al. (2020) provided values for both the residential and the non-residential sectors at a multi-regional scale. In order to show the link between material intensities and living standards, copper and aluminum intensities were differentiated with GDP/capita values in this study. The ten selected areas were segmented in three groups depending on their GDP/capita value in 2020. Maximum values were estimated based on the data summarized in Table S4. They were considered for the first group, the minimum value for the third group and an intermediate value was estimated for the second group. As described in Schipper et al. (2018), the non-residential copper intensity are half the residential values. The current regional values finally selected in this study are summarized in the Table S5.

References	Aluminum	Copper	Year	Area	Building type
Ortlepp et al. (2018)	1.07			Germany	Non-residential
Ortlepp et al. (2016)				Germany	Residential
Gontia et al. (2018)		3.3		Sweden	Residential
Hong et al. (2016)	2.02		2010	China	Residential
Hong et al. (2016)	2.2		2025	China	Residential
Hong et al. (2016)	4.9		2010	China	Commercial
Hong et al. (2016)	5.34		2025	China	Commercial
Marinova et al. (2020)	2.53	0.94		Average of studies for various countries	Residential
Deetman et al. (2020)	4.2	3.2		Average of studies for mainly developed countries	Non-residential
Dong et al. (2019)		0.11	1980	China	Residential
Dong et al. (2019)		0.43	2010	China	Residential
Dong et al. (2019)		0.59	2020	China	Residential
Dong et al. (2019)		0.88	2050	China	Residential
Dong et al. (2019)		0.05	1980	China	Non-residential
Dong et al. (2019)		0.21	2010	China	Non-residential
Dong et al. (2019)		0.3	2020	China	Non-residential
Dong et al. (2019)		0.43	2050	China	Non-residential
ADEME et al. (2018)	0.6	0.7	2020	France	Residential
ADEME et al. (2018)	3.6	0.6	2020	France	Non-residential
Kleemann et al. (2016)	0.73	0.37	1970	Austria	Residential
Kleemann et al. (2016)	0.97	1.1	2003	Austria	Hospital
Schipper et al. (2018)		0.92-3.18		Previous studies for various countries	Residential
Schipper et al. (2018)		0.46-1.59		Previous studies for various countries	Non-residential

Table S4: Summary of buildings residential and non-residential materials intensities (kg/m²).

Building type	Material	North America	Europe	China	India	ASEAN	Africa	Latin America	Other Asia	Middle East	OECD Pacific
R	Concrete	400	1000	1100	1100	1100	400	1100	1100	1100	1000
	Steel	12	37	39	37	37	14	37	37	37	33
	Copper	3.2	3.2	0.6	0.3	0.6	0.3	0.6	0.6	0.6	3.2
	Aluminum	1.3	1.3	2	0.36	0.8	0.36	0.8	0.8	0.8	1.3
NR	Concrete	900	1200	1500	1200	1400	1400	1400	1300	1300	1300
	Steel	70	72	71	68	76	71	74	68	68	69
	Copper	1.6	1.6	0.3	0.15	0.3	0.15	0.3	0.3	0.3	1.6
	Aluminum	4.2	4.2	4.2	1.2	2.7	1.2	2.7	2.7	2.7	4.2

Table S5: Summary of the current residential and non-residential material intensities considered in this study in kg/m². R = Residential, NR = Non-Residential.

Dynamic materials intensities were then modeled using a logistic function depending on GDP (eq S5). An initial value of 120 kg/m² was assumed for concrete in 1900, in accordance to values observed in Müller (2006); Ortlepp et al. (2016) and Gontia et al. (2018). Initial values of 0.05 kg/m² were considered for aluminum and copper, and 2 kg/m² and 10 kg/m² for residential and non-residential steel in 1900. A better knowledge of past steel, copper and aluminum intensities are needed to design their evolution over time. The saturation levels considered are slightly higher than the current values of Europe for concrete, with 1,200 kg/m² for residential buildings in 2100. A renewal of residual non-concrete buildings was assumed, and a small increase of concrete market share in North America. Given the lack of data, the same level than in the North America was considered in Africa for residential concrete intensity. Other materials were assumed as less structural for building, and thus higher saturation levels are observed. The same saturation levels were considered for the nine selected areas.

$$I_t = \frac{I_{sat}}{1 + \frac{I_{sat}}{I_{1900}} \cdot \exp(-\beta(GDP_t - GDP_{1900}))} \quad (S5)$$

In this equation, I_t is the material intensity in the year t , I_{sat} and I_{1900} are respectively the saturation level and the 1900 value of the material intensity, β is a parameter influencing the growth rate, GDP_t is the GDP value in the year t and GDP_{1900} is the 1900 value of GDP. The final results are displayed in Figures S4a to S4d for regional residential buildings, in Figures S5a to S5d for regional non-residential buildings. A higher steep of material intensities is observed for residential copper and non-residential aluminum. It lies in the assumption of almost no demand in 1900 and in the larger saturation values.

S2.6 Recycled aggregates concrete (RAC)

As described in the section S1.2, two types of recycled aggregates from construction and demolition waste (CDW) are considered: recycled aggregates (RA) and recycled concrete aggregates (RCA). The different applications of RA are described in Reis et al. (2021), and the authors outlined the growing number of encouraging pilot projects using recycled aggregate concrete (RAC). Silva et al. (2019) further provided a review of 31 constructions projects to demonstrate the technical and economic viability of recycled aggregates use. Among these projects, 11 are related to structural use of concrete in buildings. They carry substantial benefits through four reductions : financial, primary resources need, accumulation in landfills and environmental impacts. The RAC research field has been growing over the past years, and the specific impact of replacing natural aggregates by recycled concrete aggregates (RCA) on the properties of RAC was under focus. RCA is

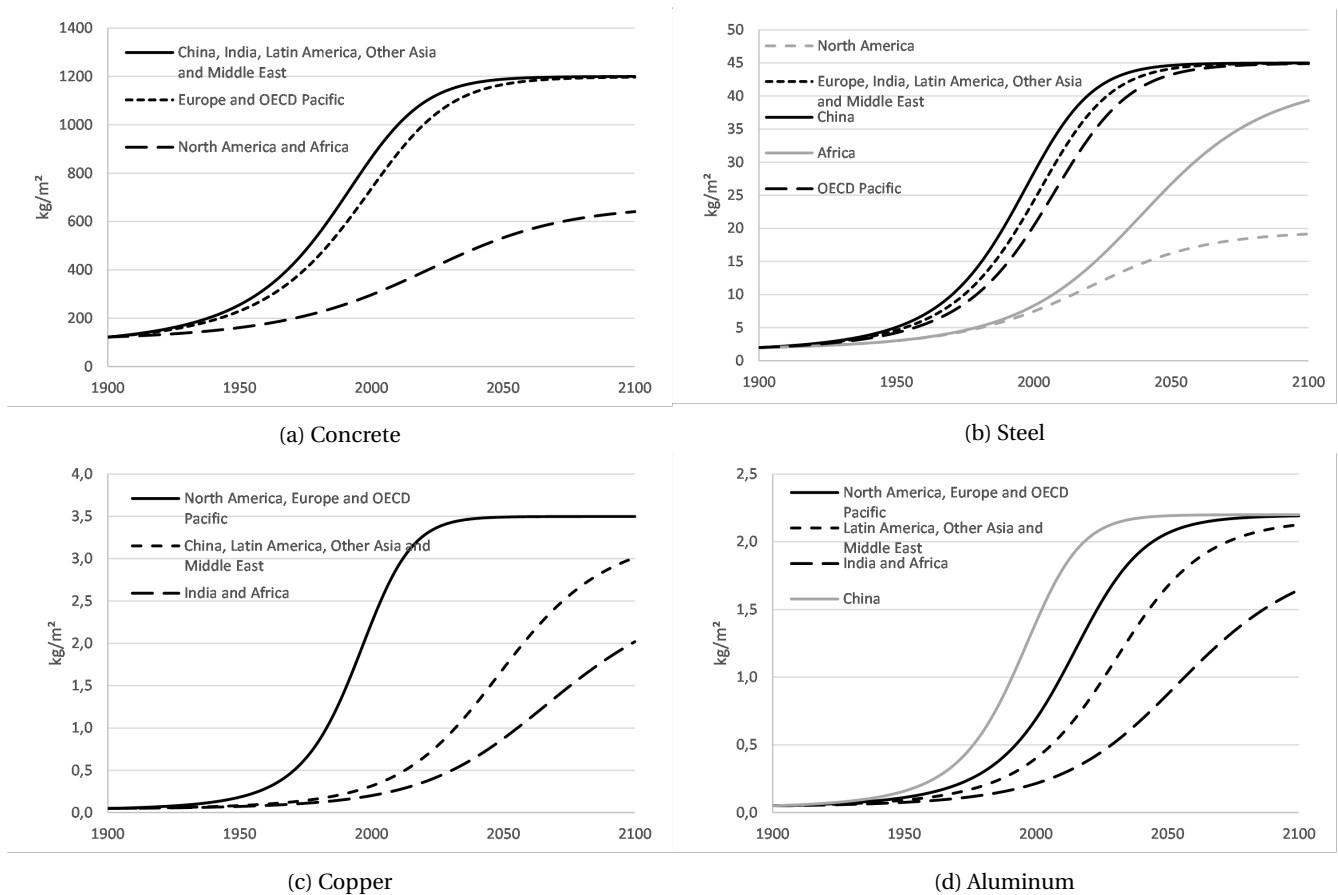


Figure S4: Dynamic and regional material intensities in concrete, steel, copper and aluminum in the residential building sector.

defined as aggregates recovered from concrete in which other materials represent less than few percents (Marinković and Carević, 2019), and it can be used for higher strength concrete than RA (Purnell and Dunster, 2010). However, Behera et al. (2014) showed lower mechanical and durability performance of RAC, and emphasized the need of further assessment of RA quality and its impact on concrete. In reviews on the RCA impacts on RAC production, Silva et al. (2014) and Verian et al. (2018) provided recommendations for an optimal use of recycled concrete aggregates (RCA) in concrete production. The authors showed different challenges depending on the use of fine and coarse aggregates¹⁸, but a potential was identified for both types. The authors further reported that some researches estimate a 30% limit in the use of coarse RCA in concrete, but state that there could be no limit in its potential share if specific process conditions are respected¹⁹. Thomas et al. (2013) further outlined the need for more assessment of RAC in different exposure conditions. For moderate conditions, Thomas et al. (2018) estimated that a 25% replacement of NA by RCA displays viable strength results. Guo et al. (2018) and Verian et al. (2018) further identified the potential to enhance recycled aggregate concrete performances, using for example CO₂ treatment and pozzolanic materials (e.g. fly ash).

S2.6.1 RCA replacement scenarios methodology

A significant increase in the EOL-CR of CDW was observed over the past decade in Europe (Villoria Sáez and Osmani, 2019). However, most of other regions display lower rates (Tam et al., 2018; Reis et al., 2021), and most of the CDW are disposed in landfills. As estimated in Section To be done, the concrete lost stock amounted to about 45 Gt in 2020 and

¹⁸Bravo et al. (2015) found that fine aggregates use hampers concrete performance, but that a limited share of coarse aggregates is viable. This higher impact of fine RCA was also highlighted in Guo et al. (2018).

