IMPERIAL

Forward-looking life cycle assessment of novel bauxite residue-incorporated cementitious materials

Maria Georgiades

Supervisors:

Dr. Rupert J. Myers

Prof. Christopher Cheeseman

A thesis submitted in total fulfilment of the requirements for the degree of Doctor of Philosophy

Department of Civil and Environmental Engineering

Imperial College London

October 2024

This page is intentionally left blank

Abstract

Bauxite residue (BR) is emerging as a raw material with significant potential for developing cementitious materials, including supplementary cementitious materials (SCMs). While laboratory and pilot-scale technologies have shown promise for incorporating BR into cement, their environmental performance remains poorly explored. Therefore, forward-looking environmental assessments are necessary to evaluate the potential benefits of BR utilisation in cements, which is explored in this thesis. First, an ex-ante life cycle assessment (LCA) was conducted to assess the environmental performance of BR vitrification technology to produce a reactive SCM at different scales. Upscaling laboratory to pilot and industrial scales reduced environmental impacts by 83% and 94%, respectively. Various upscaling approaches were explored, showing that with appropriate parameters, proxy modelling can perform comparably to complex process models, supporting its suitability in early-stage assessments with limited data. Second, prospective LCA of traditional cement production was performed, incorporating several decarbonisation measures: increasing clinker substitution, increasing alternative fuels, improving kiln efficiency, carbon capture and storage (CCS), and electricity grid decarbonisation. The results showed that clinker substitution could potentially reduce CO_2-eq. emissions by 42% between 2020 and 2050. Combining all decarbonisation measures could reduce CO_2-eq. emissions to below 0.01 kg per kg of cement by 2050. Third, ex-ante LCA was conducted on three BR processing technologies: (i) co-calcination with kaolin clay, (ii) vitrification, and (iii) reductive smelting, producing BR-based SCMs for up to 30% clinker replacement. Environmental performance was influenced by energy requirements and sources, raw material composition, and clinker replacement ratio. Composite cements were analysed by combining environmental and mechanical performance, showing that optimising fineness and material composition can reduce impacts without compromising strength. The thesis demonstrated the importance of early-stage environmental assessments in identifying hotspots, guiding optimisation, and supporting upscaling. Its findings offer benchmarks for comparing emerging BR technologies with conventional practices and quantifying their environmental benefits.

This page is intentionally left blank

Table of Contents

Abstract III
Table of Contents V
Preface X
Statement of Originality XII
Copyright Declaration XV
Acknowledgements XVII
List of Tables and Figures XXI
Chapter 1. Introduction and objectives 31
1.1 Introduction 31
1.2 Research objectives 37
1.3 Thesis overview 38
Chapter 2. Literature review 41
2.1 Bauxite ore to residue 41
2.1.1 Generation of bauxite residue 41
2.1.2 BR characteristics 45
2.1.3 Evolution of bauxite residue 48
2.2 Disposal and management 49
2.2.1 Marine and river discharge 50
2.2.2 Lagooning 50
2.2.3 Dry stacking 51
2.2.4 Mud farming 51
2.2.5 Dry disposal and filtration 51
2.3 Utilisation 52
2.3.1 BR Utilisation in construction materials 53
2.3.2 Metallurgical applications 57
2.3.3 Environmental protection applications 59
2.3.4 Conclusion 59
2.4 Barriers to reuse 61
2.5 Remediation 64
2.6 Emerging technologies for BR utilisation in composite cements 64
2.6.1 Overview of emerging BR technologies studied in this thesis 64

2.6.2 General overview of BR utilisation in LCA studies 69
Chapter 3. Methodology: Life cycle assessment of emerging, BR technologies 71
3.1 Life cycle assessment framework 71
3.1.1 Goal and scope definition 74
3.1.2 Life cycle inventory analysis 76
3.1.3 Life Cycle Impact Assessment (LCIA) 77
3.1.4 Interpretation 87
3.2 Technology readiness level (TRL) 88
3.3 Forward-oriented LCA approaches 89
3.4 Forward-looking LCA studies for emerging technologies 92
3.4.1 Forward-looking LCA framework 92
3.4.2 Summary of challenges 108
3.5 LCA studies on BR utilisation as cementitious materials 109
3.5.1 General overview 110
3.5.2 Methodological choices in existing LCA studies 111
3.5.3 Forward-looking LCA framework applied in this thesis 115
Chapter 4. Ex-ante LCA of BR vitrification technology 118
4.1 Background and motivation 118
4.2 Methodology 120
4.2.1 Description of bauxite residue vitrification technology 120
4.2.2 Life cycle assessment 123
4.3 Results and discussion 132
4.3.1 Effect of technology scale 132
4.3.2 Effect of upscaling approach 134
4.3.3 Prospective evaluation of future decarbonisation potential in the entire value chain of VBR for cementitious applications 140
4.4 Conclusions 145
Chapter 5. Prospective LCA of European cement production 146
5.1 Background and motivation 146
5.2 Methodology 150
5.2.1 Goal and scope 150
5.2.2 Foreground scenarios 153
5.2.3 Background scenarios 162
5.2.4 Uncertainty analysis 163
5.3 Results and discussion 163

5.3.1 Portland cement clinker substitution with supplementary cementitious materials 163
5.3.2 Increasing alternative fuels utilisation 165
5.3.3 Improving kiln configurations 168
5.3.4 Synergistic scenarios 169
5.3.5 Effect of background scenarios 175
5.3.6 Challenges for global decarbonisation of cement production 176
5.4 Conclusions 176
Chapter 6. Environmental and mechanical performance of composite BR Portland cements 178
6.1 Background and motivation 178
6.2 Methodology 182
6.2.1 Description of Bauxite residue technologies 182
6.2.2 Life cycle assessment 185
6.3 Results and discussion 192
6.3.1 Processing technology 192
6.3.2 Composite cement 195
6.3.3 Eco-efficiency indicators 196
6.4 Conclusions 199
Chapter 7. Conclusions 200
7.1 Recommendations for future work 203
Chapter 8. References 207
Appendix A 251
A.1 Life cycle inventory 251
A.1.1 Laboratory scale 251
A.1.2 Pilot 252
A.1.3 Industrial scale 253
A.1.3.1 Proxy technology 255
A.1.3.1.2 Proxy Selection 256
A.2 Ground granulated blast furnace slag (GGBFS) 259
A.3 Limitations in upscaling methodology 259
A.5 Furnace technology 260
A.4.1 Semi-quantitative discussion on furnace technology selection 260
A.5 Results 261
A.5.1 Effect of technology scale 261

A.5.2 Effect of upscaling approach....264
Appendix B....268
B.1 Parameter selection for scenario development....268
B.2 Life cycle inventory....272
B.2.1 Supplementary cementitious materials....272
B.2.2 Alternative fuels....275
B.2.3 Kiln type configuration (thermal efficiency)....283
B.2.4 Carbon capture and storage....284
B.2.5 Allocation....287
B.3 Uncertainty analysis....288
B.4 Contribution analysis results based on SSP2 RCP....289
B.4.1 PC clinker substitution with SCMs....289
B.4.2 Increasing alternative fuels utilisation....291
B.4.3 Synergistic scenario without CCS, following the IEA and CEMBUREAU roadmaps....292
B.4.4 Synergistic scenario with CCS, following the IEA and CEMBUREAU roadmaps....294
B.5 LCA results based on SSP2 baseline scenario....296
B.5.1 PC clinker substitution with SCMs....296
B.5.2 Increasing alternative fuels utilisation....298
B.5.3 Improving kiln configurations....301
B.6 Monte Carlo analysis....302
B.7 Global sensitivity analysis....303
B.8 Global decarbonisation approaches in cement industry....304
Appendix C....306
C.1 Life cycle inventory....306
C.2 Economic allocation for untreated BR....308
C.3 Mix design and compressive strength....309

This page is intentionally left blank

Preface

This thesis comprises a series of papers, some of which have already been published, while others are in preparation for submission to peer-reviewed journals. These papers have been reformatted and revised to enhance cohesion. Each result and discussion chapter (Chapters 4-6) concludes with a dedicated conclusion section, with an additional concluding chapter provided to tie the thesis together as a whole. The literature review includes further discussion on bauxite residue generation, current disposal and management practices, and promising industrial applications that are not covered in the publications. Chapter 3 presents the traditional LCA methodology, along with advancements in ex-ante and prospective assessment approaches. To improve clarity and flow, specific methodological details, such as the forward-looking scenarios applied to emerging technologies, are addressed separately in the results and discussion chapters.

This page is intentionally left blank

Statement of Originality

I declare that this thesis is the result of my own work and has not been submitted for the requirements of any other degree. Any work that is not my own has been properly acknowledged and referenced. Results obtained in the course of this research have been presented in a series of peer-reviewed publications and non-peer-reviewed reports.

Peer-Reviewed Journal Publications

Georgiades, M., Shah, I.H., Steubing, B., Cheeseman, C., Myers, R.J., 2023. Prospective life cycle assessment of European cement production. *Resources Conservation and Recycling194*. https://doi.org/10.1016/j.resconrec.2023.106998

Papers Under Review

Georgiades, M., Özcan Kilcan, C., Giels, M., Hertel, T., Pontikes, Y., Cheeseman, C., & Myers, R. J. Ex-ante life cycle assessment of bauxite residue vitrification technology.

Manuscripts In Preparation

Georgiades, M., Preveniou, A., Konlechner, D., Cheeseman, C., & Myers, R. J. Environmental and mechanical performance of composite bauxite residue Portland cements.

Non-Peer-Reviewed Reports in Preparation

1. **Georgiades, M.**, Kilcan, C. Ö., Tkaczyk, A., & Myers, R. J. Life cycle assessment of Bauxite residue utilisation in cement binders -Prospective LCA and LCA-NORM analysis on ReActiv technologies and novel BR-incorporated cementitious materials.

2. Termkhajornkit, P.; Ferey, F.; Bernard, P.; Pham, G.; Mathieu, B.; Moreau, C.; Kollias, K.; Evans, K.; Myers, R.J.; **Georgiades, M.**; Tkaczyk, A.H.; Kilcan, C.Ö.; Reigo, U.A.;

Hertel, T.; Giels, M.; Pontikes, Y.; Peys, A.; Martins, N.P.; Sparis, D.; Wenzel, M.; Georget, F.; Matschei, T. Pre-Dossier Cement Containing Modified Processed Bauxite.

Conference Presentations

1. **Georgiades, M.**; Shah, I.H.; Myers, R.J. "Prospective life cycle assessment of the European cement industry", 10th International Conference on Life Cycle Management, Online, 5–8 September 2021.

2. **Georgiades, M.**; Shah, I.H.; Myers, R.J. "Prospective life cycle assessment of the European cement industry", Advanced Materials for Sustainable Infrastructure Development - Gordon Research Conference, New London, NH, USA, 30–31 July 2022

3. **Georgiades, M.**; Kılcan, C.Ö.; Hertel, T.; Giels, M.; Tkaczyk, A.H.; Cheeseman, C.R.; Myers, R.J. "A sensitivity analysis on the upscaling of a novel bauxite residue vitrification technology: An ex-ante life cycle assessment case study", 42nd Annual Cement & Concrete Science Conference, Online, 11–12 September 2023.

4. **Georgiades, M.**; Kılcan, C.Ö.; Giels, M.; Hertel, T.; Pontikes, Y.; Tkaczyk, A.H.; Myers, R.J. "Ex-ante life cycle assessment of bauxite residue vitrification technology", Advanced Materials for Sustainable Infrastructure Development - Gordon Research Conference, Ventura, CA, USA, 24–25 February 2024.

5. **Georgiades, M.**; Kılcan, C.Ö.; Giels, M.; Hertel, T.; Pontikes, Y.; Tkaczyk, A.H.; Myers, R.J. "Ex-ante life cycle assessment of bauxite residue vitrification technology", SETAC Europe 26th LCA Symposium, Gothenburg, Sweden, 21–23 October 2024.

This page is intentionally left blank

Copyright Declaration

The copyright of this thesis rests with the author. Unless otherwise indicated, its contents are licensed under a Creative Commons Attribution-Non-Commercial 4.0 International Licence (CC BY-NC).

Under this licence, you may copy and redistribute the material in any medium or format. You may also create and distribute modified versions of the work. This is on the condition that: you credit the author and do not use it, or any derivative works, for a commercial purpose.

When reusing or sharing this work, ensure you make the licence terms clear to others by naming the licence and linking to the licence text. Where a work has been adapted, you should indicate that the work has been changed and describe those changes.

Please seek permission from the copyright holder for uses of this work that are not included in this licence or permitted under UK Copyright Law.

This page is intentionally left blank

Acknowledgements

This thesis marks the culmination of a four-year journey that has taken me to different parts of the world, introduced me to remarkable individuals, and shaped both my professional and personal growth. Reflecting on this journey, I am reminded of C.P. Cavafy's *Ithaka* and the mixed emotions that come with reaching the end of a long road. As he writes, *"As you set out for Ithaka, hope your road is a long one, full of adventure, full of discovery."* My journey has indeed been full of both adventure and growth. Yet, as Cavafy also reminds us, *"Keep Ithaka always in your mind. Arriving there is what you're destined for. But don't hurry the journey at all".* It is a bittersweet feeling as I reach the end of this long, often challenging, yet deeply rewarding journey. For this, I am profoundly grateful.

First and foremost, I owe my deepest gratitude to my supervisor, Dr. Rupert J. Myers. Your constant support, encouragement to take initiative, and guidance in exploring the endless possibilities ahead have been invaluable. Your thoughtful advice, work ethic, and openness to discussing both academic and personal challenges will always be appreciated. You struck the perfect balance between pushing me to grow as a researcher and providing motivation, and for that, I am truly thankful.

I am also grateful to my co-supervisor, Professor Chris Cheeseman, whose expertise and insightful feedback elevated the quality of this work. Your rigorous approach to sustainable materials sharpened my understanding of the subject and pushed me to think more critically about the broader implications of my research.

My sincere appreciation goes to the Horizon 2020 ReActiv project for the financial support that made this journey possible, and to all the ReActiv consortium partners for their collaboration. A special thanks to Ms. Cansu O.K., whose positive energy made every project more enjoyable, and whose friendship I now cherish.

To everyone I have worked with or learned from, whether through conferences, collaborations, or the papers I have read, thank you. Your contributions, whether big or small, have enriched this work in ways you may not even realise.

I am especially grateful to my colleagues, who made the ups and downs of PhD life more fun. Anusha, your support during this final stretch of the PhD has been invaluable, not to mention the conversations that helped keep me sane. Meng G., we have been on this PhD ride together since day one, from lockdowns, virtual meetings, and now at the finish line together. We have grown so much, and I am thankful to have shared this experience with you. A special thanks to my office mates Maciej, Tom, Jo, Ben, Tobuchi, Will, Jordi, Rika, Delal, Yuting and Ruiqi for making the office a place I looked forward to coming to each day.

To the incredible friends I have made in London, thank you for turning my time in the UK into an unforgettable adventure. Bruno**,** thanks for being the best gym buddy and ensuring I found healthy ways to release stress, instead of hiding under my desk (which I was very tempted to do). To Shannon, Sheenam, Ana, and Tina, thank you for keeping me socially alive with our walks, talks, and brunches, it made all the difference.

To my friends back home in Limassol, Cyprus, thank you for always keeping me rooted, no matter where I was. Our friendships have grown beyond boundaries, and I know they will last a lifetime. Cheers to all of you!

On a personal note, I am eternally grateful to my family for their love and support throughout this journey. To my mother, Panayiota, your optimism and unwavering belief in me have been a source of constant strength. To my father, Phivos, your curiosity, persistence, and love for learning have been my greatest inspiration. Your unconditional support and countless sacrifices have fueled my journey every step of the way. To my sisters, Georgia and Electra, thank you for your patience, for listening to my endless rants, and for always making me laugh.

Now for the therapy bit my dearest friends. Evelina C., Aria I., Abdullah A.B., Nafia S., Chrystalla P., Nikolas K., you have all been my lifeline, picking up the phone even when

you knew it would probably be another hour of me complaining. Your confused looks and supportive nods, when I talked about my research, were exactly what I needed to feel understood. Thank you for letting me vent, complain, and cry, it means the world to me.

Lastly, I am thankful for the small things that brought me joy during difficult moments like *Friends* and *The Big Bang Theory*, whose characters never failed to make me laugh. And with a nod to Sheldon's speech (slightly rephrased), I would like to say: "I was under a misapprehension that my accomplishments were mine alone. Nothing could be further from the truth. I have been supported, encouraged, sustained, and inspired by my family and the greatest group of friends anyone could ask for. Thank you to you all."

This page is intentionally left blank

List of Tables and Figures

List of Figures

Figure 1-1: Global cement production by region from 2014 to 2050 (Data from IEA, 2018; Van Oss. Hendrik G., 2016) 32

Figure 1-2: Schematic representation of the Portland cement production process, including decarbonisation strategies such as switching to alternative fuels, improvements in thermal efficiency, implementing carbon capture and storage, reducing Portland cement clinker with SCMs, and developing low-carbon clinkers. The decarbonisation strategies are positioned in the diagram at the processing stages where they would be implemented. 34

Figure 1-3: Thesis structure and interconnections between individual chapters. 40

Figure 2-1 Simplified flowsheet of Bayer process (adapted from Lyu et al., 2021) 42

Figure 2-2: Aerial view of the Aughinish Alumina Bauxite Residue Disposal Area (BRDA), including the nearby Shannon Estuary (Bruce, 2023) 44

Figure 2-3: Forecast of alumina production and corresponding BR production growth by 2050. Three scenarios are presented: status-quo (baseline (BS) scenario) in orange, conservative (CO) scenario in pink, and optimistic (OP) scenario in purple (CM Group, 2020; International Aluminium Institute, 2018) 49

Figure 2-4: Historical methods for BR disposal management 52

Figure 2-5: Distribution of BR utilisation in industrial applications (adapted from Klauber et al., 2011) 53

Figure 2-6: Process flow diagram of BR calcination technology (adapted from Konlechner et al., 2023) 66

Figure 2-7: Process flow diagram of BR vitrification technology (adapted from Konlechner et al., 2023) 67

Figure 2-8. Industrial-scale process flow diagram of smelted BR and pig iron (adapted from Konlechner et al., 2023) 68

Figure 3-1: Schematic diagram of iterative LCA framework showing the four phases of LCA according to ISO14040:2006 (ISO: 14044, 2006) 72

Figure 3-2: Midpoint vs endpoint impact categories covered by the ReCiPe 2016 method (Huijbregts et al., 2017)..... 82

Figure 3-3: Socio-economic pathways scenarios (SSPs) according to the challenges of mitigation and adaptation (adapted from (O'Neill et al., 2017)) 98

Figure 3-4: Methodology for generating future electricity background scenarios (adapted from (Steubing and de Koning, 2021)..... 99

Figure 3-5: Comparison of future-oriented preliminary assessment and traditional retrospective approach along technology development: The solid green and blue lines represent the projected range of environmental impacts and costs under future-oriented scenarios. The dash-dotted lines show potential outcomes in a traditional retrospective approach. The pink, red, and black lines represent the degree of uncertainty, the design flexibility, and the technological knowledge, respectively (adapted from Hetherington et al., 2014; Villares et al., 2017) 103

Figure 3-6: Schematic representation of the research framework, adapted from Villares et al. (2017). The first step (outlined in Chapter 4) involves the LCA modelling of the lab-scale system, incorporating upscaling scenarios to assess the environmental performance of the emerging BR technologies through an ex-ante LCA at an industrial scale. Simultaneously, in stage 2 (covered in Chapter 5) a prospective LCA is applied to evaluate future changes affecting the incumbent technology, PC production. Stage 3 presents the environmental profile comparison ensuring temporal consistency between the emerging and incumbent technology systems..... 115

Figure 4-1. Simplified process flow diagram for the laboratory scale BR vitrification system..... 121

Figure 4-2: Simplified process flow diagram for the pilot scale BR vitrification system. 122

Figure 4-3: Simplified process flow diagram for the industrial scale BR vitrification technology. 123

Figure 4-4. Flowchart of the VBR product system modelled across three scales (i.e., laboratory, pilot, and industrial), including foreground (red outline) and background (grey outline) activities, and upscaling approaches used in industrial scale modelling (pink boxes). 124

Figure 4-5: LCIA results of VBR production, internally normalised to the laboratory scale, across scales (i.e., laboratory, pilot, and industrial). The mean results for each impact category at the industrial scale were derived from the seven distinct upscaling approaches. For example, the global warming potential (GWP100) at the industrial scale, presents the impact distribution from different upscaling approaches. Abbreviations for ReCiPe (H) impact categories are reported in Section 2.2.1. Quantitative results are tabulated in Appendix A (Table A9)..... 132

Figure 4-6: ReCiPe 2016 midpoint impact results (absolute) for BR vitrification across all upscaling scenarios. Individual bars represent the environmental impact of each upscaling scenario i.e., process simulation (M3.1.1), linear regression (M3.2.1), power regression (M3.2.2), clinker production modelled based on Centre et al. (2013), (M3.3.1), clinker production modelled based on Ecoinvent v.3.9.1. (2007a), (M3.3.2), ceramic vitrification modelled based on European Commission. (2007) (M3.3.1), ceramic vitrification modelled based on Ecoinvent v.3.9.1. (2007b), (M3.3.2), per kg of VBR. The GGBFS bar is used as a benchmark for comparison. Abbreviations for ReCiPe (H) impact categories are reported in section 2.2.1. Quantitative results for each individual upscaling scenario are tabulated in the Appendix A (Table A10)..... 137

Figure 4-7: Contribution analysis showing CO_2-eq. emissions of VBR processed at an industrial scale based on process modelling, regression analysis, and proxy technology. "Other" represents the impact of upstream, background activities such as hard coal preparation and petroleum/natural gas production. Quantitative results are tabulated in Appendix A (Table A11)..... 139

Figure 5-1: System boundary of this prospective LCA study, including foreground changes on (i) SCMs, (ii) alternative fuels, (iii) kiln configurations with higher efficiency (η) and (iv) CCS. Background changes are included based on the SSP2 pathways (Baseline and RCP 2.6, see Section 2.3)..... 152

Figure 5-2: CO_2-eq. emissions of blended PCs with different SCMs, from 2020 to 2050, using the SSP2 RCP 2.6 background scenario, compared to reported values (for cements with a ~50% clinker replacement ratio) from traditional LCA studies in 2050 (Arrigoni et al., 2020; Miller, 2018; Miller et al., 2018; Rahla et al., 2019) 164

Figure 5-3: CO_2-eq. emissions from cement production with alternative fuels, from 2020 to 2050, using the SSP2 RCP 2.6 background scenario, compared to reported values from traditional LCA studies in 2050 (Georgiopoulou and Lyberatos, 2018; Thanos Bourtsalas

et al., 2018; Thwe et al., 2021). The percentage fuel mix ratio of each individual fuel in 2020 to 2050 is displayed in the figure legend..... 166

Figure 5-4: (a) CO_2-eq. emissions from cement production with several kiln configurations from 2020 (semi-wet/semi-dry) to 2050 (dry with preheater and precalciner), using the SSP2 RCP 2.6 background scenario. (b) Contribution analysis of the key contributing flows from 2020 to 2050 using different kiln configurations. 168

Figure 5-5. CO_2-eq. emissions of cement production with (synergistic) scenario 4 from 2020 to 2050, using SSP2 RCP 2.6, for blended PC with coal fly ash, and based on the IEA and CEMBUREAU roadmaps with no allocation and economic allocation approaches. 170

Figure 5-6: Contribution analysis for the CO_2-eq. emissions of cement production with synergistic scenarios including PC clinker substitution with coal fly ash, increasing alternative fuels utilisation, and improving kiln configurations from 2020 to 2050 (without CCS) with (economic) and without allocation, and including biogenic carbon. 171

Figure 5-7: Comparison of synergistic scenarios developed based on the IEA and CEMBUREAU roadmaps, and the GCCA roadmap. CO_2-eq. emissions of cement production from a synergistic scenario incorporating PC clinker substitution with coal fly ash (SCMs), increasing alternative fuels utilisation, and improving kiln configurations from 2020 to 2050 (without CCS). 172

Figure 5-8: Contribution analysis showing CO_2-eq. emissions of cement production developed based on the IEA and CEMBUREAU roadmaps with synergistic scenarios including PC clinker substitution with SCMs, alternative fuel use, improved kiln configurations, and CCS technologies (i.e., amine scrubbing, oxy-fuel) in 2050, with and without combined heat and power unit (CHP)..... 173

Figure 5-9. CO_2-eq. emissions of cement production from a synergistic scenario developed based on the IEA and CEMBUREAU roadmaps, incorporating PC clinker substitution with coal fly ash, increasing alternative fuels utilisation (fuel mix consists of hydrogen generated through wind power, biomass, and fossil fuels), and improving kiln configurations from 2020 to 2050 (without CCS), with SSP2 baseline (presented with an asterisk *) and SSP2 RCP 2.6 (no asterisk). 175

Figure 6-1: Process flow diagram illustrating the processing technologies for producing modified BR suitable as an SCM 183

Figure 6-2: System boundary outlining the main life cycle stages for processing BR into suitable SCMs using co-calcination, vitrification, and smelting. It includes raw material extraction, BR treatment, Portland clinker, cement and mortar production. 187

Figure 6-3**:** Split up of (a) climate change (kg CO_2 emissions per kg of cementitious material) and (b) cumulative energy demand (MJ per kg of cementitious material)..... 194

Figure 6-4: Split up of (a) CO_2-eq. emissions and (b) energy demand per kg of composite cements with BR-incorporated SCMs..... 196

Figure 6-5: Split up of (a) binder intensity (bi) versus 28-day compressive strength for mortar mixes with composite cements and (b) carbon intensity (ci) versus 28-day compressive strength for mortar mixes with composite cements. SBR is represented in blue, while CBR, VBR, and LVBR are shown using colour gradients: pink for CBR, green for VBR, and orange for LVBR. The colour gradients indicate variations in the SCM: for CBR, the pink gradient reflects changes in raw feed composition affecting CBR behaviour, while for VBR and LVBR, the green and orange gradients represent variations in material fineness for VBR and LVBR, respectively. Conventional materials are depicted with black outlines (unfilled markers)..... 198

List of Tables

Table 2-1: Chemical composition range (%) of BR main components (International Aluminium Association, 2022). 47

Table 2-2: Critical barriers hindering large-scale BR utilisation (Almeida et al., 2020; International Aluminium Institute, 2020c, 2020a, 2020b) 63

Table 3-1: Description of midpoint impact categories investigated in this thesis (adapted from Huijbregts et al., 2017). 84

Table 3-2: Overview of "forward-oriented" LCA approaches..... 90

Table 3-3: summarises the reviewed studies, focusing on key parameters and methodological choices related to the production and environmental assessment of BR-derived cementitious materials. 113

Table 4-1: Final chemical composition of VBR 121

Table 4-2. Descriptions of the nine scenarios used to investigate the environmental impacts of BR vitrification technology..... 128

Table 4-3: ReCiPe 2016 midpoint impact results of industrial scale vitrification technology (upscaled with process modelling) in 2020 and with electricity scenarios in 2050 (i.e., SSP2 RCP1.9, SSP2 RCP2.6, and SSP2 Baseline). Bold values indicate the lowest impact when comparing the SSP2 scenarios. Underlined values indicate the highest impact when comparing the SSP2 scenarios. Italic values show the lowest impact between VBR and GGBFS, both with the SSP2 RCP 1.9 scenario. 141

Table 5-1: Electricity mix of the main location-specific electricity sources..... 160

Table 6-1: Formulation of BR-incorporated composite cements. CBR is co-calcined processed bauxite/kaolinitic clay. VBR is vitrified processed bauxite. LVBR is limestone vitrified processed bauxite. GGBR is low iron smelted processed bauxite. 189

Table 6-2. Estimated market prices for product/by-product flows according to the pre-feasibility study (Efstathiou et al., 2022) 190

Table 6-3: Composition of conventional cements where limestone (L), fly ash (V), and blast furnace slag (S)..... 191

Table 6-4: Mortar composition according to EN 196–1 191

List of abbreviations

LOP	Agricultural land occupation
ASTM	American Society for Testing and Materials
BR	Bauxite residue
BRDA	Bauxite Residue Disposal Areas
BAT	Best Available Techniques
BREFs	Best Available Techniques Reference Documents
BFS	Blast furnace slag
CAC	Calcium aluminate cement
CAS	Calcium aluminate slag
CSAF	Calcium sulfoaluminate-ferrite
CCS	Carbon capture and storage
CBR	Co-calcined bauxite residue
CHP	Combined heat and power unit
CBA	Cost-benefit analysis
CRMs	Critical raw materials
CED	Cumulative energy demand
DPP	Digital Product Passport
DPI	Dynamic process inventory
DSys	Dynamic system
ESPR	Eco-design for Sustainable Products Regulation
EIA	Environmental impact assessment
EPDs	Environmental product declarations
ERA	Environmental risk assessment
CEMBUREAU	European Cement Association
EU	European Union
ELCE	Extraordinary Leuven Cement
FFP	Fossil fuel potential
FETP	Freshwater ecotoxicity potential

FEP	Freshwater eutrophication potential
FU	Functional unit
GNR	Getting the Numbers Right
GCCA	Global Cement and Concrete Association
GSA	Global sensitivity analysis
GWP	Global warming potential
GBFS	Granulated blast furnace slag
GHG	Greenhouse gas
GDP	Gross domestic product
HTPc	Human toxicity potential (carcinogenic)
HTPnc	Human toxicity potential (non-carcinogenic)
IAMs	Integrated assessment models
IPCC	Intergovernmental Panel on Climate Change
IAI	International Aluminium Institute
IEA	International Energy Agency
ISO	International Organization for Standardization
IRP	Ionising radiation potential
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
LCSA	Life cycle sustainability analysis
LiSET	Lifecycle Screening of Emerging Technologies
LC3	Limestone Calcined Clay Cement
LVBR	Limestone vitrified bauxite residue
SBR	Low-iron smelted bauxite residue
MRLs	Manufacturing Readiness Levels
METP	Marine ecotoxicity potential
MEP	Marine eutrophication potential
MFA	Material flow analysis
SOP	Mineral resource scarcity potential
MHF	Multiple hearth furnace
NGSR	Natural gas steam reforming
NORM	Naturally occurring radioactivity materials
OPC	Ordinary Portland cement

ODPinfinite	Ozone depletion potential (infinite time horizon)
PSD	Particle size distribution
FPMFP	Particulate matter formation potential
EOFP	Photochemical ozone formation potential: ecosystems
HOFP	Photochemical ozone formation potential: humans
PDFs	Probability density functions
PEF	Product Environmental Footprint
pLCA	Prospective life cycle assessment
PCC	Pulverised coal combustion
RDF	Refuse-derived fuel
RCPs	Representative concentration pathways
RSA	Residue Storage Areas
SF	Silica fume
SSPs	Socio-economic pathways scenarios
NaOH	Sodium hydroxide (caustic soda)
SEA	Strategic environmental assessment
SAF	Submerged arc furnace
SCMs	Supplementary cementitious materials
TENORM	Technologically enhanced naturally occurring radioactivity material
TRL	Technology readiness level
TAP	Terrestrial acidification potential
TETP	Terrestrial ecotoxicity potential
TDF	Tyre-derived fuel
VBR	Vitrified bauxite residue
WCP	Water consumption potential

Chapter 1. Introduction and objectives

1.1 Introduction

Concrete is the second most widely consumed material, second only to fresh water, with approximately 3 tonnes of concrete produced per person annually (Aïtcin, 2000). It is the most commonly used construction material, with its global consumption more than double that of all other building materials combined (Van Oss, 2007). Concrete is a composite material of cement, coarse and fine aggregates (e.g., crushed stones and sand), and water. When mixed with only fine aggregates and water, cement forms mortar. Cement is a "hydraulic binder" and is the primary constituent in concrete, acting as the "glue" that hardens and binds the raw materials when hydrated.

As the global population is projected to grow by 34% between 2014 and 2050, the demand for urbanisation and sustainable infrastructure is expected to increase global cement production by 12%-23% (UN DESA, 2015). Significant growth is anticipated in regions like India (tripling from its current production volume), Africa (more than tripling) and America (almost doubling), while China is expected to see a 27% decline (as seen in Figure *1-1*). These estimates are based on projections for population growth changes. China expects a 7% decrease, while India, Egypt, Pakistan, Nigeria, and Ethiopia expect the largest increase of 15%, 39%, 48%, 55% and a staggering 70%, respectively, fuelling the cement production to satisfy domestic needs (IEA, 2018).

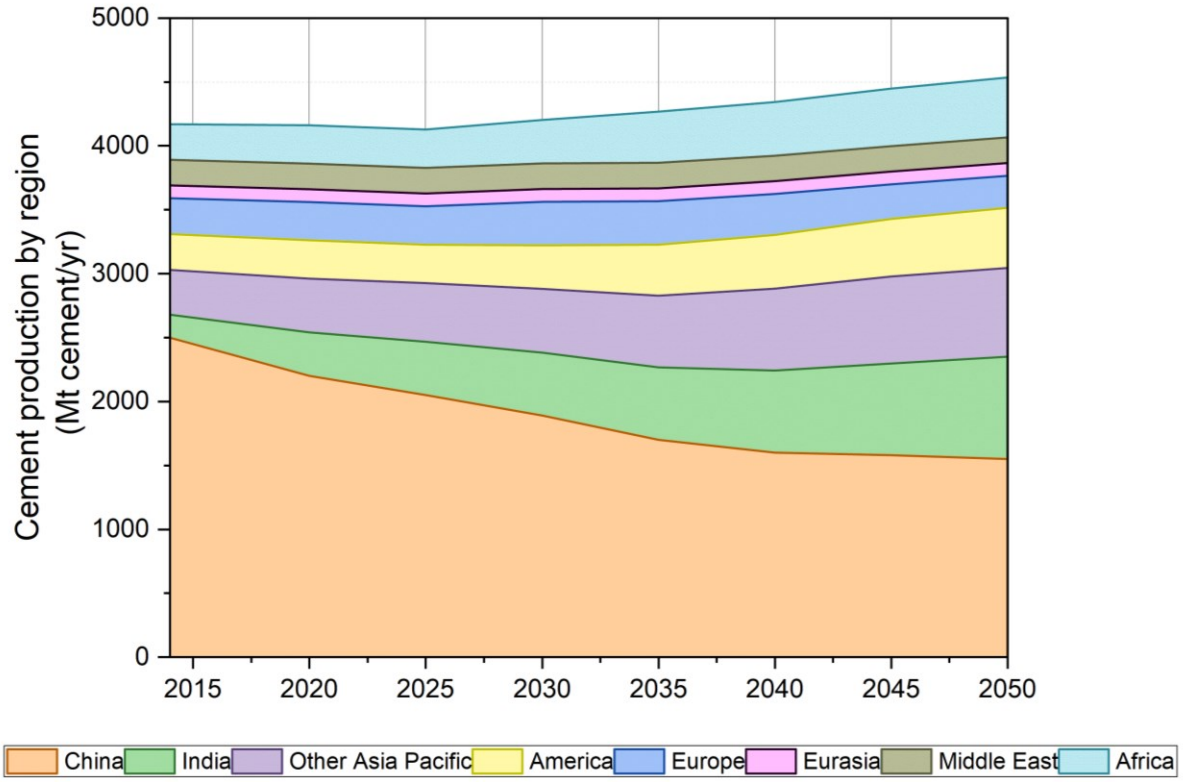


Figure 1-1: Global cement production by region from 2014 to 2050 (Data from IEA, 2018; Van Oss. Hendrik G., 2016)

Cement and concrete are important materials, with a global market value of $440 billion annually (The Business Research Company, 2019), playing an important role in the wider construction sector that accounts for 13% of the global gross domestic product (GDP) (McKinsey Global Institute, 2017). However, a drawback of conventional cement production is its climate change impact, contributing approximately 6–8% of global anthropogenic CO_2 emissions (Mikulčić et al., 2016). Overcoming the challenges of the global cement industry to reduce and mitigate its carbon footprint effectively is vital for ensuring future sustainable development.

Portland cement (PC) is by far the most commonly produced cement type, formed by heating a mixture of limestone and clay to 1450 °C (International Energy Agency (IEA), 2018). The sintered clinker is ground and blended with other minerals such as gypsum, limestone, blast furnace slag, and coal fly ash (GCCA, 2022). Cradle-to-gate CO_2 emissions from PC clinker production are approximately 798 kg (world average) and 712 kg (European average) per tonne of clinker, which mainly arise from reactions of the raw materials (i.e., process-based) and fossil fuel combustion (i.e., energy-based) (GNR, 2019a, 2019d). Emissions from PC

production depend on several factors, including the amount of clinker produced, plant location, kiln fuel, electricity mix, as well as production efficiency (Petek Gursel et al., 2014).

In line with the Paris Agreement, the IEA has set a 24% CO_2 emission reduction target (compared to pre-industrial levels) from cement production to contribute to limiting the global mean temperature rise to 2°C by the year 2100 (IEA, 2018). The European Commission's 'Green Deal' proposes a more ambitious goal for Europe to become carbon neutral by 2050, which requires limiting global temperature rise to 1.5 °C by 2100 (Cembureau, 2020). To meet these climate targets, the cement industry needs to reduce CO_2 emissions at a significantly faster pace than the historic trend between 2005 and 2018 (~0.4% reduction per year in kg CO_2 per t clinker). Typical measures that have been studied by life cycle assessment (LCA) to reduce CO_2 emissions from cement production are to replace clinker with supplementary cementitious materials (SCMs), such as coal fly ash and blast furnace slag), switching to alternative fuels for combustion (e.g., reducing petroleum coke and hard coal used as the primary kiln fuels with e.g., refuse-derived fuel, RDF), improving kiln configurations to achieve higher production efficiencies (e.g. reducing heat losses) and carbon capture and storage (CCS) (Mikulčić et al., 2016) (Figure 1-2).