¹⁹Coarse and fine aggregates are defined by the size of their particles.

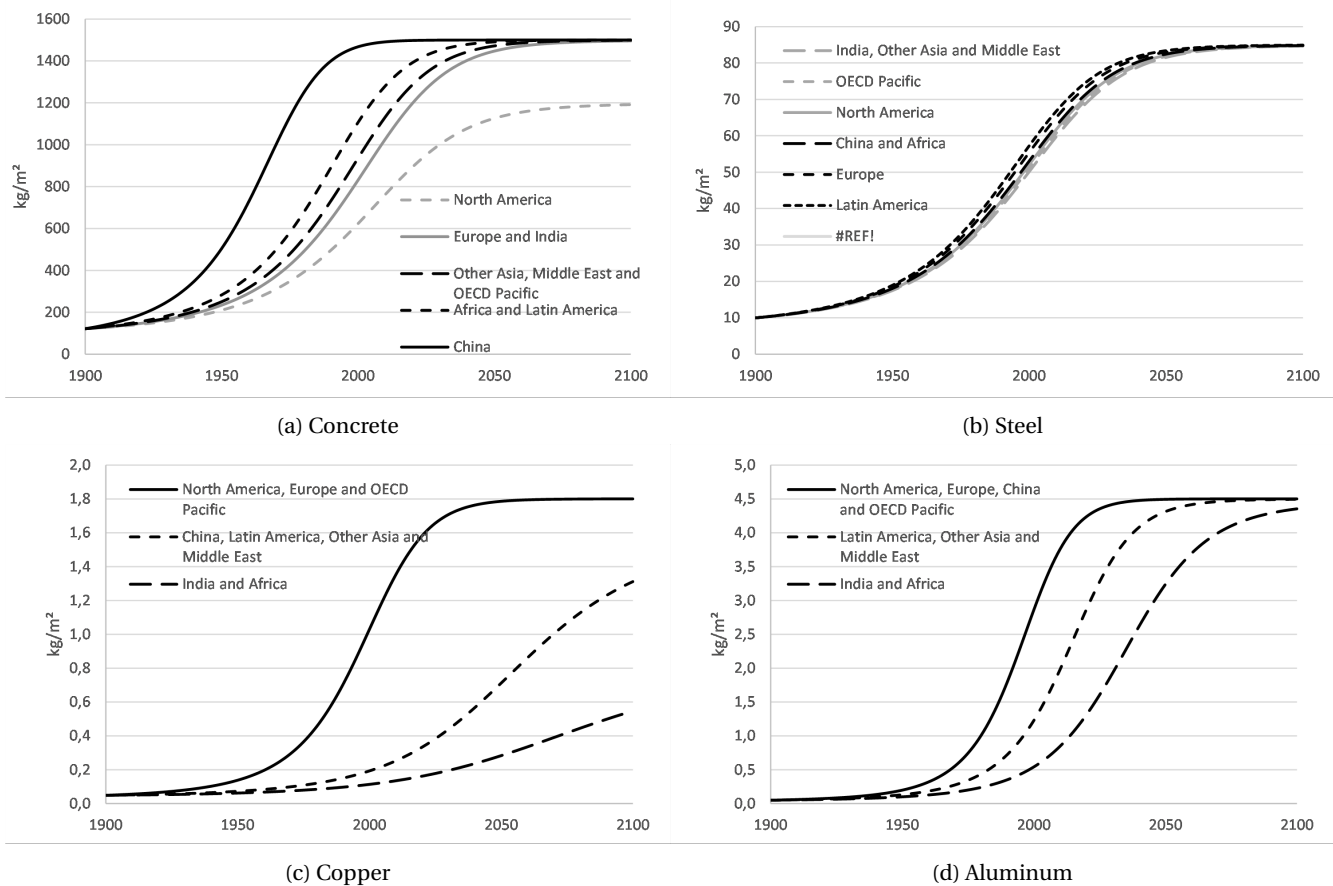


Figure S5: Dynamic and regional material intensities in concrete, steel, copper and aluminum in the non-residential building sector.

could reach 115 Gt in 2060 in the B2D and RT scenarios. The recycling of this accumulated concrete would allow to decrease the dependency on primary mineral production and a reduction of the storage burden. To provide a first-order estimate of the impact of a mass use of RCA replacement in concrete production on lost stock reduction, a methodology was developed and computed on the building stock of the [IEA \(2017\)](#) scenarios. It is based in the following assumptions :

1. No RCA replacement is observed at large scales until 2025;
2. It was assumed that both fine and coarse natural aggregates could be replaced by their related recycled aggregates;
3. The potential shares of RCA in RAC described in [Verian et al. \(2018\)](#); [Tam et al. \(2018\)](#) and [Thomas et al. \(2018\)](#) were used to develop a business-as-usual case and four scenarios of RCA replacement (Figure S6) :
 - Business-as-usual case : 0% of RCA replacement from 2025 to 2060
 - Replacement scenario 1 (RS1) : progressive increase from 0% in 2025 to 20% of RCA replacement in 2060
 - Replacement scenario 2 (RS2) : progressive increase from 0% in 2025 to 50% of RCA replacement in 2060
 - Replacement scenario 3 (RS3) : progressive increase from 0% in 2025 to 20% of RCA replacement in 2040
4. A share of 60% to 75% of aggregates in concrete was considered²⁰.
5. Most of the current annual recovered flow annual of concrete is used in non-structural applications and the quality of the remaining wastes is unknown. It was assumed that RCA will be recovered from the concrete lost stock. However, RCA represent a small share of CDW, as most of the wastes are RA ([Purnell and Dunster, 2010](#)), but no

²⁰The Portland Cement Association reports a rate of aggregates in concrete ranging from 60% to 75%.

estimation of the global overall share of RCA in CDW is available to the author's knowledge. Further estimations of RCA share in CDW will allow to calibrate the potential share of concrete inflow on which scenarios can be applied.

- The scenarios of RCA replacement are considered on 20%, 50% and 100% of the concrete inflow to exhibit the limited geographical scale of RCA replacement in landfills.

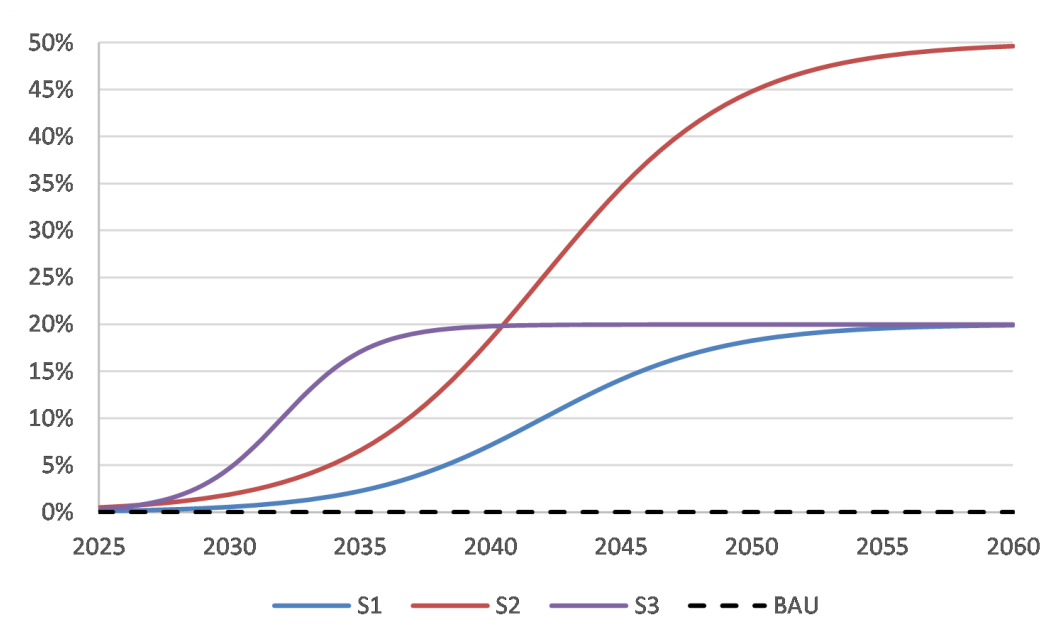


Figure S6: RCA replacement shares in the BAU case and the three replacement scenarios considered.

S2.7 "Green" concrete

S2.7.1 Fly ash generation

Coal combustion products (CCP) are the residues produced during the combustion of coal to generate electricity. They include fly ash, bottom ash, boiler slag and flue gas desulfurization materials (Kelly and Matos, 2017). Coal fly ash (CFA) represents most of the residues, and is mainly recycled to produce concrete (Luo et al., 2021; Marinina et al., 2021). Given that CFA is produced during with power generation, most countries display ash dumps and CFA is recycled for a local use. Marinina et al. (2021) reported various rate of use of the generated fly ash among countries, ranging from 10% in Russia and Africa to more than 90% in France, Japan, Germany and the UK. The future use of CFA to decrease concrete environmental impacts depends on national energy policies. A gradual shift from fossil fuels to renewable in the electricity mix will directly impact fly ash generation. CFA generation assessment in energy transition scenarios could therefore provide insights in the future viability of using CFA to decrease concrete environmental impacts. In this section, an estimation of CFA production is provided until to 2060 in the B2D and RT scenarios. It relies on :

- Historical fly ash generation data for China (Luo et al., 2021), India (Bhawan and Puram, 2020), the USA (Kelly and Matos, 2017; AACA, 2016, 2017, 2018, 2019) and the UK (UKQAA, 2016);
- Historical coal demand (BP, 2021);
- Regional or global coal demand assumptions in the B2D and RT scenarios.

Historical fly ash generation intensity was firstly evaluated for the selected areas. They provide evolution of the CFA produced per coal consumed. The countries display values ranging from about 2.5 Mt/EJ for the USA to more than 11 Mt/EJ for India, and a stabilization of their values are observed since 2005. Given that 72% of coal was consumed by those

countries in 2020 (Figure S7), these intensities allow to depict most of the global fly ash generation. The projected CFA generation was then estimated based on the evolution of coal demand in the B2D and RT scenarios (IEA, 2020, 2021) and constant future intensities were assumed until 2050. Their selected values are :

- 11 Mt/EJ for India;
- 6 Mt/EJ for China;
- 4 Mt/EJ for the UK;
- 2.5 Mt/EJ for the USA;
- 7 Mt/EJ for the rest of the world. This value was calibrated using the CFA production in 2016 reported in Jin et al. (2021).

The estimated CFA production between 2010 and 2050 is displayed in Figure S9. It shows a stabilization of CFA generation at about 1,150 Mt between 2010 and 2060 in the RTS. The reduction of coal demand in the B2DS substantially impact the CFA production. It drops to 530 Mt in 2030, and finally 250 Mt in 2060. At a regional level, the results are displayed in Figure S11 for China, Figure S12 for India and Figure S10 for the USA between 2002 and 2060 in the RT and B2D scenario. A significant increase of the fly ash production is observed between 2002 and 2020 in China. It raised from 180 Mt to 580 Mt, and then experiences a reduction to 370 Mt in 2060 in the RT scenario, and 90 Mt in the B2DS. An almost monotonous rise from 83 Mt in 2002 to 490 Mt in 2060 is showed for India in the RT scenario, while it decreases to 60 Mt in 2060 in the B2D scenario. The USA production decreases from 69 Mt in 2002 to 30 Mt in 2060 in the RT scenario, and drops to 7 Mt in 2060 in the B2D scenario (Figure S9). The utilization rate of CFA reached more than 66% in China in 2015 (Luo et al., 2021), 54% in the USA (Kelly and Matos, 2017) and 56% in India (Bhawan and Puram, 2020). Globally, this rate was only 25% in the early 2000s (Iyer and Scott, 2001), and experienced a growth to 60% in 2016 (Jin et al., 2021). Based on this evolution, a global utilization rate was estimated between 2002 and 2019 (Figure S13). Finally, the accumulation of unused CFA was calculated between 2002 and 2019 (Figure S14), and the initial lost stock in 2002 is unknown. It was found that the accumulated fly ash in landfills increased by 9.2 Gt between 2002 and 2019, which represents 15.5 years of current use, with strong regional disparities. The additional stock reached 2.8 Gt in China during the period, 1.1 Gt in India, 0.5 Gt in the USA and 4.8 Gt in the rest of the world (Figure S14). In the perspective of a progressive decrease in coal consumption, these stocks could become valuable reserves. They could provide materials to the industries many years after the shutdown of coal-fired power plants. Given that the CFA intensities are based on a stabilization assumption, the results carry high uncertainties. Moreover, lower values were observed in the past²¹ and no reliable retrospective CFA generations can be estimated from the intensities.

²¹The coal ash production in 1992 by country is provided Manz (1997)

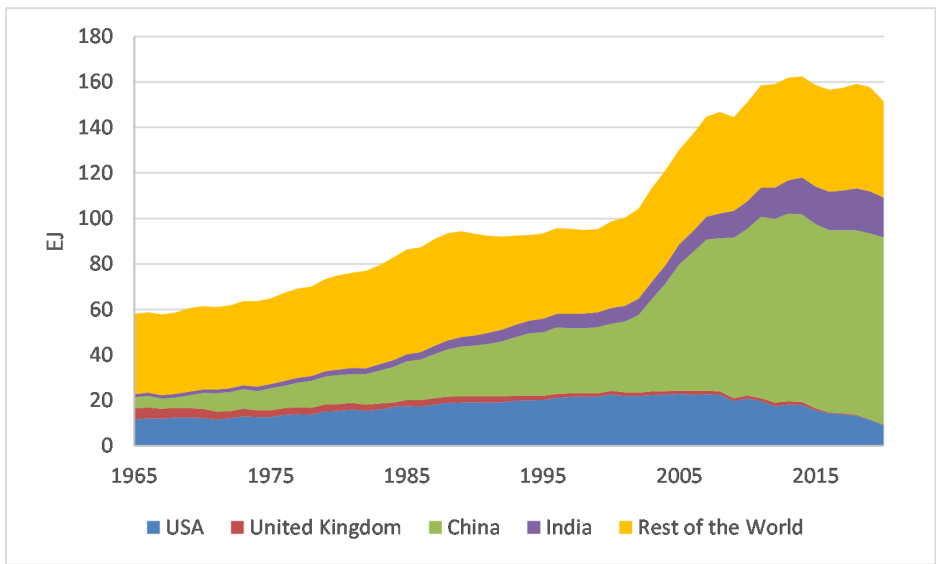


Figure S7: Historical coal consumption (BP, 2021)

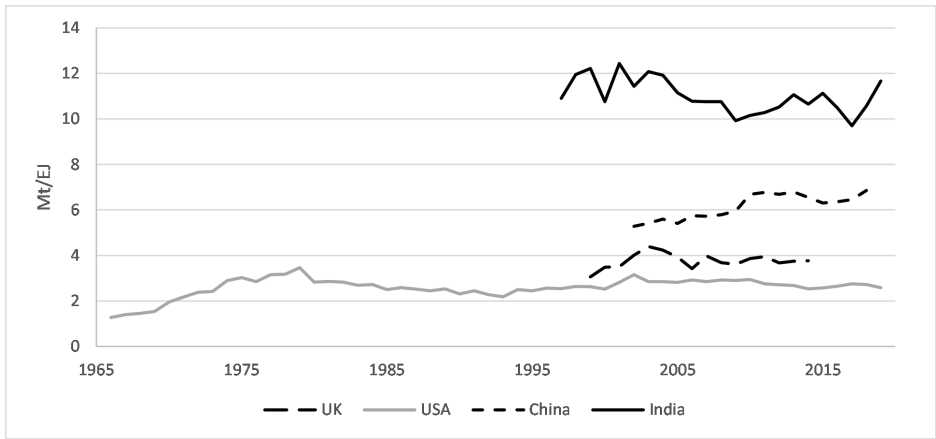


Figure S8: Historical regional coal fly ash generation per EJ of coal consumed. Estimated from (UKQAA, 2016; Luo et al., 2021; Kelly and Matos, 2017; AACA, 2017, 2018, 2019; Bhawan and Puram, 2020) and (BP, 2021).

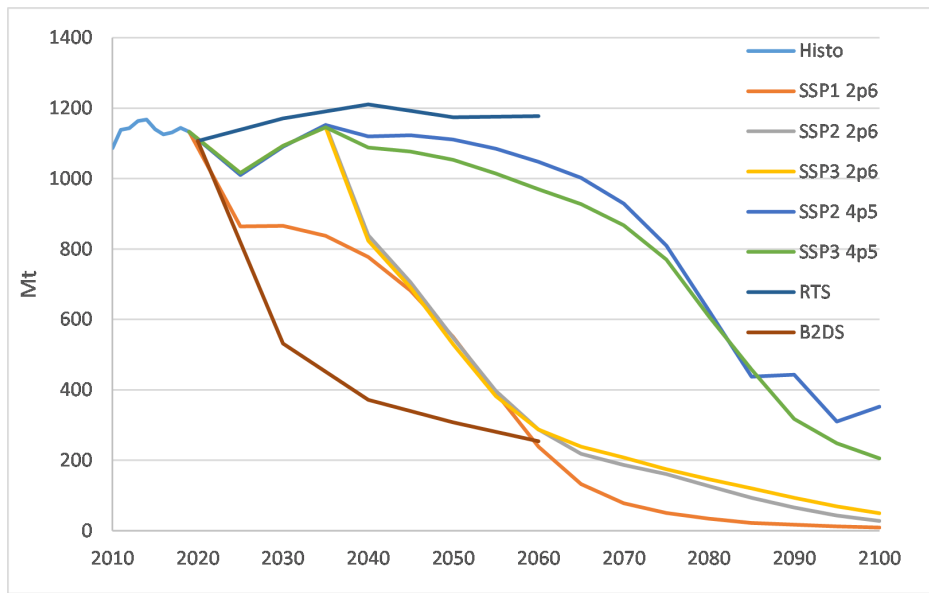


Figure S9: CFA production between 2010 and 2100. Estimated historical global data are displayed between 2010 and 2019 and prospective results are showed for the SSPs and the RTS and B2DS between 2020 and 2100 (or 2060 for the RTS and B2DS).

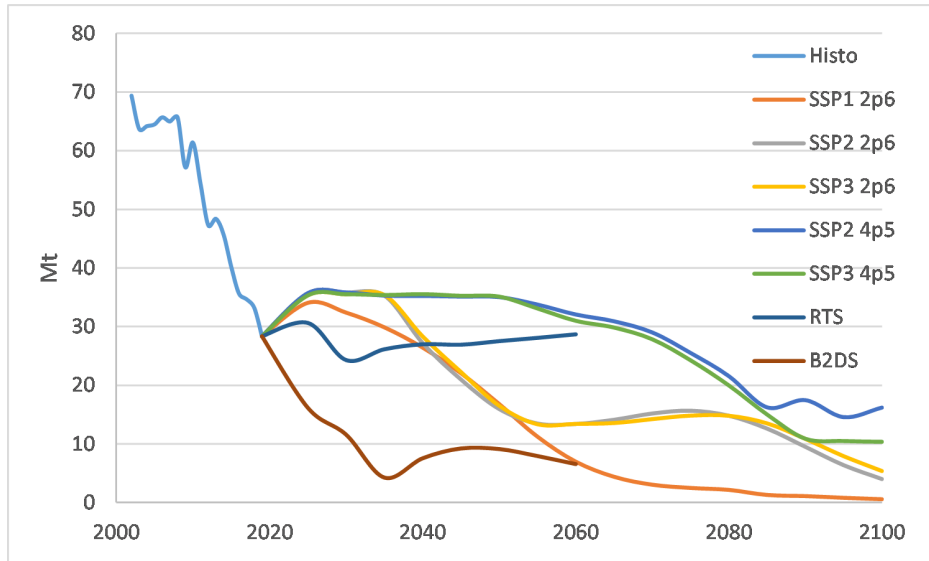


Figure S10: Estimated CFA production in the USA between 2002 and 2100. Historical data are displayed between 2002 and 2015, estimated historical data are depicted between 2016 and 2019, and modeled data are showed between 2020 and 2100.

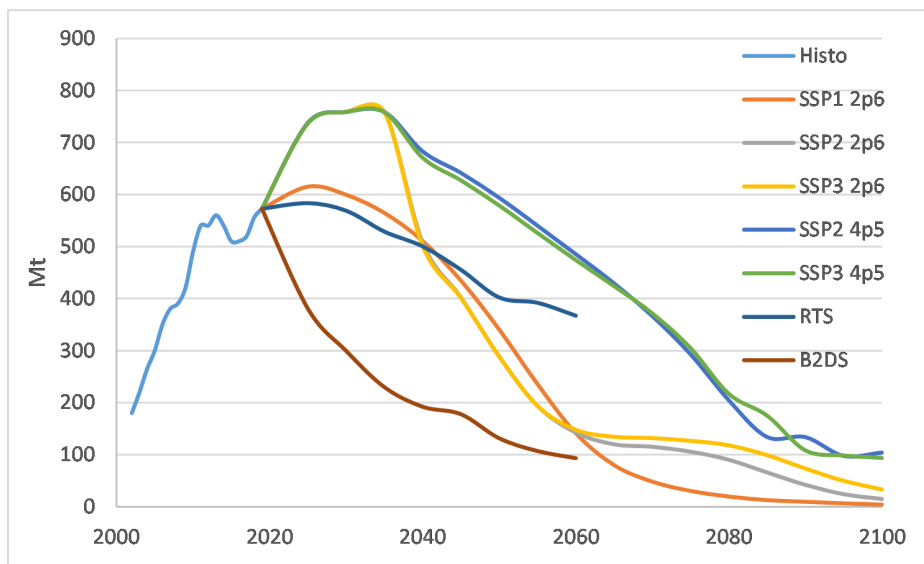


Figure S11: Estimated CFA production in China between 2002 and 2100. Historical data are displayed between 2002 and 2015 for the USA, estimated historical data are depicted in 2019 and 2019 and modeled data are showed between 2020 and 2100.