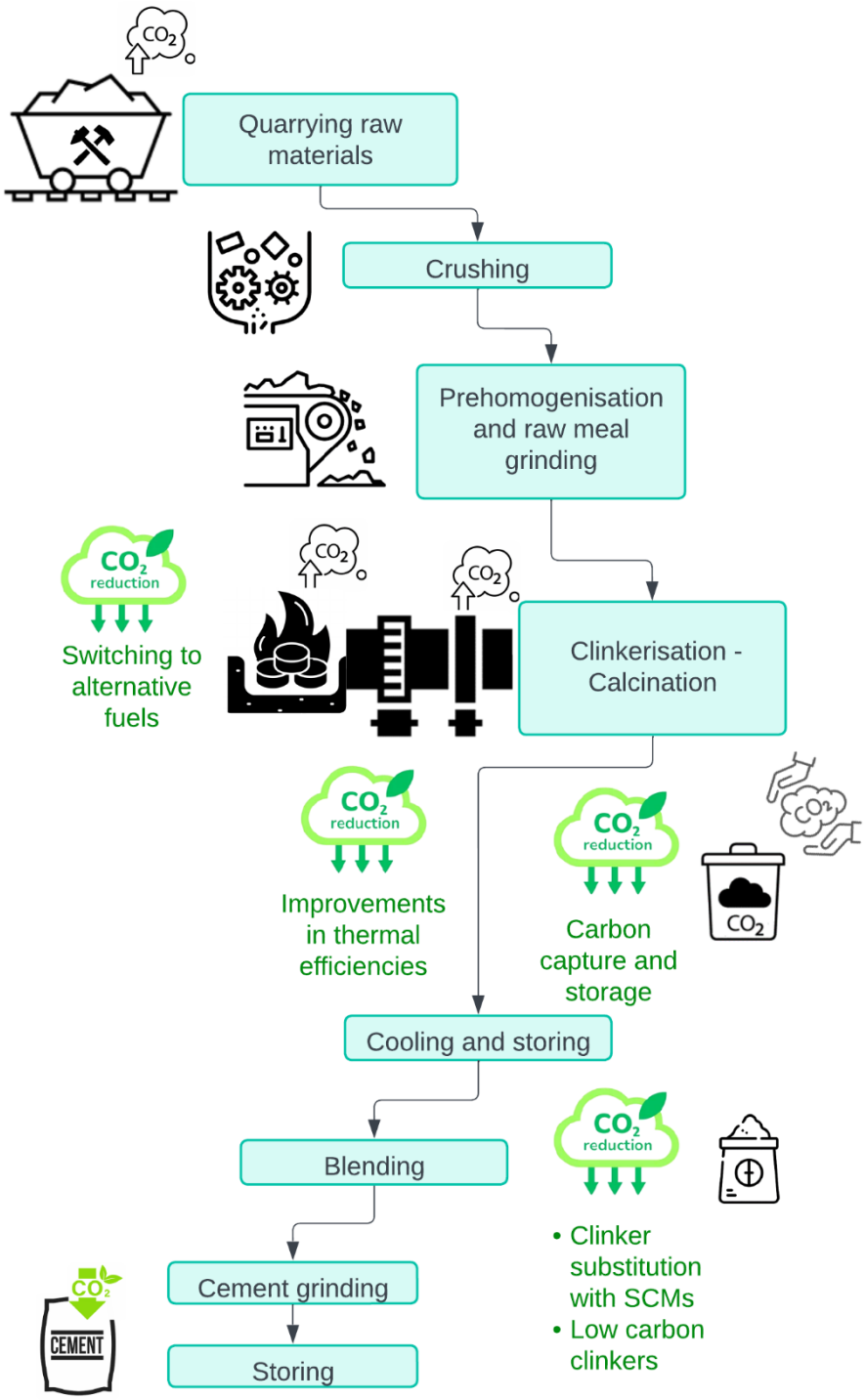


Figure 1-2: Schematic representation of the Portland cement production process, including decarbonisation strategies such as switching to alternative fuels, improvements in thermal efficiency, implementing carbon capture and storage, reducing Portland cement clinker with SCMs, and developing low-carbon clinkers. The decarbonisation strategies are positioned in the diagram at the processing stages where they would be implemented.

While implementing CCS, increasing alternative fuel consumption, and improving thermal efficiency are key strategies for decarbonising the cement industry, substituting clinker with

SCMs offers a more cost-effective approach to mitigate CO_2 emissions (Gartner and Hirao, 2015). Currently, the traditional cement industry is focused on lowering the clinker-to-cement ratio in PC by incorporating SCMs that are less energy-intensive and capable of producing blended cements which are competitive with existing cementitious products in terms of mechanical and durability performance.

There are several existing options and numerous under development. Commonly used SCMs include industrial by-products such as fly ash, blast furnace slag, silica fume, and naturally occurring pozzolans like calcined clay, calcined shale and metakaolin (Snellings et al., 2012). Pozzolans, as defined by ASTM C618, are siliceous or siliceous-aluminous materials that, even though hold little cementitious value, they react with calcium hydroxide in the presence of moisture to form compounds with cementitious properties (Mehta, 1987). The corresponding reactivity depends on several factors, such as particle size, material composition and temperature.

Currently, the availability of existing SCMs only meets 15% of the concrete demand (Maraghechi et al., 2018), with fly ash being the dominant SCM (Sakir et al., 2020). However, a significant challenge which limits the use of both natural pozzolans, and industrial by-products or wastes is the regional availability of local deposits. Additionally, the production capacity and the distance of manufacturing sites may require long-distance transportation, increasing both CO_2 emissions and costs (Snellings et al., 2012). As cement demand is projected to reach up to 6 billion tonnes by 2050 (Favier et al., 2018), the need for new, practically feasible SCMs is becoming increasingly urgent. Recent advancements in the standardisation of emerging alternatives include calcined clay and natural pozzolans, with examples including LC3 and Heidelberg Cement's Norcem Newcem, which utilises basaltic glass from Iceland (Bjerge and Brevik, 2014).

The search for viable alternatives is also partly driven by an expected shortage of current commercial SCMs, such as fly ash and granulated blast furnace slag. The G7 commitment to phasing out coal-fired power plants by 2035, except for those equipped with CCS technology (World Resources Institute, 2024), will significantly reduce fly ash availability. Evidence of this transition is already apparent, with Greece, the UK, and Denmark showing the fastest coal power reductions over 8 years (2000-2022) with 41%, 37%, and 29% respectively (Joel Jaeger, 2023). Similarly, the supply of granulated blast furnace slag is limited to primary steel

production using blast furnaces, while there is a shift towards using electric arc furnaces and submerged arc furnaces. Currently, 50% of granulated blast furnace slag in Europe is imported (Alberici et al., 2017). However, reliance on external sources such as India and China might entail greater costs, stressing the need for more environmentally and economically viable SCMs for EU construction.

As the demand for cement and concrete is expected to grow, possibly leading to increased international trade of SCMs, research has focused on identifying new SCM sources, particularly from by-products and industrial wastes. For instance, steelmaking slags, non-ferrous metallurgical slags, municipal solid waste incineration ashes, and bauxite residue (BR) have been explored as potential SCMs (Snellings et al., 2023). BR, a waste material generated during alumina refining, has been gaining attention due to its vast availability and promising chemical composition (high content of iron oxide, alumina, and silica). Recent studies have explored treatment technologies such as calcination to enhance material properties to improve reactivity, making BR a promising option for reducing clinker use and cutting CO_2 emissions in cement production (Patangia et al., 2023; Pontikes and Angelopoulos, 2013; Salim et al., 2023). In general, the aim is to develop viable SCMs processed through emerging technologies to enhance their properties and reactivity, thereby strengthening local and regional SCM availability. However, despite numerous studies investigating the feasibility of various SCM sources treated under different technologies, there is a gap in comprehensive environmental assessments (i.e., life cycle assessments (LCAs)). These assessments are essential to fully evaluate the environmental implications of processing and upscaling these emerging SCMs from laboratory to commercial scale and use them in real-world settings. Although the environmental benefits of using waste materials are highlighted in the literature, existing studies lack detailed LCAs (Singh et al., 2024; Mohammadi and Ramezanianpour, 2023; Althoey et al., 2023).

Historically, LCAs have been employed as a one-off diagnostic tool to assess the environmental impacts of industrial-scale technologies. However, when applied in the context of emerging technologies, this retrospective approach can be problematic. Even though traditional LCAs can identify critical "hotspot" areas, they cannot provide any major mitigation strategies. This is because traditional LCAs are designed for mature technologies with well-established processes and system boundaries; hence, while hotspots can be identified, there is limited room for process design iteration and optimisation. Traditional LCAs rely on historical

and current data, which is a more descriptive approach to past and current conditions and not reflective of how emerging technologies will operate when upscaled in future contexts. Emerging technologies at an early development stage offer greater flexibility in adjusting processing parameters to optimise environmental performance. While traditional LCAs were useful for existing systems, emerging technologies require a more proactive approach that anticipates and mitigates potential environmental impacts before large-scale implementation.

This thesis aims to quantitatively assess the environmental benefits and trade-offs of using BR as a raw material in cement binders treated through various emerging technologies. Employing ex-ante LCA as a technical decision-making tool and identifying the hotspots within individual treatment processes along upscaling are highlighted early on at an early stage, supporting ongoing technology development rather than performing a retrospective analysis. This research investigates the environmental performance of conventional cement production in Europe by 2050 to enable a time-consistent and forward-looking assessment of the potential of these emerging BR technologies assuming future commercialisation.

1.2 Research objectives

The overarching goal of this thesis is to deepen the understanding of how upscaling emerging technologies influence environmental performance, particularly concerning novel BR-incorporated cementitious materials. This thesis investigates the environmental impact of various emerging BR processing technologies, using the ex-ante LCA approach as a preliminary assessment tool to identify critical "hotspot" areas within the life cycle stages prior to industrial upscaling.

Ultimately, this thesis provides a thorough analysis of the environmental implications associated with novel BR-incorporated cementitious products and highlights the importance of preliminary assessments for emerging technologies, thereby paving the way for their sustainable integration into commercialised cementitious products. The objectives of this research include:

I. To assess the environmental impact of upscaling an emerging technology from laboratory to industrial scale, focusing on different upscaling approaches and their

influence on environmental performance. This includes developing an LCA model across scales, conducting hotspot analysis to identify "hotspot" areas, and examining the sensitivity of modelling assumptions as well as the influence of future changes, such as electricity grid decarbonisation, on overall environmental performance.

II. To evaluate the future environmental performance of conventional cement production by incorporating various decarbonisation scenarios. This includes strategies such as clinker substitution with supplementary cementitious materials (SCMs), increased use of alternative fuels, kiln efficiency improvements, and carbon capture and storage (CCS). Through scenario analysis of both foreground and background systems, including the decarbonisation of the electricity grid, the objective is to quantify the impact of these measures and provide a holistic framework for optimising greenhouse gas reductions in the cement industry.

III. To quantify the environmental performance of BR-derived cementitious materials using upscaled LCA models, highlighting their potential environmental benefits. Additionally, to combine the environmental performance with mechanical performance indicators, providing a comprehensive assessment of BR-incorporated materials in comparison to conventional products. The objective is to evaluate the emission reduction potential, promoting the BR utilisation in cementitious products.

1.3 Thesis overview

This thesis begins with a literature review in Chapter 2, presenting the lifecycle of bauxite ore from extraction to residue management and exploring its potential as a raw material for cementitious material development. Chapter 3 introduces the ISO-compliant LCA framework, establishing the foundation for environmental assessments and providing a comprehensive literature review of methodological advancements in evaluating emerging technologies. The results are presented in three stages: Chapter 4 examines the environmental impact of upscaling an emerging BR processing technology from laboratory to industrial scale, focusing on the influence of upscaling parameters and modelling uncertainties in preliminary environmental assessments. Chapter 5 aligns temporally with the upscaling discussed in Chapter 4; it evaluates various decarbonisation measures in traditional cement production to forecast the

environmental impact of conventional products in a future scenario where the upscaled technology is realised at an industrial scale. Chapter 6 models the environmental performance of processed BR at an industrial scale, incorporating it into cementitious materials and comparing it with conventional alternatives. The overall structure of the thesis is illustrated in Figure 1-3.

Chapter 2 provides a literature overview of the journey of bauxite ore, from extraction to residue generation, including a description of material characteristics. This is followed by a market analysis of the market evolution over time. This chapter summarises the various disposal and management methods employed historically, highlighting current practices and strategies. The discussion expands on BR utilisation, discussing potential applications and primary challenges and barriers to its large-scale reuse. Finally, this chapter concludes with an overview of the emerging BR technologies studied in this thesis.

Chapter 3 outlines the ISO-compliant LCA framework and its applications in evaluating emerging technologies for BR utilisation in the cement industry. This chapter introduces the theoretical background of LCA, including the key components such as goal and scope definition, functional unit (fu), inventory analysis, and life cycle impact assessment (LCIA). Following this, the various types of LCA (e.g., prospective, ex-ante, dynamic) identified in the existing literature are discussed while introducing the technology readiness level (TRL) and its relevance to LCA studies of emerging technologies. A comprehensive literature review on LCA applied to emerging technologies addresses critical aspects such as comparability, data challenges, scaling issues, and uncertainty. The chapter concludes with a focused literature review on LCA studies related to BR processing, setting the stage for the specific analyses conducted in this thesis.

Chapter 4 involves transitioning an BR vitrification as an exemplar of an emerging technology (from laboratory to industrial scale) and exploring model development. The environmental assessment examines the implications of upscaling to industrial-scale environmental performance, using BR vitrification as a case study while considering the influence of background changes such as the decarbonisation of the electricity grid. This chapter also examines the influence of different upscaling approaches on the environmental performance of BR vitrification, analysing the sensitivity of modelling assumptions and the uncertainty in the upscaling process as the technology transitions from laboratory to industrial scale

Chapter 5 applies prospective LCA to assess the climate change impacts of various decarbonisation measures, including clinker substitution with supplementary cementitious materials (SCMs), increasing the use of alternative fuels, kiln improvements, and implementation of carbon capture and storage (CCS). This research work considers both foreground and background system parameters. The chapter concludes with a detailed discussion of the results, highlighting the contributions of various decarbonisation measures and offering guidelines to reduce greenhouse gas intensities in conventional cement production.

Chapter 6 focuses on the LCA model development, incorporating both the decarbonisation of conventional clinker production and the integration of BR-derived cementitious materials at an industrial scale. The chapter includes an environmental performance comparison between BR-incorporated cementitious products and conventional ones. The chapter concludes with a discussion on the GHG reduction potential of BR utilisation in cementitious applications and offers guidelines for optimising these processes to achieve maximum environmental benefits.

Chapter 7 outlines the main contributions, summarises the key conclusions, and provides recommendations for future research.

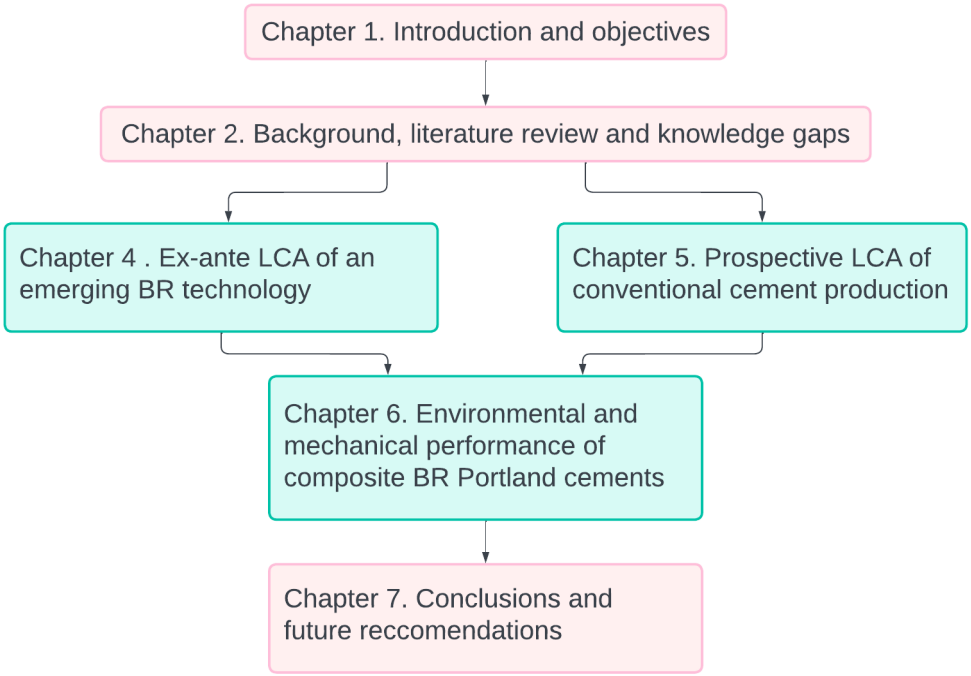


Figure 1-3: Thesis structure and interconnections between individual chapters.

Chapter 2. Literature review

Chapter 2 provides a comprehensive overview of BR generation, exploring past and current trends in disposal methods, and advancements in its application across various industrial sectors, particularly in its use as a construction material. The chapter concludes with a literature review of existing environmental assessments on BR incorporation as a cementitious material and a description of five emerging BR treatment technologies selected from academic and industrial research. Each technology is evaluated for its potential to transform BR from a waste by-product into a valuable cementitious material.

2.1 Bauxite ore to residue

2.1.1 Generation of bauxite residue

Bauxite residue, a red slurry known as "red mud," is the primary waste by-product of the Bayer process, the principal method for extracting alumina (Al_2O_3) from bauxite ore. Currently, there is no other economic alternative route to the Bayer process existing or likely to emerge within the next 50 years, hence the generation of BR will continue at a high level for many decades (International Aluminium Institute, 2020a). Depending on the quality of the bauxite ore, between 1.9 and 3.6 tonnes of bauxite are required to produce 1 tonne of alumina (Pontikes, 2005). While the production of 1 tonne of alumina generates between 1 and 1.5 tonnes of BR(Zhang et al., 2011). The Bayer process, known as the “alumina refining” stage is the first step in the aluminium production chain, where alumina is extracted from bauxite ore. The second step is the Hall–Héroult process, where the alumina is transformed into aluminium through reductive smelting (Murray, 1981).

The Bayer process involves two main processing steps; 1) digestion and pressure leaching, and 2) precipitation (Habashi, 1995). Figure *2-1* presents a simplified flowsheet of the Bayer process.

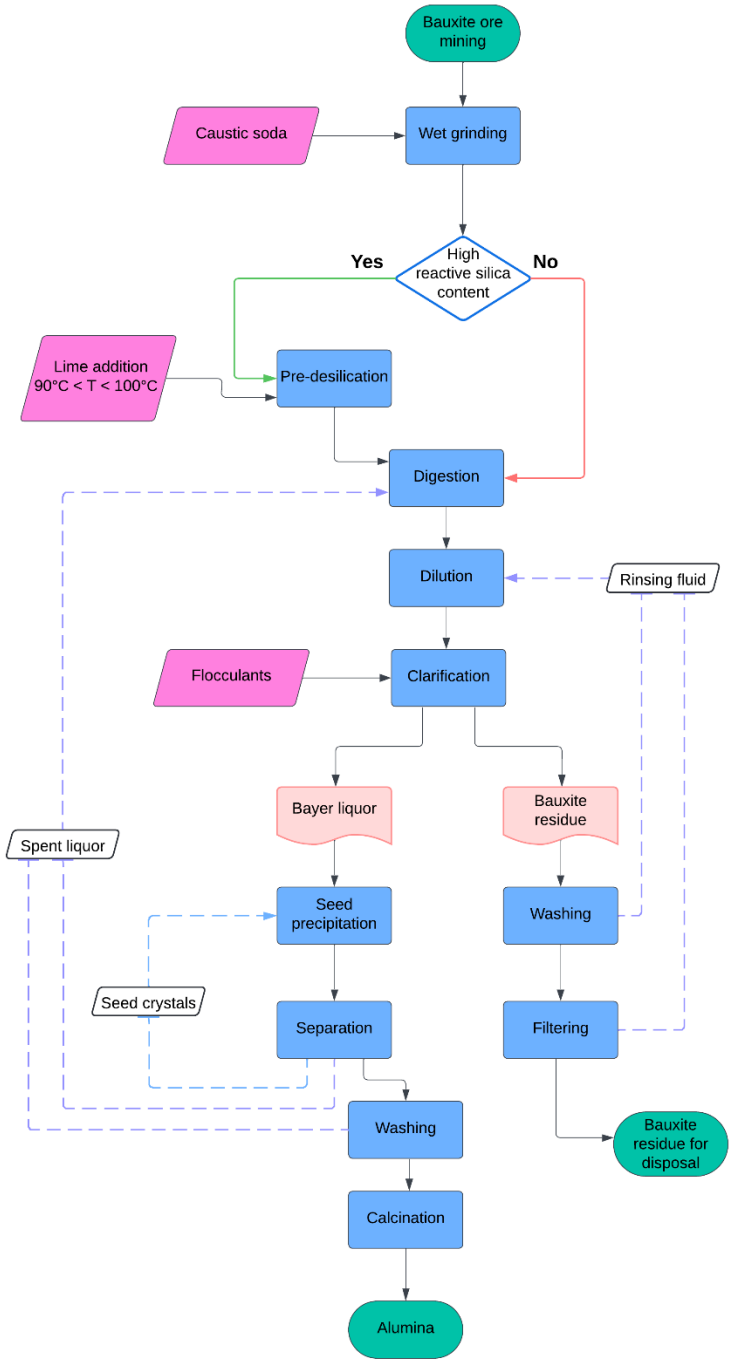


Figure 2-1 Simplified flowsheet of Bayer process (adapted from Lyu et al., 2021)

The extraction process begins with the mining of bauxite ore from both open-pit and underground mines. Mined bauxite is then mixed in a sodium hydroxide (caustic soda, NaOH) solution and water and crushed in a grinding mill. NaOH is added as a preparatory step prior to digestion to create an alkaline environment and enhance the extraction efficiency. Further grinding increases the surface area and maximises the extraction potential for alumina.

In cases of high silica content in the bauxite ore, the ground bauxite is transferred to stirred tanks with a caustic liquor at temperatures (~90-100°C) for 6-24 hours. This step ensures the reaction of caustic solution with the reactive silica present in minerals such as kaolinite or other aluminosilicates, which can otherwise react during the digestion, forming impurities like sodium aluminium silicate hydrates hindering the alumina extraction. Effectively, the "pre-desilication" minimises the impurity formation and controls the efficiency of the process enabling the reaction between alumina and caustic soda solution (Vaughan et al., 2019; Whittington et al., 1998). The first step in the Bayer process, known as "digestion and pressure leaching," involves mixing the ground bauxite with a hot concentrated NaOH solution at temperatures between 150-280°C and pressures up to 35 atmospheres. These conditions favour the reaction of alumina with the NaOH solution, forming soluble sodium aluminate (Na [Al $(OH)_4$]), while impurities remain behind as an insoluble residue known as "bauxite residue". Prior to the "clarification" step, the highly alkaline and viscous slurry, containing both the pregnant liquor (with dissolved aluminate) and undissolved BR (with additional impurities) is diluted. In the "dilution" step the concentrated slurry is mixed with water or recycled spent liquor from downstream processes. Dilution enhances the clarification and facilitates a better separation between BR and the Na [Al $(OH)_4$] solution while maintaining the liquor at optimal conditions, to ensure an effective precipitation (Brough and Jouhara, 2020; Lyu et al., 2021). The mixture is then allowed to settle in a series of clarifiers, where BR settles at the bottom and is subsequently removed.

Flocculants are added to promote the aggregation of fine particles and facilitate the clarification between the pregnant liquor and the bauxite residue. The waste BR slurry contains undissolved components (e.g., iron oxides, silicates, titanium dioxide), unrecovered liquor rich in NaOH, other elements added during the refining process (e.g., calcium from lime addition), and trace and heavy elements (e.g., including arsenic, chromium, lead; the concentration depends on the bauxite source). The BR underflow is introduced into a series of thickeners arranged in a counter-counter flow. Fresh water is introduced at the entrance of the last thickener moving in a counter-counter flow to the BR underflow, this configuration maximises the recovery of any unrecovered soluble substances (International Aluminium Association, 2022).

The Na [Al $(OH)_4$] solution (pregnant liquor) is decanted and filtered to remove any residual impurities. The solution is mixed with the recovered Na [Al $(OH)_4$] solution from the BR underflow, typically achieving a 96% recovery rate (industrial standard). In the "precipitation

step", the Na [Al $(OH)_4$ solution is cooled and seeded with fine aluminium hydroxide crystals to facilitate the precipitation of aluminium hydroxide (Al $(OH)_3$). The Al $(OH)_3$ solution is then filtered, and the crystals are heated in rotary kilns or fluidised bed calciners at temperatures around 1000-1100°C to remove water molecules and transform $Al(OH)_3$ into anhydrous alumina (Al_2O_3). The anhydrous alumina is further treated through the Hall-Héroult process to convert it into aluminium metal. The Hall-Héroult process extracts aluminium from alumina via electrolytic reduction, where alumina is dissolved in a molten electrolyte (composed of aluminium, sodium, and fluorine) at 955-965°C and electrolysed, producing aluminium at the carbon-lined cathode and oxygen gas at the graphite anodes (Kvande, 2010).

The washed BR is further treated to be suitable for disposal, this involves increasing the solid content through thickening or filtering (Dentoni et al., 2014; Power G. et al., 2011). After washing, the residue is contained in special facilities known as Bauxite Residue Disposal Areas (BRDA) (Figure *2-2*) or Residue Storage Areas (RSA).



Figure 2-2: Aerial view of the Aughinish Alumina Bauxite Residue Disposal Area (BRDA), including the nearby Shannon Estuary (Bruce, 2023).

2.1.2 BR characteristics

2.1.2.1 Composition

A large variability is observed in the composition of BR primarily influenced by the geographical origin of the bauxite ore, and secondly, as a result of the processing conditions during the Bayer process (Table *2-1*). In particular, the caustic soda recovery efficiency affects the residual alkalinity levels and pH (more details in section 2.1.2.2) (Lyu et al., 2021).

Lime addition also contributes to the variability in chemical composition, as it affects the calcium levels from forming compounds such as calcite and tri-calcium aluminate. Lime addition is optional and depends on the quality of the bauxite ore and the refinery practices. Different refineries follow different methods for managing the pH, impurity levels, and NaOH recovery. Typically, lime is added to maintain an optimal pH level, enabling the dissolution of alumina from bauxite during digestion. Additionally, lime facilitates digestion, regenerating NaOH by reacting with sodium carbonate present in the Bayer liquor.

There are two pathways for sodium carbonate formation during the Bayer process;

1. The NaOH reacts with carbon dioxide (CO_2) present in the atmosphere or from organic matter (e.g., flocculant) decomposition.

$$2NaOH + CO_2 \rightarrow Na_2CO_3 + H_2O \quad (2\text{-}1)$$

2. Sodium aluminate, formed during the digestion process, can react with CO_2 to form sodium carbonate and aluminium hydroxide

$$\mathrm{Al(OH)_3 + NaOH \rightarrow NaAl(OH)_4} \quad (2\text{-}2)$$

$$\mathrm{2NaAl(OH)_4 + CO_2 \rightarrow 2Al(OH)_3 + Na_2CO_3 + H_2O} \quad (2\text{-}3)$$

Lime (calcium oxide, CaO) is then added to the process to react with the sodium carbonate, regenerating NaOH and forming calcium carbonate ($CaCO_3$):

$$CaO + Na_2CO_3 + H_2O \rightarrow 2NaOH + CaCO_3 \quad (2\text{-}4)$$

Lime ensures a pure Bayer liquor by precipitating certain impurities, such as silica and phosphates, forming insoluble compounds. This also reduces equipment scaling and fouling.

In this reaction, calcium hydroxide ($Ca(OH)_2$ formed from lime) reacts with sodium silicate to form calcium silicate and regenerates sodium hydroxide.

- **Formation of Sodium Silicate**

$$SiO_2 + 2NaOH \rightarrow Na_2SiO_3 + H_2O \quad (2\text{-}5)$$

- **Precipitation of Calcium Silicate**

$$Na_2SiO_3 + Ca(OH)_2 \rightarrow CaSiO_3 + 2NaOH \quad (2\text{-}6)$$

Similarly in the case of phosphates, lime reacts forming insoluble calcium phosphate compounds. Here, $Ca(OH)_2$ reacts with sodium phosphate to form calcium phosphate ($Ca_3(PO_4)_2$) and regenerates sodium hydroxide.

- **Formation of Sodium Phosphate:**

$$P_2O_5 + 6NaOH \rightarrow 2Na_3PO_4 + 3H_2O \quad (2\text{-}7)$$

- **Precipitation of Calcium Phosphate:**

$$2Na_3PO_4 + 3Ca(OH)_2 \rightarrow Ca_3(PO_4)_2 + 6NaOH_2 \quad (2\text{-}8)$$

Table 2-1: Chemical composition range (%) of BR main components (International Aluminium Association, 2022).

Component	Typical range (%)
Fe_2O_3	5-60
Al_2O_3	5-30
TiO_2	0.3-15
CaO	2-14
SiO_2	3-50
Na_2O	1-10

The chemical composition and the ratio of these elements in BR play a significant role in the final properties and, hence, the ability to be utilised as a raw material for low-carbon clinker production or as an SCM (International Aluminium Institute, 2020). Further details on the varying types and sources of BR used in literature are reviewed (Putrevu et al., 2021).

2.1.2.2 Physical characteristics

The physical properties of BR, similar to the chemical composition, are influenced by the bauxite ore source and the processing conditions of each refinery (Putrevu et al., 2021). Generally, BR has a fine particle size distribution, appearing powdery when dry and clay-like when wet. It contains both coarse particles (> 100 μm), referred to as "sand", and finer particles known as "mud". Historically, sand and mud were separated to facilitate washing, but this practice is less common today.

BR slurry exhibits thixotropic behaviour, meaning it remains static when undisturbed but becomes fluid when agitated, necessitating special handling procedures. This behaviour is correlated with the solid content; as the solid content increases, the viscosity of the slurry increases exponentially. Particularly at solids content levels above 50%, the slurry becomes increasingly difficult to pump or agitate, resulting in higher yield stresses causing greater wear and tear on equipment e.g., pumps and pipelines (International Aluminium Association, 2022).

2.1.3 Evolution of bauxite residue

With the global population increasing, societal needs for transportation, construction, electricity, and renewable energy infrastructure are becoming more critical. Major trends such as rapid urbanisation, resource scarcity, demographic shifts, and technological advancements were considered to investigate their influence on the future alumina supply chain (CM Group, 2020).

Urbanisation is expected to rise by 68% by 2050, leading to increased demand for sustainable construction materials, infrastructure development, and transportation networks. Another source causing higher aluminium demand is linked to electricity grid expansion, particularly driven by the need for renewable energy infrastructure development and electric vehicle charging infrastructure, with a projected demand of 5.5 million tonnes. Technological advancements present a double-edged sword, with innovations in aluminium batteries and driverless vehicle technology promoting aluminium usage, while alternative emerging materials increase competition (Klauber et al., 2011).

China and India are expected to be major drivers of aluminium demand growth, especially as India's rapid electrification and increasing capacity for renewable energy accelerate. Two scenarios have been developed to forecast aluminium growth prospects: 1) an optimistic growth scenario assumes successful decarbonisation and CO_2 mitigation within primary aluminium production, while 2) a conservative growth scenario assumes reduced growth potential due to competition from alternative materials and slower decarbonisation efforts.

These scenarios are expected to lead to an annual aluminium production of 244-335 million tonnes by 2050, with bauxite ore resources estimated at 55 to 75 billion tonnes worldwide (Figure *2-3*). Of the bauxite ore consumed, 90% is associated with alumina production, while the remainder is used in other non-metallurgical applications. In 2009 the annual yield of BR was 120 million tonnes (Pontikes and Angelopoulos, 2013). This increased to 140 million tonnes by 2015 and 150 million tonnes by 2018 and a total inventory estimated at > 2.7 billion tonnes (Binnemans et al., 2015; Evans, 2016; Pontikes, 2016; International Aluminium Institute, 2018).

Figure *2-3* illustrates the projected increase in alumina demand and the corresponding rise in BR production by 2050.

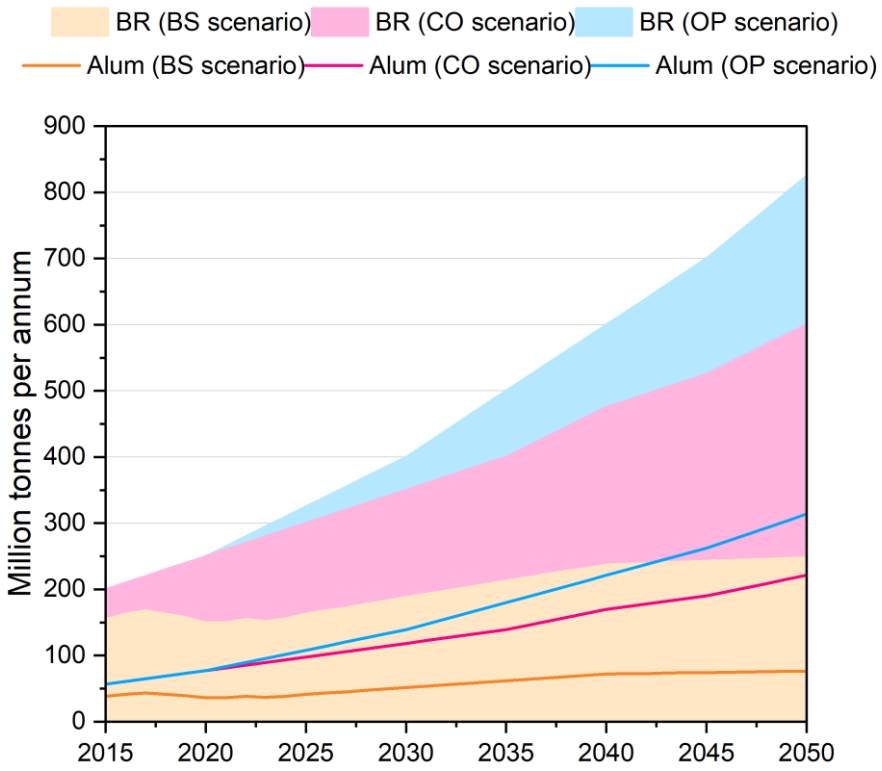


Figure 2-3: Forecast of alumina production and corresponding BR production growth by 2050. Three scenarios are presented: status-quo (baseline (BS) scenario) in orange, conservative (CO) scenario in pink, and optimistic (OP) scenario in purple (CM Group, 2020; International Aluminium Institute, 2018)

2.2 Disposal and management

BR waste management practices vary greatly across different regions and have evolved to be aligned with changes in economic and technological factors, environmental regulations, and policies. Global trends have seen improvements in BR management practises with more immediate action stemming from more consistent and harmonised legal frameworks within the European Union (EU). According to the European List of Waste, BR is currently classified as a non-hazardous material, potentially re-classifying as an inert waste if low moisture contents are kept, preventing leachability (European Commission, 2014). The residue must be landfilled in compliance with Directive 1999/31/EC (Council of European Union, 1999), with leaching control and prevention measures selected depending on the residue's classification. EU

companies can reduce their emissions and waste by implementing technologies and practices suggested by the Best Available Techniques Reference Documents (BREFs) for the Non-Ferrous Metals Industries. In the field of alumina production, these documents provide measures to minimise the alkalinity of the disposed residue, maximising the caustic solution recovery and its solids content (Cusano et al., 2017). For example, seawater, freshwater, or steam have been reported to lower the pH (Sutar et al., 2014). In cases where high-carbon emission plants such as ammonia production facilities are located near the refineries, the CO_2 stream can be fed through the residue and reduce the BR alkalinity, promoting industrial symbiosis. The pH level of the residue can reach up to 13 from the presence of alkaline sodium compounds, such as sodium carbonate and NaOH.

Despite efforts to improve management strategies and explore reuse applications, only a small fraction (2-3%) of the annual 150 million tonnes of BR produced is utilised (Evans, 2016). This section outlines the historical and current disposal methods of BR (Figure *2-4*), highlighting the progress and ongoing challenges in BR management.

2.2.1 Marine and river discharge

The most common disposal approach from the 1940s to 1960s was the “discharge” of BR directly into the sea via pipelines or ships, and into rivers and estuaries, often without neutralisation or any measures to mitigate its high alkalinity. This practice ceased entirely by the end of 2015 due to its detrimental environmental impact, destroying coastal seabeds, disrupting marine ecosystems, and pollution from toxic metals. Marine discharge has now become a "last resort" technology, accommodating less than 2-3% of global alumina production (Power G. et al., 2011).

2.2.2 Lagooning

Before 1980, Lagooning was the prevalent disposal method where BR slurry, with low solid content (18-22%), was pumped into storage areas created by dams for secure containment (European Aluminium (EA), 2015). This method posed environmental risks, including soil and groundwater contamination, which were mitigated by artificial liners and leachate control systems. Despite these measures, the risk of disasters from containment structure failures

remained high, as evidenced by incidents like the Ajka Timföldgyár spill in Hungary in 2010 (Mayes et al., 2011; Boily R., 2012).

2.2.3 Dry stacking

In the early 1980s, the inventory of BR reached 1 billion tonnes and was recognised as an environmental concern hindering the sustainable future of the industry (United Nations Industrial Development Organization, 1985). Dry stacking emerged as the preferred disposal method. This technique involved thickening and washing the residue before storage, allowing for the reuse and recovery of valuable soda and alumina in the Bayer process. In dry stacking, BR is thickened to a paste with 48-55% solid content, which can be pumped and deposited into thin layers to dry, reaching a final dry density of ~ 70% solid content (Power. et al., 2011). This method decreases environmental risks related to containment structure failure and liquor leakage, minimising land demand for lagoon storage, but introducing challenges like dust pollution and re-vegetation challenges (Dentoni et al., 2014).

2.2.4 Mud farming

In mud farming, BR (solids content ~ 50-70%) is spread in thin layers over large areas and dries through natural evaporation. The process is iterative, with layers being re-applied once the top layer dries off. Mud farming is climate-dependent, with the climate significantly influencing the degree and duration of the drying process. A solid content of 60-65% can be achieved, enabling carbonation, and reducing alkalinity through CO_2 sequestration (Svendsen, 2019; (Sutar et al., 2014; Willan and Ghataora, 2015)

2.2.5 Dry disposal and filtration

Dry disposal methods are considered the Best Available Techniques (BAT) for BR disposal treatment. Dry disposal employs filters that can provide a better degree of dewatering compared to thickeners, reaching solid contents > 65%, resulting in semi-dry cakes (Power et al., 2011). The dry, non-thixotropic BR cake can be easily transported more safely and economically to containment facilities (European Aluminium (EA), 2015; Power G. et al., 2011).

Currently, the recommended disposal practices seek the disposal of BR in designated BRDAs, preferably using dry stacking and mud-farming methods. (Evans, 2015; Gomes et al., 2016; Klauber et al., 2011).

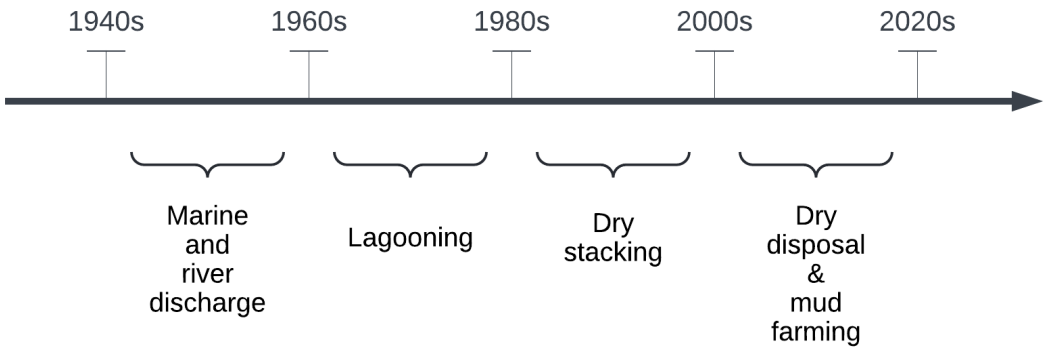


Figure 2-4: Historical methods for BR disposal management

2.3 Utilisation

The substantial quantities of BR generated, and the already significant existing inventory raised the following question of "what to do with it" (Klauber et al., 2011). As sustainable development becomes increasingly prioritised, the need to adopt a circular economy approach has been highlighted as a solution. Reusing and repurposing BR would not only address waste management concerns but also improve resource efficiency.

Numerous studies have explored and proposed BR incorporation in the construction material development (cement, bricks, tiles, road construction, concrete, geopolymers, ceramics, pigments, and fillers), agriculture (soil amendment and land reclamation), metallurgy (critical metal and rare earth element extraction), environmental management (water treatment and pollution control) as seen in Figure 2-5 (Sutar et al., 2014; Venkatesh et al., 2019; Putrevu et al., 2021). Existing reviews have analysed and expanded the discussion by investigating factors such as the influence of BR origin and variability in composition, physical and chemical properties, and processing technologies (Khairul et al., 2019).