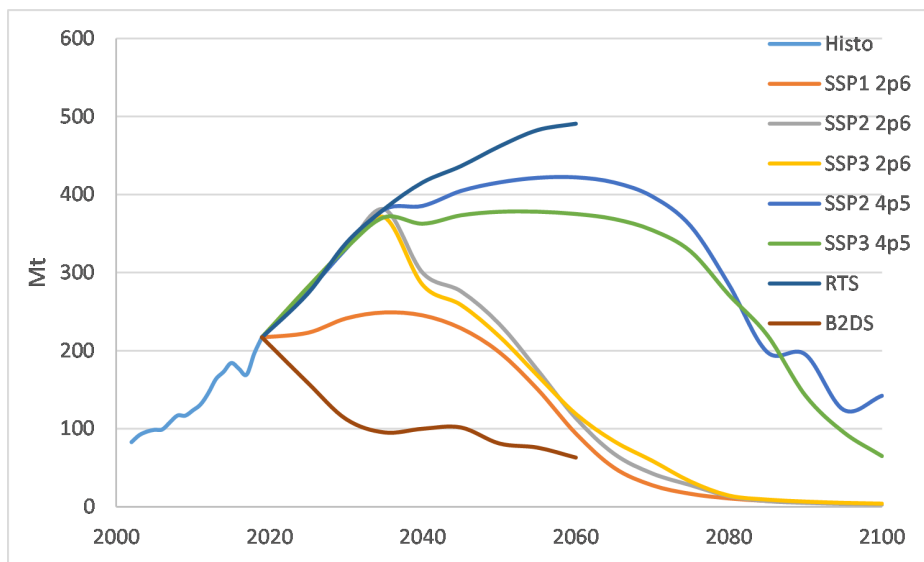


Figure S12: Estimated CFA production in India between 2002 and 2019. Historical data are displayed between 2002 and 2019 for India, estimated historical data are depicted in 2019 for India and modeled data are showed between 2021 and 2100.

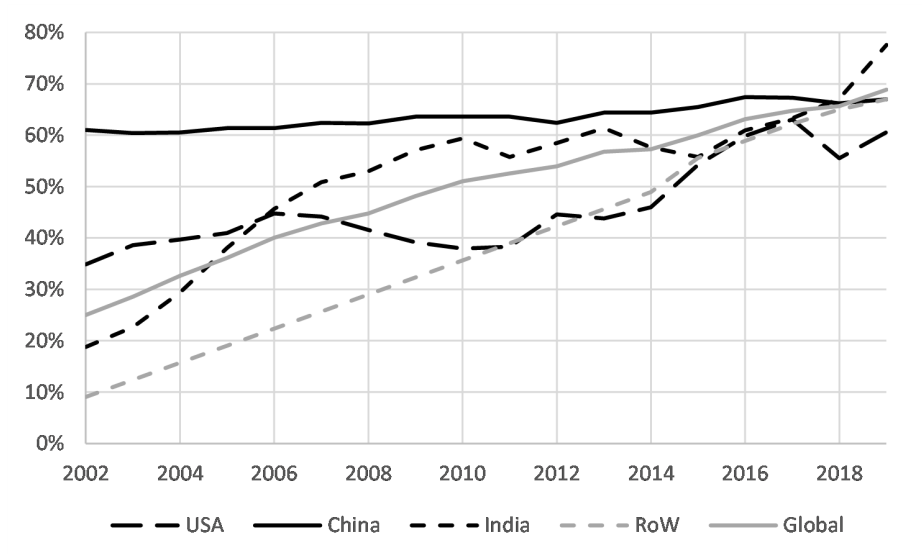


Figure S13: Historical utilization rate in China, India and the USA, and estimated values at the global level and for the rest of the world.

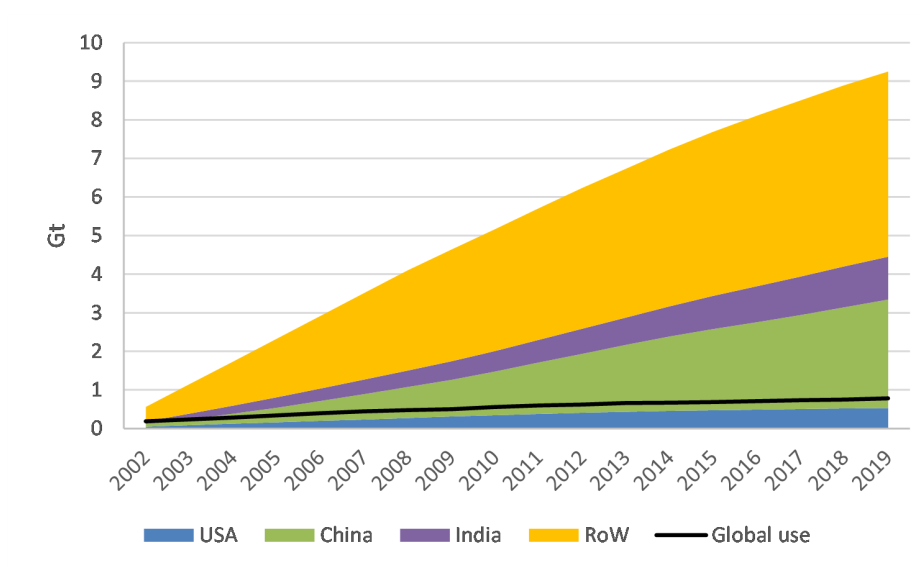


Figure S14: Estimated global additional stock of unused coal fly ash between 2002 and 2019.

S2.7.2 "Green" concrete composition

Three studies were considered to compute the energy and environmental effects of using by-products and/or recycled materials in concrete. The resulting materials is referred to as "green" concrete. Two types of replacements are considered in the studies :

- Replacement of natural aggregates, by various (fine and coarse) recycled concrete aggregates (RCA), fine RCA, coarse RCA or Electric Arc Furnace Steel Slag (EAFSS);
- Replacement of Ordinary Portland cement, by coal fly ash (CFA) or blast furnace slag (BFS).

Flower and Sanjayan (2007) considered CFA and BFS replacement, and found potential CO₂ emissions reduction by 22% for BFS and 13% to 15% for CFA. Turk et al. (2015) analysed natural aggregate replacement by EAFSS, cement replace-

ment by CFA, and combined EAFSS or CFA with RCA. The results display higher reductions for CFA replacement than for EAFSS, and a positive impact of an additional RCA use. Kurda et al. (2018) analyzed a large panel of single and combined replacements of fine RCA, coarse RCA and CFA (and additional superplasticizer not considered in this study). The authors showed significant reduction potential of CFA and a higher reduction when combined with coarse RCA than with fine RCA. The emissions decrease could reach 60% with a replacement CFA of 60% combined with a 100% coarse replacement. The results of the three studies are displayed in Table S6.

References	Concrete mix						Impact and potential reduction			
	Various RCA	Fine RCA	Coarse RCA	CFA	BFS	EAFSS	Production energy (MJ/t)	CO ₂ em. (kgCO ₂ /t)	Energy reduction	Emissions reduction
Kurda et al. (2018)							1,950	362		
		50%					1,940	361	0.5%	0.3%
		100%					1,940	360	0.5%	0.6%
				30%			1,580	269	19%	26%
		50%		30%			1,570	268	19%	26%
		100%		30%			1,560	267	20%	26%
				60%			1,210	176	38%	51%
		50%		60%			1,190	174	39%	52%
		100%		60%			1,190	174	39%	52%
				100%			1,530	331	22%	8.6%
		50%	100%				1,530	331	22%	8.6%
		100%	100%				1,530	330	22%	8.8%
				100%	30%		1,160	330	41%	34%
		50%	100%	30%			1,150	238	41%	34%
		100%	100%	30%			1,150	237	41%	35%
				100%	60%		783	145	60%	60%
		50%	100%	60%			781	145	60%	60%
		100%	100%	60%			779	144	60%	60%
Turk et al. (2015)							1,690	260		
				25%			1,300	195	23%	25%
						5.6%	1,400	250	17%	4%
		30%		25%			1,150	190	32%	27%
		28%				5.6%	1,230	242	27%	7%
Flower and Sanjayan (2007)							290-322			
				25%			253-273			13-15%
					40%		225-251			22%

Table S6: Production energy of various concrete mixes. RCA = Recycled Concrete Aggregates, CFA = Coal Fly Ash, BFS = Blast Furnace Slag and EAFSS = Electric Arc Furnace Steel Slag.

The energy and environmental impact of an increase of "green" concretes based on CFA, BFS and combined CFA and coarse RCA was computed in the B2D and RT scenario. The traditional concrete considered in Kurda et al. (2018) was used (Table S7)²². Four "green" concretes were computed (Table S8) and used in 20%, 50% and 100% of concrete demand.

²²A modification of a material used in a concrete mix display side effect on other contents (Kurda et al., 2018). For example, the fine aggregates

Materials	Fine aggregates	Coarse aggregates	Cement	Water	Total
	763	1,060	350	186	2,359

Table S7: Traditional concrete mix materials in kg for 1 m³ (Kurda et al., 2018)

Green concrete	Replacement	Production energy reduction	Emissions reduction
GC1	30% CFA	19%	26%
GC2	60% CFA	38%	51%
GC3	60% CFA and 100% coarse RCA	60%	60%
GC4	40% BFS		22%

Table S8: Considered replacements in "green" concretes.

S3 Sensitivity analysis

A local sensitivity analysis was conducted in this study. The material intensities of buildings and energy intensities of material production have linear effect in the computation. Therefore, the sensitivity analysis was only conducted on lifetime and recycling rates.

S3.1 Lifetime

Two types of sensitivity were studies on lifetimes. First, we assumed variations of $\pm 20\%$ of the mean lifetime in the normal distribution. We observe significantly higher differences in the p 20% case than in the 20% case. It is explained by the combined effects of different material intensities among regions with various socioeconomic dynamics and the larger inflow required to feed the in-use stock in a stock-driven model.

composition of a concrete mix changes when CFA is used. In this study, no side effects are considered.

Raw materials	Scenarios	2040	2060	2100
Concrete	SSP1	17%	22%	32%
Concrete	SSP2	17%	20%	24%
Concrete	SSP3	20%	22%	21%
Concrete	SSP4	19%	24%	27%
Concrete	SSP5	14%	20%	30%
Steel	SSP1	8%	7%	-1%
Steel	SSP2	9%	6%	2%
Steel	SSP3	10%	6%	1%
Steel	SSP4	9%	7%	-3%
Steel	SSP5	7%	9%	1%
Aluminum	SSP1	9%	11%	10%
Aluminum	SSP2	9%	10%	8%
Aluminum	SSP3	11%	11%	5%
Aluminum	SSP4	10%	12%	6%
Aluminum	SSP5	7%	10%	10%
Copper	SSP1	10%	11%	13%
Copper	SSP2	10%	10%	9%
Copper	SSP3	12%	11%	8%
Copper	SSP4	12%	12%	9%
Copper	SSP5	8%	10%	13%

Table S9: Sensitivity results with an increase of 20% of the mean lifetime.