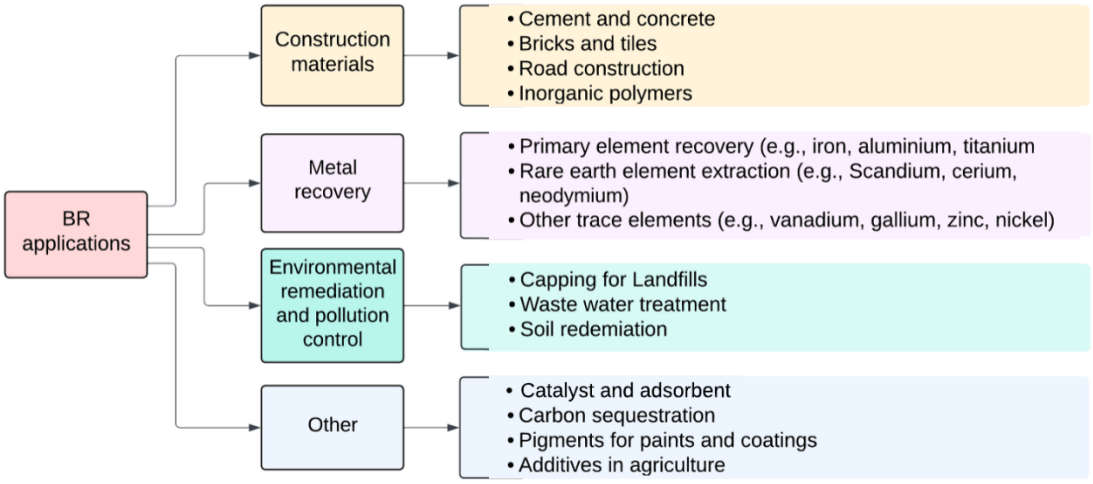


Figure 2-5: Distribution of BR utilisation in industrial applications (adapted from Klauber et al., 2011)

2.3.1 BR Utilisation in construction materials

The costs associated with BR disposal, reaching 1-2% of the price of alumina (Tsakiridis et al., 2004), along with the high global production volumes and some favourable physio-chemical properties of BR, have driven industry and academia to explore its utilisation in construction.

It can be used in the production of bricks, mortars, and ceramics, or as an additive in concrete or cement production, depending on its hydraulic and pozzolanic properties (Shinomiya et al., 2015). These applications have been shown to achieve net environmental benefits and improve environmental performance compared to conventional products. Further details of BR incorporated in cementitious materials can be found in the reviews of Venkatesh et al. (2019) and Putrevu et al. (2021). While historical trends in BR processing and utilisation as early as the 1970s are discussed in Klauber et al. (2011) and Pontikes and Angelopoulos (2013).

2.3.1.1 Supplementary cementitious material

Heavy metal presence can significantly affect the properties of BR incorporated in construction; however, combining BR with granulated blast furnace slag (GBFS), another widely generated material from the metal industry, removed 53%–77% of heavy metal ions

(Chen et al., 2019). This combination leverages the desirable physio-chemical properties of both materials, making BR a viable option as an SCM. In other work, BR was investigated for its pozzolanic properties in replacing fly ash in self-compacting concrete. At replacement ratios between 10%-40%, self-compacting concretes with comparable compressive strength were produced (Liu and Poon, 2016). Reviewing the processing technology of BR as an SCM and the corresponding influence on the mechanical, thermal and durability properties as well as microstructure, an interesting observation was found that too little or too much replacement of BR addition hinders the mechanical properties, hence optimising the mix ratio is crucial (Anirudh et al., 2020; Patangia et al., 2023). More often treated bauxite residue, undertaking calcination or sintering showed improved results in concrete, with many studies exploring the influence of system parameters such as replacement ratio and operating temperature.

For instance, in calcination, BR is thermally treated between 600-800°C to alter its chemical structure by improving the degree of polymerisation and enhancing the degree of hydration products, i.e., calcium aluminium silicate hydrate (Liu et al., 2014; Reddy et al., 2021). A 5% addition of calcined BR (treated at various temperatures ranging from 600 to 800 °C) to mortars showed similar strength values as the conventional cement mortar (Manfroi et al., 2014). Even higher replacement ratios were found feasible, giving comparable compressive strength to pure cement paste at 28 days and 30% of calcined BR to replace PC (Luo et al., 2019). Even though BR calcination at a low temperature of 600 °C for 2 hrs showed higher pozzolanic activity, leading to higher compressive strengths compared to untreated BR, it only allowed for a 10% replacement ratio instead of only 5% untreated (Venkatesh et al., 2020). Reviews have shown that studies exploring the processing of BR through calcination observed enhanced physical properties and mechanical performance compared to untreated BR allowing for higher replacement ratios of 20-30%. Still, the potential replacement ratio remains dependant on the original composition, processing technology, and operating conditions (Putrevu et al., 2021).

Further work on laboratory investigations involving the use of BR as a pozzolanic material or as a filler in cement blends is presented in (International Aluminium Institute, 2020a)

2.3.1.2 BR-derived specialty cements

Several BR-derived specialty types of cement were developed, such as ferrite aluminate cement (consisting of lime, bauxite, and BR) and sulfoaluminate (consisting of lime, bauxite, BR, and gypsum). In both cases, cements with high BR content (5-55%) were investigated. For ferrite aluminate cement, the BR content and firing temperature were identified as the primary parameters influencing the compressive strength, with replacement ratios between 25-35% and the firing temperature at 1250°C as the optimal conditions. Similarly, for sulfoaluminate ferrites the firing temperature was identified as a key parameter and the raw mix composition. Higher lime contents resulted in considerable higher compressive strengths, caused by the larger formation of C_4AF phase due to higher BR content (Liu and Zhang, 2011. The BR content up to 50% were investigated in the raw meal, showing the optimal proportions at 30–35 wt.% BR, 15–20 wt.% bauxite, 7.5–10 wt.% gypsum and 45–50 wt.% lime, depending on the type of cement desired. The firing temperature of 1250°C is lower than the temperature used in OPC production (1450 °C) whereas higher firing temperatures (up to 1300 °C) resulted in cements with higher compressive strengths (Singh et al., 1996). Three raw meal compositions consisting of lime, BR, spent bed ash, pulverised coal combustion (PCC) fly ash and bauxite were developed to investigate the influence of raw meal composition on calcium sulfoaluminate-ferrite cement performance. BR content varied between 15 to 38 wt.% in the raw meal achieving moderate mechanical performance (Duvallet and Jewell, 2009).

2.3.1.3 BR incorporation in PC clinker

Concrete made with cement produced from clinker that included 1% BR in the kiln feed showed no effects on the mechanical properties of the clinker, but high alkalinity of BR was noted as a potential concern for crack initiation, spalling and subsequently, loss of strength (Tsakiridis et al., 2004). Further work, investigating filter pressed BR (water content 26-28%) was added at 1, 3 and 5%, respectively, into the raw meals composed of limestone and sandstone to produce OPC clinker and fired to 1550°C. A slightly higher BR content (3.5%) was investigated, mixed with 74.8% limestone, 11.4% schist, 3% bauxite and 7.3% sand as a raw mix composition to prepare Portland cement clinker. The mix was sintered at 1450°C and the produced clinker was further mixed with gypsum (5%) to form the final cement product. The chemical composition of the produced clinker was similar to that of OPC clinker,

and the incorporation of BR at 3.5% did not affect the mineralogical composition not any physical properties (i.e., mechanical strength, grindability, and setting time) (Singh et al., 1997, 1996). The BR-incorporated clinker was then mixed with 5% gypsum producing comparable physical properties (specific surface area, setting time and water demand) with conventional OPC (Vangelatos et al., 2009). Even though the feasibility of BR as a raw meal was presented, at least in terms of mechanical properties, the 1-5% replacement ratio remains low for the vast inventory of BR and a more suitable high-volume application is needed. With an effort to increase the BR content, untreated BR at 50% replacement showed a compressive strength loss but an overall good pozzolanic activity, suitable for non-structural applications (Ribeiro et al., 2011).

2.3.1.4 Alkali activate material, inorganic polymer, geopolymer

Geopolymers, is a form of inorganic polymers, where materials with aluminosilicates are reacted with an alkaline solution to form a material suitable in cementitious applications, high tech ceramics and composite applications such as defence, space and marine products, and even petrochemical treatments (Promentilla et al., 2016; Sutar et al., 2014). Research at Leuven University has explored the use of BR in producing geopolymers as a cementitious binder in combination with various metallurgic slags. The Extraordinary Leuven Cement (ELCE) product demonstrates BR's potential in high-performance cementitious applications (Cusano et al., 2017). BR-derived inorganic polymers were studied by firing BR at 1100 °C mixed at 88.6 wt.% BR, 1.4 wt.% coke, and 10.0 wt.% silica, showed promising results in terms of mechanical performance (Hertel et al., 2016). Studies on repolymerisation showed satisfactory compressive strength results, and good interaction of BR with fly ash, showing a potential replacement ratio between 5-20% (Giannopoulou et al., 2009) (Kumar and Kumar, 2013). Promising results were seen in BR sintering with blast-furnace slag and a solid alkali activator (such as solid water glass and sodium aluminate clinkers). The clinker-free alkali-activated cement possessed high compressive strength (56 MPa at 28 days and 65 MPa at 90 days) and excellent resistance against chemical attacks and freezing–thawing cycles, mainly due to the denser microstructure and lower total porosity in the hydrated paste in comparison with those of the reference OPC (Pan et al., 2003, 2002)

A key limitation with the wide use of geopolymers is the variability in BR composition, hindering the production of consistent geopolymer properties, and also the influence of potential impurities which can interfere with the geopolymerisation process. Another limitation is the lack of long-term monitoring and durability evaluation of BR-derived geopolymers under various environmental conditions. Additional challenge is the higher cost of geopolymerisation process due to the requirement for high-purity alkaline activators which are considerably more expensive than the raw materials used in conventional cement production. Finally, the lack of standardised protocols and guidelines hinders the broader acceptance and regulatory approval of geopolymer technology, delaying the introduction of these materials into the market (Van Deventer et al., 2012; Reddy et al., 2021; Salim et al., 2023; Wang et al., 2019)

2.3.1.5 BR as a filler in road construction

BR has been successfully incorporated in road construction applications such as a filler in bituminous mixtures or in pavement base layers. The material often meets or exceeds national guidelines, providing performances superior to conventional materials (Lima et al., 2017). Its use in this sector not only capitalises on its inert properties but also offers a sustainable alternative to traditional road construction materials. However, environmental concerns such as leachability of contaminants and high alkalinity, as well as high variability in BR composition, local regulations, costs, and public perception restrict its large-scale use.

2.3.2 Metallurgical applications

Numerous technologies have been explored for the recovery of raw materials and recycling valuable metals (Al, Fe, Na, Ti, Sc, Ga, V) from BR, such as direct magnetic separation, smelting in a blast furnace, and pyrometallurgical and hydrometallurgical recovery (Liu and Naidu, 2014; Wang et al., 2019) as well as different extracting agents (Ujaczki et al., 2017). The typical amounts of Ti, Si, Fe, Na and Al in BR are 2–12%, 1–9%, 14–45%, 1–6% and 5–14%, respectively (Kumar et al., 2006). The recovery of critical raw materials (rare earth elements and valuable metals) from BR has been a topic of great significance reviewed in several studies (Binnemans et al., 2015; (Borra et al., 2016); Liu and Li, 2015; Pan et al., 2023; Ujaczki et al., 2018).

Among these, direct magnetic separation has emerged as a promising iron (Fe) recovery technique. The working principle depends on the magnetic properties of ferromagnetic materials, such as iron, to separate them from non-magnetic materials, also known as paramagnetic. The literature highlights that direct magnetic separation has a satisfactory separation efficiency and reduces energy consumption compared to traditional pyrometallurgical recovery processes (Xiang, 2001). Subsequent work confirmed the feasibility of this method, enabling effective iron recovery, while the non-magnetic residue can be repurposed in construction materials (Hammond et al., 2013). In addition to Fe recovery, titanium (Ti) recovery from BR has been studied using both pyrometallurgical and hydrometallurgical approaches. For example, pigment-grade titanium oxide (TiO_2), alongside aluminium oxide (Al_2O_3) and pig iron, can be efficiently recovered from Turkish BR through sintering and smelting processes that incorporate dolomite and coke (Erçağ and Apak, 1997).

Recovery methods can be broadly categorised into direct leaching, leaching following pyrometallurgical or mechanical operations, and hybrid approaches combining these techniques. In direct hydrometallurgical leaching, acids such as H_2SO_4, HCl, or HNO_3 are employed to extract critical raw materials (CRMs). However, this method can be less selective and requires significant acid consumption. Leaching following pyrometallurgical treatment achieves a better selective extraction efficiency, although it is often associated with higher energy requirements and increased operational costs (Binnemans et al., 2015; Borra et al., 2016; Guo et al., 2009). Several studies have explored the selective extraction of specific elements, such as nickel (Ni) and cobalt (Co) from Fe-rich lateritic ores (Koutsoupa et al., 2021; Önal et al., 2015; Rivera et al., 2018; Tanvar et al., 2023), and neodymium (Nd) from Nd–Fe–B magnets (Önal et al., 2015; Reid et al., 2017).

Post-leaching, the dissolved materials undergo further purification and concentration using techniques such as solvent extraction, ion exchange, carbon adsorption, precipitation, and nanofiltration. For example, nanofiltration, a pressure-driven process that utilises fixed charges and steric effects, has been identified as a promising alternative for CRM recovery. However, the practicality of this technique remains limited due to the lack of large-scale testing (Yaroshchuk et al., 2009; Zimmermann et al., 2014). The use of non-aqueous solvents compared to conventional water and organic solvent systems has been investigated, with some studies exploring ion exchange potentials, such as for removing vanadium from synthetic BR leachate (Liu and Naidu, 2014; Pontalier et al., 1997).

During the pyrometallurgical approach, BR undergoes an initial thermal treatment known as carbothermic reduction, where the iron oxides present are transformed into magnetite or metallic iron, both exhibiting stronger magnetic properties. The transformation significantly improves their recovery potential. This method also enables the recovery of other valuable metals, such as aluminium, titanium, and rare earth elements (Kumar et al., 1998). The high concentrations of iron and aluminium in BR position it as a promising secondary ore, with studies reporting recovery rates as high as 86% for iron and 88% for aluminium (Sutar et al., 2014).

2.3.3 Environmental protection applications

As an inert material, BR has been explored as a surface cover or capping material in landfill applications. It can also serve as a geological liner to limit leachate or enhance drainage. The versatility of BR allows for its application in various types of landfills (Wang et al., 2019). However, compared to using clay, conventional capping material, heavy metal leaching and dust formation and spreading are higher for BR. Nevertheless, experimental studies have shown that the leachable potential toxic elements are within the permissible limits set by environmental protection agencies, causing no harm to the environmental or human health (Evans, 2016; Reddy et al., 2021).

BR has demonstrated the capability of absorbing various heavy metal ions, making it useful for pollution control. It can be applied in wastewater treatment, soil remediation, and the removal of sulfur dioxide from waste gases (Ciccu et al., 2003).

2.3.4 Conclusion

Despite these advancements, the industrial utilisation of BR remains minimal, with only 2-3% of the annual production currently being exploited. Currently, BR holds no inherent market value and incurs disposal costs that vary depending on the chosen method, site location, and transportation logistics. The general goal is to identify a service or application of BR in a market with a gap for high-volume products. Key considerations include the environmental health and safety risks associated with implementing and upscaling this technology or

application (European Aluminium, 2015; Evans, 2016). More importantly, to ensure economic viability considering current disposal expenses and potential savings from co-use with other industrial by-products. Finally, BR-derived products should aim for at least comparable performance to be competitive with alternative materials in areas such as quality, costs, and risk (Klauber et al., 2011). Typically, the potential applications of BR are in an already mature market, with conventional, readily available, and inexpensive materials, necessitating a highly competitive competitor. Stakeholder consultation and regulatory challenges are additional issues, as community acceptance and legal responsibilities can postpone or terminate progress. While utilising other industrial waste streams alongside BR is gaining attention, the task of developing economically feasible and environmentally acceptable solutions remains complex, requiring coordinated efforts from research laboratories and industry.

Moreover, the chemical and physical characteristics of BR, including its composition, mineralogy, and particle size, differ significantly between sites. These variations complicate handling and processing, thereby impacting the effectiveness and efficiency of recovery technologies and influencing the overall costs associated with BR management (Power et al., 2011). In addition, the lack of comprehensive environmental assessments and risk analyses prevents a full evaluation of the true impact of these techniques, particularly when compared to conventional mining technologies (Ujaczki et al., 2018; Gomes et al., 2016).

The transition towards greater use of BR in cement has involved increasingly active participation from refineries to support research, collaborate across disciplines, and build a supply chain to transform BR from waste into a marketable product. Although BR usage in cement production was initially given moderate priority in the Technology Roadmap for BR Treatment and Utilization by The Aluminum Association (2000), the alumina industry has recognized the need for higher efficiency being reflected in the Alumina Technology Roadmaps of 2001, 2006, and 2010, which set a goal for 20% BR utilization by 2025. Building on this foundation, the International Aluminium Institute (IAI) has published a series of comprehensive reports and technical guidelines, emphasizing the use of BR in supplementary cementitious materials (SCMs), specialty cement, and Portland cement clinker production to achieve up to 30% of global BR production utilisation in cement and concrete by 2050 (International Aluminium Institute, 2020a, 2020b, 2020c).

BR disposal and management practices are followed according to local laws, regulations, and

company-specific practices. These operations follow the engineering and operational standards of the individual plant as there are no globally binding laws or standards for BR management. Hence, deviating from current management practices, by repurposing BR into a new (cementitious) product, raises concerns amongst the stakeholders due to the potential risks involved. For such a change to be considered, it must meet strict criteria related to the relevant economic, environmental, social, and legal factors. Even then, successfully implementing these changes faces significant barriers, including the need to demonstrate that the new approach offers greater value and lower risk than current practices.

Practical challenges associated with geographical, jurisdictional, and operational factors further complicate the transition from storage to utilisation. These challenges include securing funding, establishing processing and distribution systems, and gaining stakeholder acceptance. Moreover, any solution must be compatible with the regional economy—what works in China or India may not be suitable for Australia or Brazil.

2.4 Barriers to reuse

As discussed in Section 2.3, BR has been extensively researched for its potential utilisation in industry. Apart from finding an economically viable and easily applicable technology to large-scale operations, challenges remain concerning cost and risk. Risk is raised when stakeholders are involved, with a key question which remains valid today “What would a given alumina producer pay, at the refinery gate, to have the residue taken away and have no or limited future liability?”. Other barriers hindering its large-scale integration arise from its processing with caustic solution forming this alkaline and toxic nature. This complicates large-scale utilisation due to concerns about heavy metal leaching, radioactivity, high alkalinity, hazardous ratings, moisture level and transportation. Table *2-2* discusses further these barriers. BR can be neutralised prior to disposal, minimising any damaging effects. In cases where BR is not neutralised, safety and environmental hazards could be crucial if contact of humans and wildlife with alkaline liquor and mud, and contamination of surface and ground waters by leaching caustic liquor and other contaminants. Specifically, issues regarding heavy metals leaching and radioactivity are particularly important for its use as a construction material.

Review on BR utilisation as a construction material highlighted key areas of improvement. Areas such as material and technological optimisation have been extensively studied, investigating BR's properties and performance (physical and chemical properties) under varying processing conditions and its synergistic effect with other cementitious materials (e.g., fly ash, silica fume, limestone). However, the is a lack of work on assessing the environmental impacts of using BR on a large-scale and analysing the cost-effectiveness, potential market adoption, and social acceptance (Pontikes and Angelopoulos, 2013).

Table 2-2: Critical barriers hindering large-scale BR utilisation (Almeida et al., 2020; International Aluminium Institute, 2020c, 2020a, 2020b)

Barriers	
Heavy metal leaching	Problematic at extreme pH levels, particularly the leaching of chromium and arsenic contaminating construction materials and posing environmental and health risks.
Radioactivity	Bauxite ore contains low levels of naturally occurring radioactivity materials (NORM) such as ^{238}Uranium and ^{232}Thorium. Bauxite processing into BR enhances the radioactivity levels defining BR as a technologically enhanced naturally occurring radioactivity material (TENORM). Even though direct measurements of radioactivity are performed to ensure the levels are within the EU guidelines, public perception of radioactivity and long-term exposure risks in buildings hinder the application and further progression (Nuccetelli et al., 2015).
Alkalinity	High pH levels can affect the durability and stability of BR-incorporated construction materials reducing the weathering resistance, potentially leading to degradation or compatibility issues with other materials. Filter press, carbonation and acid neutralisation are a few mitigation methods found in literature (Cooling, 2007).
Hazardous rating	In certain jurisdictions, the hazardous characteristics of BR (which depend on composition, heavy metal concentration and pH levels) increase the cost of handling, requiring regulatory compliance, and testing to increase safety in building materials. Implementing press filtering reduces the pH and soda content to a level that avoids skin and eye irritation criteria as judged by the appropriate tests.
Moisture level	A low moisture content reduces the cost associated with the transportation and energy required for the drying. Methods such as using flocculants and press filters can reduce the moisture levels to < 30%.
Transport costs	Due to its bulk and potentially hazardous classification, transporting BR to manufacturing sites can be expensive, decreasing the cost-benefit of incorporating it in construction. Whilst the high alkalinity does not impose a problem with steel corrosion, it can cause pitting of aluminium which is a part of the UN transport code. Use of neutralised products would minimise this issue.

2.5 Remediation

Remediation and rehabilitation of BR sites are undertaken when a site has reached its maximum capacity and cannot be expanded, or when a BR facility closes, often due to the shutdown of an alumina refinery, economic challenges, depletion of bauxite reserves, or regulatory issues.

As BR facilities reach maximum capacity, remediating and rehabilitating are in order. In some cases, the site is left to naturally rehabilitate through the colonisation of wind-dispersed and animal-transported plant species. Alternatively, capping methods, which involve covering the residue with topsoil or synthetic materials are employed. The choice of capping materials and techniques is influenced by local regulations, material availability, and the intended future use of the rehabilitated site (Lyew-Ayee et al., 2011). Manure and gypsum are examples of alternative materials used to promote vegetation and restore the ecological value of the site (Bray et al., 2018). The type of vegetation planted is selected based on the local climate and soil conditions. After a facility is closed, long-term monitoring is conducted to ensure the effectiveness of the remediation measures and that the site remains stable and safe.

2.6 Emerging technologies for BR utilisation in composite cements

2.6.1 Overview of emerging BR technologies studied in this thesis

Multiple options were proposed for the utilisation of BR in several industries. Critical raw materials and rare earth element extraction were initially given attention due to their potential high-value market. However metallurgical transformation of BR, even if successful, the considerable costs remain as the stumbling block hindering any further development and implementation. Hence, research has focused on lower-risk opportunities, such as the implementation in the construction and material sector. For instance, the development of BR-derived SCMs, low-carbon PC clinkers, and specialty cements. Additionally, BR can be used as a component in light-weight aggregates, bricks, and blocks, or as a filler, and pigment (Klauber et al., 2011).

Building on the shift towards more viable and lower-risk applications of BR, this thesis focuses on emerging technologies such as calcination, vitrification, sintering, and smelting to explore their potential for producing BR-derived cementitious materials, including SCMs, binders, and clinkers. Existing studies have shown promising results, with high substitution rates (>25%) and good mechanical properties (Balomenos et al., 2024; Bullerjahn and Mehringskötter, 2021; Giels, 2023; Giels et al., 2023, 2022; Hertel et al., 2023; Peys et al., 2022). These technologies could offer a solution by creating synergistic value, reducing waste from the alumina industry by developing novel cementitious materials and lowering the environmental impact of conventional cement production. The proposed LCA framework is applied to five case studies derived from both academic and industrial research and development, part of the EU HORIZON 2020 project 'ReActiv' (Grant Agreement No. 958208). These case studies are briefly introduced below and will be examined in detail in Chapter 5.

The first three technologies focus on converting BR into a supplementary cementitious material, aiming to achieve at least 30% clinker replacement in cement binders. The remaining two case studies explore the development of new, alternative to OPC, BR-derived clinkers for specialty cements, specifically calcium sulfoaluminate clinker and calcium aluminate slag. These studies investigate the potential of converting BR into reactive cementitious materials, creating a synergistic value chain between the alumina and cement industries, and aiming to reduce environmental impact through industrial symbiosis. This contributes to several UN 17 sustainability development goals by (1) sustainable consumption and production promoting resource and energy efficiency, (2) urgent action to combat climate change and its impacts, (3) building resilient infrastructure, sustainable industrialisation and fostering innovation.

2.6.1.1 Co-calcination of BR with Kaolin clay

The industrial-scale model involves blending of unmodified filter-pressed BR (~25% moisture content) with dry kaolin clay, assuming a similar particle size (Figure *2-6*). If only coarse material is available, milling is required before mixing. Calcination is conducted in a multiple hearth furnace (MHF) at 700 °C for better temperature control. The dust emissions are filtered, and the hot off-gas stream is directed through a heat exchanger to preheat the air used in the furnace. The co-calcined material is then cooled and ready for immediate use as an SCM.

Currently, the technology has only reached the pilot scale for testing. The pilot test is being conducted at the MYTILINEOS aluminium production facility in Greece, employing an indirectly heated rotary kiln with a maximum feed rate of 300 kg per hour. The co-calcination process is currently operating at TRL 6, where the system is tested under conditions closely resembling those in full-scale deployment. The pilot process has achieved stable operation at this stage, but it is not yet running at full capacity, with larger-scale trials needed to validate its performance under maximum load conditions.

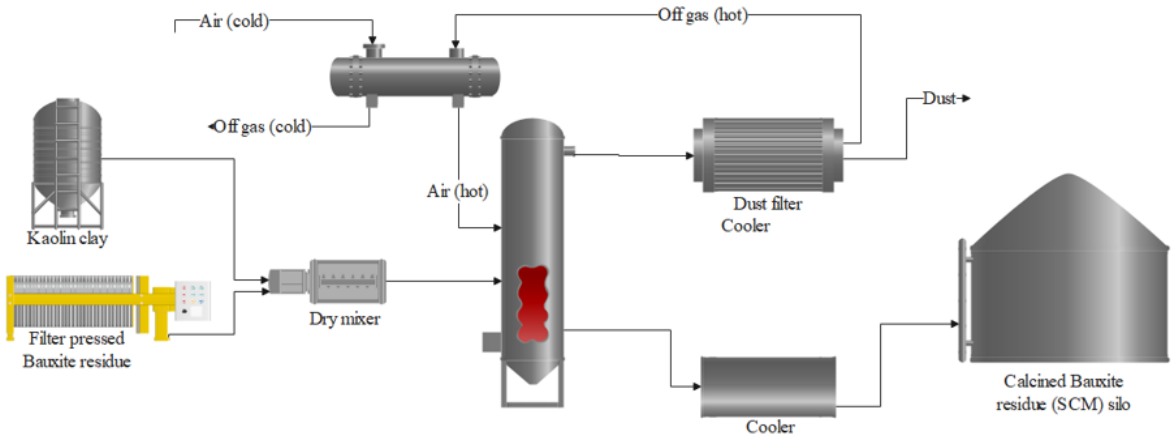


Figure 2-6: Process flow diagram of BR calcination technology (adapted from Konlechner et al., 2023)

2.6.1.2 Vitrification of BR

The industrial-scale model involves blending of unmodified filter-pressed BR mixed with quartz-rich sand (silica source), milled coke (carbon source) and limestone (calcia source) to achieve the desired final chemical composition for the produced SCM (Figure *2-7*). This targeted composition, comprising CaO (18 wt.%) and SiO_2 (20 wt.%), facilitates the formation of an alkaline glassy material that can effectively substitute Portland clinker. The resulting mixture undergoes pelletisation and is then transferred to a direct-fired rotary kiln via a Lepol preheater to reduce dust formation. Vitrification occurs at temperatures ranging from 1200 to 1350 °C, in a slightly reducing atmosphere producing a partly to fully molten mixture. Off-gases are filtered and can be used for heat recovery, to dry input material (i.e., BR). Rapid cooling of the vitrified BR melt maximizes the formation of a glassy and reactive phase. The produced glassy material is then milled to attain a specific surface area of 4000-4500 cm^2/g,

At this stage, two pilot plants are employed to test the upscaling of BR vitrification into an SCM. The first plant, located at KU Leuven in Belgium, uses a rotary induction furnace and is expected to produce 100-150 kg of slag per day. However, due to delays caused by technical modifications (such as installing a new coil), a second pilot plant was set up at UCT Prague in Čížkovice, Czech Republic. This plant uses a rotary kiln with a minimum feed rate of 10 kg per hour and targets a 5 ton production. The vitrification process is currently at TRL 6, where it is being tested under conditions similar to those expected in full-scale operation. The pilot plants are running steadily but are not yet operating at full capacity.

Additional research investigates vitrification as a technology to produce calcium sulfoaluminate-ferrite (CSAF) cement by varying the raw mix composition. Specifically, by incorporating 35 wt.% bauxite residue (BR) and using a clinkering temperature of 1300°C, vitrification can generate CSAF clinker, a lower-carbon alternative to PC clinker. This strategy enhances the flexibility of vitrification as a technology, as it allows for tailored cement compositions. Compared to OPC, CSAF cement has a higher amount of alumina and sulfate-rich minerals, making it suitable for applications where high early strength is required.

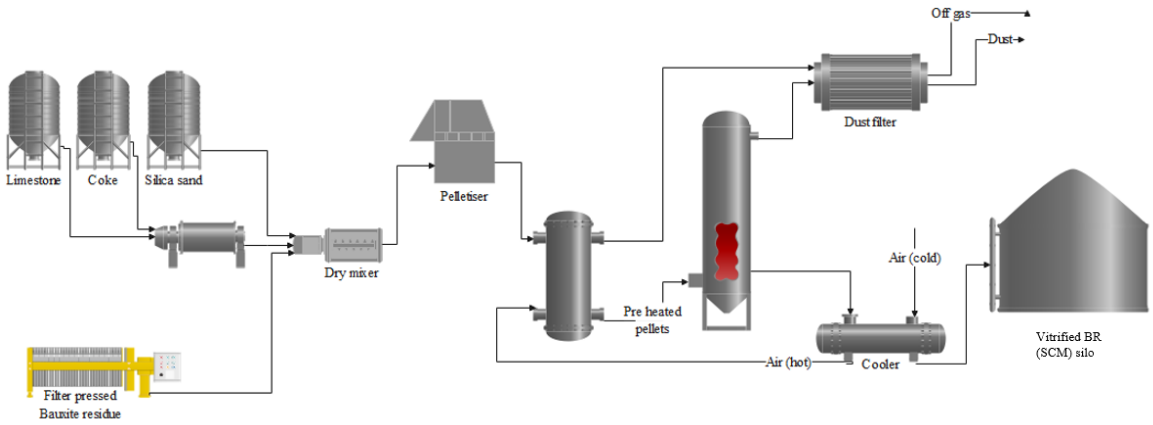


Figure 2-7: Process flow diagram of BR vitrification technology (adapted from Konlechner et al., 2023)

2.6.1.3 Reductive smelting of BR and pig iron production

The industrial-scale model involves blending of unmodified filter-pressed BR mixed with quartz-rich sand (silica source), milled coke (carbon source) and limestone (calcia source). The resulting mixture undergoes pelletisation and is then transferred to and calcined in a direct-fired rotary kiln via a Lepol preheater to reduce dust formation. A secondary heating step is

included in a submerged arc furnace (SAF) after the rotary kiln (Figure *2-8*). In the rotary kiln, further preheating and iron pre-reduction take place, followed by final melting and phase separation in the SAF. The carbothermic smelting of BR yields pig iron and slag. The resulting iron and slag are then separated, with the slag undergoing air-quenching and granulation to form glassy, reactive aggregates. The slag can be milled and be used as an SCM. Further milling is performed to adjust particle size and product properties.

Compared to conventional blast furnace slag, the BR slag has a higher concentration of alumina. Proper flux addition during smelting can control the composition of the final slag. Furthermore, slag cooling rates can affect the morphology of the slag, with slower cooling rates leading to crystallin reactive slags. By the proper fluxing (SiO_2, CaO additions), control of the reductive conditions, and fast cooling rates, the amorphous BR slag results in composition and properties similar to the Granulated Blast Furnace Slag (GBFS) used currently as SCM in blended cements.

The pilot test is being conducted at the MYTILINEOS aluminium production facility in Greece, employing an electric arc furnace with a targeted production rate of 500 kg per week. The reductive smelting process is currently operating at TRL 6, and the system is being tested under conditions that closely simulate full-scale deployment. Initial tests have successfully produced a few kilograms of granulated slag, demonstrating the operation of the pilot-scale system. However, further testing at full capacity is required to validate the technology's performance on a larger scale.

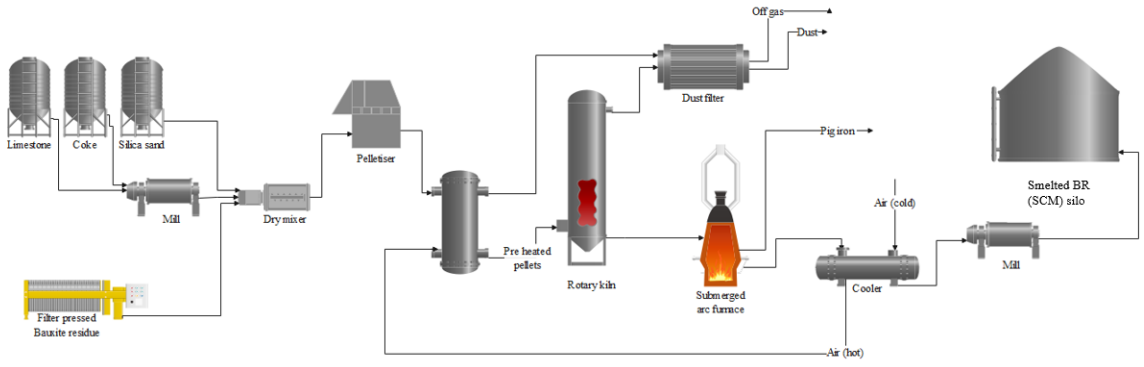


Figure 2-8. Industrial-scale process flow diagram of smelted BR and pig iron (adapted from Konlechner et al., 2023)

2.6.2 General overview of BR utilisation in LCA studies

Incorporating BR into cementitious materials has shown significant potential for reducing environmental impacts. However, existing research on BR utilisation remains limited in terms of LCA studies, with most research focusing on material synthesis and mechanical property testing. Only a few studies have conducted detailed assessments of the environmental performance, particularly when these processing technologies are upscaled, creating a gap in understanding the full environmental impacts of BR-incorporated cementitious materials.

Studies have explored various BR applications, demonstrating environmental benefits for BR-derived inorganic polymers, geopolymers, and supplementary cementitious materials (SCMs). For instance, BR-derived inorganic polymers and geopolymers have shown notable reductions in environmental impacts. However, processing stages such as the firing process, which involves fossil fuel combustion, and the combustion of the precursor were identified as the primary contributors to environmental burdens. The firing stage accounted for 89% of climate change and 87% of acidification. To mitigate these impacts, optimising processing parameters such as the fuel mix and the type of precursor is crucial (Joyce, 2018). Synthesising clinkers with high BR content and inorganic polymer binders has shown significant reduction in environmental burdens, including (976 and 903 kg CO_2 eq.), acidification (2.24 and 2.10 H+ eq.), and freshwater ecotoxicity (549 and 1217 CTUe) per tonne of processed BR (Joyce and Björklund, 2019). Utilising BR-derived inorganic geopolymers, reduces the input of OPC, leading to net environmental benefits across all impact categories, preventing 3.26 $MtCO_2$ associated with OPC production (Sterpellone, 2020). Further research into alkali-activated materials using BR as precursors, demonstrated that increasing BR content in blends with blast furnace slag (BFS) led to larger impact reductions. For instance, BR-based inorganic mortars with BR content ranging between 30% and 70% achieved impact scores between 11.18 and 25.01, compared to 24.41 for the reference OPC mortar (Palmero et al., 2022).

Research on BR incorporation in lightweight concrete and blended cement, also showed that increasing the replacement ratio of BR resulted in significant environmental impact reductions (Di Mare et al., 2021; Morsali and Yildirim, 2024). For instance, a 25 wt.% BR replacement in lightweight concrete led to a 26% reduction in climate change impact, reducing CO_2 emissions from 556.8 kg CO_2 eq. per m^3 to 409.9 kg CO_2 eq. per m^3. The BR replacement also led to

additional reductions in other gas emissions, including CO (32.5%), NO_x (17.1%), Pb (31.8%), and SO_2 (22.4%), due to minimised clinker pyro processing (Nikbin et al., 2016). Additional work on the integration of BR with additional cementitious materials and their synergistic effect has shown promising results with reduced environmental impacts and improved performances. In ready-to-use lightweight concrete produced using waste materials, including 27.83 wt.% BR demonstrated significant improvements in most impact categories. The synergistic effect of calcined BR with fly ash and carbide slag resulted in a 52% reduction in climate change impact, 50% in acidification, and 78% in ionising radiation (Yao et al., 2019). Similarly, the combination of BR with silica fume showed that even higher BR replacement ratios could be used leading to even greater reductions (Ghalehnovi et al., 2022). Expanding on the synergistic effects of BR in blended cements, mixing BR with silica fume at varying ratios (BR: 10%, 15%, 20%; SF: 5%, and 7.5%) could reduce CO_2 emissions by 5% to 30%. This suggests that the strategic use of BR with additional materials and the formulation can be optimised to maximise the environmental performance of the blended cement (Ghalehnovi et al., 2022).

Despite the promising preliminary results in terms of environmental performance, there is a lack of long-term tracking and investigation of BR-derived construction materials. Comprehensive studies spanning 50-70 years are imperative to thoroughly assess the applicability, particularly in terms of durability, safety, and environmental performance of these materials (Wang et al., 2024). The primary challenge lies in the early stage of material and technology testing and development and has not yet progressed beyond laboratory and pilot testing.

While preliminary LCA studies have begun to shed light on the environmental benefits of BR utilisation in the cement industry, significant gaps remain, particularly in terms of scalability and real-world application. This thesis applies a forward-looking LCA framework to address these gaps and evaluate the environmental implications of three emerging BR processing technologies (co-calcination, vitrification, and reductive smelting), specifically concerning the challenges of scaling these technologies and their commercial, large-scale use in BR-incorporated cements.

Chapter 3. Methodology: Life cycle assessment of emerging, BR technologies

This chapter introduces an ISO-compliant LCA framework and examines its application in the context of innovative BR treatment technologies. It provides a thorough overview of the theoretical background and key methodological choices in LCA studies related to forward-looking LCA frameworks, emerging technologies, and, more particularly, emerging BR technologies considered for their use in the cement industry.

3.1 Life cycle assessment framework

As environmental concerns are rising, the urgent need for global action has become central to policymakers, businesses, and individuals and their target of mitigating the threat of climate change. Integrating environmental considerations into societal decisions has become crucial to accelerate the transition towards carbon neutrality. Numerous tools and indicators have been developed through the years notably; life cycle assessment (LCA), strategic environmental assessment (SEA), environmental impact assessment (EIA), environmental risk assessment (ERA), cost-benefit analysis (CBA), material flow analysis (MFA), and ecological footprint (Ness et al., 2007). These tools assess and benchmark environmental impacts to ensure data transparency to the public and facilitate better-informed decision-making. LCA holds particular significance and is embedded in EU legislation, such as the Integrated Product Policy. (ISO: 14044, 2006).