Raw materials	Scenarios	2040	2060	2100
Concrete	SSP1	-11%	-14%	-22%
Concrete	SSP2	-11%	-13%	-16%
Concrete	SSP3	-13%	-15%	-14%
Concrete	SSP4	-13%	-16%	-18%
Concrete	SSP5	-9%	-13%	-21%
Steel	SSP1	-7%	-7%	-2%
Steel	SSP2	-7%	-6%	-3%
Steel	SSP3	-9%	-7%	-2%
Steel	SSP4	-9%	-7%	0%
Steel	SSP5	-5%	-7%	-4%
Aluminum	SSP1	-7%	-8%	-10%
Aluminum	SSP2	-7%	-8%	-8%
Aluminum	SSP3	-8%	-9%	-5%
Aluminum	SSP4	-8%	-9%	-7%
Aluminum	SSP5	-5%	-8%	-11%
Copper	SSP1	-7%	-8%	-13%
Copper	SSP2	-7%	-8%	-8%
Copper	SSP3	-8%	-8%	-7%
Copper	SSP4	-8%	-9%	-8%
Copper	SSP5	-5%	-7%	-12%

Table S10: Sensitivity results with an decrease of 20% of the mean lifetime.

Secondly, we assumed an alternative methodological choice by considering a logit-based mean lifetime. We observe a small difference between scenarios but a significant difference between raw materials because of the considered evolution of raw material intensities (Figure S11). The detailed methodology to set dynamic mean lifetimes is described below.

Raw materials	Scenarios	2040	2060	2100
Concrete	SSP1	6%	9%	11%
Concrete	SSP2	6%	8%	9%
Concrete	SSP3	7%	8%	8%
Concrete	SSP4	6%	9%	9%
Concrete	SSP5	5%	9%	11%
Steel	SSP1	1%	1%	4%
Steel	SSP2	2%	1%	0%
Steel	SSP3	2%	0%	0%
Steel	SSP4	1%	0%	2%
Steel	SSP5	2%	2%	4%
Aluminum	SSP1	3%	5%	2%
Aluminum	SSP2	3%	4%	3%
Aluminum	SSP3	4%	4%	2%
Aluminum	SSP4	4%	4%	2%
Aluminum	SSP5	3%	5%	2%
Copper	SSP1	3%	4%	4%
Copper	SSP2	3%	4%	4%
Copper	SSP3	4%	3%	3%
Copper	SSP4	4%	4%	3%
Copper	SSP5	3%	4%	4%

Table S11: Sensitivity results with logit-based mean lifetime.

In order to obtain dynamic lifetimes, we use a Gompertz-modeled mean value over time and a regional normal distribution. The selected methodology thus intends to provide regional dynamic lifetimes in accordance with the previously depicted conclusions. However it does not consider local exceptions, such as a diminishing average lifetime in Norway described in [Brattebø et al. \(2009\)](#). As previously stated, the building stock was segmented into two categories : residential and non-residential. Due to a lack of data about the non-residential building stocks lifetime characteristics, similar values are considered for both categories. The methodology relies in three successive steps :

1. Estimating current average regional lifetimes;
2. Evaluating logistic-based profiles for those lifetime between 1900 and 2100;
3. Calibrating regional normal distributions with dynamic mean values.

The selected average lifetime were then made dynamic over time. A four-parameter logistic function was used (eq S1). The parameters were designed assuming initial lifetimes 30% lower than current values in 1900 and saturation values 10% higher than the current values (Table S2). The resulting lifetimes are displayed in Figure S15.

$$L_t = L_{sat} + \frac{L_{init} - L_{sat}}{1 + \frac{t}{t_i}^\beta} \quad (S1)$$

In this equation, L_t is the lifetime in the year t , L_{sat} is the saturation level of the lifetime, L_{init} its initial value, t_i is the inflection year and β influence the growth rate of the curve.

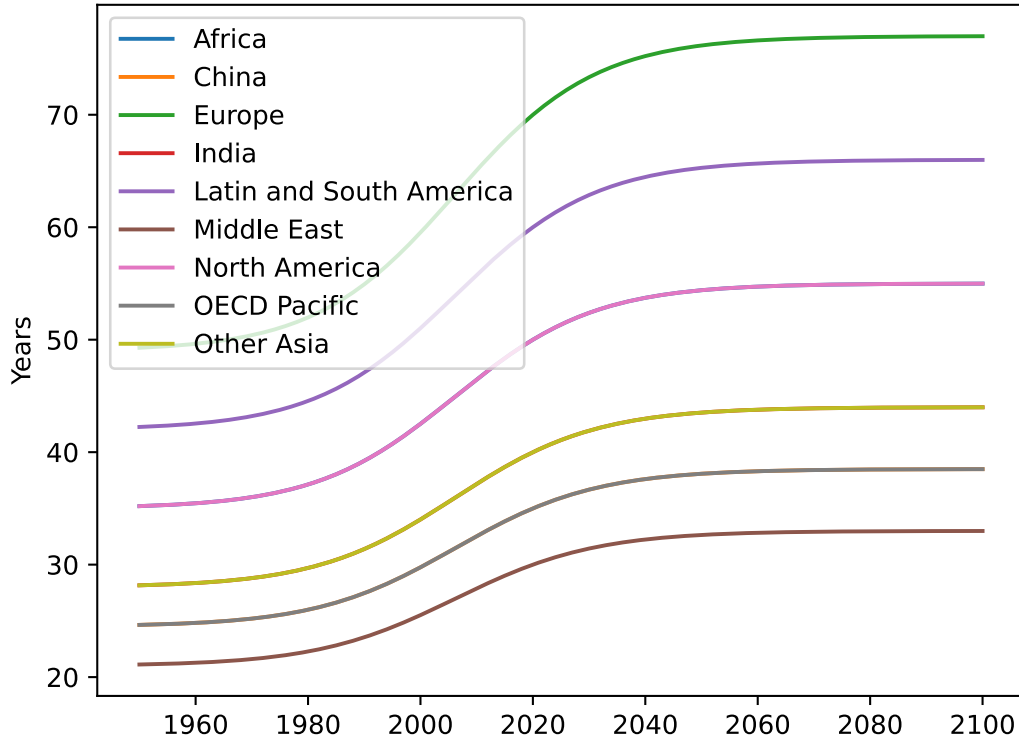


Figure S15: Regional average lifetimes between 1900 and 2100.

In a third and last step, normal distribution were applied on the logistic-based average lifetimes (eq S2). As depicted in Table S3, a wide panel of standard deviation values are used in the existing studies. It also shows an evolution with the mean value, and the ratio $\frac{\mu}{\sigma}$ of the selected studies ranges from 2 to 4.6. The data of Zhou et al. (2019) and Aktas and Bilec (2012) were considered, respectively for China and North America, because of their estimation from historical data. Given the lack of data for the other areas, an average value of $0.3 \cdot \mu$ was selected in this study. This is however a first order estimation, and it does not provide any overall pattern. Given the dynamic nature of the mean lifetimes, the standard deviation also varies over time.

$$f(L) = \frac{1}{\sigma \cdot \sqrt{2 \cdot \pi}} \cdot \exp\left(-\frac{1}{2} \cdot \left(\frac{L - \mu}{\sigma}\right)^2\right) \quad (S2)$$

With $f(L)$ the probability density function depending on the lifetime, σ the standard deviation and μ the mean lifetime. The final probability density functions of Europe, China, India and North America in 2000, 2020 and 2050 are displayed in Figures S16a to S16d.

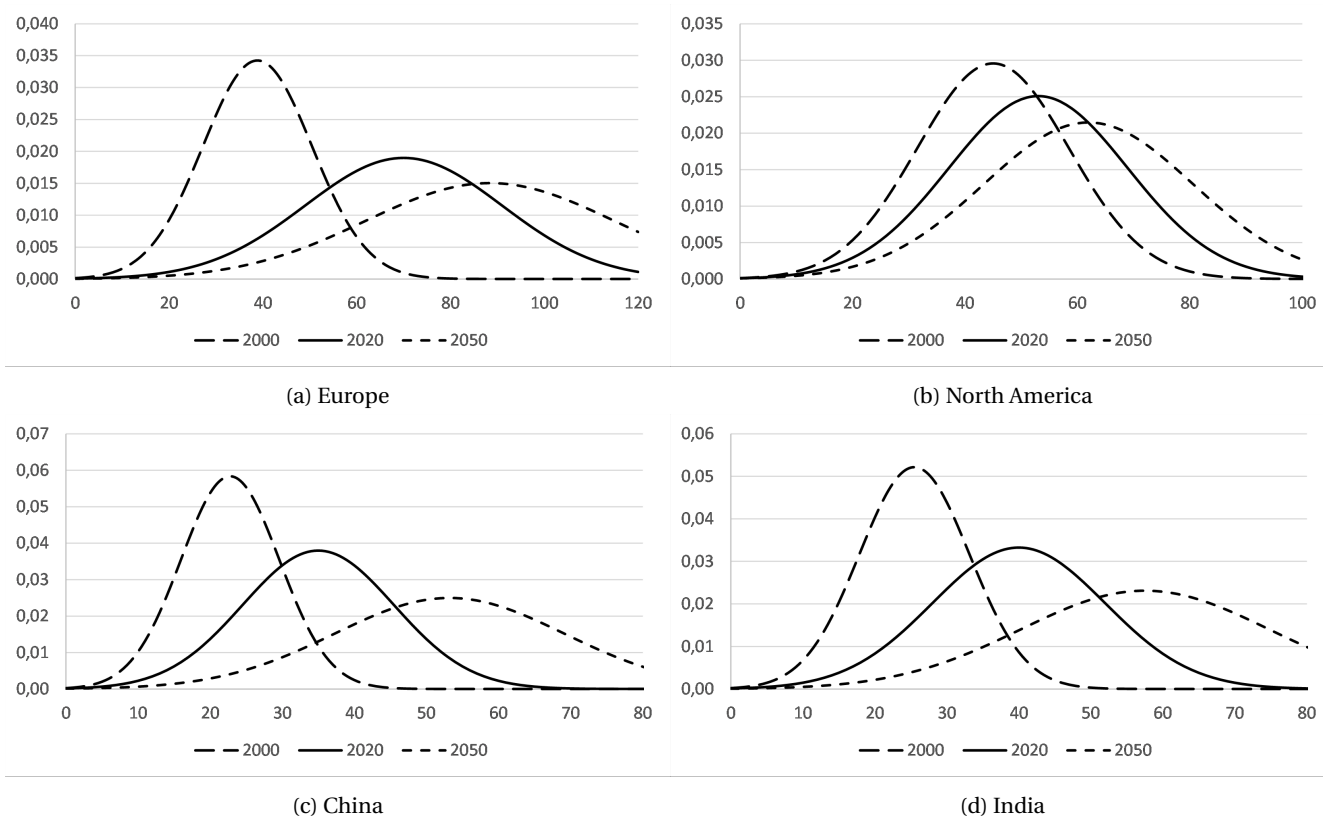


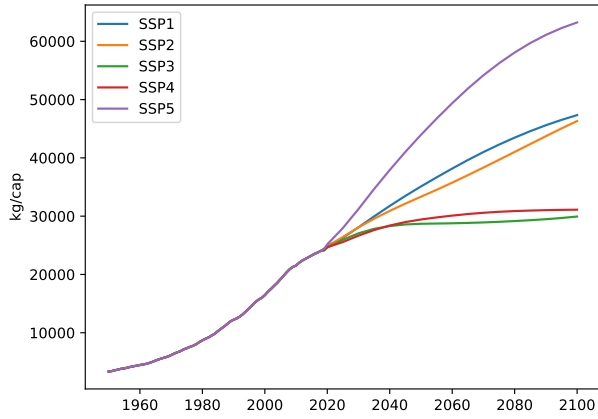
Figure S16: Normal distribution of the European, North American, Chinese and Indian buildings lifetimes in 2000, 2020 and 2050.