In LCA analysis, the aim is to evaluate the environmental burdens of a product, technology, or service throughout its entire life cycle (from raw material extraction to disposal). By considering the entire life cycle of a product, LCA enables the identification of primary contributing stages “hotspots”, by analysing resource use and emissions at each stage. Highlighting the primary hotspots supports sustainable optimisation in product development, improving the eco-design potential (ISO: 14044, 2006).

Prior to discussing the LCA framework, aspects such as LCA perspectives must be introduced first. LCA can be divided into two main perspectives: attributional and consequential. An

attribution LCA focuses on evaluating the emissions and resource flows associated with a specified product/service. Typically, this is performed using average data, ideally producer-specific, that represent actual physical flows within a specified temporal window. In contrast, the consequential LCA perspective rather than dealing with "the environmental impacts of a product or service" models the influence of marginal changes such as demand, production methods and policies on the broader system and environmental impacts (Curran et al., 2005; Finnveden et al., 2009).

The standardised LCA framework is structured into four critical phases (as seen in Figure *3-1*): (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation (ISO: 14040, 2006). Compiling a detailed life cycle inventory (LCI) of relevant material and energy flows addresses the scope of the study and is directly used as the data feed for assessing the contributions to environmental categories such as climate change, eutrophication, and resource scarcity (Hauschild et al., 2018). LCA is typically considered an iterative process; starting from the goal and scope and proceeding to the impact assessment and interpretation phases, new information is produced that can refine and narrow down the initial scope.

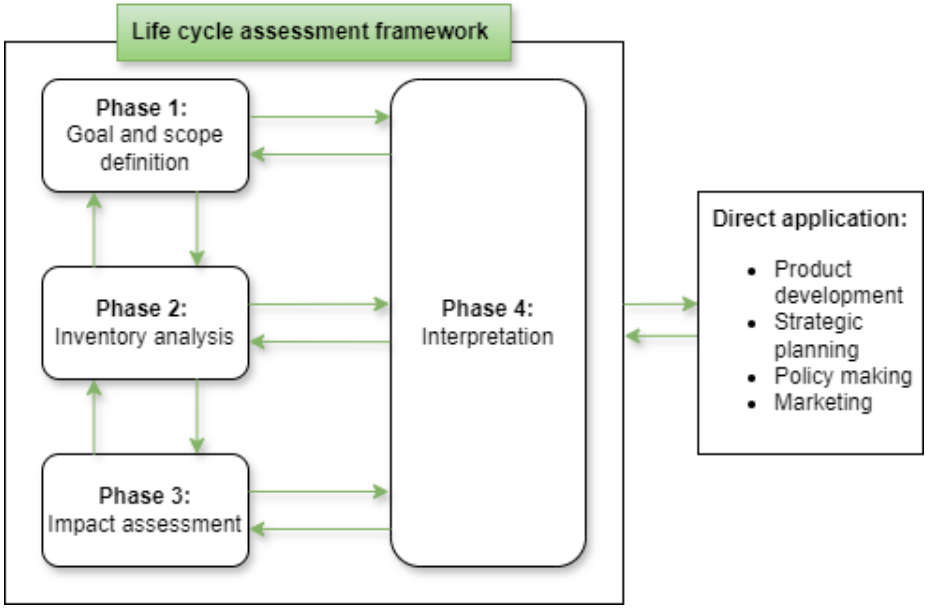


Figure 3-1: Schematic diagram of iterative LCA framework showing the four phases of LCA according to ISO14040:2006 (ISO: 14044, 2006)

The iterative nature is particularly observed in emerging technologies, where data scarcity is a key challenge. Hence, preliminary iterations often rely on generic or average data for many background and foreground aspects (European Commission, 2010). More representative and

reliable assessments can be achieved as emerging technologies mature with additional information from testing and expert judgements available. It is important to note that LCA results are not intended to be used as absolute values; instead, they facilitate the comparison of alternatives, evaluation of potential impacts, and identification of opportunities for improvement. Considering a comprehensive scope, rather than narrowing it to an individual environmental category, allows for a better understanding of the potential for burden-shifting and problem-shifting across different life cycle phases, regions, or issues.

Emerging technologies have risen in response to the urgent need to address the climate crisis. Consequently, LCA methods must adapt to and enable the assessment of not only existing products and services but also incorporate forward-looking perspectives. Known organisations in LCA development include SETAC, ISO, UNEP, and academic collaboration.

The technical nature of LCA remains an inherent challenge when communicating with non-expert stakeholders and consumers. Recent advancements in communication tools, such as environmental product declarations (EPDs), EU Digital Product Passport (DPP), and material passports, have been gaining attention and are increasingly adopted by the LCA community to effectively communicate to consumers and stakeholders, enabling informed decision-making and improving stakeholder engagement (EN 15804, 2019; ISO 14025, 2006). EPDs provide third-party verified, LCA-based environmental impact data, allowing for comparison between products within the same category. These declarations follow international standards, such as ISO 14025 and EN 15804, to ensure a high level of consistency and comparability between generated results. The DPP is an integral component of the EU's Eco-design for Sustainable Products Regulation (ESPR) and functions as a unique digital product tag that provides detailed information about a product's environmental impact and lifecycle across the entire value chain. The target is to make DPP mandatory, particularly for high-impact sectors, to enhance transparency and traceability for all products entering the European market. DPPs are designed to assist consumers in making informed, sustainable choices but also prevent greenwashing by requiring companies to support their environmental claims with verified data (European Parliament and Council of the European Union, 2024). These tools are particularly beneficial for emerging technologies and products. They allow for an easier demonstration of environmental performance and comparison with existing alternatives, assisting in securing further investment and development while also promoting wider acceptance and adoption in the market.

3.1.1 Goal and scope definition

Defining the goal and scope of the LCA sets the framework and system boundaries. The goal involves establishing the aim and outlining the key objectives, specifying the intended application and target audience. The scope defines the product system, describing the intended function quantified by the functional unit. This stage identifies the processes and activities to be considered within the system boundary and aspects such as allocation principles, data quality requirements, and the choice of impact categories and assessment methods (ISO: 14040, 2006). Even though the goal and scope stage is often underestimated, a clear initial definition is crucial as it lays the foundation for the entire LCA process (Curran, 2017)

3.1.1.1 Functional unit

The functional unit (FU) quantitatively describes the function or service being assessed and is the basis for scaling the inventory data. Selecting the functionality is a crucial aspect of the analysis, as it can vary widely depending on the intended goal and scope. The choice of the functional unit must accurately reflect the defined goal as it directly influences the study's outcome, whether the focus of the study is to investigate the performance of the technology or whether the aim is to compare alternative methods of delivering a function. Hence, the functional unit is selected to represent an equivalent basis for these comparisons.

3.1.1.2 System Boundaries

Establishing the system boundary involves setting the geographical areas, time horizons, and the boundaries between the life cycle of the studied product and the life cycles of indirectly related products (Tillman et al., 1994). Within the system boundary, it is essential to distinguish the technological system from nature by identifying and tracking the input and output flows associated with the life cycle stages of the product or service. In particular, to pinpoint the flows taken from or released into the environment and not associated with any human activity. These flows are known as elementary flows. Additionally, consider the technosphere flows, representing the material and energy flows caused by human transformation activities. When setting the system boundary, the target is to identify the technosphere flows associated with the life stages of the specified product service and link them back to their associated elementary

flows. Only then can a clear picture of the natural resource requirements and emissions released be formed. Practical challenges such as time constraints, system complexity, and data limitations often make this impractical. Hence, it is necessary to prioritise flows directly linked to the product system to simplify and narrow down the data requirements of the study.

Given the numerous flows typically associated with a product or service, the technical system can be divided into foreground and background systems to distinguish the importance of the flows. The foreground system consists of activities directly influenced by the product system. At the same time, the background encompasses all other processes interacting with the foreground, usually by supplying or receiving materials or energy (Clift et al., 2000). For example, a common background activity is the electricity supply in society, which is typically purchased in a market without the option to choose specific suppliers. The key distinction between the foreground and background systems lies in data quality. The foreground system is modelled using primary data collected first-hand by the LCA practitioner, while the background relies on LCI databases based on existing literature, established databases and industry data (Hauschild et al., 2018). As the background data are considered secondary quality, they might not fully align with various aspects of the study, such as geographical areas and time horizons

LCA terms like cradle-to-gate and cradle-to-grave indicate life stages considered in the foreground and background system, from the origin to a production stage or the end life. For example, a cradle-to-gate study examines the life cycle of a product system from the origin, including all upstream processes associated with raw material extraction or feedstock supply, up to the processing stage producing the functional unit.

Foreground data requires more case-specific information, typically referring to processes under the control of the decision-maker. Background data refers to resources delivered to the foreground system as aggregated datasets. A well-defined system boundary and clear distinction between different systems (i.e., foreground, background) is essential to correctly identify the data source and quality needs.

3.1.2 Life cycle inventory analysis

According to ISO 14040, the life cycle inventory (LCI) stage is responsible for the "data collection and calculation procedures to quantify relevant inputs and outputs of a product system" (ISO: 14040, 2006). The LCA practitioner compiles and quantifies all relevant input and output flows within the foreground that serve the functional unit and subsequently relate to the reference flow. Compiling the LCI can be the most challenging part of an LCA study due to data scarcity and the significant time and effort required to measure or obtain high-quality primary data (e.g., due to high costs or confidentiality issues) (Hauschild et al., 2018).

Due to the iterative nature of LCA, the inventory stage is revised frequently to ensure that data is up-to-date and consistent with recent advancements. This is crucial for emerging and existing technologies to ensure the study goals are met and remain relevant to the desired temporal setting. A comprehensive LCI is necessary to provide a basis for evaluating comparative environmental impacts and identify potential improvements (Curran, 2017).

The LCI process can be divided into three sections: data collection, calculation, and allocation of flows and emissions in multifunctional systems. Starting from the functional unit, the foreground system is constructed, and the associated flows are determined. Subsequently, linked flows physically embodied in or have a supporting function to the functional unit are addressed, followed by other upstream or downstream flows with a more indirect purpose. The flows collected can be grouped into four major categories: energy inputs, raw material inputs, ancillary inputs, and other physical inputs; products, co-products, and waste; emissions to air, discharges to water and soil; and other environmental aspects (e.g., radiation)(Hauschild et al., 2018). The data collected are validated and linked to each foreground activity, relating it to the reference flow of the functional unit. This ensures that all associated activities are considered and can be quantitatively assessed, providing a comprehensive inventory for each unit process.

To simplify the system, "cut-offs" may be used for unit processes that are not quantitatively significant. Cut-offs imply not accounting for certain activities if they do not account for more than 5% of the total environmental impacts. However, this is a paradoxical approach, as a rough estimation of the total impact is required. In comparative LCA, the cut-off criteria must be

defined consistently across the systems being compared, based on their differences and similarities.

Overall, the LCI stage involves a detailed and iterative process of data collection, calculation, and allocation. It ensures that all relevant inputs and outputs are accurately quantified and linked to the functional unit, thereby providing the foundation for evaluating the environmental contributions.

3.1.2.1 Multifunctionality

When an industrial process produces more than one product (multifunctionality), the ISO 14044 standard outlines a hierarchy of solutions to make product systems functionally comparable and to represent a single product system (Hauschild et al., 2018).

1. Subdivision of unit process: This approach improves the degree of detail of the model by breaking down the multifunctional unit process into smaller units to determine if the production of the main product can be separated from that of the co-product.
2. System expansion: If subdivision cannot be applied, system expansion compares the primary process to the most likely alternative method of providing the secondary function.
3. Allocation: If neither subdivision nor system expansion can be applied, allocation is used. This method distributes inputs and outputs based on underlying causal physical relationships or economic values.
    - Physical allocation: Inputs and outputs are allocated according to physical parameters affected by changes in product quantities.
    - Economic allocation: Inputs and outputs are allocated based on the economic value of the products.

3.1.3 Life Cycle Impact Assessment (LCIA)

The life cycle impact assessment evaluates the magnitude and significance of impacts arising from activities of the product system throughout the life cycle stages considered (ISO: 14040, 2006). This is achieved by translating the inventory flows, which describe activities within the

product system boundary, into environmental impact scores using a set of characterisation factors. These impact scores are used as a common ground for comparing alternative products or as a measure of environmental behaviour for further analysis, e.g., to identify the key contributing areas. The key steps in impact assessment include;

- The selection of characterisation models, considering the impact categories and category indicators of interest
- The assignment of the LCI results to selected impact categories. This classification links the elementary flows from the activities of the product system to relevant impact categories, assuming their contribution to the environmental problem of concern.
- The calculation of the impact category indicator results, each elementary flow that influences an impact category is multiplied by a characterisation factor, which describes the flow's contribution to that specific impact category. The impact categories depend on the chosen characterisation method, but generally, all methods include scores related to categories such as; abiotic resource depletion, acidification, climate change, ecotoxicity, eutrophication, human toxicity, land use, particulate matter formation, photochemical ozone formation, stratospheric ozone depletion and water use.

3.1.3.1 Optional impact assessment steps: normalization and weighting

Optional steps in LCIA include normalisation and weighting and are described in Chapter 3, sections 3.1.3.1.1 and 3.1.3.1.2

3.1.3.1.1 Normalisation

Normalisation considers the understanding of the relative importance of impacts within a system or process under study (Pacheco-Torgal et al., 2013). Given that impact categories are measured in different units, normalisation helps in judging the severity of these impacts by comparing the magnitude of category indicator results against a reference value. Normalisation is performed by dividing the indicator results by the selected reference value. This allows for a consistent comparison amongst the same impact categories of comparable product systems

and provides a better understanding of the relative magnitude of each indicator result (ISO: 14040, 2006)

3.1.3.1.2 Weighting

Weighting converts and can aggregate indicator results across various impact categories by applying numerical factors based on value judgments. (ISO: 14040, 2006). This optional step is usually applied to endpoint categories to prioritise the most critical impacts. Weighting factors are assigned to normalised indicator results, reflecting the relative importance of each impact category. This step aggregates multiple impact category results into a single score, simplifying interpretation, especially when trade-offs exist between different impact categories.

A clear documentation of the rationale and the processing steps followed to develop the weighting factors must be disclosed, as value judgments can vary widely amongst stakeholders and LCA practitioners. Additionally, weighting factors depend on the geographical and temporal scope of the study, and societal conditions must be considered and how they can change over time. Sensitivity and scenario analysis can be employed to minimise the inherent uncertainty of using weighting factors; hence, applying a range of values for weighting factors can lead to a better understanding of the influence on environmental performance.

3.1.3.2 Selecting the impact assessment method

The impact assessment stage connects the inventory data with environmental impact categories and indicators and estimates their contribution. The impact categories, indicators, and characterisation models are selected according to the study's goal and scope. The LCA practitioner has the flexibility to make these selections, assuming they align with the ISO criteria.

ISO 14040 criteria given as requirements and recommendations state that the choice of impact categories should ensure that:

- Each LCIA element is distinct and clearly defined.
- The goal and scope definition phase of an LCA can address each LCIA element separately.

- Quality assessment of LCIA methods, assumptions, and decisions can be conducted for each element.
- LCIA procedures, assumptions, and operations can be made transparent for critical review and reporting within each element.
- Value choices within each element can be made transparent for critical review and reporting.

Typically, category indicators based on specific characterisation models are integrated into sets or LCIA methods. With increasing LCIA sets and indicators available, selecting the most suitable methods requires a deep understanding of the main characteristics. It is also important to consider the evolution of these methods as they are subject to continuous updates and merging. Since the conception of LCA in the 1990s, a series of LCIA methods have been developed, including:

- The CML method was developed by the Institute of Environmental Sciences of the University of Leiden in 1992. Regular updates are taking place to refine and assure up-to-date impact categories and characterisation factors. CML focuses on midpoint indicators, such as global warming potential, acidification, and eutrophication (Van Oers and Guinée, 2016).
- The ILCD method was developed by the European Commission and published as part of the International Reference Life Cycle Data System (ILCD) in 2010. It provides a standardised framework for consistent, robust, and quality-assured life cycle data and studies. The ILCD focuses on guiding practitioners in conducting detailed LCAS and ensuring consistency with ISO 14040 and 14044 standards. It supports both midpoint and endpoint indicators and is used in various Sustainable Consumption and Production (SCP) instruments such as ecolabelling, ecodesign, and carbon footprinting. (European Commission, 2010)
- Recipe is a characterisation model developed in 2008 (updated in 2016) through a collaboration between the National Institute for Public Health and the Environment, Netherlands, Radboud University, and PRé Sustainability. This method integrates both midpoint and endpoint indicators, making it a robust tool for assessing the potential environmental impacts across various impact categories (Goedkoop et al., 2008; Huijbregts et al., 2017).

Recipe is divided into numerous perspectives with the primary being Hierarchical (H), Egalitarian (E), and Individualist (I), which represent different viewpoints on the environmental impact assessment. Each perspective represents a different approach to dealing with uncertainties and time horizons in environmental impacts:

- Hierarchical (H): This perspective balances between short-term and long-term impacts. It provides a middle ground that reflects the immediate and future environmental impacts, integrating both midpoint and endpoint indicators. The Hierarchical perspective is the default choice in many LCA studies due to its balanced nature.
- Egalitarian (E): Focuses on long-term impacts and adopts a precautionary principle, suitable for stakeholders who prefer a conservative approach to environmental protection, taking strong measures now to prevent possible future environmental damage. This perspective is the most precautionary among the ReCiPe models.
- Individualist (I): Focuses on the short term and is preferred by stakeholders who favour immediate solutions. It takes an optimistic perspective and focuses on impacts that are more certain and less precautionary. This perspective assumes that technological advancements will solve many environmental issues in the future.

- The Intergovernmental Panel on Climate Change (IPCC) consists of characterisation factors for global warming potential (GWP) starting in 1990, with the most recent revision in 2013 and 2021. This model provides characterization factors specifically for GWP over different time horizons (e.g., 20, 100, 500 years), widely used for assessing climate change impacts in LCA. The IPCC (GWP) characterisation factors are used in various other characterization models such as CML, PEF, and Recipe, providing a standardized method for assessing the climate change impacts of greenhouse gas emissions and ensuring consistency across different LCA studies and tools. By using IPCC factors, these models align with international standards and enhance the comparability of environmental impact assessments.

The majority of characterisation models focus on the evaluation of impacts up until the midpoint, which is a point in the cause-effect chain of a particular impact category, meaning only looking at the effect of a single impact category such as climate change. When using midpoint indicators, aggregation and contribution analysis of different impact categories are only possible after normalisation and weighting. However as discussed in Recipe, evaluation up to the endpoint allows for further aggregation, grouping specific environmental problems into three main areas of protection (AoP): damage to human health, damage to ecosystems and damage to resource availability. The difference between the midpoint and endpoint is illustrated in Figure *3-2*, which also describes the damage pathways connecting the two points, as described in the ReCiPe 2016 model.

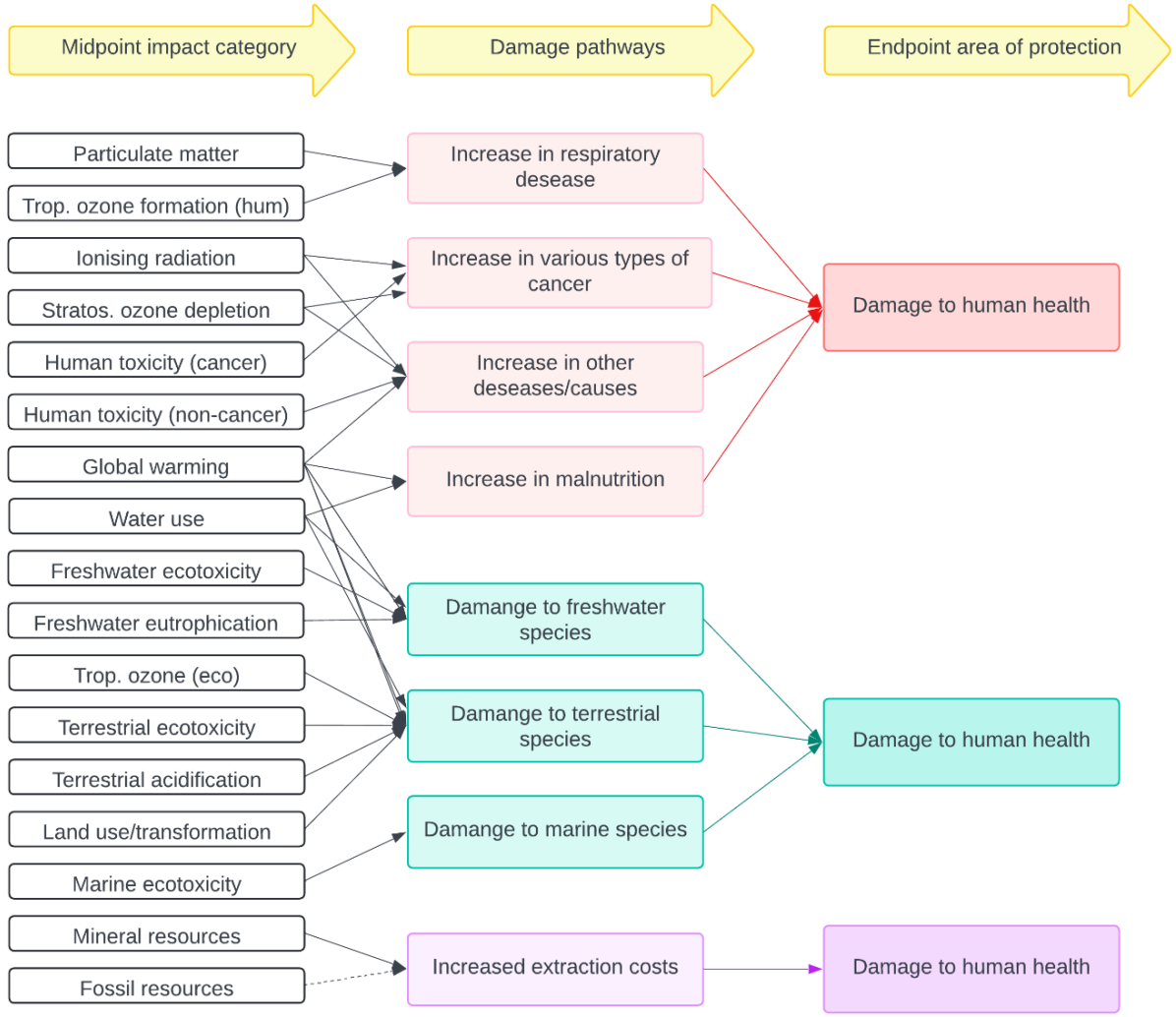


Figure 3-2: Midpoint vs endpoint impact categories covered by the ReCiPe 2016 method (Huijbregts et al., 2017).

The choice between midpoint and endpoint indicators depends on the defined goals of the assessment and the target audience. Midpoint indicators provide detailed insights focusing on the intermediate stage of an environmental mechanism, addressing each environmental category separately. This manages the cause-effect chain more clearly. However, the results can be complex to interpret for non-expert audiences and offer an incomplete picture of ultimate effects. Endpoint indicators measure the final impacts on AoP i.e., human health, ecosystem quality, and resource availability, and simplify the communication of results by providing a holistic overview of the environmental impacts, by calculating a single damage score. This is beneficial for communication with non-experts and decision-makers. However, endpoint indicators include higher statistical uncertainty due to the complexity of modelling impacts further down the cause-effect chain. An overview of midpoint impact categories examined in this thesis (Table *3-1*).

In the context of emerging technologies, current impact categories face the limitation of potentially being inadequate to address potential environmental impacts, often highlighted in ex-post LCA studies (Guinee, 2002; Huijbregts et al., 2017). The lack of characterisation factors during the LCIA is highlighted as a key challenge (Tufvesson et al., 2013; Wender et al., 2014). This limitation can obscure the true environmental performance of emerging technologies compared to existing ones. This arises as many biosphere flows with potential environmental impacts remain unclassified due to inadequate models and data, leaving critical questions unanswered: 1) Do current LCIA methods cover all impacts associated with emerging technologies, or are there unforeseen impacts? 2) How can characterization factors relevant to both incumbent and emerging technologies evolve over time to ensure consistency? (Van der Giesen et al., 2020).

Although creating new impact categories and characterisation is vital for advancing LCA methodology in terms of emerging technologies and products, this would require overcoming significant challenges, such as data limitations, modelling complexity, and uncertainties related to long-term environmental effects. As this is beyond the scope of this thesis, the focus here is on evaluating how existing methods perform when applied to emerging technologies.

Table 3-1: Description of midpoint impact categories investigated in this thesis (adapted from Huijbregts et al., 2017).

Impact category	**Unit**	**Description**
Terrestrial acidification potential (TAP)	kg SO_2-eq.	TAP measures the potential for acidification of soil and water taking place through the transformation of air pollutants into acidic compounds (such as SO_2, NO_x, NH_3). It is associated with the ability of substances to build and release H^+ ions, quantified in sulphur dioxide (SO_2) equivalents, commonly measured over a 100-year horizon.
Global warming potential (GWP)	kg CO_2-eq.	GWP measures the potential increase in infrared radiative forcing caused by greenhouse gases in the atmosphere. The contribution of GHG to GW is calculated relatively to CO_2, over a time horizon, typically 100 years, with the option for 20-year and 500-year horizons. It is calculated in kg CO_2 equivalents.
Freshwater ecotoxicity potential (FETP)	kg 1,4-D, C, B-eq.	Ecotoxicity measures the effect of chemical emissions released into freshwater, marine and terrestrial environments. Ecotoxicity potential is measured in kg 1,4-dichlorobenzene (D, C, B) -equivalent, a carcinogenic compound.
Marine ecotoxicity potential (METP)	kg 1,4-D, C, B-eq.	
Terrestrial ecotoxicity potential (TETP)	kg 1,4-D, C, B-eq.	
Fossil fuel potential (FFP)	kg oil-eq.	FFP quantifies the depletion of fossil fuel resources. FFP is calculated based on the ratio between the energy content of a fossil fuel source and that of crude oil; FFP is measured in kg of oil equivalent, reflecting the impact of long-term extraction and consumption.

Impact category	**Unit**	**Description**
Freshwater eutrophication potential (FEP)	kg P-eq.	FEP assesses the potential for phosphorous (P) emissions to cause nutrient enrichment and its negative effect on the aquatic ecosystem.
Marine eutrophication potential (MEP)	kg N-eq.	FEP assesses the potential for nitrogen (N) emissions to cause nutrient enrichment and its negative effect on marine biodiversity
Human toxicity potential (HTPc)	kg 1,4-D, C, B-eq.	HTPc and HTPnc reflect the potential exposure of chemicals in humans and the potential to cause carcinogenic, and non-carcinogenic health effects, respectively.
Human toxicity potential (HTPnc)	kg 1,4-D, C, B-eq.	
Ionising radiation potential (IRP)	kg Co-60-eq.	IRP refers to the ionising radiation exposure to human health and the environment. It is measured in kg Cobalt-60 (Co-60) equivalents.
Agricultural land occupation (LOP)	m^2 a crop-eq.	LOP represents the agricultural land occupied by activities related to the lifecycle of the product, measured in square meters per year of crop-equivalents (m^2 a crop-eq.).
Mineral resource scarcity potential (SOP)	kg Cu-eq.	SOP measures the depletion of mineral resources, consumed during the lifecycle of the product. SOP is measured in kg of copper (Cu) equivalents.
Ozone depletion potential (ODPinfinite)	kg CFC-11-eq.	ODP reflects the potential of a substance to destroy the stratospheric ozone layer, compared to the contribution of chlorofluorocarbon-11 (reference substance)
Particulate matter formation potential (FPMFP)	kg PM2.5-eq.	PMFP evaluates the change in particulate matter concentration in the atmosphere. It is measured in kg of particles less than 2.5 µm equivalents, considering emissions from precursors such as $PM_{2.5}$, NH_3, NO_x, and SO_2
Photochemical ozone formation potential: humans (HOFP)	kg NO_x-eq.	

Impact category	**Unit**	**Description**
Photochemical ozone formation potential: ecosystems (EOFP)	kg NO_x-eq.	HOFP and EOFP evaluate the potential of volatile organic compounds and NO_x to interact resulting in the formation of photochemical ozone affecting both human health and the ecosystem
Water consumption potential (WCP)	m^3	WCP quantifies the amount of water consumed during the lifecycle of the product, measured in cubic meters (m^3)

3.1.4 Interpretation

Interpretation of results is the final phase in the LCA framework. The findings from the inventory analysis and the impact assessment are examined to identify key limitations and make final recommendations in accordance with the defined goal and scope of the study. It is important to stress that the findings should reflect that LCIA results are relative and indicate potential environmental effects rather than actual impacts on environmental categories. The interpretation phase includes 3 main elements;

- The identification of any significant issues based on the LCI and LCIA through a series of further analyses; hotspot analysis (identifies the stages, processes, or components of a product system that contribute the most to its overall environmental impact), contribution analysis (evaluates the degree of confidence in the results), and sensitivity analysis (assesses how variations in input data affect the outcomes).
- The evaluation of the reliability of the results. This is usually established by performing completeness (verifying that the inventory data, impact categories, and other necessary elements are fully accounted for and that no significant gaps exist), sensitivity (evaluates how changes in key assumptions or input data affect the outcomes of the study) and consistency (assesses whether the methodology applied throughout the study is consistent) checks.
- The conclusions should be supported by the data, the limitations should be acknowledged, and recommendations should be practical and relevant.

3.1.4.1 Reporting

The conclusions, limitations and recommendations summarised during the "Interpretation" phase should be unbiased and available to the target audience. The data collected (inventory) and assumptions shall be presented transparently, allowing the reader to understand the complexities in the LCA. The results and interpretation should be reported in a manner that aligns with and satisfies the objectives of the study (Scientific Applications International Corporation (SAIC), 2006)

3.2 Technology Readiness Level (TRL)

TRL measures the maturity of technology, ranging from scientific breakthrough (TRL 1) to full commercialisation (TRL 9) (Moni, 2020). Generally, larger production scales (higher maturity levels) reduce environmental impacts per unit output due to improved material and energy efficiency (Gavankar et al., 2015). However, it is important to distinguish that TRLs primarily indicate functional readiness (i.e., how close a technology is to being deployed in real-world applications, not manufacturing readiness. Manufacturing Readiness Levels (MRLs) are thus used alongside TRLs to address the readiness and maturity of the manufacturing processes, infrastructure, and supply chain to support the production of the technology (Gavankar et al., 2015). For an accurate evaluation of environmental impacts at a mass production scenario (MRL 9 or 10), the technology should be at least at TRL 9. As inventory data for emerging technologies are often at lab-scale stages (TRL 2-5), scale-up assumptions are required to accurately represent environmental performance (Harris et al., 2019).

The TRL framework differentiates between the innovation phase (TRL 1-5) and the industry phase (TRL 6-9) (Alberts et al., 2015). During the innovation phase, technologies undergo process improvements, exploring safer and more cost-effective production techniques (van der Hulst et al., 2020). At TRL 5, the technological system undergoes upscaling considerations to advance into higher TRLs (Padilla-Rivera et al., 2023). In the industry phase, TRL 6-7 involves scaling from laboratory to engineering scale and testing full-scale production (Ma et al., 2018; Ren and Toniolo, 2018; Ribeiro et al., 2020; Valente et al., 2019). Finally, the TRL 8-9 represents commercial implementation, with technologies ready for low-rate initial production and full-scale operation (Azapagic et al., 2016; Henzler et al., 2020; Hoque et al., 2020; Keller et al., 2015). Another important consideration noted by (Bergerson et al., 2020; van der Giesen et al., 2020) is the distinction between technology maturity and market maturity, which often developed in parallel over time since the formation of a market is a significant factor in the technology's development (Bergek et al., 2008).

3.3 Forward-oriented LCA approaches

LCA is an ISO-standardised approach widely used to evaluate the environmental of existing systems, through ex-post analyses that require detailed inventory data. Traditionally, LCA studies focus on assessing technologies post-implementation. However, this standardised LCA framework is inadequate for emerging technologies due to upscaling and data deficiency limitations.

Although there is no standardised framework for assessing emerging technologies, several attempts have provided useful recommendations and guidelines. Due to the novelty of the field and the inconsistent use of terminology this causes challenges when trying to identify limitations (van der Giesen et al., 2020). Numerous studies have worked to define and streamline a more suitable methodology, leading to various definitions and proposed methods (Adrianto et al., 2021; Arvidsson et al., 2024, 2023, 2018; Buyle et al., 2019; Cucurachi et al., 2018; Guinée et al., 2018; Moni et al., 2020; Thonemann et al., 2020a). These offer reliable and accurate predictions for emerging technologies under different design possibilities, by considering parameter changes that could affect performance, this approach optimises design before scale-up. The ultimate goal is to introduce emerging technologies and ensure their competitiveness with conventional incumbent technologies. Variations of these predictive LCA studies are summarised in Table *3-2*.

A great inconsistency in the field of emerging technology assessment is the terminology used across different studies. For example, Arvidsson et al. (2018) defined prospective LCA similarly to the ex-ante LCA definitions given by other studies (Cucurachi et al. 2018; van der Giesen et al. 2020), while in other work on pLCA, the concept of emerging technologies is not discussed (Guinée et al., 2018; Sandén and Karlström, 2007). Arvidsson et al. (2024) has recently aimed to clarify this, particularly for future oriented LCA types such as ex-ante, prospective, and anticipatory studies. For a more detailed discussion on ex-ante LCA, see Arvidsson et al. (2023) for a review of definitions and clarification of future-oriented LCA terminology; Buyle et al. (2019) for a framework to address methodological inconsistencies in prospective LCA; Cucurachi et al. (2018); for a definition of ex-ante LCA, its key features and challenges; and van der Giesen et al. (2020) for an analysis of the main challenges in applying prospective LCA to emerging technologies and recommendations.

Table 3-2: Overview of "forward-oriented" LCA approaches

Type of assessment	**Definition and description given in the literature**
Prospective LCA (pLCA)	This approach predicts future developments and provides preliminary insights to guide further technological development, investment, and implementation. Prospective LCAs use future scenarios to anticipate changes and optimise design (Hummen and Kastner, 2014; Mendoza Beltran et al., 2018; Miller and Keoleian, 2015; Cucurachi et al., 2018). This thesis follows the definition given by (Arvidsson et al., 2024, 2023), in which pLCA models the product system at a future point in time relative to the study's timeframe.
Ex-ante LCA	Ex-Ante LCA conducts environmental analyses of technologies still in the research and development (R&D) phase and modelling them at a more advanced future state (e.g., from small-scale to large-scale production) to guide sustainable upscaling (Arvidsson et al., 2017; Hesser et al., 2017; Schrijvers et al., 2014; Tecchio et al., 2015). Rather than predicting the future impacts of a technology, ex-ante LCA aims to explore various development scenarios to optimise technology design prior to scale-up, ensuring that the new technology will remain competitive with the incumbent (Celik et al., 2016; Roes and Patel, 2011; Villares et al., 2017, 2016). In this thesis, the ex-ante LCA definition is selected as modelling a product system at a future point in time that also involves an increase in technology maturity (Arvidsson et al., 2024, 2023)
Dynamic LCA	This approach incorporates the rapidly evolving nature of technologies, products, and production processes, considering temporal variations in parameters and development pathways, to improve the accuracy of assessments (Alfaro et al., 2010; Cucurachi et al., 2018; Pehnt, 2006)
Anticipatory LCA	Anticipatory LCA involves stakeholders in modelling decisions to enhance social engagement and credibility of the LCA. This is non-predictive method exploring a range of future environmental scenarios (i.e., highly possible to extreme cases) to identify significant uncertainties and engage R&D decision-makers (Cucurachi et al., 2018; Wender et al., 2014)

Type of assessment	**Definition and description given in the literature**
Consequential LCA	Consequential LCA aims to understand the potential effects of policies on market response, concentrating on environmental burdens arising from decisions, typically represented by changes in product demand or technological landscape i.e., the introduction of new technologies or policies(Cucurachi et al., 2018; Ekvall, 2019; Guinée et al., 2018).
Life cycle sustainability analysis (LCSA)	LCSA extends traditional LCA by incorporating social and economic impacts alongside environmental factors, scaling technology to societal levels, and creating future scenarios (Guinée et al., 2011; Van Der Giesen et al., 2013)

3.4 Forward-looking LCA studies for emerging technologies

3.4.1 Forward-looking LCA framework

In this section, a comprehensive literature review on the LCA of emerging technologies is performed, with an emphasis on the primary challenges encountered during early-stage environmental assessments and the guidelines proposed to address these issues (Moni et al., 2020; Thonemann et al., 2020a). The review not only addresses traditional LCA challenges, such as incomparable functional units and data deficiencies, but also expands the discussion to include critical considerations specific to prospective and ex-ante LCAs, such as comparability, scaling issues, data quality, and uncertainty. This critical review served as the foundation for the methodological framework and modelling assumptions used in assessing the environmental performance of emerging BR technologies.

3.4.1.1 Goal and scope

3.4.1.1.1 Comparability

Comparability challenges in forward-looking LCAs arise primarily when defining the study aim, functional unit, and system boundary (Thonemann et al., 2020a). In ex-ante LCA, two main LCA types are identified: a comparative study of the emerging technology with existing "incumbent" mature technologies, or alternatively, a comparison of the same technology at different TRL stages (Arvidsson and Molander, 2017). In the former case, the study aims to assess the potential environmental benefits of implementing the emerging technology relative to the incumbent. Each approach targets a unique goal, in the former to evaluate the potential environmental benefits while in the latter to identify early-stage hotspots for technology developers (Gear et al., 2018).

3.4.1.1.2 Functional unit

Defining the functional unit at the early stages of technology development presents another challenge, as the function of the technology or product being assessed may not be fully understood. Hence, as a technology evolves, it is proposed to restate its functionality according to its development stage (Hetherington et al., 2014; Miller and Keoleian, 2015). However, this introduces variability that can affect the significance of the LCA results. To address this, Hischier et al. (2017) proposed using a range of functional units to assess their influence on environmental impacts. Nonetheless, several studies assume the same functional unit across different development stages of the emerging technology (Piccinno et al., 2016; Shibasaki et al., 2007). Despite the recognised importance of the functional unit, few investigations have been conducted on this matter (Arvidsson et al., 2015, 2014; Wender et al., 2014).

These challenges are interconnected due to the direct link between the functional unit and the system boundary. Varying the functional unit can complicate the definition of a single system boundary. Product application and end-of-life stages are typically included in the system boundary, but for emerging technologies, data on these life stages are often unknown (Hetherington et al., 2014). The scope of the study is subject to change. For instance, depending on whether the study aims to evaluate the environmental performance of an emerging technology or compare it with a conventional technology for a specified application (Hischier et al., 2017), this can lead to the need for multiple functional units, each requiring different system boundaries and assumptions. Consequently, two approaches have been proposed for selecting functional units: focusing on the environmental behaviour of the technology across different development stages to account for varying functionalities or defining a single functional unit while investigating multiple technologies or alternatives (Arvidsson and Molander, 2017).

3.4.1.1.3 System boundary

The choice of system boundary and co-product inclusion may lead to different conclusions between alternative processes, thereby impacting decision-making during the development stages (Cucurachi et al., 2018; McManus and Taylor, 2015; Wardenaar et al., 2012). Processing stages can vary along the TRL scale (from lab to commercial scale), while the end-of-life stage

remains unknown (Hetherington et al., 2014). This makes comparative LCA very challenging and simplifying the study by focusing on parts of the production chain most affected by technological change has been suggested as a practical approach, assuming the new technology is a direct replacement of an existing one and simplifications are reasonable to be made (Hospido et al., 2010). Co-product use may not be specified at low TRLs for emerging technologies, potentially influencing future impacts. For example, the potential GHG reduction of fast pyrolysis-derived transportation fuel is highly sensitive to co-product scenarios and allocation methods, which can shift LCA results from meeting to exceeding policy requirements, such as the renewable fuel standard emission reduction threshold for cellulosic biofuels (Zaimes et al., 2015).

3.4.1.2 Life cycle inventory

3.4.1.2.1 Scenario analysis

Incorporating scenario analysis enables the assessment of emerging technologies while considering variations in temporal and geographical context, co-production, and allocation. Explicitly stating any assumptions made during this process is essential for appropriately interpreting LCA results. In traditional LCA studies, primary data are used to model the foreground system. In future-oriented LCA, the foreground system is modelled at a future time to represent a scaled-up version of the emerging technology (Arvidsson and Molander, 2017). Defining the time and scale parameters is a necessary step in influencing the inventory data (Gavankar et al., 2015). The inventory for each scenario should be supplemented with the corresponding upscaling assumptions to simulate the commercial-scale scenario of the emerging technology.

3.4.1.2.2 Data limitations

A key challenge in assessing emerging technology is the scarcity of primary data typically required in an LCA study. Data limitations arise because technologies at early stages (laboratory and pilot) do not yet have representative industrial-scale data. Hence, various secondary data approaches are commonly used to fill in the data gaps, such as proxy technologies, data extrapolation, learning curves, empirical calculations, and artificial neural

networks (Wernet et al., 2009;Cucurachi et al., 2018; Thonemann et al., 2020a; al-Swaidani et al., 2022; Upadhyayula et al., 2018). However, these methods have limitations, particularly in their applicability to specific technologies such as chemical processes. The lack of upscaling results post-implementation indicates a need for future studies to validate LCA predictions against actual industrial-scale operations (Tsoy et al., 2020).

3.4.1.2.3 Inventory modelling: a dynamic approach

To address these data challenges, the LCI can be modelled as a dynamic process inventory (DPI) or a dynamic system (DSys). Modelling the inventory as a DPI means focusing on the dynamic changes in the foreground system, whereas selecting a DSys incorporates changes in the background, providing a more comprehensive evaluation of impacts across the life cycle (Sohn et al., 2020). The foreground system can be modelled using predictive scenarios, which involve known information such as learning curves and proxy technologies to model a highly possible future state. Alternatively, a range of scenarios considering extreme conditions can be explored, resulting in a range of impacts, including worst and best-case scenarios (Arvidsson and Molander, 2017;Wender et al., 2014b; Bisinella et al., 2021). While comparing lab-scale LCA results to mature technologies can be problematic, lab-scale LCA remains valuable for identifying hotspots early in the technology development process and guiding technology developers (Pallas et al., 2020a, 2020b)

3.4.1.2.4 Temporal consistency and upscaling

Ensuring temporal consistency between foreground and background data is another critical challenge, as both systems are expected to evolve over time. Background system modelling typically relies on secondary data from established databases, but this can lead to temporal mismatches if the data do not adequately reflect future conditions (Arvidsson and Molander, 2017; Hospido et al., 2010). Using scenarios with average data representing future times in line with the foreground system could mitigate the risk of temporal mismatch between the two systems (Arvidsson and Molander, 2017; Pesonen et al., 2000; Sandén et al., 2005). An additional consideration is the scale of adoption of the studied technology, which is case-specific, with the extent of adoption and the degree of influence parameters to be defined. This is particularly important for emerging technologies in electricity production, which could be

widely adopted in the electricity grid, with their consumption considerably increasing with time (Hospido et al., 2010).

3.4.1.2.5 Foreground system

Foreground data, specific to the technology within the system boundary, is typically sourced from experimental work, especially during the early stages of emerging technologies (Aryapratama and Janssen, 2017). As these technologies scale up, additional foreground data is required, often drawn from various sources, including literature, patents, expert interviews, unpublished results, chemical equations, and process simulations. However, using data from multiple sources can introduce inconsistencies in the LCI (Moni et al., 2020).

3.4.1.2.6 Background system

Background data, on the other hand, represent the broader context of the study, including upstream supply chains necessary for the technology to perform its selected functions (van der Giesen et al., 2020). Typically provided by LCI databases such as ecoinvent (Wernet et al., 2016), background data is often aggregated and reflects average market processes (Thonemann et al., 2020a). While including the background system in LCA studies is crucial for aligning with traditional LCA practices, it introduces additional challenges. One key challenge is ensuring temporal alignment between changes in the background and foreground systems, as inconsistencies can significantly influence LCA results (Arvidsson and Molander, 2017; Hospido et al., 2010; Sandén et al., 2005)

3.4.1.2.6.1 Integrated assessment models (IAMs) and socio-economic pathways scenarios (SSPs)

To overcome the temporal challenges associated with background data during the upscaling of emerging technologies in the foreground, integrated assessment models (IAMs) have been proposed, such as the Integrated Model to Assess the Global Environmental (IMAGE) PBL Netherlands Environmental Assessment Agency., 2014). IAMs incorporate socio-economic factors such as population growth, technological development, and policy changes to provide projections energy supply and demand, land use, GHG emissions, air pollution, mitigation costs, prices that influence relevant processes (Mendoza Beltran et al., 2020).

These future socio-economic changes are quantified by IAMs and converted into numerical scenarios that show how these factors interact with the natural environment over time. For example, in IMAGE, population growth is translated into a numerical scenario estimating the increase in electricity demand. This is processed by energy system models, such as TIMER (model in IMAGE), which use mathematical equations to simulate future energy demand, energy sources, and associated emissions. IMAGE incorporates various energy sources (e.g., bioenergy, nuclear, solar, and wind) and policy measures (e.g., carbon pricing and incentives) to estimate the potential transition to cleaner energy (Riahi et al., 2017; van Vuuren et al., 2017). As electricity demand rises due to population growth, IMAGE can simulate the necessary investment in renewable energy and supporting infrastructure to meet the demand. By modelling these transitions, IAMs help to bridge the gap between existing background systems and emerging foreground technologies.

Databases used in IAMs are often based on shared socioeconomic pathways (SSPs) that describe broad future scenarios for technological and socio-economic change (O'Neill et al., 2017). There are five distinct SSPs, namely "sustainability" (SSP1), "middle of the road" (SSP2), "regional rivalry" (SSP3), "inequality" (SSP4), and "fossil-fuelled development" (SSP5), each of which contains narratives for factors affecting demographics, human development, economy, technology, and the environment (Figure *3-3*) (O'Neill et al., 2017).

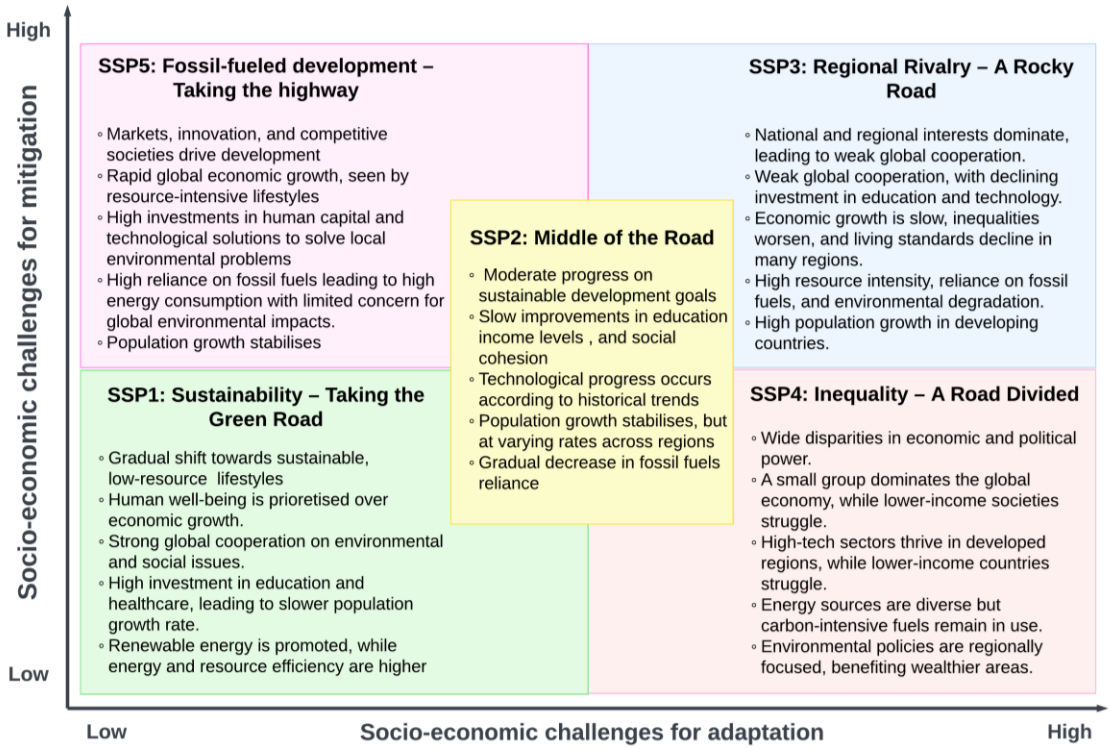


Figure 3-3: Socio-economic pathways scenarios (SSPs) according to the challenges of mitigation and adaptation (adapted from (O'Neill et al., 2017))

3.4.1.2.6.2 SSP "baseline" scenarios and representative concentration pathways (RCPs)

SSPs have been used to develop 'baseline' climate scenarios in the absence of any climate change mitigation policies (Riahi et al., 2017; van Ruijven et al., 2014; van Vuuren and Carter, 2014). In addition to SSPs baseline scenarios, representative concentration pathways (RCPs) have been applied to describe different levels of radiative forcing (Riahi et al., 2017). For example, SSP2 baseline projects a 3.8-4.2°C rise in global mean surface temperature by the end of the century without further mitigation efforts, while the integration of RCP 2.6 targets in the same socioeconomic pathway could constrain temperature rise to 2°C by 2100 (O'Neill et al., 2014).

3.4.1.2.6.3 Future background scenarios for electricity generation

In the context of electricity generation, IAMs such as IMAGE (Stehfest et al., 2014), have been integrated to model the effects of SSPs on future LCIs (Figure *3-4*). Future LCIs for electricity generation had been developed by merging projection data from the IMAGE model into the ecoinvent database (v3.6 cut-off model) (Steubing et al., 2016). This enabled the projection of

future energy system changes, facilitating a more accurate representation of prospective electricity generation scenarios across various time periods (e.g., 2020, 2025, 2030). These future-based versions of the ecoinvent database were then merged into a single 'superstructure database', which encompassed all processes, intermediate flows, and environmental exchanges across the various databases, while a 'scenario difference file' was generated to capture the differences in the flow values for all time steps and selected scenarios (e.g., SSPs, RCPs). The superstructure database can be converted into any of the original pLCI databases with the data contained in the scenario difference file. This enables the LCA practitioner to use a single background database (the superstructure database) instead of multiple ones to represent all scenarios and time steps (Steubing and de Koning, 2021). The scenario difference file can be simultaneously used to incorporate both foreground and background scenario data.

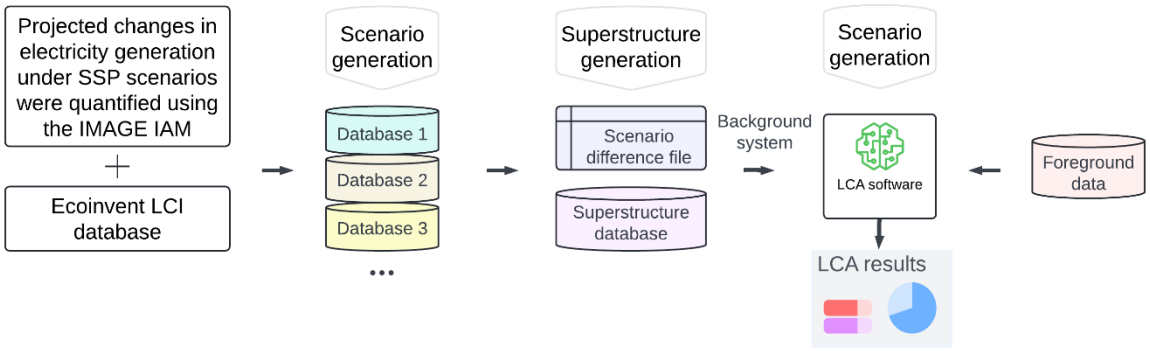


Figure 3-4: Methodology for generating future electricity background scenarios (adapted from (Steubing and de Koning, 2021).

These models have been effectively employed to adjust the electricity sector in background systems, providing a more accurate examination of the environmental burdens associated with future technologies, such as battery electric vehicles (Cox et al., 2018). Climate mitigation scenarios have further been employed as background scenarios to investigate long-term environmental impacts, providing a holistic view of future outcomes (Gibon et al., 2015). Incorporating future projections into the background system of environmental assessments is necessary for preventing the overestimation or underestimation of impacts. Recent studies emphasise the importance of accounting for variations in factors like electricity grid compositions, which are not typically considered in traditional attributional LCA studies (Maes et al., 2022; Hren et al., 2023; Thonemann and Schulte, 2019; Arvidsson and Molander, 2017; Battiston et al., 2023). For example, Hren et al. (2023) demonstrated that climate change impacts from electricity sources could vary significantly, depending on the energy mix, ranging

from 0.25 to 0.8 kg CO_2 per kg of product. This variability highlights the necessity for sensitivity analyses to reflect future scenarios accurately (Battiston et al., 2023; Hren et al., 2023; Maes et al., 2022).

3.4.1.3 LCA software and tools

Several LCA software tools, such as GaBi, SimaPro, and OpenLCA, have been developed, incorporating extensive databases that enable users to model and assess the environmental impacts of various products and processes. However, these tools face significant limitations when it comes to modelling future scenarios, particularly for evolving background systems e.g., future energy mixes, as their databases have been designed for current or past conditions. This makes it challenging to incorporate dynamic parameters such as those arising from SSP or RCP climate scenarios. To address these limitations, more flexible tools like Brightway2 and the Activity Browser have been introduced for modelling emerging technologies. Brightway2, an open-source, Python-based framework, allows for LCA model development, facilitating advanced and non-standard approaches (Steubing et al., 2020). The Activity Browser, built on the Brightway2 framework, offers dynamic solutions by allowing users to adjust parameters for both foreground (e.g., emerging technologies) and background (e.g., evolving energy systems) scenarios. Additionally, “premise”, an open-source Python library, can be integrated into Brightway2 and the Activity Browser, enabling the direct incorporation of prospective SSP-based LCIs into LCA models (Sacchi et al., 2022). This streamlines the modelling process, significantly reducing the time and effort required by the LCA users. By facilitating the application of dynamic, future oriented LCA scenarios, “premise” in combination with Brightway2 or the Activity Browser, becomes a powerful tool for assessing emerging and prospective technologies.

3.4.1.3.1 Limitations and the need for dynamic models in assessing future environmental impacts

Research consistently indicates that relying solely on current data can lead to significant misestimations in environmental impact assessments. Excluding such sensitivity analyses can misrepresent the assessment of CO_2 emissions (Maes et al., 2022). Hence, there is a need to integrate dynamic models that consider future changes.

While existing literature has evaluated background effects such as electricity scenarios, a more comprehensive evaluation is necessary to capture their influence on the overall environmental performance. (Hren et al., 2023; Thonemann and Schulte, 2019) assessed the impact of electricity scenarios but did not consider impact categories such as ionising radiation, metal depletion, and land use. This gap points to the need for holistic evaluations that include a broader range of impact categories to understand potential trade-offs.

The literature highlights the need for a future-oriented database developed through online collaborative platforms to better predict the societal impacts of emerging technologies. Although this initiative remains in the proposal stage, it is crucial to address the unique challenges that LCSA of emerging technologies faces compared to mature technologies. These challenges include:

1. Identifying reliable sources and methods for obtaining social foreground data to model both early-stage and commercial-scale technologies.
2. Ensuring that background data is accurate and representative of the evolving scope of emerging technologies.
3. Assessing whether the integration of background data, rather than relying solely on foreground data, will enhance the sustainability performance of these technologies as they scale.

Building on these concepts, Hung et al., (2020) introduced the Lifecycle Screening of Emerging Technologies (LiSET) framework. This framework complements the need for adaptable LCSA methodologies by employing screening and streamlined LCA methods for efficient environmental evaluation at low TRLs, such as during the lab scale or design phase. The LiSET approach, which transitions from screening to streamlined to full LCA, is beneficial for assessing the feasibility of new technologies early in their development and identifying promising candidates for further advancement.

3.4.1.3.2 Challenges and opportunities during technology transition

As emerging technologies transition from lab-scale or pilot-scale processes to industrial-scale production, the differences in process equipment, operating parameters, and performance data become increasingly significant (González-García et al., 2018; Piccinno et al., 2018; Salas et al., 2018). Industrial processes are generally much more efficient due to the implementation of complex equipment and advanced features, e.g., heat recovery systems and waste recycling loops. To perform meaningful LCAs on emerging technologies, the lack of industrial-scale data can be resolved by upscaling the lab-scale or pilot-scale systems and data to projected industrial-scale levels using explicit methods and scenarios (Tsoy et al., 2020).

Future-oriented preliminary assessments play a crucial role in minimising environmental impacts and avoiding unnecessary financial costs by identifying key impact-contributing areas and optimising technology design prior to scale-up. These assessments utilise scenarios that incorporate future projections, resulting in a range of potential environmental impacts and costs (depicted by the solid green and blue lines, respectively). These ranges offer valuable insights into processing parameters and conditions, guiding early-stage optimisation and selecting the most sustainable design, which minimises costs and environmental impacts. The dash-dotted green and blue lines illustrate how a traditional retrospective approach might reveal that lower impacts could have been achieved under different conditions. At higher TRL levels, although data uncertainty decreases, the flexibility to make significant changes diminishes. This underscores the importance of a strong early understanding to adjust parameters and optimise design more effectively when greater flexibility and degrees of freedom are available during the initial stages of technology development (Figure *3-5*).

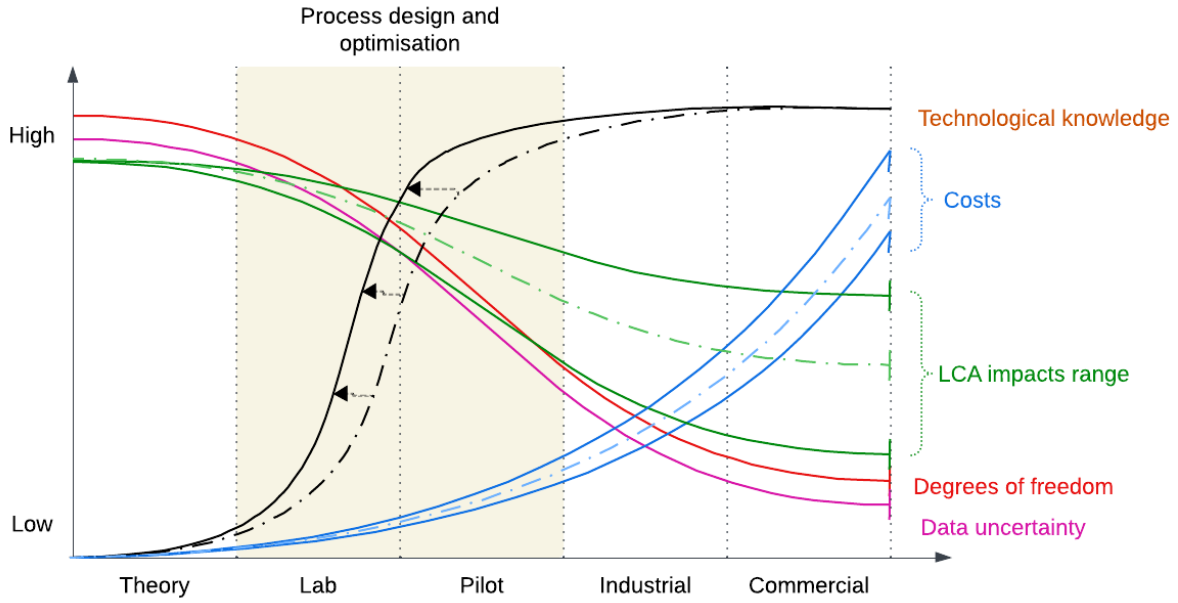


Figure 3-5: Comparison of future-oriented preliminary assessment and traditional retrospective approach along technology development: The solid green and blue lines represent the projected range of environmental impacts and costs under future-oriented scenarios. The dash-dotted lines show potential outcomes in a traditional retrospective approach. The pink, red, and black lines represent the degree of uncertainty, the design flexibility, and the technological knowledge, respectively (adapted from Hetherington et al., 2014; Villares et al., 2017)

3.4.1.3.3 Framework development for up-scaling emerging technologies

A robust scale-up framework is required to model and evaluate an emerging technology's environmental performance effectively. Numerous studies have explored methods (e.g., thermodynamics and scientific principles, engineering perspective, use of power law, proxy technologies) to scale up inventory data (Balgobin and Evrard, 2020; Caduff et al., 2014; Miller and Keoleian, 2015; Piccinno et al., 2016; Shibasaki et al., 2006).

3.4.1.3.3.1 Proposed frameworks and upscaling approaches

A structured approach was proposed by (Walczak et al., 2014) to manage the scale-up process, which involved a 6-step procedure to transition from lab to full commercial operations. Building on this, Shibasaki et al. (2007) introduced a systematic, target-oriented analysis to upscale chemical processes from pilot to industrial scale. Key production or life cycle stages contributing the environmental impacts could be identified as facilitating early-stage optimisation. (Simon et al., 2016) advanced the discussion by proposing a theoretical framework for process scale-up, combining analyses of functions, dimensions, and similarities

to develop a new life cycle inventory based on lab-scale information, providing a pathway to extrapolate lab-scale data to predict industrial-scale impacts. Other studies focused on developing techniques for carrying out environmental LCA of emerging technologies and identifying significant challenges such as cross-study comparability, data availability and quality, scale-up issues, uncertainty, and uncertainty communication (Bergerson et al., 2019; Thonemann et al., 2020). (Piccinno et al., 2016) proposed an engineering-based framework that estimates future impacts at an industrial scale, supporting comparisons with existing technologies. This theoretical approach, however, relies on the existence of similar processes at an industrial scale. Complementing this, (Caduff et al., 2014, 2011) used empirical data to establish scaling laws, particularly power-law relationships, to estimate key properties and life cycle impacts of various energy equipment. Their work highlights the importance of distinguishing between technological learning and scaling effects when applying these relationships in LCA. However, Piccinno et al. (2018) noted that the relative contributions of individual equipment or steps could change significantly after scale-up, which simple scaling procedures might not fully account for. Additionally, using economies of scale for individual equipment often ignores the synergies achievable by integrating different equipment in the production line.

3.4.1.3.3.2 Upscaling challenges

Upscaling poses a significant challenge in the preliminary pLCA of emerging technologies. This phase involves making several assumptions to generate scenarios regarding future development and the transition to a commercial scale. Reliable and consistent up-scaled models require data on potential future changes, such as increases in physical size and improvements in energy efficiency (Arvidsson et al., 2018; Gavankar et al., 2015). Laboratory steps are typically not directly linked, and the equipment used is not equivalent to industrial reactors and machinery (Piccinno et al., 2016). Efficiency measures, such as the reuse and recycling of raw materials, the use of waste heat, and continuous processing instead of energy-intensive batch processing, become feasible only beyond a certain production volume. For instance, an LCA on wood torrefaction revealed approximately a 99% reduction in cradle-to-gate environmental impacts when scaling from a laboratory to an industrial scale (Tsalidis and Korevaar, 2022). Consequently, LCAs based on lab-scale data may overestimate environmental impacts, as process yield and efficiency gains are not comparable until higher production levels are achieved (Khanna et al., 2008).

3.4.1.3.3.3 Predictive scenarios and extreme ranges

Several approaches have been employed to address scaling challenges, including predictive scenarios and scenario ranges (Thonemann et al., 2020a) to generate inventory data at an industrial scale. Predictive scenarios investigate environmental performance based on plausible future developments considering the status quo. The use of scenario ranges determines the maximum environmental impact under extreme conditions (Moni et al., 2020) and has been validated as a suitable approach for addressing uncertainty around future developments (Arvidsson et al., 2018; Thonemann et al., 2020a) Data for these predictive scenarios and ranges can be sourced from scientific articles, patents, expert interviews, unpublished experimental data, and process modelling (Arvidsson et al., 2014; Arvidsson and Molander, 2017; Villares et al., 2017). Valsasina et al. (2017) employed extreme scenario analysis along with power laws to predict potential impacts at future industrial scales compared to current pilot scales. This approach ensures a broader consideration of possible outcomes. Furthermore, (Arvidsson and Molander, 2017) emphasised the importance of considering different foreground and background scenarios to ensure temporal robustness. In their prospective LCA study of epitaxial graphene production, the electricity production dominated overall environmental impacts, indicating that the upscaling had a minor contribution to the impact reduction.

3.4.1.3.3.4 Guidance on selecting appropriate scaling procedures

Reviews on LCA of emerging technologies have also touched upon the topic of upscaling, highlighting the need for guidance on selecting appropriate scaling procedures considering factors like time-intensiveness or complexity (Buyle et al., 2019; Moni et al., 2020; Thonemann et al., 2020a). Approximations are commonly used across all technology types, with process engineering prevalent for chemicals and biotechnologies. Simple extrapolation is often combined with these methods. Additionally, there is a potential relationship between simulations and areas such as chemicals, bio-products, and waste treatment (Erakca et al., 2024). Additional upscaling techniques include learning curves, algorithms, stoichiometric relations, and scaling factors (Bergesen and Suh, 2016; Caduff et al., 2014; Sousa et al., 2001; Tecchio et al., 2016).

3.4.1.3.3.5 Limitations and considerations

However, a significant limitation in the assessment of new technologies or materials is the lack of characterisation factors during the environmental impact assessment phase (van der Giesen et al., 2020). This challenge arises because impact assessment methodologies often lag behind the rapid development of new materials, making it difficult to accurately assess their environmental impacts. Additional challenges involve data quality, lack of transparency, and data variability are common problems that can affect the reliability of LCA results (Hetherington et al., 2014).

Although LCA impact assessment models have advanced significantly, technology development has outpaced these models, leading to unclassified and uncharacterised flows that cannot be accounted for in future oriented LCAs. This gap can hinder early intervention in the design process to mitigate environmental burdens (Tufvesson et al., 2013). Different characterisation factors for the same impact category, risk assessment, and management should be applied to cover most environmental impact categories and avoid burden shifting. Failing to do so could leave many environmental, social, and economic impacts unclassified, potentially leading to incomplete or misleading assessments (Tsang et al., 2017).

In their prospective LCA study, Huijbregts et al. (2003) classified uncertainties into three main categories: input data (parameter uncertainty), scenario uncertainty, and model uncertainty. Scenario uncertainty arises from assumptions made during the goal and scope definition phase and scale-up scenario development, while model uncertainty is inherent to the mathematical models used (Miller and Keoleian, 2015). Inadequate capture of variability and uncertainty can undermine the integrity and effectiveness of a study, potentially leading to misguided technology development (Heijungs and Huijbregts, 2004; Hetherington et al., 2014; Lloyd and Ries, 2007; McManus and Taylor, 2015). Prospective LCA analyses of emerging technologies encounter different and often greater uncertainties compared to traditional retrospective LCAs.

3.4.1.3.4 Addressing uncertainty and sensitivity considerations

Several methodologies have been proposed to address these uncertainties in LCA. Common indicators include intervals (lower and upper bounds), variance, probability distribution, and possibility distribution (or fuzzy intervals). These indicators, combined with multiple scenario

analyses, help address uncertainties related to model structure and context (Igos et al., 2019). Several strategies have been identified to minimise uncertainty from input data and scenarios. Techniques such as probability distributions, statistical testing, and global sensitivity analysis are widely discussed in the literature (Cooper and Gutowski, 2020; Marini and Blanc, 2014). Among these, Monte Carlo simulations, which use probability distributions to address parameter uncertainty, are the most commonly employed (Gerber et al., 2011; Weyand et al., 2019). Global sensitivity analysis is also gaining attention for its ability to assess the sensitivity of LCA outputs to the entire input system (Lacirignola et al., 2017). Ravikumar et al. (2018) extended these insights with an anticipatory decision-driven sensitivity analysis approach tailored for the comparative assessment of emerging technologies. This approach identifies factors most influencing the relative performance of technology alternatives, aiding in selecting the most promising options for further research and development. It applies internal normalisation, similar to the stochastic multi-attribute analysis proposed by (Prado and Heijungs, 2018), to prevent bias from external normalisation and enhance the benchmarking of the studied technology against competitors.

Another critical consideration in LCA is external normalisation to select preferable alternatives. It is essential to ensure that the normalisation reference is consistent with the context of the results, accurately reflecting regional differences and uncertainties. If this alignment is not maintained, the normalisation process may fail to represent the environmental performance of the compared technologies (Prado and Heijungs, 2018; Prado-Lopez et al., 2016).

Conducting sensitivity and uncertainty analyses is crucial for enhancing the transparency and integrity of LCA results. (Miller and Keoleian (2015) emphasised the importance of recognising uncertainties in both foreground and background systems, with (Arvidsson et al. (2018) highlighting the influence of the temporal scale of technology development on the degree of uncertainty. As LCA results are increasingly used by policymakers, stakeholders, funding agencies, and organisations to manage and report their environmental responsibilities, effectively communicating these uncertainties is essential. This ensures the transparency and credibility of LCA studies and helps prevent biased interpretations by non-expert stakeholders (Gavankar et al., 2015; Igos et al., 2019).

Global sensitivity analysis is another approach for evaluating the sensitivity of LCA outputs across the entire input space, offering a comprehensive understanding of how input variations affect results. For further details on global sensitivity analysis (GSA), refer to Cucurachi et al. (2015) for an outline of best practises and methodological guidance, and Lacirignola et al. (2017) for managing high input variability in the context of emerging technologies.

3.4.2 Summary of challenges

The reviewed ex-ante LCA studies faced several challenges, particularly in future scenario and data modelling, and the integration of new LCA methodologies. For instance, technology scenario definitions were included but without stating whether technology experts or stakeholders are involved. While simplified process diagrams rather than full LCA flowcharts are typically used, potentially leading to oversights in system modelling (Tsoy et al., 2020). Manual calculations and stoichiometry were the most commonly used methods, especially in chemical technology studies, while process simulations and machine learning-based surrogate models (based on neural networks) are still rarely used in ex-ante LCA due to their complexity and higher data requirements.

To improve the accuracy of ex-ante LCA, it is important to involve technology experts in scenario definitions and clearly document their involvement. Detailed LCA flowcharts should be prepared and presented for transparency. Emission estimation methods must be thoroughly documented, and TRLs should be explicitly provided to provide more precise insights into technology maturity (Tsoy et al., 2020).

Combining future scenarios with LCA methodologies introduces several conceptual and methodological challenges in LCI modelling. While prospective LCA aims to account for the consequences of future changes, formal guidance on future oriented LCA data modelling remains limited (Bisinella et al., 2021). Challenges include predicting the operational maturity of new technologies, defining functional units that align with both new and incumbent technologies, and identifying appropriate incumbent technologies for comparison (Arvidsson and Molander, 2017).

Further challenges involve predicting future performance, obtaining representative LCI data, and adapting to future developments in incumbent technologies and background systems. Additionally, addressing new impacts not covered by existing LCIA methods, adapting characterisation factors for future technological changes, and managing increased uncertainty in modelling new technologies are critical (van der Giesen et al., 2020).

In conclusion, assessing emerging technologies through LCA presents numerous challenges, from insufficient expert involvement to data gaps and an over-reliance on simplified models. As the field progresses, there is a pressing need for tailored methodologies that address the unique characteristics of BR utilisation processes. The following section will look into the environmental assessment of emerging technologies within BR utilisation, highlighting the approaches necessary to capture their environmental implications.

3.5 LCA studies on BR utilisation as cementitious materials

After an overview of LCA studies on BR utilisation in cementitious materials was provided (Section 2.6.2) and the methodological framework for conducting traditional LCA studies was introduced (Section 3.1), recommendations and guidelines for performing ex-ante LCA for emerging technologies were discussed (in Sections 3.3 and 3.4). This section provides a critical review of existing LCA studies for BR processing and valorisation as a cementitious material. This review aims to identify trends, methodological choices, and key limitations across the existing literature, and how these were used to guide the modelling choices and scenarios developed within this thesis. A detailed summary of the reviewed studies is provided in Table 3-3.

Additionally, it introduces the forward-looking LCA framework applied in this thesis, which aims to assess the environmental potential of three emerging BR technologies and their scalability for large-scale use in BR-incorporated cements.

3.5.1 General overview

An early LCA study on the valorisation of BR in cementitious applications was published by Joyce et al. (2018), which evaluated paving blocks made from geopolymers with high BR content using a cradle-to-grave system boundary. In this study, the firing stage was identified as the main contributor (accounting for 89% of the climate change impact and 87% of acidification), highlighting the importance of optimising process energy parameters. Building on this, Joyce and Björklund (2019) expanded the scope to include BR-based cementitious binders produced at industrial scale, focusing on the potential benefit from avoided burdens (FU: 1 kg of avoided product). Their study reported GWP reductions of up to 900 kg CO_2-eq per tonne of BR material, although it relied on several assumptions regarding high BR replacement ratios and mechanical performance, which remain uncertain on a commercial scale. Additional studies by Sterpellone (2020) and Palmero et al. (2022) further demonstrated the environmental advantages of BR valorisation in the production of inorganic geopolymers and alkali-activated binders.

Nikbin et al. (2016) investigated lightweight concrete with up to 25% untreated BR substitution. Their cradle-to-gate analysis showed a 26% reduction in GWP, lowering emissions from 556.8 to 409.9 kg CO_2 eq. per m^3, and also demonstrated significant reductions in CO (32.5%), NOx (17.1%), Pb (31.8%) and SO_2 (22.4%), attributed to reduced energy-intensive clinker pyro-processing. More recent work has explored the synergistic effects of BR blended with other SCMs. For example, Yao et al. (2019) demonstrated that the use of calcined BR in lightweight concrete, in combination with fly ash and carbide slag, resulted in a reduction of 52% in the impact of climate change, 50% in acidification and 78% in ionising radiation. Similarly, Ghalehnovi et al. (2022b) examined BR with silica fume mixtures and found that even higher BR replacement levels (10-20 wt%) could be achieved without compromising performance, resulting in CO_2 reductions of 5–30%, depending on the mixture ratios. These findings highlighted the critical role of raw material selection and mix design optimisation in cement blending. By carefully tailoring formulations, it is possible to achieve higher clinker replacement rates and significantly improve environmental performance without compromising mechanical properties. Complementary findings by Di Mare et al. (2021) and Morsali and Yildirim (2024) demonstrated that integrating untreated BR into composite

cements, particularly at higher replacement ratios, consistently led to reductions in environmental impacts.

While the environmental benefits of incorporating BR into cementitious materials, particularly as an SCM have been demonstrated in the existing literature, most studies are limited to the laboratory scale. This restricts the ability to assess the true environmental performance and feasibility of BR utilisation at commercial scale or in comparison with conventional alternatives. Few studies have evaluated BR specifically as an SCM, with many focusing on untreated BR. Although untreated BR has been shown to reduce environmental impacts such as global warming potential, it often compromises mechanical performance. In contrast, calcination technology has shown promising results in balancing both environmental and mechanical performance, but it remains the only BR processing technology assessed to date. The absence of comparative studies with alternative processing technologies hinders the benchmarking of its performance against other potential BR processing routes.

This highlights a clear research gap: the need for a comprehensive, industrial-scale assessment of multiple BR processing routes to better understand their environmental performance, optimisation potential, and associated trade-offs. Rather than focusing solely on identifying a single best option, it is essential to benchmark these technologies to explore their relative advantages, limitations, and suitability for SCM applications. While some comparisons between BR-based cementitious materials and conventional products have been conducted, a prospective LCA approach has not been adopted. As a result, these comparisons fail to account for the temporal dynamics such as sectoral decarbonisation, that is likely to shape future cement production. Addressing this gap, the current thesis introduces a forward-looking LCA that evaluates BR-incorporated cementitious materials against incumbent alternatives while explicitly incorporating temporal changes. This approach enables a more realistic assessment of BR technologies in future industrial context.

3.5.2 Methodological choices in existing LCA studies

Most studies reviewed in Table 3-3 focus on the cradle-to-gate system, examining stages from raw material extraction to the manufacturing of the final product (Joyce, 2018). However, many exclude life cycle stages, such as the use phase and end-of-life disposal, due to the emerging

state of BR-derived cementitious materials and the lack of available data for these later stages. This omission neglects a full evaluation of BR-cementitious materials across the entire value chain. The studies also exhibit inconsistencies in FUs, with some using FUs based on the weight of avoided product (Ghalehnovi et al., 2022) while others use volume-based units such as per m^3 of concrete (Nikbin et al., 2016). These variations make it difficult to directly compare results and draw unified conclusions regarding the environmental impacts of BR utilisation, highlighting the need for assessment across many stages of the value chain.

The use of various LCIA methods, such as ReCiPe 2016 Endpoint and Midpoint, ILCD, and TRACI impact category weighting factors, introduces inconsistencies in assessed impact categories. While many studies focus on global warming potential (GWP), other relevant impact categories, such as resource depletion and toxicity, are often neglected (Ghalehnovi et al., 2022; Yao et al., 2019; Joyce and Björklund, 2019). This selective approach risks burden-shifting, with potential environmental trade-offs remaining hidden. Additionally, even if foreground data were typically collected from lab or pilot scale experiments, background data rely on secondary databases such as Ecoinvent. While Ecoinvent provides consistency, it introduces additional uncertainty, as these databases may not capture regional and technological variations.

Table 3-3: summarises the reviewed studies, focusing on key parameters and methodological choices related to the production and environmental assessment of BR-derived cementitious materials.