S4 Further results

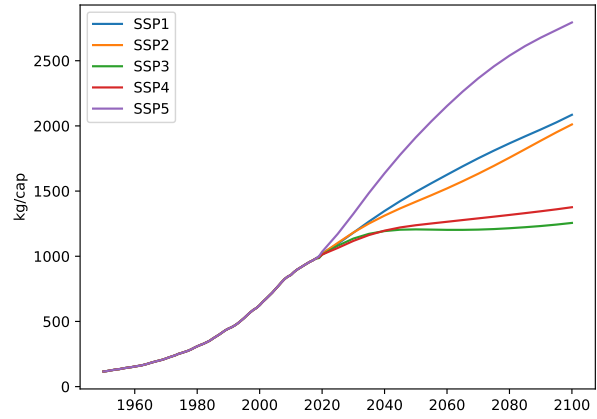
S4.1 Global in-use stocks

S4.2 Regional in-use stocks

S4.3 Lost and recovered stocks

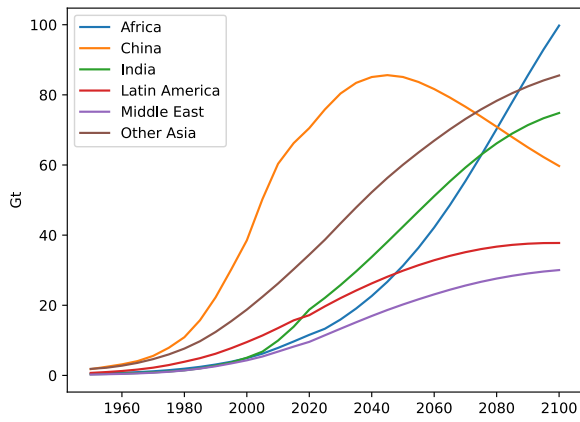


(a) Concrete

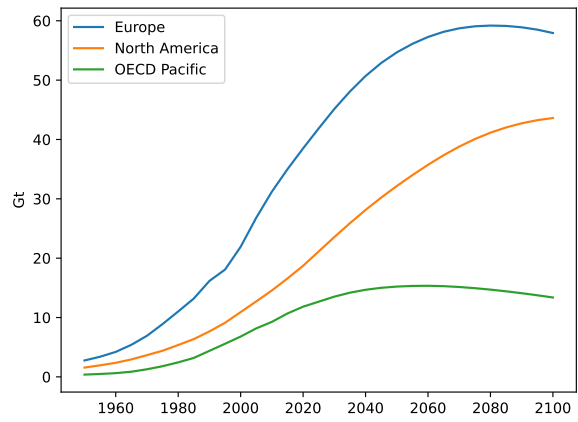


(b) Copper

Figure S17: Stock per capita of concrete and steel by scenarios in the building sector.



(a) Industrializing regions

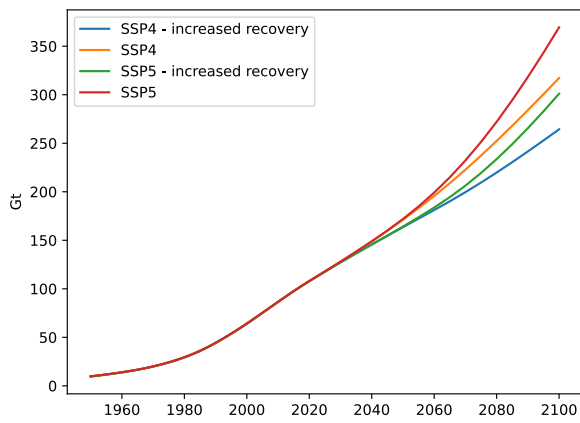


(b) Industrialized regions

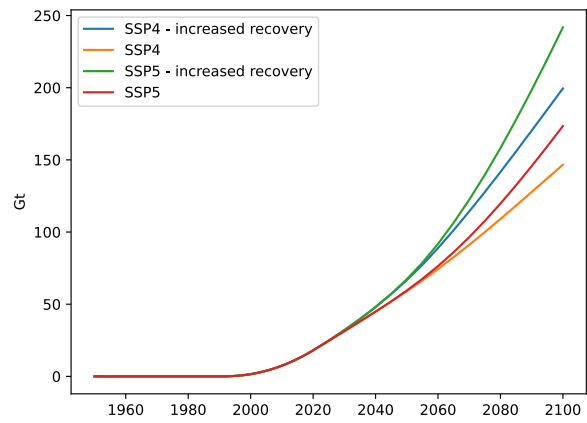
Figure S18: Total in-use stocks of base materials by regions in the building sector for the SSP2 scenario.

S5 Discussion

S5.1 Comparison of materials results with other studies



(a) Lost stock



(b) Recovered stock

Figure S19: Global concrete lost and recovered stocks in the building sector for the selected scenarios. Only the two scenarios with the lowest and the highest lost stocks are displayed. The filled lines show the lost stocks for increasing share of recovered concrete, and constant 2020 shares are presented by the dashed lines. The residential and non-residential segments are aggregated.

References	Year	Area	Perimeter	Scenario	Concrete	Steel	Copper	Aluminum
Stock (Gt)								
Pauliuk et al. (2013)	2008	Global	B&I			17		
Glöser et al. (2013)	2010	Global	B&I				0.2	
Hu et al. (2010)	2010	China	Residential			0.7		
Dong et al. (2019)	2015	China	Building				0.03	
This study	2018	China	Building		66	3	0.03	0.14
Marinova et al. (2020)	2018	Global	Residential		240	12	0.2	1.2
Deetman et al. (2020)	2018	Global	Building		267	15	0.31	1.3
This study	2018	Global	Building		211	9	0.27	0.39
Deetman et al. (2020)	2050	Global	Building	SSP2	406	27	0.56	1.8
Pedneault et al. (2022)	2050	Global	B&I	SSP2				1.07
This study	2050	Global	Building	SSP2	355	16	0.54	0.8
Flow (Mt/yr)								
Gao et al. (2018)	2010	China	Building			210		
Hu et al. (2010)	2010	China	Residential			56		
Dong et al. (2019)	2015	China	Building				2.2	
Yin and Chen (2013)	2010	China	Building			190		
This study	2018	China	Building		2,200	87	4.9	1.6
Moynihan and Allwood (2012)	2006	Global	Building			290		
Glöser et al. (2013)	2010	Global	B&I				7	
Deetman et al. (2020)	2018	Global	Building		8,900	600	12	44
This study	2018	Global	Building		8,100	330	16	10
Deetman et al. (2020)	2050	Global	Building	SSP2	11,900	769	18	52
Pedneault et al. (2022)	2050	Global	B&I	SSP2				20.8
This study	2050	Global	Building	SSP2	7,500	310	17	14

Table S12: Results of stocks and flows of bulk materials in previous studies. B&I is Building and Infrastructures

References

- AACA (2016): “Production & Use survey report,” Tech. rep., AACA.
- (2017): “Production & Use survey report,” Tech. rep., AACA.
- (2018): “Production & Use survey report,” Tech. rep., AACA.
- (2019): “Production & Use survey report,” Tech. rep., AACA.
- ADEME, P. LEONARDON, CSTB, S. LAURENCEAU, AND M. LOUERAT (2018): “Prospective de consommation de matière pour les bâtiments neufs aux horizons 2035 et 2050,” Tech. rep., ADEME, CSTB.
- AIMCC (2015): “GE04-Fin de vie des bâtiments,” Tech. rep., Ministère de la transition écologique.
- AKTAS, C. B. AND M. M. BILEC (2012): “Impact of lifetime on US residential building LCA results,” *The International Journal of Life Cycle Assessment*, 17, 337–349.
- ARORA, M., F. RASPALL, L. CHEAH, AND A. SILVA (2019): “Residential building material stocks and component-level circularity: The case of Singapore,” *Journal of Cleaner Production*, 216, 239–248.
- BEHERA, M., S. BHATTACHARYYA, A. MINOCHA, R. DEOLIYA, AND S. MAITI (2014): “Recycled aggregate from C&D waste & its use in concrete – A breakthrough towards sustainability in construction sector: A review,” *Construction and Building Materials*, 68, 501–516.
- BERGSDAL, H., R. A. BOHNE, AND H. BRATTEBØ (2008): “Projection of Construction and Demolition Waste in Norway,” *Journal of Industrial Ecology*, 11, 27–39.
- BERGSDAL, H., H. BRATTEBØ, R. A. BOHNE, AND D. B. MÜLLER (2007): “Dynamic material flow analysis for Norway’s dwelling stock,” *Building Research & Information*, 35, 557–570.
- BHAWAN, S. AND R. K. PURAM (2020): “Annual report,” Tech. rep., Central Electricity Authority, Ministry of Power, Government of India.
- BP (2021): “Statistical Review of World Energy 2021 - 70th edition,” Tech. rep., BP.
- BRADLEY, P. E. AND N. KOHLER (2007): “Methodology for the survival analysis of urban building stocks,” *Building Research & Information*, 35, 529–542.
- BRATTEBØ, H., H. BERGSDAL, N. H. SANDBERG, J. HAMMERVOLD, AND D. MÜLLER (2009): “Exploring built environment stock metabolism and sustainability by systems analysis approaches,” *Building Research & Information*, 37, 569–582.
- BRAVO, M., J. DE BRITO, J. PONTES, AND L. EVANGELISTA (2015): “Mechanical performance of concrete made with aggregates from construction and demolition waste recycling plants,” *Journal of Cleaner Production*, 99, 59–74.
- CAI, W., L. WAN, Y. JIANG, C. WANG, AND L. LIN (2015): “Short-Lived Buildings in China: Impacts on Water, Energy, and Carbon Emissions,” *Environmental Science & Technology*, 49, 13921–13928.
- CAO, Z., G. LIU, H. DUAN, F. XI, G. LIU, AND W. YANG (2019): “Unravelling the mystery of Chinese building lifetime: A calibration and verification based on dynamic material flow analysis,” *Applied Energy*, 238, 442–452.
- CAO, Z., L. SHEN, S. ZHONG, L. LIU, H. KONG, AND Y. SUN (2018): “A Probabilistic Dynamic Material Flow Analysis Model for Chinese Urban Housing Stock: A Probabilistic Dynamic MFA Model,” *Journal of Industrial Ecology*, 22, 377–391.
- CHEMILLIER, P. (1997): *Panorama des techniques du bâtiment 1947-1997*, Paris: Centre scientifique et technique du bâtiment (CSTB), oCLC: 636478229.
- CIACCI, L., W. CHEN, F. PASSARINI, M. ECKELMAN, I. VASSURA, AND L. MORSELLI (2013): “Historical evolution of anthropogenic aluminum stocks and flows in Italy,” *Resources, Conservation and Recycling*, 72, 1–8.
- COCHRAN, K., T. TOWNSEND, D. REINHART, AND H. HECK (2007): “Estimation of regional building-related C&D debris generation and composition: Case study for Florida, US,” *Waste Management*, 27, 921–931.
- DAVIS, J., R. GEYER, J. LEY, J. HE, R. CLIFT, A. KWAN, M. SANSOM, AND T. JACKSON (2007): “Time-dependent material flow analysis of iron and steel in the UK,” *Resources, Conservation and Recycling*, 51, 118–140.
- DEETMAN, S., H. DE BOER, M. VAN ENGELENBURG, E. VAN DER VOET, AND D. VAN VUUREN (2021): “Projected material requirements for the global electricity infrastructure – generation, transmission and storage,” *Resources, Conservation and Recycling*, 164, 105200.