Study	**Cementitious material parameter**			**LCA methodological parameters**			
	Material	Processing technology	Scale	FU	System boundary	LCIA method	LCI data source
Joyce et al. (2018)	Paving blocks from novel high BR content inorganic polymers	Alkali activation	Laboratory scale	1 m² of paving block	Cradle-to-grave	ILCD	Ecoinvent 3.3 database
Joyce and Björklund (2019)	high BR content cements, and inorganic polymers	Untreated	Industrial scale	1 kg of avoided product produced	Cradle-to-gate	ILCD	Ecoinvent 3 database
Nikbin et al. (2016)	Composite cement with BR (for lightweight concrete)	Untreated	Laboratory scale	1 m³ of concrete	Cradle-to-gate	TRACI	Ecoinvent v2.0 database (Green concrete LCA tool)
Ghalehnovi et al. (2022)	Supplementary cementitious material in composite cement	N/A	Laboratory scale	1 kg concrete	Cradle-to-gate	GWP 100	Ecoinvent 3 database

Palmero et al. (2022)	Inorganic alkali-activated components (e.g., bricks and blocks)	Alkali activated	Laboratory scale	1 m^3 of mortar	Cradle-to-gate	ReCiPe 2016 endpoint method	Ecoinvent v.3 databases
(Morsali and Yildirim. (2024)	BR as a partial replacement for cement in concrete production	Untreated	Laboratory scale	1 m^3 of concrete	Cradle-to-gate	ReCiPe midpoint method	Ecoinvent 3.8 database
Yingtao. (2021)	Producing inorganic binder for concrete	Untreated/dried	N/A	1 tn of dry BR consumed	Cradle-to-gate	ReCiPe 2016 endpoint method	Ecoinvent v3.5 database
Yao et al. (2019)	Ready-to-use lightweight porous concrete	Calcined BR	Laboratory scale	1 m^3 block with a density of 600 kg/m^3	Cradle-to-gate	ReCiPe 2016 midpoint method	Ecoinvent 3 database
Sterpellone. (2020)	Bauxite residue-derived geopolymer production	Alkali activated	Industrial scale	1.333 tn of BR-derived geopolymer	Cradle-to-gate	ReCiPe 2016 v1.03 midpoint method,	Ecoinvent v3.5 database

3.5.3 Forward-looking LCA framework applied in this thesis

The limitations and inconsistencies in existing LCA studies on BR utilisation, outlined in Section 3.5.1, underscore the need for standardised environmental assessments for emerging technologies and novel materials. Conducting preliminary assessments during the transition from laboratory to industrial scales is crucial for identifying key environmental "hotspots" and informing technological development. This approach would provide a more comprehensive understanding of the environmental performance and potential benefits of BR processing and utilisation in cementitious materials. To bridge these gaps, this thesis applies a forward-looking LCA framework to assess the environmental performance of BR technologies at an industrial scale. Figure *3-6* illustrates the research framework applied in this thesis adapted from Villares et al. (2017), which is applied in three stages across Chapters 4, 5, and 6.

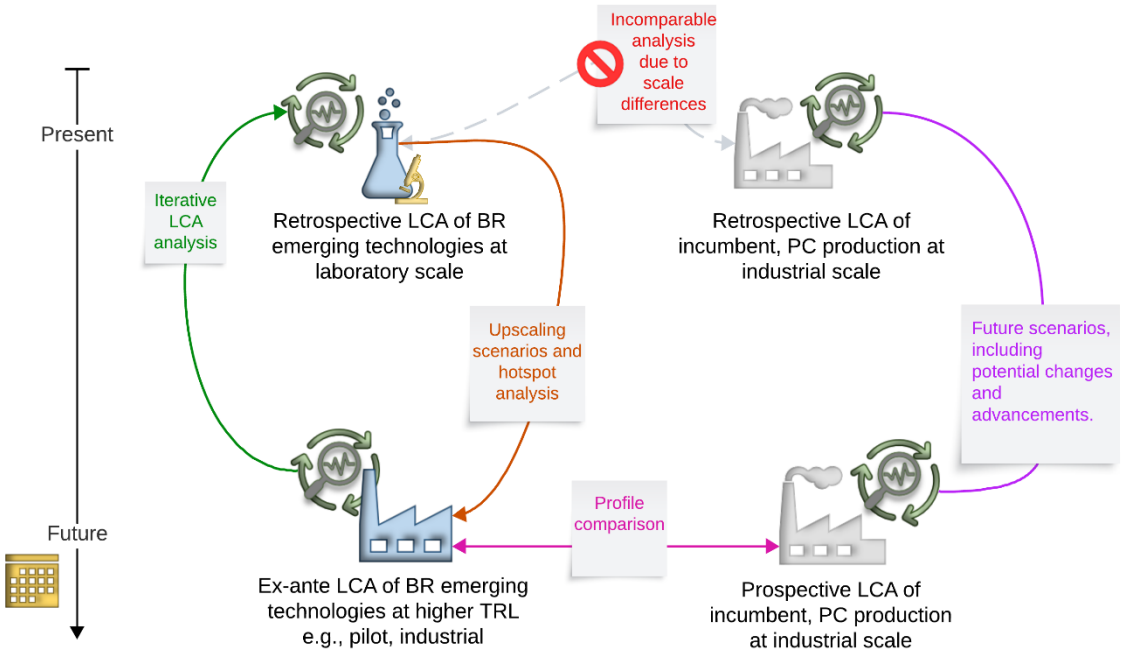


Figure 3-6: Schematic representation of the research framework, adapted from Villares et al. (2017). The first step (outlined in Chapter 4) involves the LCA modelling of the lab-scale system, incorporating upscaling scenarios to assess the environmental performance of the emerging BR technologies through an ex-ante LCA at an industrial scale. Simultaneously, in stage 2 (covered in Chapter 5) a prospective LCA is applied to evaluate future changes affecting the incumbent technology, PC production. Stage 3 presents the environmental profile comparison ensuring temporal consistency between the emerging and incumbent technology systems.

The analysis in Chapter 4 focused on the upscaling of an emerging BR vitrification technology, and how LCA can be applied across laboratory, pilot and industrial scales. Initially, LCA was conducted at the laboratory scale using primary experimental data provided by technology developers and project partners. Similarly, for the pilot scale, preliminary experimental data were obtained and used to evaluate the performance of the vitrification technology. Conducting LCA across multiple scales, enabled the identification of environmental "hotspots" and key areas for improvement. However, as previously discussed, laboratory and pilot data are not directly comparable to mature, incumbent technologies. Hence, industrial-scale models were developed through upscaling approaches i.e., process modelling, regression analysis and proxy technology. This iterative, scale-sensitive LCA highlighted the need for an upscaled LCA model to reflect future industrial applications.

At the same time in Chapter 5, the incumbent technology selected for PC production was investigated, incorporating future technological and operational changes. A prospective LCA was applied to assess the climate change impacts of various decarbonisation measures affecting both the foreground and background systems. This step ensured temporal consistency not only within the incumbent system but also against the emerging BR technologies. This analysis projected the climate change impacts of European PC production, serving as a benchmark for comparing the upscaled industrial results of the emerging BR technologies.

Building upon the industrial-scale modelling from Chapter 4 and clinker production modelling in Chapter 5, Chapter 6 evaluated the environmental performance of BR-incorporated cementitious material, using industrial scale models of three treatment technologies; vitrification, calcination and reductive smelting. Data for the vitrification process were based on the process modelling upscaling approach developed in Chapter 4 and was selected in coordination with project partners as the most suitable approach. While data for calcination and reductive smelting were provided by partners. The performance of BR-incorporated cements was compared to conventional cement types, considering the decarbonisation potential of clinker substitution. The chapter concluded by quantifying the climate change mitigation potential of BR utilisation in cementitious applications and offers guidelines for optimising these processes to achieve maximum environmental benefits.

This thesis was carried out as part of the ReActiv project (Grant Agreement No. 958208), which aimed to valorise BR by enabling its use as a supplementary cementitious material, targeting a

30% clinker replacement rate. While various project partners focused on material characterisation, technology development and testing of BR processing technologies at laboratory and pilot scales, the work presented in this thesis introduced a novel approach for assessing the environmental implications of not only early stage technology scales, but also estimating the large-scale implementation of these technologies using a forward-looking LCA.

The novelty and distinction of this research within the wider project were demonstrated through the following aspects:

- Detailed, laboratory, pilot and large-scale LCA models were developed for emerging BR technologies, extending beyond the laboratory-scale assessments (e.g., material characterisation, technology testing, products evaluation) undertaken in other work packages.
- Process-specific data, including material and energy consumption, emissions, and waste generation, were integrated into the modelling framework. These data had not been previously incorporated in other LCA efforts within the project.
- A scenario-based environmental trade-off analysis was conducted to evaluate the impacts of technology upscaling, an aspect not addressed by other studies within the project, which typically evaluated technologies at the scale tested.

Model development and validation were supported through ongoing collaboration with project partners. For example, process flow diagrams and material and energy balances for BR processing technologies were reviewed in consultation with technology developers, ensuring alignment with technical specifications and experimental outcomes. Additionally, methodological choices such as functional units, system boundaries, and allocation, were discussed during project meetings, facilitating methodological consistency across the different work packages.

Chapter 4. Ex-ante LCA of BR vitrification technology

This section outlines the development of an LCA model for upscaling an emerging technology, BR vitrification from laboratory to industrial scale. Existing literature gaps are identified, followed by a detailed description of the developed LCIs at various scales (laboratory, pilot, and industrial) and incorporating upscaling scenarios according to three upscaling approaches (process modelling, regression analysis, and proxy technologies). Decarbonisation scenarios for electricity generation are also implemented. Finally, the results are presented and discussed, evaluating the prospective impacts of BR by 2050.

4.1 Background and motivation

The rapid expansion of the global population and increasing consumption rates present significant challenges for sustainable industrial development. In particular, the cement industry must change to meet the net-zero emissions target by 2050 (GCCA, 2021). Among the various strategies to mitigate CO_2 emissions in the cement industry, the substitution Portland clinker with SCMs is considered one of the most feasible and cost-effective approaches (Georgiades et al., 2023; Habert et al., 2020; Miller et al., 2021; Pamenter & Myers, 2021). Other strategies, such as increasing alternative fuel consumption and implementing CCS have been extensively studied for their environmental benefit. However, challenges related to material availability, infrastructure demands, retrofitting potential, and primarily higher costs complicate large-scale implementation. These issues could lead to higher cement prices (Driver et al., 2024), posing a challenge to economic growth and affordable housing (European Commission, 2024).

Waste utilisation is increasingly promoted to reduce CO_2 emissions and support cement decarbonisation while fostering a circular economy and industrial symbiosis. Many studies have explored the use of BR as a construction material, with vitrification emerging as a promising technology for its potential to produce a highly reactive material suitable as an SCM. Vitrified BR (VBR) is one of a landscape of emerging technologies to recycle BR into construction materials, with extensive research focusing on applications in cement (Peys et al., 2022; Di Mare et al., 2021; Feng et al., 2022; Giels et al., 2022; Hertel et al., 2021; X. Liu et

al., 2011; Pontikes & Angelopoulos, 2013; Tsakiridis et al., 2004), alkali-activated materials (Hertel et al., 2016; Hertel & Pontikes, 2020; Koshy et al., 2019; Perez-Cortes & Escalante-Garcia, 2020; Toniolo et al., 2018), concrete (Ghalehnovi et al., 2019, 2022; R. X. Liu & Poon, 2016; Nikbin et al., 2016; Tang et al., 2018), bricks (Kim et al., 2019; S. Liu et al., 2017), ceramics (Pérez-Villarejo et al., 2012; Yang et al., 2008), and road base and subgrade materials (Mukiza et al., 2019; Pontikes & Angelopoulos, 2013). However, despite the development of these technologies, most remain at the laboratory or pilot scale, and their environmental performance and economic viability are still poorly understood.

Several ex-ante LCA studies have explored upscaling approaches for developing industrial scale inventories (Caduff et al., 2011; Cucurachi et al., 2018; Piccinno et al., 2016; Shibasaki et al., 2006; Simon et al., 2016; Villares et al., 2016). Mainardis et al. (2021) applied industrial scale data on sludge characteristics, and operational parameters from wastewater treatment plants, along with proxy data on waste sludge treatment. This study indicated that ultrasonication of sewage sludge could reduce greenhouse gas (GHG) emissions by 22% compared to conventional anaerobic digestion. Hybrid approaches that combine proxy technologies with additional steps were explored to address up-scaling challenges. In battery cell production, Simon et al. (2016) integrated experimental data with knowledge of mature processes to estimate the working characteristics of theoretical upscaled models, reducing environmental impact by 40-70%. Van der Hulst et al. (2020) proposed combining proxy technologies and regression analysis to address process changes and size scaling, reporting an 83% reduction in GHG emissions for producing copper indium gallium (di)selenide photovoltaic laminate. Scaling factors linked to economies of scale, where higher production scales lead to reduced operating costs, can also be used for upscaling. These factors can be derived from power law relationships for inventory scaling (Caduff et al., 2011, 2014). Tsalidis & Korevaar (2022) used process modelling with Aspen Plus software (Aspen Technology, 2013) to model wood torrefaction at an industrial scale, showing a 99% reduction in environmental impacts from bench to commercial scale.

Laboratory, pilot, and industrial-scale studies serve different functions: lab-scale tests assess the feasibility under controlled conditions, pilot-scale studies simulate real-world environments on a smaller scale for process optimisation and risk assessment, and industrial-scale aims to validate the performance of new technologies in actual production settings. Effective upscaling

facilitates a smooth transition from concept to market, although challenges with inadequate data and inconsistent frameworks remain (Hischier et al., 2005; Cooper et al., 2013).

In this research, an ex-ante LCA study of BR vitrification technology was completed to evaluate the influence of scale (laboratory, pilot, and industrial) and upscaling approach (process modelling, regression analysis, and proxy technology) on environmental performance. The work addresses a critical gap in the existing literature by highlighting the scarcity of robust data for BR recycling technologies and the lack of standardised methods in analytical up-scaling frameworks. The aim is to enhance understanding and expand on the practical application of ex-ante LCA methodology, to guide the development of BR vitrification technology for improved environmental performance.

4.2 Methodology

4.2.1 Description of bauxite residue vitrification technology

As already described in Section 2.6.1.2, BR vitrification, modelled based on the untreated BR chemical composition provided by Hertel et al. (2016), begins with the blending of additives such as limestone ($CaCO_3$), silica sand (SiO_2), and carbon (for reducing Fe(III) to Fe(II)) to achieve a VBR target composition with 65 wt.% amorphous phase. The final chemical composition of VBR modelled for the vitrification process is shown in Table 4-1. This facilitates the formation of a semi-glassy material that can partially substitute Portland clinker in composite cement. The blended mixture is thermally treated at 1,200-1,300 °C, rapidly cooled, and milled. The resulting material is partly amorphous and significantly more reactive than raw BR in cementitious systems (Giels et al., 2023; Hertel et al., 2023). The BR vitrification process is investigated at three different production scales here: laboratory (Figure *4-1*), pilot (Figure *4-2*), and industrial (Figure *4-3*).

Table 4-1: Final chemical composition of VBR

Chemical composition	**Percentage (%)**
FeO	34.7
Al_2O_3	18.7
SiO_2	18
CaO	17.5
Na_2O	7
TiO_2	2.8
Others	1.3

4.2.1.1 Laboratory-scale process description

At the laboratory scale (Figure *4-1*), raw BR is dried at 105 °C for 24 hours. Limestone, silica (fine quartz sand), and carbon (coke) are mixed with the dried residue. The resulting mixture is homogenised and heated to 1,200-1,300 °C in an induction furnace, using indirect heating by using a susceptor crucible surrounding a ceramic crucible. The melt is rapidly cooled via water-quenching to produce granules. The granules are dried at 105 °C for 24 hours to remove moisture. The final drying step is performed for practicality in further experimental testing, as moisture can greatly impact the reliability and accuracy of characterisation analysis methods (e.g., thermogravimetric analysis, X-ray diffraction, scanning electron microscopy etc.) (Ortiz Ortega et al., 2022). Finally, the dried granules are milled to attain a specific surface area of 4,000-4,500 cm²/g, producing VBR, that can then be used as an SCM.

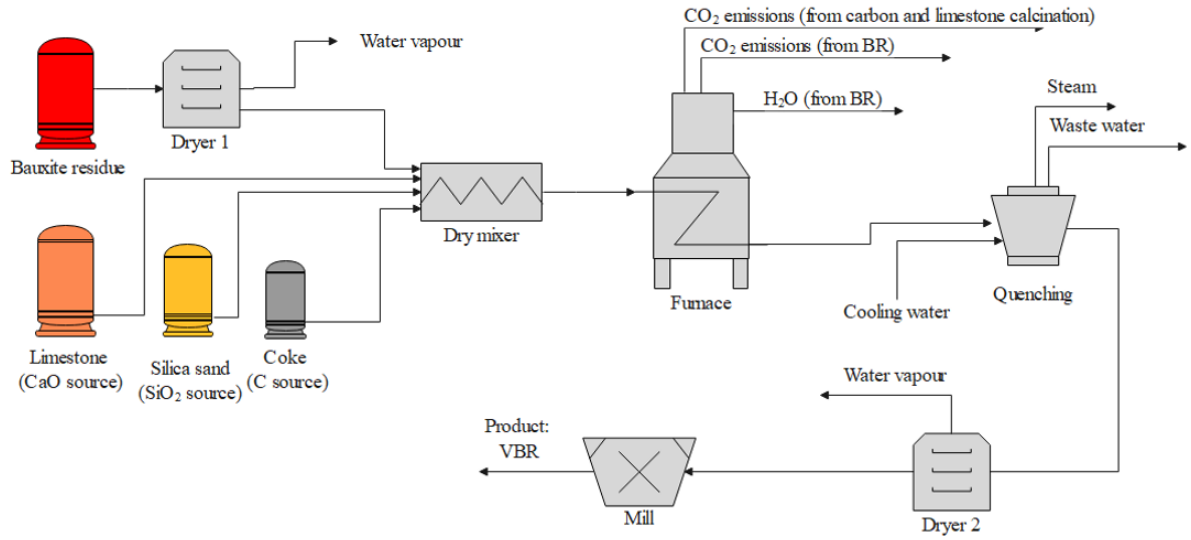


Figure 4-1. Simplified process flow diagram for the laboratory scale BR vitrification system.

4.2.1.2 Pilot-scale process description

At the pilot scale (Figure *4-2*), raw BR is mixed with limestone, silica (fine quartz sand), and carbon (coke). The mixture is heated to 1,200-1,300 °C in an induction furnace using indirect heating, by utilising a susceptor crucible surrounding a ceramic crucible, forming a melt. Following heating, the melt is quenched through air granulation and then milled, producing VBR that can be utilised as an SCM.

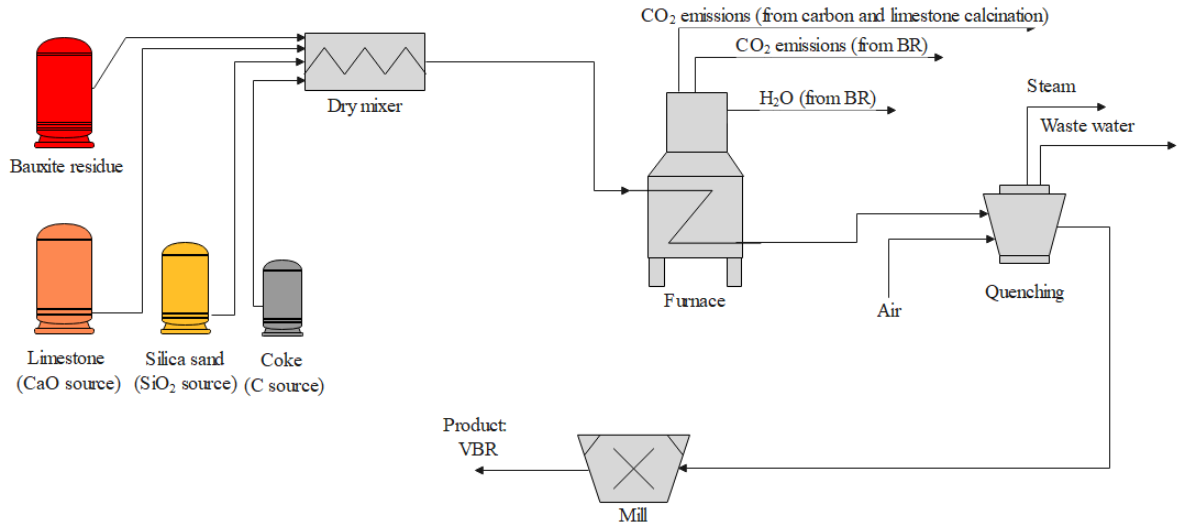


Figure 4-2: Simplified process flow diagram for the pilot scale BR vitrification system.

4.2.1.3 Industrial-scale process description

At industrial scale (Figure *4-3*), raw BR is mixed with limestone, silica (fine quartz sand), and carbon (coke). The blend is subjected to a pelletisation process with a binder addition (i.e., bentonite) to obtain stable pellets with a diameter of approximately 1-2 cm, to reduce dusting during vitrification. The pellets are preheated in a heat exchanger utilising waste heat from the cooling stage. The preheated pellets are heated to 1,200-1,300 °C in an electric-powered furnace. The partial melt is rapidly cooled and then milled.

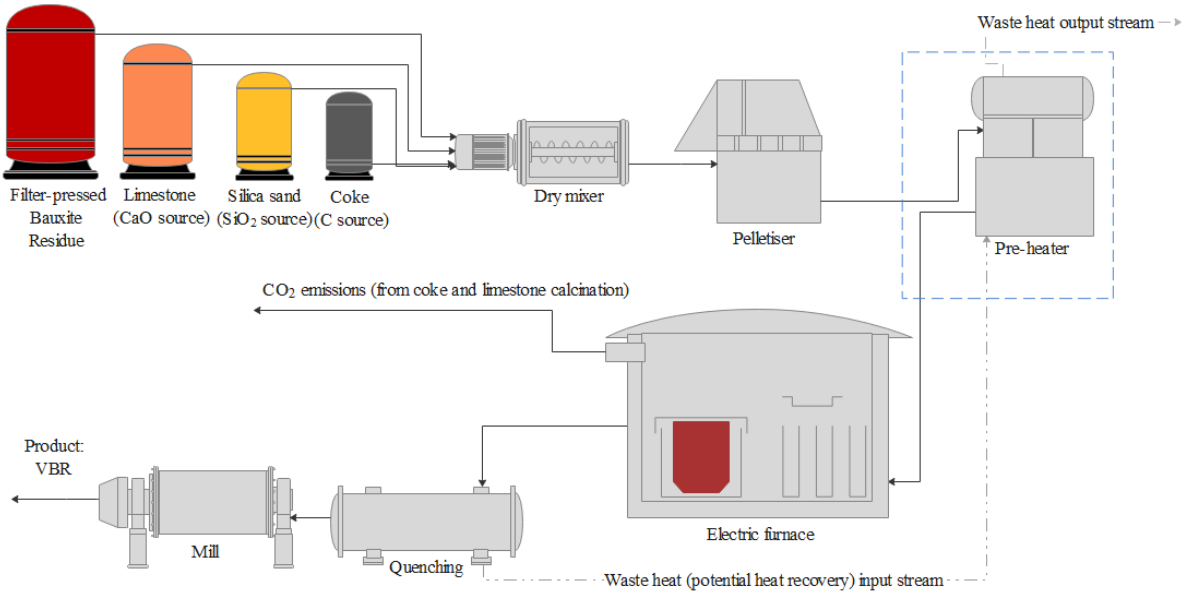


Figure 4-3: Simplified process flow diagram for the industrial scale BR vitrification technology.

A generic electric-powered furnace was selected to model the up-scaled vitrification furnace technology. Alternative furnace technologies, such as the rotary kiln used in cement production, could also be considered. Adopting a rotary kiln would necessitate a shift in the energy source, prompting consideration of alternative fuel mixes, including biomass and hydrogen to replace fossil fuels at least partially. The choice of furnace technology and process parameters such as throughput, residence time, and heat losses play a pivotal role in determining energy requirements (further details on the furnace technology selection are provided in the Appendix A, section A4.1). Although the specific furnace type (e.g., electric arc furnace, induction furnace) can affect energy demand and product properties, this analysis is beyond the scope of our study, due to a lack of data for VBR production using multiple furnaces at pilot scale.

4.2.2 Life cycle assessment

4.2.2.1 Goal and scope

This chapter evaluates the environmental impacts of novel vitrification technology for processing BR into an SCM, referred to as VBR. The influence of technological scales, such as laboratory, pilot, and industrial, and data upscaling approaches, i.e., process modelling,

regression analysis, and proxy technology, on environmental performance was assessed. The functional unit was defined as 1 kg of VBR, suitable for immediate use as an SCM at the factory gate.

The system boundary is defined as a cradle-to gate, including all foreground activities related to the production of VBR: raw material extraction and preparation (i.e., BR, coke, limestone, and silica sand), vitrification technology, and transportation. The vitrification system is disaggregated into key unit operations (i.e., drying. mixing, pelletising, thermal treatment, quenching, and milling) tailored to each technology scale. Electricity production is considered a background activity (Figure *4-4*). Europe was selected as the geographical scope due to its relevance to the ReActiv project, high availability of data and significant potential for BR utilisation in the European cement market. This aligns with our objective to maintain a generic rather than company or site-specific scope.

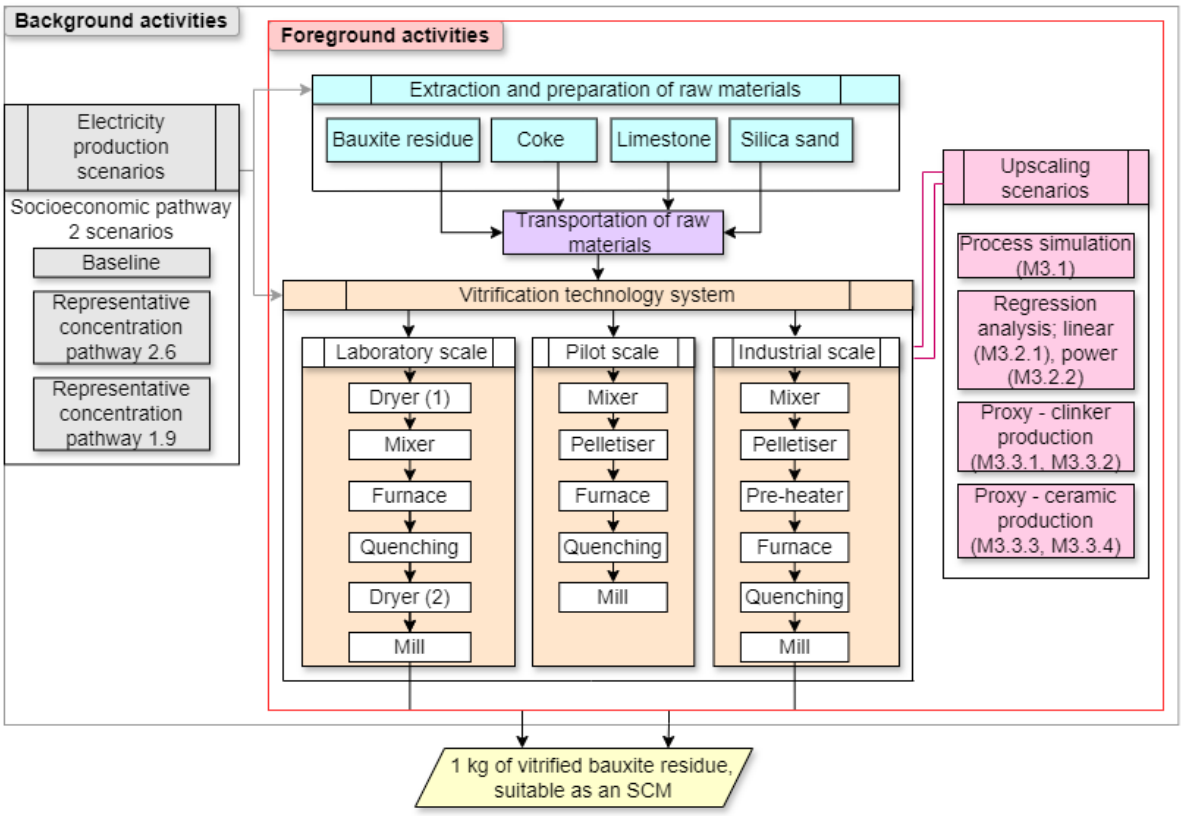


Figure 4-4. Flowchart of the VBR product system modelled across three scales (i.e., laboratory, pilot, and industrial), including foreground (red outline) and background (grey outline) activities, and upscaling approaches used in industrial scale modelling (pink boxes).

4.2.2.2 Life cycle inventory

Foreground data were obtained from experimental results, from existing literature and databases, and generated through upscaling approaches such as process modelling, regression analysis, and proxy technology. Laboratory and pilot scale data were provided by project partners and technology developers based on primary experimental results. For the industrial scale, various upscaling approaches were used to estimate industrial scale inventory data, and to assess the sensitivity of the results to the upscaling approach. Data associated with the construction of the processing plant were excluded, as the environmental impact was assumed to be negligible compared to operational impacts.

The open-source life cycle assessment (LCA) software Activity Browser (v2.8) was used with the ecoinvent database (v3.9.1 cut-off system) as the background database. European electricity production flows were used to model the background system while considering future shifts towards renewable energy sources due to potential decarbonisation measures.

4.2.2.2.1 Foreground system

Upscaling data from laboratory and pilot scales to the industrial scale follows the framework proposed by Buyle et al. (2019) and Tsalidis & Korevaar. (2022). Both foreground (i.e., vitrification technology) and background (i.e., electricity generation) changes were considered to present a realistic implementation of the technology at a future time, using three upscaling approaches: process modelling, regression analysis, and proxy technology.

4.2.2.2.1.1 Process modelling

Process modelling simulates an industrial process using thermodynamic principles and material/energy balances. It predicts system behaviour under varying process conditions and optimises the design through iterative calculations. The “Solids” thermodynamic package in Aspen Plus v.11 software was used to model BR vitrification, estimating energy data across unit processes (e.g., mixer, pelletiser, preheater, furnace, cooler, and mill). Input material flows, such as bauxite residue, limestone, pet coke, and silica sand, were modelled based on their chemical composition.

The energy demand of the pelletiser was calculated based on a normal distribution of particle size distribution, with an average particle size of 1.5 cm. For the preheating step, a counter-current heat exchanger was modelled. The furnace was represented by a stoichiometric reactor to simulate the chemical reactions, such as limestone decomposition, decarbonation (at 600 - 800 °C), and the formation of sodium aluminosilicates, melilite, and spinel (at 600 - 1050 °C). This resulted in a phase composition consisting of 65% amorphous phase, 21% hercynite-magnetite, and 9% magnetite as primary phases. Phase change due to partial melting was empirically calculated and added separately. The hot melt exiting at ~1200 °C, was cooled through a series of coolers to reduce the temperature to 50 °C, with the recovered heat integrated into the preheater. The mill was modelled based on an average particle size distribution (D50) of 10μm.

4.2.2.2.1.2 Regression analysis

Regression analysis models the relationship between independent and dependent variables, enabling prediction and quantification of the dependent variable based on changes in the independent variable. Linear and power law relationships were applied to estimate energy demands for each unit process (independent variable) based on material throughput (dependent variable), where correlation were derived using existing literature and industrial data sheets for unit operations such as the dryer, mixer, pelletiser, furnace, cooler, and mill. These correlations were graphically represented and used to explore the influence of different scaling assumptions on industrial energy demand.

4.2.2.2.1.3 Proxy technology

Proxy technology uses well-established technologies that employ current and/or future unit process operations (e.g., equipment) with comparable process conditions. Clinker production and ceramics vitrification for wall and floor tiles were deemed suitable proxy technologies due to comparable operating conditions, such as firing temperature, unit operations, product application, industry relevance, and data availability. Further information on the proxy technology selection is provided in Appendix A, Section A1.3.1.2. This includes energy demand and material flow data, thermal and power consumption profiles, and process-specific assumptions (e.g., kiln temperature, fuel type, and kiln configuration) used to model the clinker

and ceramic tile production. Two clinker production scenarios (M3.3.1 and M3.3.2) and two ceramic tile production scenarios (M3.3.3 and M3.3.4) were considered: one modelled according to the BAT reference documents and the other based on the Ecoinvent v3.9.1 database. Further detailed inventories are listed in Tables A7-A8. The proxy technologies were selected considering various vitrification technologies across industries (i.e., cement, traditional ceramics, sintered glass, multiphase technical ceramics) to ensure similarities in operating conditions, raw materials, and product characteristics with BR vitrification technology.

4.2.2.2.1.4 Upscaling scenarios

Nine scenarios were selected (Table 4-2) to investigate the effects of the upscaling and upscaling approach on system parameters. This has specifically focused on the influence of variations in operating conditions such as electricity demand per (kg) output in the furnace on the environmental performance. Further details on the inventories for each scenario, including material and energy flows are shown in Appendix A (Tables A1-A6).

Table 4-2. Descriptions of the nine scenarios used to investigate the environmental impacts of BR vitrification technology.

Scenario number	Scale	Data source or type of upscaling approach	Upscaling approach detail	TRL level	Production output (kg/ hr)	Energy demand per output (kWh kg ^{1})
1	M1 Laboratory	Experimental data		4	~2	26
2	M2 Pilot	Theoretical estimations/ (experimental results to be considered)		6	~120	4
3	M3 Industrial	M3.1 Process modelling and thermodynamic calculations	M3.1.1	9	~ 140,000	0.6
4		M3.2 Regression analysis	M3.2.1 Linear relationship			0.57
5			M3.2.2 Power law relationship			0.55
6		M3.3 Proxy technology	M3.3.1 Clinker production[1]			1.0
7			M3.3.2 Clinker production[2]			0.72
8			M3.3.3 Ceramic tile production[1]			1.3
9			M3.3.4 Ceramic tile production[2]			2.0

[1] Proxy technology was modelled according to the best available technique reference document (BREF)(Centre et al., 2013; European Commission, 2007)

[2] Proxy technology was modelled using datasets in ecoinvent v.3.9.1 (clinker production (Ecoinvent v.3.9.1, 2007a), ceramic tile production (Ecoinvent v.3.9.1, 2007b)

4.2.2.2.1.5 Uncertainty and sensitivity analysis considerations

Process modelling was conducted using Aspen Plus simulations to estimate the energy demand of BR vitrification at an industrial scale. As BR was not pre-defined within the software, it was manually modelled based on available chemical composition data. Nevertheless, this approach introduced input data uncertainty, as BR characteristics such as mineralogy, moisture content, and chemical composition, can vary significantly depending on refinery processes, storage methods, and geographic origin. In this work, a defined composition and raw material blend were used to simulate a product with appropriate reactivity as an SCM. However, future studies are recommended to examine how such variations may influence process performance and associated environmental impacts.

Further model structure uncertainty was introduced due to software limitations in simulating solid-state reactions characteristic of high-temperature vitrification processes. For instance, the partial melting of BR had to be calculated and manually integrated into the model.

Regression analysis was employed to investigate the sensitivity of environmental impacts to different scaling relationships. Both linear and power-law models were applied based on published energy consumption data for industrial induction furnaces. This compared how different mathematical relationships affect the LCA outcomes. Parametric uncertainty was also presented, as the regression relationships were derived from energy data not specific to BR vitrification. However, this was mitigated by selecting induction furnace data from ceramic vitrification processes, previously identified as a suitable proxy technology.

To assess uncertainty in technology choice, four proxy scenarios were developed, involving clinker production and ceramic vitrification, each modelled with data from both the Ecoinvent and BREF databases. This allowed for a scenario-based sensitivity analysis, examining how environmental performance shifts with different process assumptions and data sources, especially with respect to fuel mix and process efficiency. For example, differences between electrified kiln systems (used in the process modelling and regression scenarios) and fossil fuel-based configurations (used in the proxy scenarios) were analysed to highlight the impact of energy source assumptions. To address long-term variability, electricity decarbonisation scenarios were incorporated, reflecting potential future reductions in grid carbon intensity.

In addition to scaling and modelling assumptions, the inherent heterogeneity of BR represents a critical source of uncertainty in environmental performance assessments. As highlighted in the literature, BR varies considerably in chemical composition, mineralogy, pH, particle size distribution, and morphology depending on the origin of the bauxite ore, the refining process, and subsequent treatment methods (Evans, 2016; International Aluminium Institute, 2020). These variations influence not only the processability, such as handling characteristics and moisture content, but also the potential for use in cementitious applications. For example, higher levels of iron, alumina, and silica can be advantageous in clinker substitution, whereas elevated sodium content may be detrimental to cement performance and durability. The wide variability in chemical composition is presented in Chapter 2, Table 2-1.

In this study, a fixed chemical composition representative of a selected BR source was used in the modelling of vitrification to ensure a reactive product, suitable as an SCM. While this ensured consistency across scenarios, it also limits the generalisability of the findings. Future work should therefore investigate how variation in BR composition across different refineries and regions could affect energy demand during processing and the resulting environmental impacts.

4.2.2.2.2 Background system

Since electricity is the primary energy source for vitrification and significantly contributes to the environmental performance of VBR (Joyce et al., 2018), various electricity grid scenarios were used to explicitly determine the effects of using lower-carbon electricity in the future. Future electricity production scenarios were based on a modified ecoinvent 3.9.1 database. Three versions of the modified ecoinvent 3.9.1 database were generated, each incorporating different European electricity grids. The year 2050 was selected as the future time step and used three shared socioeconomic pathway (SSP) 2 scenarios: SSP2 baseline (3.8-4.2°C warming by 2100 relative to 1980), SSP2 representative climate pathway (RCP) 2.6 (~2°C warming by 2100 relative to 1980), and SSP2 RCP 1.9 (~1.5°C warming by 2100 relative to 1980) (Bauer et al., 2017).

Full details of the background database modelling setup are provided in Section 3.4.1.2.6. This approach ensures temporal consistency between the foreground and background systems,

reflecting the evolving environmental scope over time alongside technological development and industrial implementation of BR vitrification.

4.2.2.3 Life cycle impact assessment

Life cycle impact assessment was carried out using the ReCiPe 2016 mid-point level impact assessment method with the hierarchist (H) perspective. The impact categories studied are terrestrial acidification (TAP), climate change (GWP100), freshwater ecotoxicity (FETP), marine ecotoxicity (METP), and terrestrial ecotoxicity (TETP), fossil resource scarcity (FFP), freshwater eutrophication (FEP), marine eutrophication (MEP), human toxicity; carcinogenic (HTPc), human toxicity; non-carcinogenic (HTPnc), ionising radiation (IRP), land use (LOP), mineral resource scarcity (SOP), ozone depletion (ODPinfinite), fine particulate matter formation (PMFP), photochemical oxidation formation, human health (HOFP); photochemical oxidation formation, terrestrial ecosystem (EOFP), water use (WCP).

All 18 midpoint impact categories from the ReCiPe 2016 (H) method were included to capture potential environmental trade-offs, which is particularly relevant in early-stage assessments of emerging technologies. While certain categories, such as human toxicity and ionising radiation, are associated with higher uncertainty in relation to emerging BR processing technologies, their inclusion was especially relevant for electricity grid decarbonisation scenarios, where background system changes directly influence several impact categories beyond climate change. This broader perspective enabled the identification of system-level trade-offs and supported a more comprehensive understanding of how evolving energy systems affect the environmental performance of BR processing technologies. In this context, the selection of impact categories reflected the use of a forward-looking LCA approach, requiring consideration of both foreground and background-related impacts.

4.3 Results and discussion

4.3.1 Effect of technology scale

Upscaling BR vitrification technology from the laboratory to pilot or industrial scale significantly reduces its environmental impact by ~83% (66-91%) at the pilot scale (M2) and ~97% (94-99%) at the industrial scale (M3.1-3.3) across all impact categories (Figure 4-5).