- DEETMAN, S., S. MARINOVA, E. VAN DER VOET, D. P. VAN VUUREN, O. EDELENBOSCH, AND R. HEIJUNGS (2020): "Modelling global material stocks and flows for residential and service sector buildings towards 2050," *Journal of Cleaner Production*, 245, 118658.
- DIEFENDORE, J. M. (1989): "Urban Reconstruction in Europe After World War II," *Urban Studies*, 26, 128–143.
- DONG, D., A. TUKKER, AND E. VAN DER VOET (2019): "Modeling copper demand in China up to 2050: A business-as-usual scenario based on dynamic stock and flow analysis," *Journal of Industrial Ecology*, 23, 1363–1380.
- DRAKONAKIS, K., K. ROSTKOWSKI, J. RAUCH, T. GRAEDEL, AND R. GORDON (2007): "Metal capital sustaining a North American city: Iron and copper in New Haven, CT," *Resources, Conservation and Recycling*, 49, 406–420.
- EUROPEAN ENVIRONMENT AGENCY (2021): "Construction and demolition waste: challenges and opportunities in a circular economy," .
- EUROPEAN UNION (2017): "Construction and demolition waste," .
- FISHMAN, T., H. SCHANDL, H. TANIKAWA, P. WALKER, AND F. KRAUSMANN (2014): "Accounting for the Material Stock of Nations," *Journal of Industrial Ecology*, 18, 407–420.
- FLOWER, D. J. M. AND J. G. SANJAYAN (2007): "Green house gas emissions due to concrete manufacture," *The International Journal of Life Cycle Assessment*, 12, 282–288.
- GAO, C., H. NA, M. TIAN, Z. YE, AND Z. QI (2018): "Analyzing Structure and Driving Force of Steel Consumption in China," *Journal of Environmental Accounting and Management*, 14.
- GLÖSER, S., M. SOULIER, AND L. A. TERCERO ESPINOZA (2013): "Dynamic Analysis of Global Copper Flows. Global Stocks, Postconsumer Material Flows, Recycling Indicators, and Uncertainty Evaluation," *Environmental Science & Technology*, 47, 6564–6572.
- GONTIA, P., C. NÄGELI, L. ROSADO, Y. KALMYKOVA, AND M. ÖSTERBRING (2018): "Material-intensity database of residential buildings: A case-study of Sweden in the international context," *Resources, Conservation and Recycling*, 130, 228–239.
- GU, L. AND T. OZBAKKALOGLU (2016): "Use of recycled plastics in concrete: A critical review," *Waste Management*, 51, 19–42.
- GUO, H., C. SHI, X. GUAN, J. ZHU, Y. DING, T.-C. LING, H. ZHANG, AND Y. WANG (2018): "Durability of recycled aggregate concrete – A review," *Cement and Concrete Composites*, 89, 251–259.
- HABERT, G. (2014): "Assessing the environmental impact of conventional and 'green' cement production," in *Eco-efficient Construction and Building Materials*, Elsevier, 199–238.
- HASHIMOTO, S., H. TANIKAWA, AND Y. MORIGUCHI (2007): "Where will large amounts of materials accumulated within the economy go? – A material flow analysis of construction minerals for Japan," *Waste Management*, 27, 1725–1738.
- (2009): "Framework for estimating potential wastes and secondary resources accumulated within an economy – A case study of construction minerals in Japan," *Waste Management*, 29, 2859–2866.
- HATAYAMA, H., I. DAIGO, Y. MATSUNO, AND Y. ADACHI (2009): "Assessment of the Recycling Potential of Aluminum in Japan, the United States, Europe and China," *MATERIALS TRANSACTIONS*, 50, 650–656.
- HATAYAMA, H. AND K. TAHARA (2016): "Using decomposition analysis to forecast metal usage in the building stock," *Building Research & Information*, 44, 63–72.
- HEEREN, N., C. L. MUTEL, B. STEUBING, Y. OSTERMEYER, H. WALLBAUM, AND S. HELLWEG (2015): "Environmental Impact of Buildings—What Matters?" *Environmental Science & Technology*, 49, 9832–9841, publisher: American Chemical Society.
- HOLLAND, R., K. KURTIS, AND L. KAHN (2016): "Effect of different concrete materials on the corrosion of the embedded reinforcing steel," in *Corrosion of Steel in Concrete Structures*, Elsevier, 131–147.
- HONG, L., N. ZHOU, W. FENG, N. KHANNA, D. FRIDLEY, Y. ZHAO, AND K. SANDHOLT (2016): "Building stock dynamics and its impacts on materials and energy demand in China," *Energy Policy*, 94, 47–55.
- HU, M., S. PAULIUK, T. WANG, G. HUPPES, E. VAN DER VOET, AND D. B. MÜLLER (2010): "Iron and steel in Chinese residential buildings: A dynamic analysis," *Resources, Conservation and Recycling*, 54, 591–600.

- HUANG, B., F. ZHAO, T. FISHMAN, W.-Q. CHEN, N. HEEREN, AND E. G. HERTWICH (2018): "Building Material Use and Associated Environmental Impacts in China 2000–2015," *Environmental Science & Technology*, 52, 14006–14014.
- HUANG, T., F. SHI, H. TANIKAWA, J. FEI, AND J. HAN (2013): "Materials demand and environmental impact of buildings construction and demolition in China based on dynamic material flow analysis," *Resources, Conservation and Recycling*, 72, 91–101.
- IEA (2017): "Energy Technology Perspectives 2017," Tech. rep., IEA.
- (2018): "Global Status Report 2018," Tech. rep., IEA.
- (2020): "World total energy supply by source, 1971-2018," .
- (2021): "Net Zero by 2050 - A Roadmap for the Global Energy Sector," Tech. rep., IEA.
- IYER, R. S. AND J. A. SCOTT (2001): "Power station fly ash — a review of value-added utilization outside of the construction industry," *Resources, Conservation and Recycling*, 12.
- JIANG, Y., T.-C. LING, C. SHI, AND S.-Y. PAN (2018): "Characteristics of steel slags and their use in cement and concrete—A review," *Resources, Conservation and Recycling*, 136, 187–197.
- JIN, S., Z. ZHAO, S. JIANG, J. SUN, H. PAN, AND L. JIANG (2021): "Comparison and Summary of Relevant Standards for Comprehensive Utilization of Fly Ash at Home and Abroad," *IOP Conference Series: Earth and Environmental Science*, 621, 012006.
- JRC (2011): "Development of European Ecolabel and Green Public Procurement criteria for office buildings – Economical and market analysis," Tech. rep., JRC.
- KAPUR, A., G. KEOLEIAN, A. KENDALL, AND S. E. KESLER (2008): "Dynamic Modeling of In-Use Cement Stocks in the United States," *Journal of Industrial Ecology*, 12, 539–556.
- KELLY, T. D. AND G. R. MATOS (2017): "Mineral and Material Commodities in the United States," Tech. rep., USGS.
- KLEEMANN, F., J. LEDERER, P. ASCHENBRENNER, H. RECHBERGER, AND J. FELLNER (2016): "A method for determining buildings' material composition prior to demolition," *Building Research & Information*, 44, 51–62.
- KNOERI, C., E. SANYÉ-MENGUAL, AND H.-J. ALTHAUS (2013): "Comparative LCA of recycled and conventional concrete for structural applications," *The International Journal of Life Cycle Assessment*, 18, 909–918.
- KOBAYASHI, M. (2016): "The Housing Market and Housing Policies in Japan," *SSRN Electronic Journal*.
- KOUTAMANIS, A., B. VAN REIJN, AND E. VAN BUEREN (2018): "Urban mining and buildings: A review of possibilities and limitations," *Resources, Conservation and Recycling*, 138, 32–39.
- KURDA, R., J. D. SILVESTRE, AND J. DE BRITO (2018): "Life cycle assessment of concrete made with high volume of recycled concrete aggregates and fly ash," *Resources, Conservation and Recycling*, 139, 407–417.
- LI, J., Z. DING, X. MI, AND J. WANG (2013): "A model for estimating construction waste generation index for building project in China," *Resources, Conservation and Recycling*, 74, 20–26.
- LIU, J., M. WANG, C. ZHANG, M. YANG, AND Y. LI (2020): "Material flows and in-use stocks of durable goods in Chinese urban household sector," *Resources, Conservation and Recycling*, 158, 104758.
- LUO, Y., Y. WU, S. MA, S. ZHENG, Y. ZHANG, AND P. K. CHU (2021): "Utilization of coal fly ash in China: a mini-review on challenges and future directions," *Environmental Science and Pollution Research*, 28, 18727–18740.
- MANZ, O. E. (1997): "Worldwide production of coal ash and utilization in concrete and other products," *Fuel*, 6.
- MARININA, O., M. NEVSKAYA, I. JONEK-KOWALSKA, R. WOLNIAK, AND M. MARININ (2021): "Recycling of Coal Fly Ash as an Example of an Efficient Circular Economy: A Stakeholder Approach," *Energies*, 14, 3597.
- MARINKOVIĆ, S. AND V. CAREVIĆ (2019): "Comparative studies of the life cycle analysis between conventional and recycled aggregate concrete," in *New Trends in Eco-efficient and Recycled Concrete*, Elsevier, 257–291.
- MARINKOVIĆ, S., V. RADONJANIN, M. MALEŠEV, AND I. IGNJATOVIĆ (2010): "Comparative environmental assessment of natural and recycled aggregate concrete," *Waste Management*, 30, 2255–2264.
- MARINOVA, S., S. DEETMAN, E. VAN DER VOET, AND V. DAIIOGLOU (2020): "Global construction materials database and stock analysis of residential buildings between 1970-2050," *Journal of Cleaner Production*, 247, 119146.