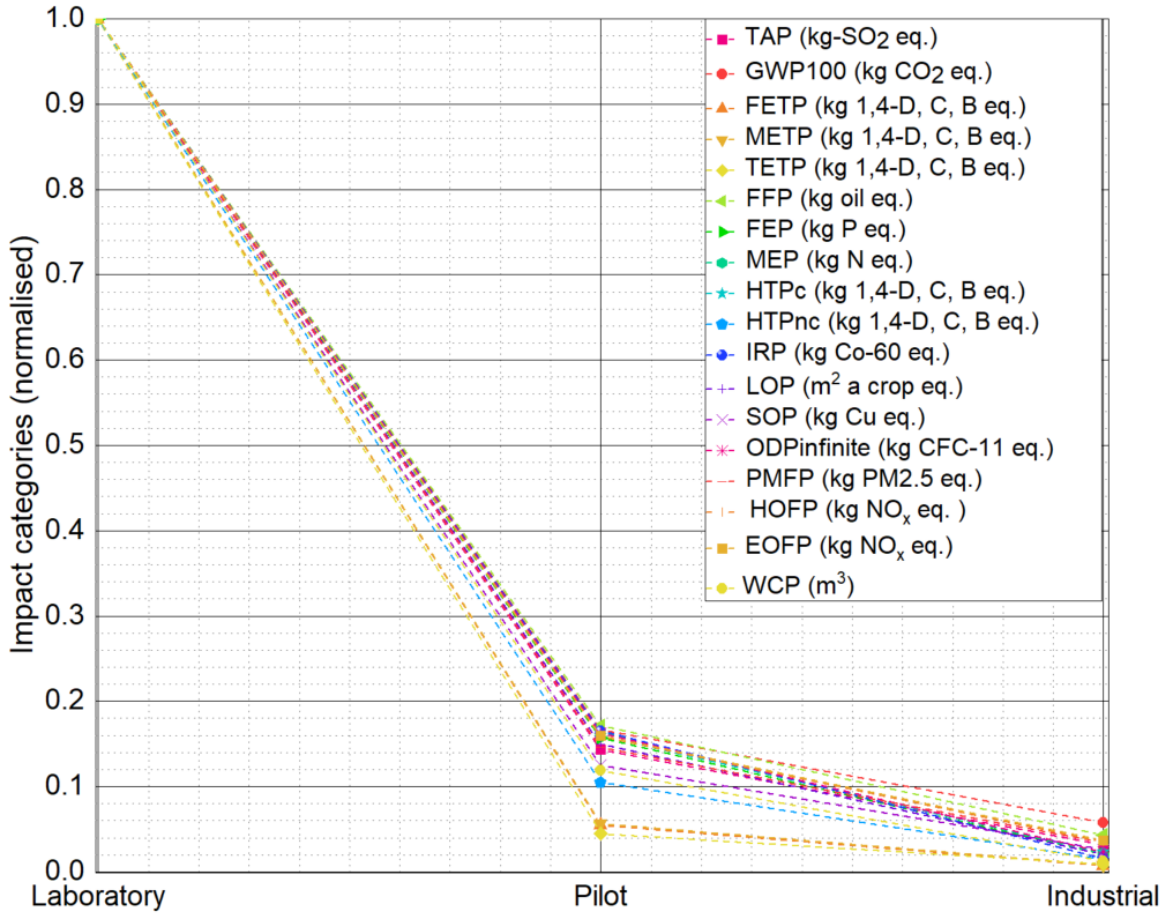


Figure 4-5: LCIA results of VBR production, internally normalised to the laboratory scale, across scales (i.e., laboratory, pilot, and industrial). The mean results for each impact category at the industrial scale were derived from the seven distinct upscaling approaches. For example, the global warming potential (GWP100) at the industrial scale, presents the impact distribution from different upscaling approaches. Abbreviations for ReCiPe (H) impact categories are reported in Section 2.2.1. Quantitative results are tabulated in Appendix A (Table A9).

In the context of climate change, the laboratory scale impact (10.01 kg CO_2-eq. emissions) is substantially reduced to 1.68 kg CO_2-eq. emissions at the pilot scale and further reduced to a mean of 0.58 kg CO_2-eq. emissions at an industrial scale (ranging between: 0.44 kg CO_2-eq.

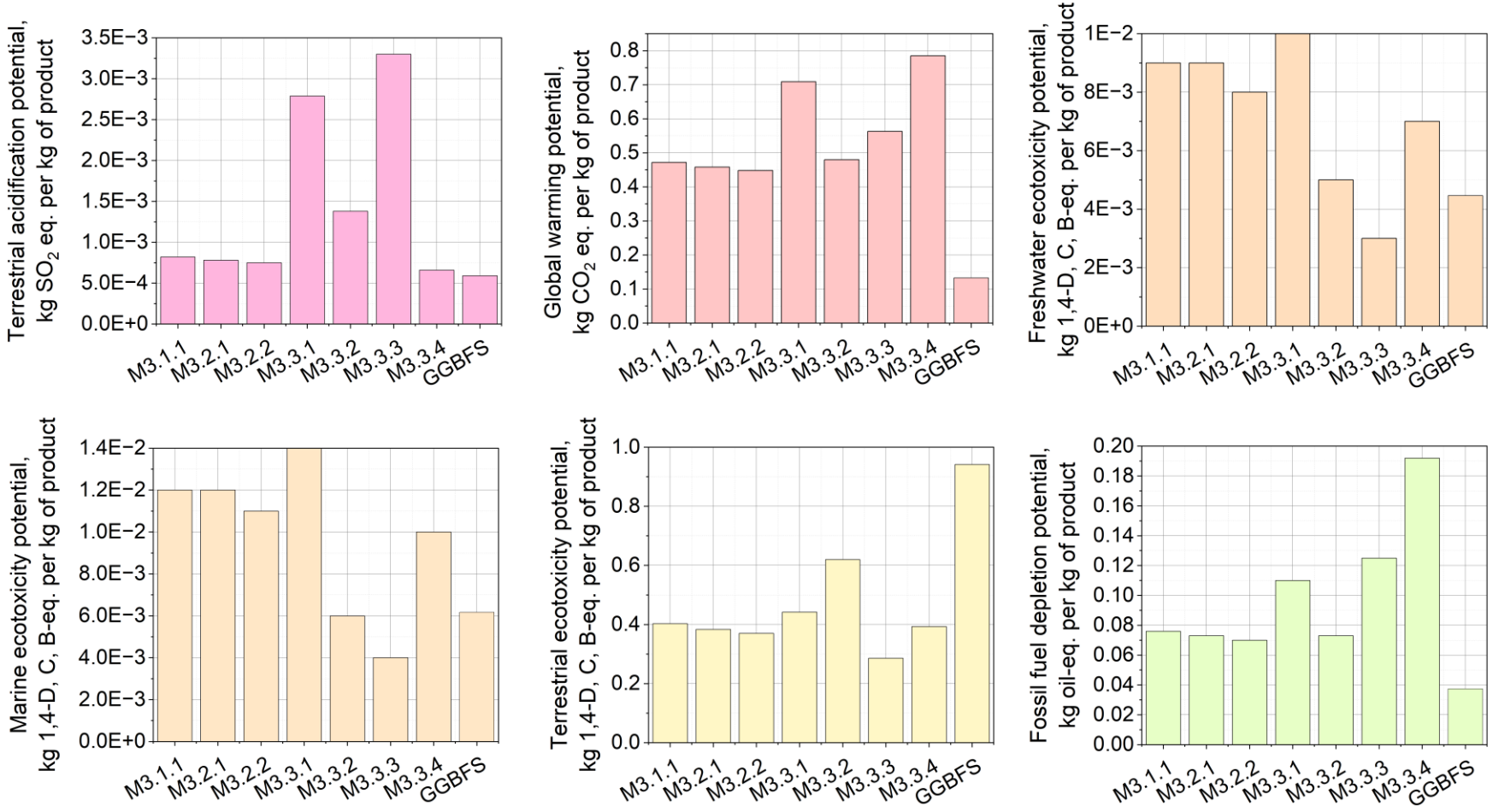
to 0.84 kg CO_2-eq.). The industrial-scale BR vitrification technology has 94% lower CO_2-eq. emissions compared to the laboratory scale. This reduction is due to the high energy inefficiency (14.7 kWh kg^{1}) at the laboratory scale. At the laboratory scale, small batch sizes prioritise technical feasibility, whereas larger, industrial-scale production volumes benefit from economies of scale, leading to process efficiency improvements (0.49-0.75 kWh kg^{-1}) and reduced environmental impacts (Tsalidis & Korevaar, 2022). At the industrial scale, individual unit operations and equipment (e.g., size and geometry), such as milling and drying, are optimised resulting in lower environmental impact contributions. Improvements beyond efficiency are necessary to further reduce the environmental impacts, for instance reducing direct process emissions from heating raw materials, such as coke and limestone.

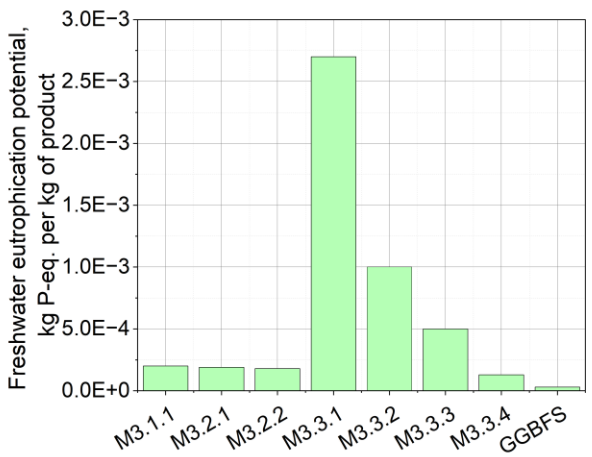
The reduction in environmental impact when upscaling from laboratory to pilot or industrial scales aligns with ex-ante LCA studies of other industrial processes (Tan et al., 2018; Van der Hulst et al. 2020; Tsalidis & Korevaar, 2022). Tan et al. (2018) observed over an 80% reduction in non-renewable energy use, greenhouse gas emissions, and agricultural land depletion when upscaling cellulose nanocrystal foam production from laboratory to industrial scale. Van der Hulst et al. (2020) reported an 83% reduction in greenhouse gas emissions for photovoltaic lamination when upscaling from laboratory to commercial scale. Similarly, Röder et al. (2022) found >95% impact reduction in the direct upscaling of a protein production process across several impact categories e.g., climate change (98.1%), ionising radiation (98.2%). Battiston et al. (2023) demonstrated that increasing alumina pellet production rates by fivefold reduced impacts such as climate change and mineral resource scarcity by over 90%. Tsalidis & Korevaar (2022) reported a ~99% reduction in wood torrefaction from bench to commercial scale, indicating an even greater reduction potential with at a larger scale difference. However, results from Arvidsson and Molander (2017) suggest that the effect of scale-up on emerging technologies may vary, as laboratory and pilot results for graphene production showed similar orders of magnitude in environmental impacts. These results emphasise the significance of additional design parameters, such as raw material selection, as seen in the work of Gavankar et al. (2015). At larger production rates, environmental impact reductions occurred from improved raw material efficiency achieved through measures such as reusing and recycling raw materials. Therefore, as the technology readiness level (TRL) increases, environmental impacts are reduced, with the extent of this reduction variable depending on the specific technology and areas of contribution (e.g., thermal efficiency, material efficiency).

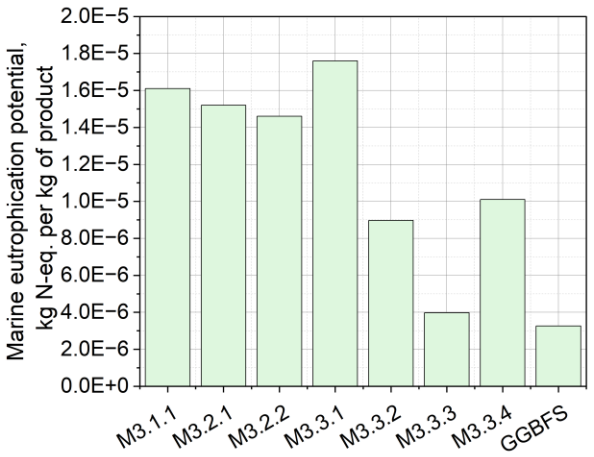
4.3.2 Effect of upscaling approach

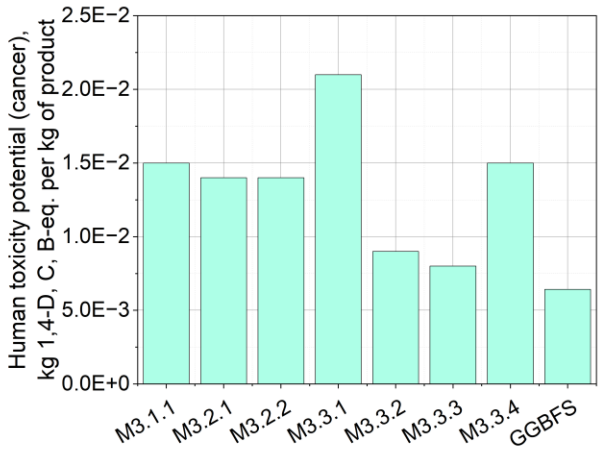
Figure 4-6 shows the impact scores for VBR production, determined across various environmental impact categories using different upscaling approaches, compared to GGBFS production (Figure 6, black connected squares). GGBFS and VBR are comparable SCMs. VBR has higher impact scores compared to GGBFS in categories such as climate change impact (0.34 kg CO_2-eq. emissions per kg product), terrestrial ecotoxicity (0.33 kg 1.4 D, C, B-eq. per kg product), and non-carcinogenic human toxicity (0.34 kg 1.4 D, C, B-eq. per kg product), primarily due to its greater energy demand (0.55-2.00 kWh per kg VBR compared to 0.095 kWh per kg GGBFS). For climate change, VBR showed a ~70% higher impact score than GGBFS. The relative climate change performance of VBR and GGBFS is expected to change in the future as the carbon footprint of the energy source (electricity) decreases (see Section 3.3).

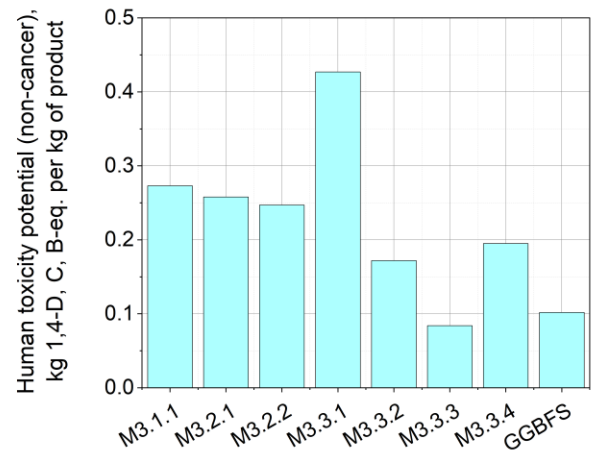
The upscaling approach significantly influences the environmental impact. Climate change impact (ranges from 0.48 to 0.85 kg CO_2-eq., with a variance of 2.80%), terrestrial ecotoxicity (ranges from 0.29 to 0.6 kg 1.4 D, C, B-eq., with a variance of 1.77%), and non-carcinogenic human toxicity (ranges from 0.084 to 0.4 kg 1.4 D, C, B-eq., with a variance of 0.96%) show the highest degree of variability, indicating substantial differences between the upscaling methods. Ozone depletion (ranges from 4.44×10^{-8} to 1.97×10^{-8} kg CFC-11-eq., with a variance of 8.15×10^{-11}%) and marine eutrophication (ranges from 4.00×10^{-6} to 1.62×10^{-5} kg N-eq., with a variance of 2.14×10^{-9}%) display the lowest variability, suggesting that the different upscaling approaches have a minor effect on these impacts. Other impact categories show moderate variability, with variances lower than 2.80×10^{-1} %. Overall, the effect of the upscaling approach is minor across most environmental impact categories since the associated variances are low to moderate. However, the relative environmental performance of VBR and BFS is sensitive to categories with higher variances, such as climate change (with a variance of 2.80%and terrestrial ecotoxicity (with a variance of 1.77%).

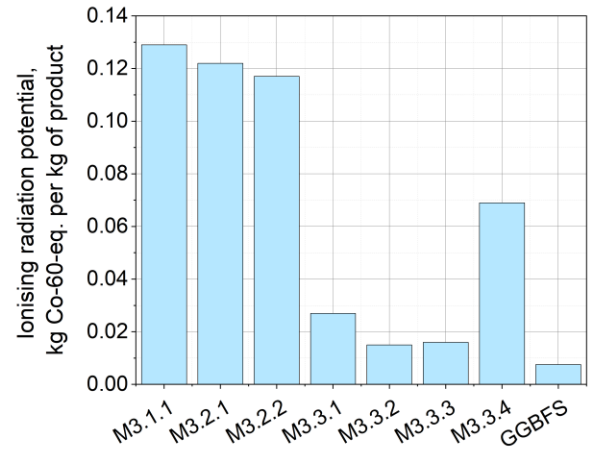

Terrestrial acidification potential, kg SO_2 eq. per kg of product
3.5E−3
3.0E−3
2.5E−3
2.0E−3
1.5E−3
1.0E−3
5.0E−4
0.0E+0
Global warming potential, kg CO_2 eq. per kg of product
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
Freshwater ecotoxicity potential, kg 1,4-D, C, B-eq. per kg of product
1E−2
8E−3
6E−3
4E−3
2E−3
0E+0
Marine ecotoxicity potential, kg 1,4-D, C, B-eq. per kg of product
1.4E−2
1.2E−2
1.0E−2
8.0E−3
6.0E−3
4.0E−3
2.0E−3
0.0E+0
Terrestrial ecotoxicity potential, kg 1,4-D, C, B-eq. per kg of product
1.0
0.8
0.6
0.4
0.2
0.0
Fossil fuel depletion potential, kg oil-eq. per kg of product
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
M3.1.1
M3.2.1
M3.2.2
M3.3.1
M3.3.2
M3.3.3
M3.3.4
GGBFS

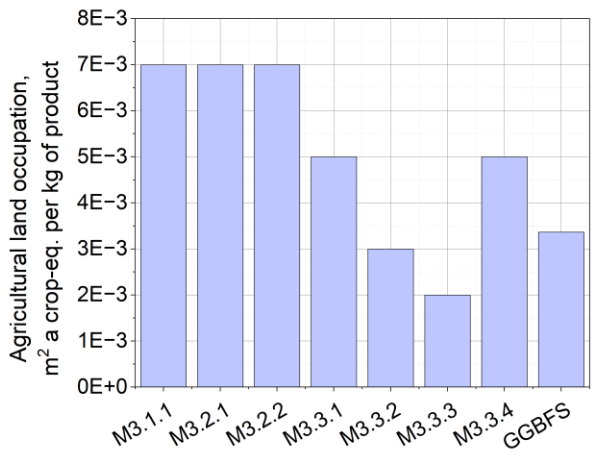

Freshwater eutrophication potential, kg P-eq. per kg of product
3.0E−3
2.5E−3
2.0E−3
1.5E−3
1.0E−3
5.0E−4
0.0E+0
M3.1.1
M3.2.1
M3.2.2
M3.3.1
M3.3.2
M3.3.3
M3.3.4
GGBFS


Marine eutrophication potential, kg N-eq. per kg of product
2.0E−5
1.8E−5
1.6E−5
1.4E−5
1.2E−5
1.0E−5
8.0E−6
6.0E−6
4.0E−6
2.0E−6
0.0E+0
M3.1.1
M3.2.1
M3.2.2
M3.3.1
M3.3.2
M3.3.3
M3.3.4
GGBFS


Human toxicity potential (cancer), kg 1,4-D, C, B-eq. per kg of product
2.5E−2
2.0E−2
1.5E−2
1.0E−2
5.0E−3
0.0E+0
M3.1.1
M3.2.1
M3.2.2
M3.3.1
M3.3.2
M3.3.3
M3.3.4
GGBFS


Human toxicity potential (non-cancer), kg 1,4-D, C, B-eq. per kg of product
0.5
0.4
0.3
0.2
0.1
0.0
M3.1.1
M3.2.1
M3.2.2
M3.3.1
M3.3.2
M3.3.3
M3.3.4
GGBFS


Ionising radiation potential, kg Co-60-eq. per kg of product
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
M3.1.1
M3.2.1
M3.2.2
M3.3.1
M3.3.2
M3.3.3
M3.3.4
GGBFS


Agricultural land occupation, m2 a crop-eq. per kg of product
8E−3
7E−3
6E−3
5E−3
4E−3
3E−3
2E−3
1E−3
0E+0
M3.1.1
M3.2.1
M3.2.2
M3.3.1
M3.3.2
M3.3.3
M3.3.4
GGBFS

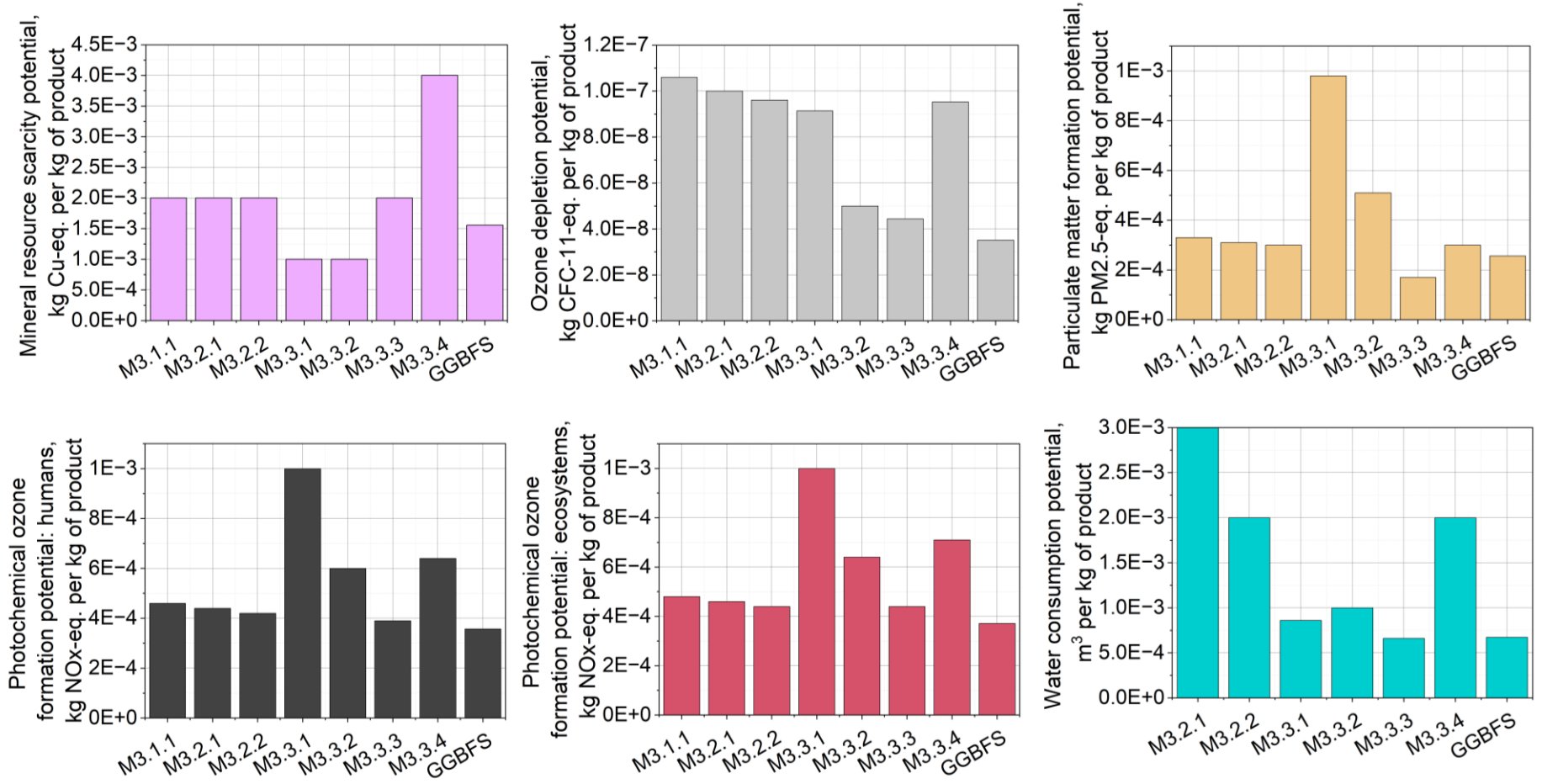


Figure 4-6: ReCiPe 2016 midpoint impact results (absolute) for BR vitrification across all upscaling scenarios. Individual bars represent the environmental impact of each upscaling scenario i.e., process simulation (M3.1.1), linear regression (M3.2.1), power regression (M3.2.2), clinker production modelled based on Centre et al. (2013), (M3.3.1), clinker production modelled based on Ecoinvent v.3.9.1. (2007a), (M3.3.2), ceramic vitrification modelled based on European Commission. (2007) (M3.3.1), ceramic vitrification modelled based on Ecoinvent v.3.9.1. (2007b), (M3.3.2), per kg of VBR. The GGBFS bar is used as a benchmark for comparison. Abbreviations for ReCiPe (H) impact categories are reported in section 2.2.1. Quantitative results for each individual upscaling scenario are tabulated in the Appendix A (Table A10).

Industrial upscaling resulted in climate change impacts ranging between 0.45 and 0.78 kg CO_2-eq. emissions per kg VBR (Figure *4-7*). Proxy models caused the upper bound, specifically ceramic vitrification (ecoinvent v.3.9.1 "ceramic tile production - RoW - ceramic tile" with 0.78 kg CO_2-eq. emissions per kg ceramic tile) and clinker production (modelled according to Centre et al., 2013, with 0.71 kg CO_2-eq. emissions per kg clinker). Despite the ceramic vitrification technology having double the energy demand per kg of product than clinker (see Table 4-2), the use of natural gas as its primary energy source instead of pet coke (the primary energy source in clinker production) yields comparable climate change impacts for the two technologies. The data source and modelling assumptions, particularly the power source, significantly influence the environmental impact during upscaling. This finding is consistent with LCA studies on BR recycling (Joyce & Björklund, 2019; Joyce et al., 2018). The other scenarios (proxy technology ecoinvent v3.9.1 "clinker production - Europe without Switzerland", (M3.3.2), and ceramic tile production, (M3.3.3) (European Commission, 2007)), regression analysis [linear (M3.2.1), power law (M3.2.2)], and process modelling (M3.1.1)) resulted in less variable climate change impact, ranging between 0.45 and 0.54 kg of CO_2-eq. emissions per kg of VBR. These results show the importance of selecting a suitable proxy technology in ex-ante LCA studies, particularly concerning key modelling assumptions i.e., power source.

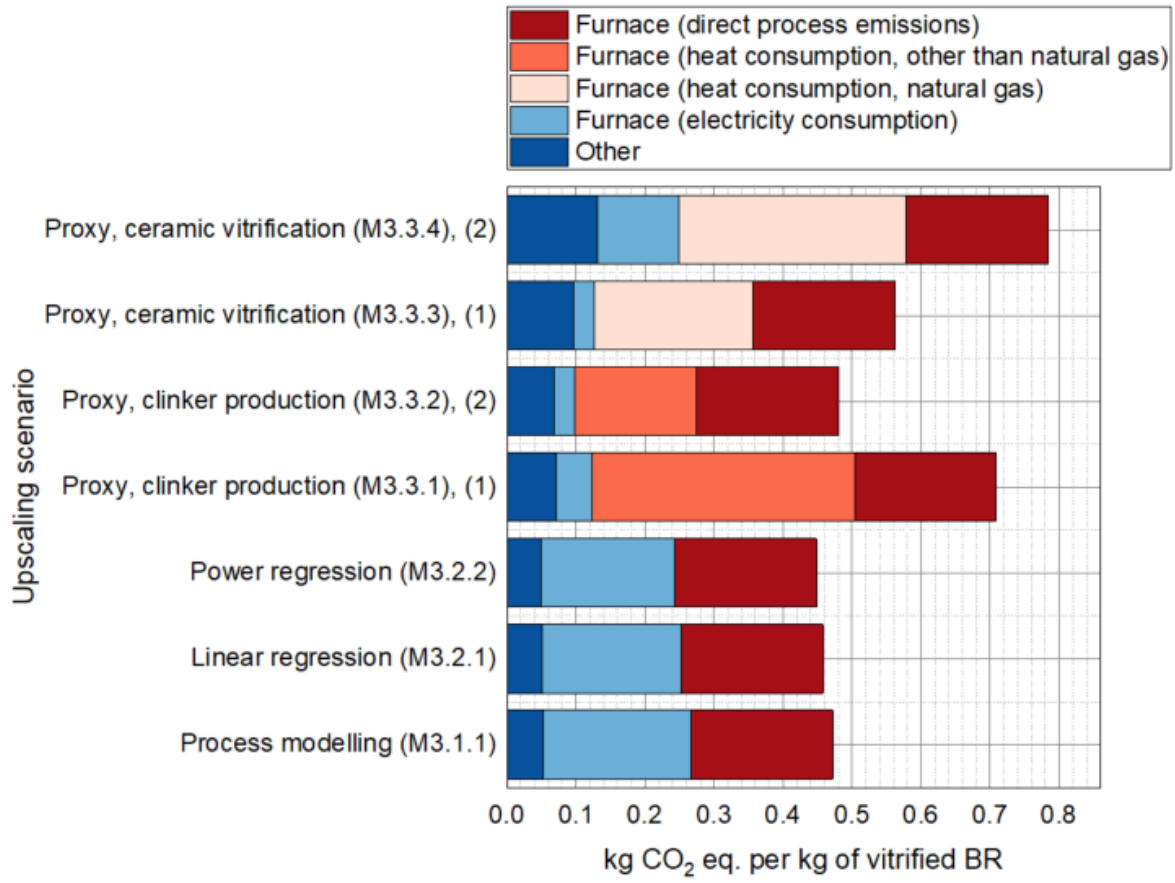


Figure 4-7: Contribution analysis showing CO_2-eq. emissions of VBR processed at an industrial scale based on process modelling, regression analysis, and proxy technology. "Other" represents the impact of upstream, background activities such as hard coal preparation and petroleum/natural gas production. Quantitative results are tabulated in Appendix A (Table A11).

In the case that appropriate proxy technologies are chosen (M3.3.2, and M3.3.3), the results are comparable to those obtained via regression analysis (M3.2.1, and M3.2.2) and process modelling (M3.1.1). This finding is notable since process modelling is significantly more complex than proxy technology and regression analysis approaches. A key benefit of using process modelling for ex-ante LCA is its higher physical significance and thus reliability from its use of thermodynamic data, chemical reactions, and material properties. Proxy technology and regression analysis are beneficial in ex-ante LCA studies where time, budgets, data availability, and experience and expertise in process modelling are limited (Buyle et al., 2021; Alexandra C. Hetherington et al., 2014).

In systems where environmental impacts primarily arise from contributions other than energy demand, e.g., waste flows and raw materials, process modelling may lead to more reliable

results than proxy technology and regression analysis approaches, as it more accurately models energy and material flows by considering the thermodynamic and chemical reactions of the process. For instance, increasing the recycling rate of ethanol (to 98%) in the upscaling of cellulose nanocrystal foam production reduced greenhouse gas emissions by 11% by lowering ethanol demand (Tan et al., 2018). Further research is needed to test the hypothesis that generic upscaling relationships can be developed for emerging technologies, such as bauxite residue (BR) vitrification, where primary hotspots are energy related. Investigating the feasibility of developing such relationships could lead to widely adopted and streamlined upscaling methods applicable across various industries. For other technologies, where impacts stem from raw materials or waste flows, a deeper understanding of the effects of upscaling is needed.

4.3.3 Prospective evaluation of future decarbonisation potential in the entire value chain of VBR for cementitious applications

At the industrial scale (upscaled with process modelling), electricity consumption was the primary hotspot in BR vitrification causing ~52% of the total climate change impact. Direct emissions accounted for ~43% of the total climate change impact due to the contribution of coke and limestone calcination, while 5% was associated with other background activities. Hence, a sensitivity analysis was performed to analyse the effects of decarbonising future electricity generation on the environmental impacts of VBR production.

In 2020, VBR underperformed (i.e., had higher impact category indicator scores) compared to GGBFS across all impact categories, except terrestrial ecotoxicity (Table 4-3). However, by 2050, its relative performance improved considerably under the SSP2 RCP1.9 scenario, benefiting from the transition to more renewable energy sources. In 2050, VBR underperformed GGBFS in 10 out of 18 impact categories (including climate change impact), achieved similar performance in 4 categories, and performed better in 4 impact categories. However, the transition of the iron and steel industry from traditional blast furnaces and basic oxygen furnaces to electric arc furnaces or submerged arc furnaces, powered increasingly by natural gas, hydrogen, and renewable energy sources is expected to reduce GGBFS supply. This transition may lead to higher costs for GGBFS, accelerating the need for an emerging, suitable, and cost-effective SCM (Alberici et al., 2017; Andreas Ehrenberg, 2023).

Table 4-3: ReCiPe 2016 midpoint impact results of industrial scale vitrification technology (upscaled with process modelling) in 2020 and with electricity scenarios in 2050 (i.e., SSP2 RCP1.9, SSP2 RCP2.6, and SSP2 Baseline). Bold values indicate the lowest impact when comparing the SSP2 scenarios. Underlined values indicate the highest impact when comparing the SSP2 scenarios. Italic values show the lowest impact between VBR and GGBFS, both with the SSP2 RCP 1.9 scenario.

		2020		**2050**			**2050 GGBFS**
Impact categories	**Units**	**SSP2 default**	**GGBFS default**	**SSP2 baseline**	**SSP2 RCP 2.9**	**SSP2 RCP 1.9**	**SSP2 RCP 1.9**
Terrestrial acidification potential (TAP)	kg SO_2-eq.	1.11×10^{-3}	5.90×10^{-4}	$\underline{1.36 \times 10^{-3}}$	3.75×10^{-4}	**3.01×10^{-4}**	3.59×10^{-4}
Global warming potential (GWP100) [d]	kg CO_2-eq.	5.30×10^{-1}	1.30×10^{-1}	$\underline{5.80 \times 10^{-1}}$	3.80×10^{-1}	**3.58×10^{-1}**	7.10×10^{-2}
Freshwater ecotoxicity potential (FETP)	kg 1,4-D, C, B-eq.	1.12×10^{-2}	4.92×10^{-3}	$\underline{1.56 \times 10^{-2}}$	7.12×10^{-3}	**6.70×10^{-3}**	5.10×10^{-3}
Marine ecotoxicity potential (METP)	kg 1,4-D, C, B-eq.	1.51×10^{-2}	8.45×10^{-3}	$\underline{2.10 \times 10^{-2}}$	9.13×10^{-3}	**8.60×10^{-3}**	8.20×10^{-3}
Terrestrial ecotoxicity potential (TETP)	kg 1,4-D, C, B-eq.	$\underline{6.75 \times 10^{-1}}$	3.95	**4.86×10^{-1}**	5.80×10^{-1}	6.44×10^{-1}	3.22
Fossil fuel potential (FFP)	kg oil-eq.	8.87×10^{-2}	3.67×10^{-2}	$\underline{1.09 \times 10^{-1}}$	4.48×10^{-2}	**4.10×10^{-2}**	2.10×10^{-2}

		2020		2050			2050 GGBFS
Impact categories	**Units**	**SSP2 default**	**GGBFS default**	**SSP2 baseline**	**SSP2 RCP 2.9**	**SSP2 RCP 1.9**	**SSP2 RCP 1.9**
Freshwater eutrophication potential (FEP)	kg P-eq.	2.76×10^{-4}	3.27×10^{-5}	4.43×10^{-4}	3.24×10^{-5}	**1.73×10^{-5}**	6.80×10^{-6}
Marine eutrophication potential (MEP)	kg N-eq.	2.11×10^{-5}	3.17×10^{-6}	2.89×10^{-5}	4.42×10^{-6}	**3.91×10^{-6}**	1.29×10^{-6}
Human toxicity potential (HTPc)	kg 1,4-D, C, B-eq.	1.91×10^{-2}	7.16×10^{-3}	2.79×10^{-2}	8.40×10^{-3}	**7.30×10^{-3}**	5.00×10^{-3}
Human toxicity potential (HTPnc)	kg 1,4-D, C, B-eq.	3.68×10^{-1}	1.27×10^{-1}	5.21×10^{-1}	9.88×10^{-2}	**8.60×10^{-2}**	9.60×10^{-2}
Ionising radiation potential (IRP)	kg Co-60-eq.	1.18×10^{-1}	7.51×10^{-3}	**1.75×10^{-2}**	3.47×10^{-2}	5.38×10^{-2}	7.80×10^{-3}
Agricultural land occupation (LOP)	m^2 a crop-eq.	4.70×10^{-3}	3.39×10^{-3}	**3.10×10^{-3}**	3.80×10^{-3}	5.20×10^{-3}	5.20×10^{-3}
Material resources: metals/minerals | surplus ore potential (SOP)	kg Cu-eq.	1.65×10^{-3}	1.51×10^{-3}	**1.60×10^{-3}**	2.17×10^{-3}	2.15×10^{-3}	1.67×10^{-3}

		2020		2050			2050 GGBFS
Impact categories	**Units**	**SSP2 default**	**GGBFS default**	**SSP2 baseline**	**SSP2 RCP 2.9**	**SSP2 RCP 1.9**	**SSP2 RCP 1.9**
Ozone depletion potential (ODPinfinite)[e]	kg CFC-11-eq.	8.10×10^{-8}	3.51×10^{-8}	9.08×10^{-8}	3.62×10^{-8}	**3.26×10^{-8}**	1.62×10^{-8}
Particulate matter formation potential (PMFP)[f]	kg PM2.5-eq.	3.57×10^{-4}	2.57×10^{-4}	4.32×10^{-4}	1.24×10^{-4}	**1.01×10^{-4}**	1.29×10^{-4}
Photochemical oxidant formation potential: humans (HOFP)	kg NO_x-eq.	5.15×10^{-4}	3.57×10^{-4}	4.93×10^{-4}	2.31×10^{-4}	**2.20×10^{-4}**	2.20×10^{-4}
Photochemical oxidant formation potential: ecosystems (EOFP)	kg NO_x-eq.	5.35×10^{-4}	3.71×10^{-4}	5.12×10^{-4}	2.47×10^{-4}	**2.36×10^{-4}**	2.36×10^{-4}
Water consumption potential (WCP)	m^3	2.77×10^{-3}	6.72×10^{-4}	2.19×10^{-3}	1.98×10^{-3}	**1.87×10^{-3}**	1.87×10^{-3}

[d] Reported using global warming potential over a 100-year horizon (GWP100).

[e] Measured in kg CFC-11-eq.; Chlorofluorocarbons (CFCs)

[f] Measured in kg PM2.5-eq.; Particulate matter (PM)

SSP2 RCP scenarios (i.e., 2.6, and 1.9) both led to reductions in impact category indicator values (Table 4-3). A 28% and 33% reduction was observed in climate change impact, due to decreased fossil fuel (i.e., lignite) consumption in SSP2 RCP scenarios (2.6 and 1.9 respectively), minimising the CO_2 emissions released during combustion. This result aligns with Cossutta et al. (2017) and Salas et al. (2018), who reported a ~80% reduction in climate change impact when transitioning to a more renewable energy grid for graphene production. In 2050, the SSP2 baseline scenario negatively impacts most categories, primarily due to the absence of climate policies and mitigation measures. This scenario leads to a ~65% increase in lignite and hard coal consumption as primary fuels for electricity generation compared to 2020, resulting in higher environmental impacts than 2020 and SSP2 RCP scenarios.

In 2050, VBR production in the SSP2 RCP1.9 scenario has lower environmental impacts in fourteen of the eighteen impact categories compared to those for the SSP2 baseline and SSP2 RCP 2.6 scenarios (Table 4-3).