- MARSH, R. (2017): "Building lifespan: effect on the environmental impact of building components in a Danish perspective," *Architectural Engineering and Design Management*, 13, 80–100.
- MIATTO, A., H. SCHANDL, AND H. TANIKAWA (2017): "How important are realistic building lifespan assumptions for material stock and demolition waste accounts?" *Resources, Conservation and Recycling*, 122, 143–154.
- MOYNIHAN, M. C. AND J. M. ALLWOOD (2012): "The flow of steel into the construction sector," *Resources, Conservation and Recycling*, 68, 88–95.
- MURAKAMI, S., M. OGUCHI, T. TASAKI, I. DAIGO, AND S. HASHIMOTO (2010): "Lifespan of Commodities, Part I: The Creation of a Database and Its Review," *Journal of Industrial Ecology*, 14, 598–612.
- MÁLIA, M., J. DE BRITO, M. D. PINHEIRO, AND M. BRAVO (2013): "Construction and demolition waste indicators," *Waste Management & Research: The Journal for a Sustainable Circular Economy*, 31, 241–255.
- MÜLLER, D. (2006): "Stock dynamics for forecasting material flows—Case study for housing in The Netherlands," *Ecological Economics*, 59, 142–156.
- MÜLLER, E., L. M. HILTY, R. WIDMER, M. SCHLUEP, AND M. FAULSTICH (2014): "Modeling Metal Stocks and Flows: A Review of Dynamic Material Flow Analysis Methods," *Environmental Science & Technology*, 48, 2102–2113.
- NOVÁKOVÁ, I. AND K. MIKULICA (2016): "Properties of Concrete with Partial Replacement of Natural Aggregate by Recycled Concrete Aggregates from Precast Production," *Procedia Engineering*, 151, 360–367.
- O'CONNOR, J. (2004): "Survey on actual service lives for North American buildings," *Woodframe Housing Durability and Disaster Issues conference*, 9.
- OLAYA, Y., F. VÁSQUEZ, AND D. B. MÜLLER (2017): "Dwelling stock dynamics for addressing housing deficit," *Resources, Conservation and Recycling*, 123, 187–199.
- ORTLEPP, R., K. GRUHLER, AND G. SCHILLER (2016): "Material stocks in Germany's non-domestic buildings: a new quantification method," *Building Research & Information*, 44, 840–862.
- (2018): "Materials in Germany's domestic building stock: calculation model and uncertainties," *Building Research & Information*, 46, 164–178.
- PARIS, J. M., J. G. ROESSLER, C. C. FERRARO, H. D. DEFORD, AND T. G. TOWNSEND (2016): "A review of waste products utilized as supplements to Portland cement in concrete," *Journal of Cleaner Production*, 121, 1–18.
- PAULIUK, S. AND N. HEEREN (2021): "Material efficiency and its contribution to climate change mitigation in Germany: A deep decarbonization scenario analysis until 2060," *Journal of Industrial Ecology*, 25, 479–493.
- PAULIUK, S., T. WANG, AND D. B. MÜLLER (2013): "Steel all over the world: Estimating in-use stocks of iron for 200 countries," *Resources, Conservation and Recycling*, 71, 22–30.
- PEDNEAULT, J., G. MAJEAU-BETTEZ, S. PAULIUK, AND M. MARGNI (2022): "Sector-specific scenarios for future stocks and flows of aluminum: An analysis based on shared socioeconomic pathways," *Journal of Industrial Ecology*, 26, 1728–1746.
- PURNELL, P. AND A. DUNSTER (2010): "Recycling of concrete," in *Management, Recycling and Reuse of Waste Composites*, Elsevier, 569–591.
- REINERS, J. AND S. PALM (2015): "Technical report - Closing the loop: what type of concrete re-use is the most sustainable option?" Tech. rep., European Cement Research Academy, Düsseldorf.
- REIS, G. S. D., M. QUATTRONE, W. M. AMBRÓS, B. GRIGORE CAZACLIU, AND C. HOFFMANN SAMPAIO (2021): "Current Applications of Recycled Aggregates from Construction and Demolition: A Review," *Materials*, 14, 1700.
- SANDANAYAKE, M., Y. BOURAS, R. HAIGH, AND Z. VRCELJ (2020): "Current Sustainable Trends of Using Waste Materials in Concrete—A Decade Review," *Sustainability*, 12, 9622.
- SCHEBEK, L., B. SCHNITZER, D. BLESINGER, A. KÖHN, B. MIEKLEY, H. J. LINKE, A. LOHMANN, C. MOTZKO, AND A. SEEMANN (2017): "Material stocks of the non-residential building sector: the case of the Rhine-Main area," *Resources, Conservation and Recycling*, 123, 24–36.
- SCHILLER, G., F. MÜLLER, AND R. ORTLEPP (2017): "Mapping the anthropogenic stock in Germany: Metabolic evidence for a circular economy," *Resources, Conservation and Recycling*, 123, 93–107.

- SCHIPPER, B. W., H.-C. LIN, M. A. MELONI, K. WANSLEEBEN, R. HEIJUNGS, AND E. VAN DER VOET (2018): "Estimating global copper demand until 2100 with regression and stock dynamics," *Resources, Conservation and Recycling*, 132, 28–36.
- SEO, S. AND Y. HWANG (1999): "An Estimation of Construction and Demolition Debris in Seoul, Korea: Waste Amount, Type, and Estimating Model," *Journal of the Air & Waste Management Association*, 49, 980–985.
- SERRAND, M., D. MONFORT-CLIMENT, S. VAXELAIRE, P. MICHEL, AND A.-L. GAUTIER (2013): "Projet ANR ASURET - Méthodologie d'aide à la décision: Evolution prospective des besoins en matériaux et de la disponibilité des ressources primaires et secondaires - Exemple de la Ville d'Orléans (tâche 5)," Tech. Rep. Rapport final BRGM/RP-62586-FR, BRGM.
- SHANKS, W., C. DUNANT, M. P. DREWNIOK, R. LUPTON, A. SERRENHO, AND J. M. ALLWOOD (2019): "How much cement can we do without? Lessons from cement material flows in the UK," *Resources, Conservation and Recycling*, 141, 441–454.
- SHI, F., T. HUANG, H. TANIKAWA, J. HAN, S. HASHIMOTO, AND Y. MORIGUCHI (2012): "Toward a Low Carbon-Dematerialization Society: Measuring the Materials Demand and CO₂ Emissions of Building and Transport Infrastructure Construction in China," *Journal of Industrial Ecology*, 16, 493–505.
- SIDDIQUE, R. (2014): "Utilization of Industrial By-products in Concrete," *Procedia Engineering*, 95, 335–347.
- SILVA, R., J. DE BRITO, AND R. DHIR (2014): "Properties and composition of recycled aggregates from construction and demolition waste suitable for concrete production," *Construction and Building Materials*, 65, 201–217.
- (2019): "Use of recycled aggregates arising from construction and demolition waste in new construction applications," *Journal of Cleaner Production*, 236, 117629.
- STEELE, W. (2017): "Constructing the Construction State: Cement and Postwar Japan," *The Asia-Pacific Journal*, 11.
- TAM, V. W. (2009): "Comparing the implementation of concrete recycling in the Australian and Japanese construction industries," *Journal of Cleaner Production*, 17, 688–702.
- TAM, V. W., M. SOOMRO, AND A. C. J. EVANGELISTA (2018): "A review of recycled aggregate in concrete applications (2000–2017)," *Construction and Building Materials*, 172, 272–292.
- TANIKAWA, H., T. FISHMAN, K. OKUOKA, AND K. SUGIMOTO (2015): "The Weight of Society Over Time and Space: A Comprehensive Account of the Construction Material Stock of Japan, 1945-2010: The Construction Material Stock of Japan," *Journal of Industrial Ecology*, 19, 778–791.
- THOMAS, C., J. SETIÉN, J. POLANCO, P. ALAEJOS, AND M. SÁNCHEZ DE JUAN (2013): "Durability of recycled aggregate concrete," *Construction and Building Materials*, 40, 1054–1065.
- THOMAS, J., N. N. THAICKAVIL, AND P. WILSON (2018): "Strength and durability of concrete containing recycled concrete aggregates," *Journal of Building Engineering*, 19, 349–365.
- TURK, J., Z. COTIĆ, A. MLADENVIĆ, AND A. ŠAJNA (2015): "Environmental evaluation of green concretes versus conventional concrete by means of LCA," *Waste Management*, 45, 194–205.
- UKQAA (2016): "UKQAA Ash Availability Report," Tech. rep., UKQAA.
- VAN BEERS, D. AND T. GRAEDEL (2007): "Spatial characterisation of multi-level in-use copper and zinc stocks in Australia," *Journal of Cleaner Production*, 15, 849–861.
- VERIAN, K. P., W. ASHRAF, AND Y. CAO (2018): "Properties of recycled concrete aggregate and their influence in new concrete production," *Resources, Conservation and Recycling*, 133, 30–49.
- VILLORIA SÁEZ, P. AND M. OSMANI (2019): "A diagnosis of construction and demolition waste generation and recovery practice in the European Union," *Journal of Cleaner Production*, 241, 118400.
- VISHWAKARMA, V. AND S. UTHAMAN (2020): "Environmental impact of sustainable green concrete," in *Smart Nanococoncretes and Cement-Based Materials*, Elsevier, 241–255.
- VÁSQUEZ, F., A. N. LØVIK, N. H. SANDBERG, AND D. B. MÜLLER (2016): "Dynamic type-cohort-time approach for the analysis of energy reductions strategies in the building stock," *Energy and Buildings*, 111, 37–55.

- WANG, J., Y. ZHANG, AND Y. WANG (2018): "Environmental impacts of short building lifespans in China considering time value," *Journal of Cleaner Production*, 203, 696–707.
- WANG, T., D. B. MÜLLER, AND S. HASHIMOTO (2015a): "The Ferrous Find: Counting Iron and Steel Stocks in China's Economy: Counting Iron and Steel Stocks in China's Economy," *Journal of Industrial Ecology*, 19, 877–889.
- WANG, T., X. TIAN, S. HASHIMOTO, AND H. TANIKAWA (2015b): "Concrete transformation of buildings in China and implications for the steel cycle," *Resources, Conservation and Recycling*, 103, 205–215.
- WIEDENHOFER, D., J. K. STEINBERGER, N. EISENMENGER, AND W. HAAS (2015): "Maintenance and Expansion: Modeling Material Stocks and Flows for Residential Buildings and Transportation Networks in the EU25," *Journal of Industrial Ecology*, 19, 538–551, _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1111/jiec.12216>.
- WIJAYASUNDARA, M., R. H. CRAWFORD, AND P. MENDIS (2017): "Comparative assessment of embodied energy of recycled aggregate concrete," *Journal of Cleaner Production*, 152, 406–419.
- WITTMER, D., T. LICHTENSTEIGER, AND D. WITTMER (2007): "Exploration of urban deposits: long-term prospects for resource and waste management," *Waste Management & Research: The Journal for a Sustainable Circular Economy*, 25, 220–226.
- WUYTS, W., A. MIATTO, R. SEDLITZKY, AND H. TANIKAWA (2019): "Extending or ending the life of residential buildings in Japan: A social circular economy approach to the problem of short-lived constructions," *Journal of Cleaner Production*, 231, 660–670.
- XICOTENCATL, B. M. (2017): "Scenarios for concrete-rubble recycling in the Netherlands," Tech. rep., Leiden University.
- YANG, W. AND N. KOHLER (2008): "Simulation of the evolution of the Chinese building and infrastructure stock," *Building Research & Information*, 36, 1–19.
- YIN, X. AND W. CHEN (2013): "Trends and development of steel demand in China: A bottom–up analysis," *Resources Policy*, 38, 407–415.
- YOUSEFI, F. AND Y. GHOLIPOUR (2018): "Life cycle assessment of a real residential building in Tehran," *Honar-Ha-Ye-Ziba: Memary Va Shahrsazi*, 23, 81–92, publisher: University of Tehran University College of Fine Arts.
- ZHANG, L., J. YANG, Z. CAI, AND Z. YUAN (2015): "Understanding the Spatial and Temporal Patterns of Copper In-Use Stocks in China," *Environmental Science & Technology*, 49, 6430–6437.
- ZHOU, W., A. MONCASTER, D. M. REINER, AND P. GUTHRIE (2019): "Estimating Lifetimes and Stock Turnover Dynamics of Urban Residential Buildings in China," *Sustainability*, 11, 3720.