Transitioning to more renewable energy sources increased impact scores in four categories: terrestrial ecotoxicity, ionising radiation, agricultural land use, and material consumption. This is caused by higher demand for biomass (~30%) and nuclear energy (~60%) in the SSP2 RCP 1.9 electricity scenario, causing increased agricultural land occupation and ionising radiation impacts, respectively. The higher impact on mineral resources and terrestrial ecotoxicity was associated with increased demand for the production, distribution, and storage of renewable energy (e.g., infrastructure development for wind turbines) and transportation (brake wear emissions) of alternative fuels, i.e., biomass. This shows the complex interconnections between different impact categories (e.g., terrestrial ecotoxicity, ionising radiation, agricultural land use, and material consumption) and how the choice of power source affects each category differently, leading to varied overall outcomes.

4.4 Conclusions

This study assessed the environmental impacts of a novel BR vitrification technology to produce an SCM. Results showed significant reductions in environmental impacts at pilot (~83%) and industrial (>94%) scales compared to the laboratory scale. Overall, the effect of the upscaling across most environmental impact categories showed low to moderate variances. Proxy scenarios with considerably higher energy demands and fossil fuels power systems caused large variances in impact categories such as climate change and terrestrial ecotoxicity, highlighting the critical importance of modelling assumptions and data quality. If appropriate proxy technology was selected, proxy technology and regression analysis were found comparable to process modelling. These approaches are thus suitable for early-stage assessment and environmental hotspot analysis, particularly when time constraints, limited data availability, and a lack of expertise prevent the use of process modelling.

Incorporating future changes to the electricity mix in the industrial scale BR vitrification system significantly reduced impact category indicator values (up to 94% for freshwater eutrophication), emphasising the importance of background considerations in ex-ante LCA studies of BR recycling. These changes also demonstrated the burden-shifting potential related to impact categories hindered by a lower carbon electricity mix, such as the 30% increase in agricultural land occupation from biomass-based energy generation. Hence, holistic evaluations must consider the complex interactions between key processing parameters and their influence on environmental performance more broadly than climate change impacts.

Chapter 5. Prospective LCA of European cement production

This section outlines the development of an LCA model incorporating decarbonisation measures for Portland cement production within a European context. Existing literature gaps are identified, followed by a detailed description of the developed LCI and the various decarbonisation scenarios studied. Finally, the results are presented and discussed, evaluating the prospective impacts of conventional European cement production by 2050, considering established and emerging technologies. This study has been peer-reviewed and published in Resources, Conservation and Recycling (Georgiades et al., 2023).

5.1 Background and motivation

Cement production, a major contributor to global anthropogenic CO_2 emissions, accounts for ~7-8% of the total due to the inherent process emissions from raw materials and fossil fuel combustion (Mikulčić et al., 2016). The cement industry faces an urgent need for decarbonisation to align with international climate targets. For instance, to be in line with the Paris Agreement, the International Energy Agency (IEA) has set a goal to reduce CO_2 emissions from cement production by 24% compared to pre-industrial levels, aiming to limit global temperature rise to 2°C by 2100 (IEA, 2018). Furthermore, the European Commission's 'Green Deal' proposes an even more ambitious objective for Europe to achieve carbon neutrality by 2050, which would limit the global surface temperature rise to 1.5°C by the end of the century (Cembureau, 2020).

Meeting these climate targets necessitates a faster pace of CO_2 reduction than the historic trend of approximately 0.4% per year observed between 2005 and 2018. To this end, the industry must explore and evaluate a range of measures such as replacing clinker with supplementary cementitious materials (SCMs) i.e., coal fly ash and blast furnace slag, switching to alternative fuels such as biomass and refuse-derived fuel (RDF) instead of petroleum coke and hard coal, enhancing kiln configurations to improve production efficiency, and implementing carbon capture and storage (CCS) technologies (Mikulčić et al., 2016).

Assessing these measures establishes benchmarks for the future implementation of emerging technologies aimed at decarbonising cement production. This approach ensures that new technologies are evaluated against projected benchmarks rather than current emission levels, providing a realistic measure of their potential impact and avoiding overestimation.

Many LCA studies have investigated the climate change impact caused by clinker replacement with SCMs such as blast furnace slag, coal fly ash, metakaolin, limestone, rice husk ash, and volcanic ash. These studies have shown that composite PC containing such SCMs can have significantly reduced CO_2-eq. emissions footprints (reductions of 15-88% at clinker-to-cement ratios of 0.85-0.1) relative to ordinary PC (Miller, 2018; Rahla et al., 2019; Arrigoni et al., 2020). The major CO_2-eq. emissions reduction effect is due to reducing PC clinker content, which decreases CO_2 emissions from both calcination and fuel consumption.

The use of alternative fuels in cement kilns is reported to have a potential to reduce CO_2 emissions from cement production by 20-30% [including refuse- and tyre-derived fuels (Dziok et al., 2021; Georgiopoulou and Lyberatos, 2018), sewage sludge ((Dziok et al., 2021; Husillos Rodríguez et al., 2013)), non-recycled plastics and paper (Thanos Bourtsalas et al., 2018), and natural gas (Thwe et al., 2021)]. Incorporating waste fuels in the mix (e.g., waste tires, agricultural waste, solid waste, and oil waste) utilises otherwise lost energy (from fuels being landfilled) and reduces CO_2-eq. emissions associated with fossil fuel mining, transportation, and combustion (ICF International, 2008; Mokrzycki et al., 2003).

Energy use in cement production can be reduced in a variety of ways, leading to CO_2 emissions reductions. Pyro-processing heat losses can be as much as 20% of energy inputs and cause 8% CO_2 emissions of cement production (Engin and Ari, 2005), heat integration can provide up to 35% of cement plant thermal energy requirements (Khurana et al., 2002), and using higher thermal efficiency kiln configurations such as shifting from a wet process to a dry process with a precalciner, irrespective of the state of the cement plant (new or retrofitted), can reduce energy use by 50% and CO_2 emissions by 20% (IEA, 2018, 2021). The extent of CO_2 emissions reduction depends highly on the changes in process equipment and changing the cement kiln configuration has a reported potential to reduce CO_2 emissions by 15-20%.

Carbon capture and storage (CCS) is a versatile technology that is applicable in various industries (iron and steel, cement, petrochemicals, etc.) (Kuramochi et al., 2012). CCS aims to

capture process-based CO_2 emissions prior or after fuel combustion (pre- and post- combustion technologies, respectively). Calcium looping is a type of CCS technology in which the CO_2 in the flue gas is removed using a calcium oxide sorbent to form a high purity, concentrated CO_2 stream. (Colelli et al., 2022; Tilak and El-Halwagi, 2018). LCA studies investigating CCS technologies such as oxy-fuel and post-combustion (i.e., amine scrubbing) show a wide range of CO_2-eq. emissions reduction potentials (15-78%) that are affected by several parameters (i.e., CO_2 capture technology, CO_2 capture efficiency energy consumption, and energy source) (García-Gusano et al., 2015; Hills et al., 2016; Miller et al., 2018; Volkart et al., 2013). These parameters include capture efficiency (e.g., 85%), technology type (e.g., post-combustion), and fuel source (e.g., natural gas) (An et al., 2019; Chen and Yang, 2021; Wunderlich et al., 2021). Importantly, CCS has only been tested in pilot and demonstration plant scales (International Association of Oil & Gas producers, 2022) i.e., in the Norcem cement plant in Brevik, Norway (Bjerge & Brevik, 2014) and in the HeidelbergCement plant in Lixhe, Belgium (LEILAC, 2017), where the energy source powering the process was highlighted as a key parameter affecting the CO_2 emissions mitigation potential. The follow-on project in HeidelbergCement aims to accelerate the implementation of CCS at the commercial scale, by testing the performance of a demonstration plant to capture 20% of the CO_2 emissions from a conventional cement plant (LEILAC 2, 2021). Other recent projects involving CCS include e.g., Cleankerk in Buzzi Unicem cement plant in Vernasca, Italy (Fantini et al., 2020), Lighthouse Carboneras in the LafargeHolcim cement plant in Almeria, Spain (Carbon Clean, 2020), Carbon2Business in the LafargeHolcim Lägerdorf cement plant in Germany (Holcim, 2020), ANRAV in the HeidelbergCement Devnya cement plant in Bulgaria (HeidelbergCement, 2022), and a full-scale CCS system by HeidelbergCement and Cementa in the Slite cement plant in Sweden, which has a target to capture the plant's total process CO_2 emissions (1.8 Mt CO_2 per year) by 2030 (CO_2-eq. emissions arising from other life cycle stages are not included in this target) (HeidelbergCement, 2021).

LCA studies that aim to assess the environmental performance of product systems in the future, which is important for emerging technologies on which decarbonisation measures are based, should use a prospective LCA approach. Prospective LCA accounts for the upscaling of emerging technologies to industrial application and or an existing technology to a future state where changes may have occurred on both the specific technology under study (i.e., focusing on existing PC production) as well as the wider technological surroundings (i.e., electricity

mix). If these changes are not captured, the reliability of the results may be compromised (Miller and Keoleian, 2015; Villares et al., 2017) and limit their application to analyse future scenarios. The main challenge in a prospective LCA study, however, is dealing with the epistemological uncertainty (due to lack of case specific data) which is inherent in future models (Joyce and Björklund, 2019). To overcome this uncertainty, it has been suggested to use future scenarios that describe plausible future states of the product system(s) under analysis (Mendoza Beltran et al., 2020; Pesonen et al., 2000; Spielmann et al., 2005; Villares et al., 2017). Future scenarios should be developed based on direct (foreground) and indirect (background) system parameters for case studies where indirect parameters are deemed important. Examples of the system parameters considered in this study are the kiln type (foreground) and electricity mix (background).

Few studies have assessed future life cycle environmental impacts of cement production in combination with multiple decarbonisation measures and the prospective LCA approach. Among the relevant studies published, Yao et al. (2020) assessed the environmental performance of a 3D-printed geopolymer concrete panel compared to a conventional PC concrete panel. Maes et al., (2021) studied an emerging electro-mass separation technology to separate fly ash into several fractions and use the resulting product as a substitute for silica fume in ultra-high-performance concrete. This electro-mass separation technology was environmentally assessed at the pilot and the industrial scale to evaluate the environmental benefits of separating fly ash prior to its use. The future environmental assessment considered background changes by investigating the influence of changes to the electricity mix, based on projections from the European Commission on energy, transport and greenhouse gas (GHG) emissions. The influence of electricity mix (background changes) and scale-up drastically reduced the life cycle CO_2 emissions of the emerging electro-mass separation technology (~ –4,000 kg CO_2 eq.) compared to the incumbent (~ –1,000 kg CO_2 eq.) method (direct use of Class F coal fly ash in cement). Prospective LCA has not yet been used to investigate the influence of direct (foreground) and indirect (background) system parameters on the CO_2-eq. emissions of cement production.

In this chapter, prospective LCA is applied to comprehensively investigate the climate change impacts of decarbonisation measures for cement production, including the influence of both foreground and background (decarbonisation in electricity production) system parameters. Modelling the effects that decarbonising electricity generation has on the CO_2-eq. emissions

footprint of cement production, including kiln electrification, is important given the growing interest in this topic (Ellis et al., 2020; GCCA, 2021). The focus is on the European cement production and greenhouse gas emissions-induced climate change impacts until 2050. Six foreground scenarios were developed to assess important cement-related developments (i.e., clinker substitution, alternative fuels, kiln improvements, and carbon capture and storage) but also include prospective life cycle inventory background scenarios (i.e., electricity mix) that indirectly affect cement production, and that have been developed based on broad and global storylines of socio-economic pathways (SSPs, i.e., SSP2).

5.2 Methodology

5.2.1 Goal and scope

The primary goal is to investigate the potential changes in greenhouse gas emissions induced climate change impacts (in terms of CO_2-eq. emissions over a 100-year horizon) from incorporating decarbonisation measures into cement production until 2050. These interventions include: (i) PC clinker substitution with SCMs, (ii) increased use of alternative fuels in cement production, (iii) improvements in kiln efficiency, and (iv) implementing CCS. This analysis is intended to benchmark the reduction potential of these decarbonisation measures and serve as a reference point for assessing emerging technologies and cementitious materials. By providing a clearer picture of how the environmental performance of conventional cement may evolve, the study supports later evaluations in this thesis (see Chapter 6) that explore whether and under what conditions new technologies could offer environmental benefits.

For (i) PC clinker substitution with SCMs, evaluation was performed solely on the influence of the SCM upstream/pre-treatment processing on the climate change impact and no physical properties (e.g., compressive strength) were considered. The functional unit is 1 kg cement produced at the factory gate (Figure *5-1*). It is important to note that this study only considers the climate change impact category, as this is the primary environmental concern for the cement industry and the main area influenced by the selected decarbonisation measures. Additional research is needed to evaluate the potential impacts on other impact categories e.g., land use (further details can be found in Appendix B, section B2.2.).

A functional unit incorporating physical properties (i.e., compressive strength) in terms of 1 kg binder per m^3.MPa concrete, as defined by Damineli et al. (2010)as the binder intensity, could result in a closer comparison between the various blended cement types studied. This addition to the functional unit enables blended cement to be assessed in terms of both mechanical and environmental performance. However, depending on the strength class desired and type of SCM used, the clinker-to-cement ratio can vary significantly. This makes it difficult to analyse the effect of changing the clinker-to-cement ratio on the CO_2-eq. emissions footprint of cement. Hence, the focus is on analysing this effect of changing the clinker-to-cement ratio using relatively simple single SCM systems, leaving comparisons of more complex composite PCs for future work. It is noted that composite PCs can have both relatively low clinker-to-cement ratios and comparable or faster compressive strength development than ordinary PC, e.g., limestone calcined clay cement (clinker-to-cement ratio = 0.5) (Maraghechi et al., 2018).

A cradle-to-gate system boundary is used that includes raw material extraction, raw material preparation, clinker production, cement formulation, and transportation among the foreground activities. The system accounts for background changes based on the Shared Socioeconomic Pathway 2 (SSP2) scenarios (Baseline and RCP 2.6, see Section 5.2.3), which reflect shifts in energy systems. Europe was selected as the geographical scope due to the high availability of data for this region, enabling this work to become a reliable reference point for future prospective LCA studies of cement production in other regions including globally. This also aligns with the significant interest and policy momentum toward decarbonisation in Europe, along with the growing potential for BR and other SCM utilisation in the European cement market. This choice of scope does not capture more local conditions, e.g., governmental regulations, electricity mix, market size, raw material availability and logistics in each cement plant, which makes this work generic rather than specific to individual production sites. While this approach provides a useful regional benchmark, future work should explore localised case studies, as cement production practices and background conditions (e.g., electricity grids, kiln technologies, SCM availability) can vary substantially by location.

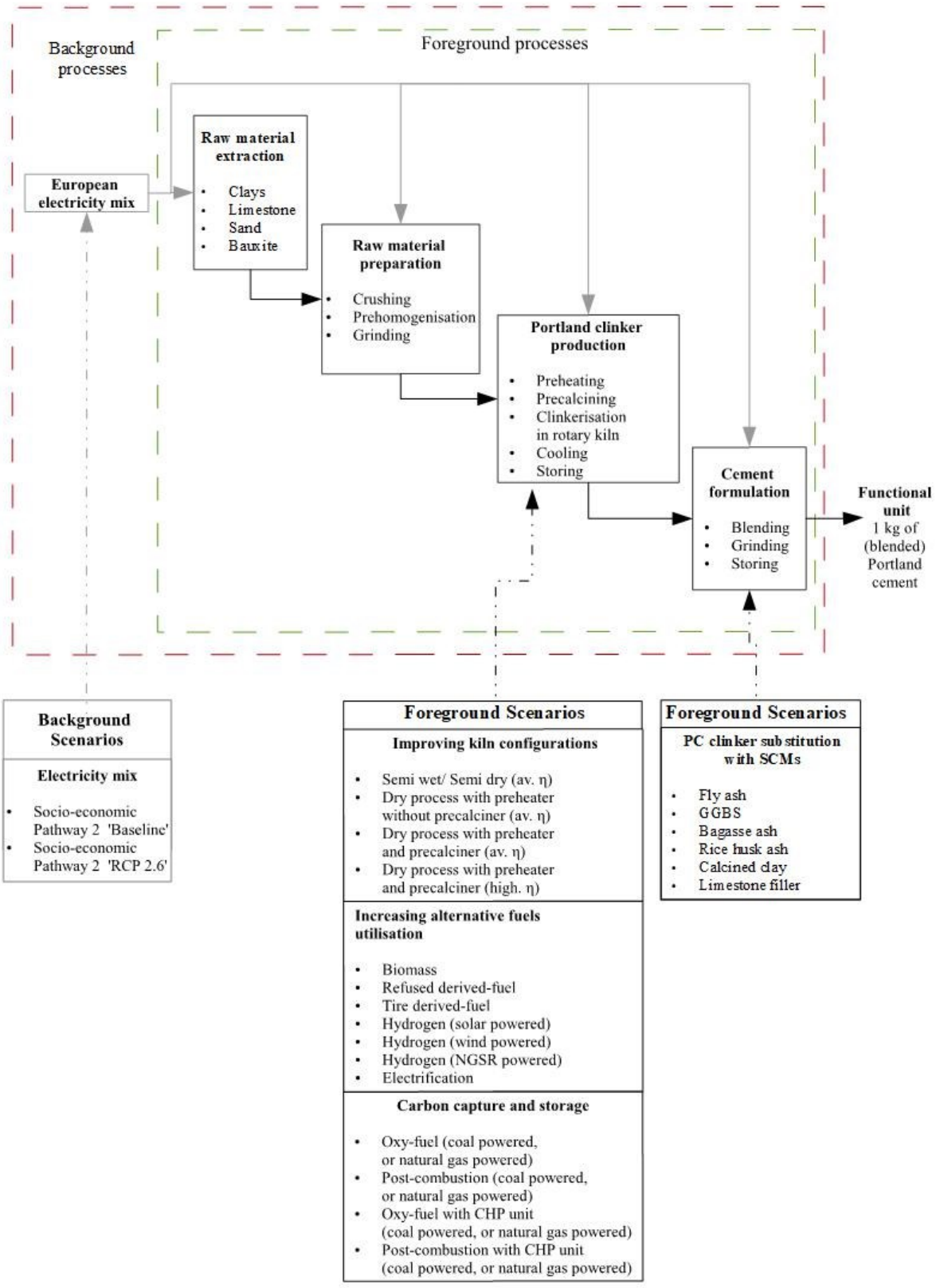


Figure 5-1: System boundary of this prospective LCA study, including foreground changes on (i) SCMs, (ii) alternative fuels, (iii) kiln configurations with higher efficiency (η) and (iv) CCS. Background changes are included based on the SSP2 pathways (Baseline and RCP 2.6, see Section 2.3).

Climate change impacts were analysed based on the 100-year global warming potential (GWP), using the IPCC 2013 method (Tun et al., 2020). It should be noted that the IPCC 2013 method was the most current version available considering the software version at the time of this analysis. Activity Browser software (version 2.6.3) and its implementation of the superstructure approach was used to carry out this prospective LCA study (Steubing et al., 2020; Steubing and de Koning, 2021).

Results from the existing literature are used in a non-harmonized manner to compare to the results of this study, meaning that they do not use the same methodological choices. For example, some studies use the IPCC impact assessment method (e.g., Arrigoni et al., 2020; Miller et al., 2018), whereas others use CML 2001 (i.e., Rahla, Mateus and Bragança, 2019).

5.2.2 Foreground scenarios

The foreground system describes PC production and comprises four steps: (1) raw material extraction, (2) raw material preparation, (3) clinker production, and (4) cement formulation (Figure *5-1*). The raw materials for clinker production are limestone and a range of materials in smaller quantities including clays, sand, bottom ashes, and bauxite to make up the required feedstock composition. The primary raw materials are quarried and then transported to the cement plant, where they are crushed, milled, and/or mixed to form a homogenous fine powder. Depending on the kiln configuration, the fine powder is processed through a series of heat exchangers, heating the material up to ~900 °C during which limestone is decomposed to lime, releasing CO_2 emissions. This calcined material is then fired in a kiln to high temperatures (~1,450 °C) to produce PC clinker. The produced PC clinker exits the kiln and is rapidly cooled before being further processed into cement, which is typically performed by milling and mixing the material with ~5 mass% gypsum (and sometimes additional materials e.g., limestone, quicklime depending on the final cement properties) to form ordinary PC (IEA, 2018). In Europe, ordinary PC is standardised as CEM I (ISO EN 197-1, 2011). Secondary raw materials such as blast furnace slag, coal fly ash, limestone can be blended to replace a fraction of clinker in the produced cement.

Six (1-3, ‘individual’; 4-6, ‘synergistic’) foreground scenarios were developed for CEM I production that correspond to the implementation of decarbonisation measures. Both individual

and synergistic scenarios were developed using both International Energy Agency (IEA) and European Cement Association (CEMBUREAU) roadmaps (Cembureau, 2020; IEA, 2018). The parameters and values used in these scenarios are shown in Appendix B. The scenarios are:

1. PC clinker substitution with SCMs (Section 5.2.2.1)
2. Increasing alternative fuels utilisation (Section 5.2.2.2)
3. Improving kiln configurations (Section 5.2.2.3)
4. Effect of allocation type: synergistic scenario incorporating all three of these individual decarbonisation measures, based on IEA and CEMBUREAU roadmaps investigating the influence of the allocation approach taken, to allocate the upstream impacts in cases of by-product production (none, and economic) (Section 5.2.2.4)
5. Effect of technology roadmap: synergistic scenarios incorporating all three of these individual decarbonisation measures, based on IEA and CEMBUREAU roadmaps compared to GCCA roadmap (Section 5.2.2.4)
6. Effect of carbon capture and storage: synergistic scenario incorporating all three of these individual decarbonisation measures, based on IEA and CEMBUREAU roadmaps, and additionally CCS (Section 5.2.2.4).

These six scenarios were developed according to projections in International Energy Agency (IEA) and European Cement Association (CEMBUREAU) roadmaps (Cembureau, 2020; IEA, 2018), specifically following changes in the clinker-to-cement ratio, alternative fuels in the kiln fuel mix, and kiln efficiency improvements to 2050. The IEA cement roadmap presents a realistic transition pathway for cement production to achieve a 24% reduction of direct CO_2 emissions by 2050, which is consistent with the Paris Agreement target of limiting the global temperature rise to less than 2 °C above pre-industrial levels (1980) (UNFCCC, 2015). The CEMBUREAU roadmap promotes a more ambitious pathway for the European cement industry targeting the objectives of the Green Deal to contribute to limiting the global temperature rise to 1.5 °C by 2050, with a 63% direct CO_2 emissions reduction for cement production by 2050. The influence of the technology roadmap on climate change impact was determined through a fifth synergistic scenario (i.e., incorporating PC clinker substitution, alternative fuels, and improved kiln configurations), which was developed according to the GCCA roadmap, to compare against the IEA and CEMBUREAU roadmaps. The GCCA roadmap sets out a pathway to achieve net zero CO_2 emissions in cement production by 2050,

with an intermediate target of 25% CO_2 emissions reduction by 2030 (Cembureau, 2020). Similarly, to the CEMBUREAU roadmap, the GCCA roadmap (GCCA, 2021) is based on limiting the global mean surface temperature rise to 1.5 °C above pre-industrial levels. These roadmaps use the same decarbonisation measures (PC substitution with SCMs, improving kiln type configurations, and implementing CCS). A sixth synergistic foreground scenario was developed based on the IEA and CEMBUREAU roadmaps that includes CCS in addition to the other individual decarbonisation measures (clinker substitution, alternative fuels, improved kiln configurations).

The inventory analysis input data for CEM I production were extracted from the environmental product declaration tool by the Global Cement and Concrete Association (GCCA) (WBCSD, 2016), which is based on data from the Getting the Numbers Right (GNR) database (Dauriat et al., 2019). The scope of the GCCA environmental product declaration tool data includes raw material extraction and preparation, transportation, clinker production, and cement formulation. Input data associated with pre-treatment of SCMs, and alternative fuels production were extracted from the existing literature [ground granulated blast furnace slag, coal fly ash (Gursel, 2014), rice husk ash, bagasse ash (Jittin et al., 2021), calcined clay (Salvi Malacarne et al., 2021), hydrogen production (Dufour et al., 2011; Cetinkaya, Dincer and Naterer, 2012), sewage sludge (Dufour et al., 2011), tyre-derived fuel (Gursel, 2014), refused-derived fuel Grzesik and Malinowski, 2016), and biomass (Sgarbossa et al., 2020).

5.2.2.1 PC clinker substitution with supplementary cementitious materials

The PC clinker substitution with SCMs foreground scenario models substituting PC clinker with primary and secondary cementitious materials such as industrial and agricultural by-products. The PC clinker substitution rate was assumed to increase linearly from 20% in 2020 (WBCSD, 2016) to 30% in 2030, 40% in 2040, and 50% in 2050. Currently, the most dominant SCMs are limestone, coal fly ash, blast furnace slag, and silica fume (Juenger et al., 2012). Other SCMs that are receiving relatively high attention include rice husk ash, bagasse ash, and calcined clay. The following SCMs were incorporated in binary PC-SCM composite cements and treatment processes in the foreground system boundary:

- Class F coal fly ash, which is produced in power plants during coal combustion and undergoes drying prior to transport to cement plants (Chen et al., 2010; Gursel, 2014). Both drying and transportation are included here. To investigate the effect of emissions allocation (economic compared to no allocation) on the LCA results, upstream process emissions from primary production were included, i.e., electricity production at coal power plants. Allocation was only performed for the case of coal fly ash.
- Blast furnace slag, which is produced during iron production in blast furnaces. The slag is treated through quenching and granulation, dewatering and/or drying, grinding, storage, and is then transported from iron to cement plants (CTL, 2003; Petek Gursel et al., 2014) Quenching, dewatering, crushing, grinding, storage, and transportation were included in the inventory.
- Bagasse ash, which is generated via combustion of bagasse residue from sugar production. Bagasse ash can be ground and then transported to cement plants to be used as an SCM (Jittin et al., 2021). Both grinding and transportation are included here.
- Rice husk ash is produced via combustion of rice husks, which is a by-product from rice production. Rice husk ash is typically dried and then transported to cement plants (Jittin et al., 2021). Both drying and transportation are included here.
- Calcined clay is produced by heating kaolinite-rich clay to between 650 °C and 750 °C. Its production includes kaolinite extraction, production, calcination, and transportation to the cement plant (Salvi Malacarne et al., 2021). Raw material extraction and processing, and transportation are accounted for.
- Fine limestone is produced by extracting limestone rock, then crushing and finely grinding the material (Rebello et al., 2019).

Although coal fly ash and blast furnace slag are currently highly used SCMs, their future supplies are expected to decrease consistent with reduced operation of coal-fired electricity power stations and blast furnaces. The use of other SCMs like calcined clay and ground limestone is expected to increase. The use of alternative SCMs such as rice husk ash may also increase, e.g., due to the relatively high local supply (GCCA, 2021; IEA, 2018). A range of such SCMs are modelled here.

The individual influence of the above SCMs to substitute PC clinker on the climate change impact of cement production (i.e., binary SCM-PC systems) using consistent substitution (i.e.,

15%, 25%, 35%, 45% for 2020, 2030, 2040, 2050 respectively) ratios for all SCMs is studied. The 2020 results present a theoretical value of what would have been the results with these scenarios, while 2030-2040-2050 data present potential future states.

5.2.2.2 Increasing alternative fuels utilisation

The alternative fuels foreground scenario models increased utilisation of alternative fuels in cement kilns. In 2020, alternative fuels provided 45% (of which 16% was from biomass e.g., wood pellets and agricultural waste) of the thermal energy requirements in cement kilns, with the remaining 55% covered by fossil fuels. This scenario models an increase in alternative fuel use to 60% (including 30% biomass) by 2030, 75% (including 40% biomass) by 2040, and 90% (including 50% biomass) by 2050. Petroleum coke and hard coal were the fossil fuels replaced by alternative fuels including in a separate, complete electrification scenario variant. The following alternative fuels and treatment processes were included in the foreground system boundary.

- Biomass is organic matter that can be used as fuel for combustion. Wood pellets produced from sawdust were used as the feedstock to represent biomass, since this is the most used feedstock in Europe. Forest operations, transportation, processing units such as milling, drying, pelletising, packaging, and distribution are all included (Sgarbossa et al., 2020).
- Refuse-derived fuel (RDF) is generated from several waste types. Here, RDF produced from mechanical-biological treatment of municipal solid waste was used, including collection and transportation of municipal solid waste, separation, shredding, and bio-drying, as well as final product loading and transport (Grzesik and Malinowski, 2016).
- Tyre-derived fuel (TDF) is shredded scrap tyres. Even though both whole and shredded tyres can be burned in cement plants, both tyre transportation and shredding were included (Petek Gursel et al., 2014).
- Hydrogen can be produced from a variety of feedstocks. Two main production routes were considered here. The first and most common route is natural gas steam reforming, in which methane from natural gas is steam heated (usually with a catalyst), to generate a hydrogen-carbon monoxide mixture (syngas). Carbon monoxide is reacted further with steam (in a water-gas shift reaction with a catalyst) to produce carbon dioxide and

additional hydrogen. A purified hydrogen stream can then be produced by removing carbon dioxide and any other impurities. For natural gas steam reforming, the processes included in the inventory are raw material extraction, steam reforming plant construction and operation, distribution of electricity and natural gas, and landfilling (Cetinkaya et al., 2012). The second production route considered is hydrogen generation via water electrolysis powered by either wind or solar energy. Water electrolysis is a method of hydrogen production in which direct current is passed through an electrolyser to decompose water into oxygen and hydrogen gas (Rajeshwar, Mcconnell and Licht, 2008). In wind-powered electrolysis, electricity would be provided from a wind turbine, and in solar-powered electrolysis, electricity is generated through photovoltaic panels. For wind-powered electrolysis, the processes included are the manufacturing and operation of the wind turbines, water-electrolysis operation for hydrogen production, and hydrogen storage and compression. For solar-powered electrolysis, the processes included are raw materials and manufacturing of photovoltaic panels, transportation and installation, operation and maintenance, water-electrolysis operation for hydrogen production, and hydrogen storage and compression (Dufour et al., 2011; Cetinkaya et al., 2012).

Eight scenarios were developed with different fuel mix designs using various alternative fuels and electrification. The scenarios represent plausible future mix designs, as the single use of a fuel (e.g., biomass) is considered challenging due to market competitiveness and limited regional availability. The scenarios (listed as 1-8 below) are presented according to the fuel mix ratio in 2050:

1. Hydrogen (providing 40% of the thermal demand, generated through wind power), biomass (providing 50% of the thermal demand), and fossil fuels (providing 10% of the thermal demand).
2. Hydrogen (providing 40% of the thermal demand, generated through solar power), biomass (providing 50% of the thermal demand), and fossil fuels (providing 10% of the thermal demand).
3. Hydrogen (providing 40% of the thermal demand, generated through natural gas steam reforming (NGSR) power), biomass (providing 50% of the thermal demand), and fossil fuels (providing 10% of the thermal demand).

4. Biomass (providing 50% of the thermal demand), TDF (providing 40% of the thermal demand), and fossil fuels (providing 10% of the thermal demand).
5. Biomass (providing 50% of the thermal demand), RDF (providing 40% of the thermal demand), and fossil fuels (providing 10% of the thermal demand).
6. Biomass (providing 50% of the thermal demand), RDF (providing 20% of the thermal demand), RDF (providing 20% of the thermal demand), and fossil fuels (providing 10% of the thermal demand).
7. Biomass (providing 90% of the thermal demand), and fossil fuels (providing 10% of the thermal demand).
8. Complete electrification of the cement kiln based on the European grid (with main electricity sources i.e., Denmark (15%), France (15%), Great Britain (9%), Italy (9%), Spain (7%) in 2020).

Overall, the 8 scenarios vary the energy source in pyro-processing. Scenarios 1-7 present plausible fuel mixes to satisfy the thermal demands of the kiln. Scenario 8 assumes complete electrification of the rotary kiln, preheater, and precalciner. Scenario 8 was developed to estimate the potential climate change benefits of a complete transition to electricity as the main source of energy in a cement plant. Complete electrification of the cement kiln is modelled using base-load electricity supplied from the European grid, with country-specific energy sources detailed in Table *5-1*). This scenario uses electricity to satisfy the overall energy demand of the kiln configuration (e.g., dry kiln with preheater and precalciner), at the same energy efficiency as conventional fuel-based operation. Therefore, the electrification scenario provides a theoretical estimate of the potential performance of electrified cement production based on future decarbonisation of electricity generation. The overall energy demands per PC clinker for the kiln configurations modelled here are 1.13 kWh in 2020, 1.08 kWh in 2030, 1.00 kWh t in 2040, and 0.90 kWh in 2050. Table *5-1* presents the types of electricity and their corresponding shares in national electricity mixes that are used in Scenario 8.

Table 5-1: Electricity mix of the main location-specific electricity sources.

	Country				
Electricity flow	Denmark (15%)	France (15%)	Great Britain (9%)	Italy (9%)	Spain (7%)
Electricity production, natural gas, combined cycle power plant	48.2%	48.2%	48.2%	48.2%	48.2%
Electricity production, natural gas, conventional power plant	17.6%	17.6%	17.6%	17.6%	17.6%
Electricity production, hard coal	10.5%	21.0%	21.0%	10.5%	10.5%
Electricity production, lignite	10.5%	-		10.5%	10.5%
Electricity production, oil	-	3.4%	3.4%%	-	-
Electricity production, hydro, run-of-river	-	-	5.5%	-	-

5.2.2.3 Improving kiln configurations

The kiln efficiency scenario describes the use of different kiln types regardless of the original cement plant state (new or retrofitted). The average European cement production in 2020 is represented by the semi-wet/semi-dry kiln process with a thermal demand of 3.87 MJ per kg PC clinker. A dry kiln with preheater and without pre-calciner was selected for 2030 (3.68 MJ per kg PC clinker) while the average thermal efficiency for the dry kiln with preheater and pre-calciner was used for 2040 (3.39 MJ per kg clinker). The highest kiln efficiency, which is obtained using a dry kiln with preheater and pre-calciner, was used for 2050 (3.02 MJ per kg clinker). The modelling of changes in kiln configuration includes changes in energy demand and the requirements to construct the kiln in a new cement plant.

5.2.2.4 Synergistic scenario

The three individual scenarios (Sections 2.2.1-2.2.3) address different parts of the PC production process, and the decarbonisation technologies can be combined to further reduce CO_2-eq. emissions. Therefore, synergetic scenarios incorporating all three decarbonisation measures, without and additionally with CCS were developed. Due to the numerous ways that the decarbonisation measures can be implemented (e.g., use of different SCMs and alternative fuels), three synergistic scenarios were investigated to explore (a) the maximum potential CO_2 emissions reduction without CCS, (b) the effect of roadmap selection, and (c) the influence of CCS:

a. Synergistic scenario without CCS, consistent with the IEA and CEMBUREAU roadmaps, to determine the largest CO_2-eq. emissions per kg cement reduction potential (in 2050) without CCS. This scenario contains 90% alternative fuel use including 20% hydrogen from wind-powered electricity generation and 45% biomass (the remaining 25% is covered by 20% waste plastic, 3% waste solvent, 1% sawdust, and 1% waste oils), a dry kiln type with preheater and precalciner (3.02 MJ per kg clinker) and uses SCMs at a constant clinker-to-cement mass ratio of 0.5.
b. Synergistic scenario without CCS, following the GCCA roadmap, to analyse the effect of roadmap selection on modelled CO_2 emissions (in 2050). This scenario contains 90% alternative fuel use (40% natural gas, 35% biomass, 7.5% hydrogen (wind-powered electrolysis) and 7.5% electrification; the remaining 10% consists of oil products and non-renewable waste, i.e., no coal), a dry kiln with preheater and pre-calciner (3.02 MJ per kg clinker), and 50% PC clinker substitution using SCMs. The key difference between synergistic scenarios 1 and 2 and thus the IEA/CEMBUREAU and GCCA roadmaps is the relative share of alternative fuels in the fuel mix (e.g., biomass is 45% in scenario a, and 35% in scenario b).
c. Synergistic scenario with CCS, with 90% alternative fuel use (45% biomass; 20% hydrogen; 20% waste plastic, 3% waste solvent, 1% sawdust, and 1% waste oils are used to satisfy the thermal energy requirement), a dry kiln process including a pre-calciner and preheater (3.02 GJ per t of clinker), fly ash as an SCM (50% PC clinker, 45% fly ash and 5% gypsum), and CCS (oxyfuel combustion and separately amine scrubbing, with and without a combined heat and power unit) implemented in 2050.

This year of industrial scale implementation was chosen since CCS is currently only implemented at pilot scale.

In the third synergistic scenario (with CCS, synergistic scenario c), the amine scrubbing CO_2 capture unit was modelled with monoethanolamine (chemical absorbent) with an 85% capture efficiency. The same CO_2 capture efficiency (85%) was assumed for the oxyfuel combustion process, and here an air separation unit was additionally included. Both technologies were modelled according to Chen and Yang (2021). These CCS technologies were modelled to be implemented without and with a combined heat and power unit modelled according to An, Middleton and Li (2019), to satisfy their thermal demands. Two energy sources – hard coal and natural gas – were compared for both amine scrubbing and oxyfuel processes.

5.2.3 Background scenarios

When performing prospective LCA, it is important to ensure temporal consistency between foreground and background systems (Arvidsson and Molander, 2017). Here, prospective life cycle inventory (pLCI) databases were used, generated based on an integration of the ecoinvent database (Wernet et al., 2016) and data from the integrated assessment model IMAGE (Stehfest et al., 2014). Full details of the background database modelling setup are provided in Section 3.4.1.2.6. The background system in this study only accounts for the influence of future changes in electricity generation on the climate change impacts of cement production. Data for two scenarios, the SSP2 baseline (3.8-4.2°C warming by 2100 relative to 1980) and SSP2 RCP 2.6 (~2°C warming by 2100 relative to 1980) were available.

Four-time steps were accounted for (2020, 2030, 2040, and 2050), which resulted in a total of 8 pLCI databases (four for each scenario). In order to facilitate the practical modelling, the superstructure approach was applied to merge the individual pLCI databases into a single 'superstructure' LCI database and an associated 'scenario difference file' (Steubing and de Koning, 2021). The scenario difference file was also simultaneously used to accommodate foreground (e.g., cement production) and background (e.g., electricity generation) scenario data (Figure *5-1*). The open source LCA software Activity Browser was used, and the superstructure modelling approach was implemented with a modified ecoinvent database (v3.6, cut-off system model) (Steubing et al., 2020).

5.2.4 Uncertainty analysis

The quantitative uncertainty of the results was investigated using Monte Carlo analysis, focusing on the influence of PC clinker because it is the key contributor to cement production CO_2-eq. emissions. The mass of PC clinker used in cement formulations was classified using data quality indicators in a pedigree matrix based on a scoring system (scale 1-5), which assesses judgements on data integrity, reliability, temporal correlation, geographical, and technological uncertainty (as seen in Table B20) (Weidema B.P et al., 2013). These indicator scores were used to generate probability density functions (PDFs) using log-normal distribution (Yu, Liu, and Gu, 2018). PDFs were generated based on a 10% uncertainty in the mass of PC clinker and a standard deviation of 0.084 for both 2020 and 2050 (for further details refer to Appendix B, Section B3). These PDFs were randomly sampled 1,000 times to evaluate the influence of LCA model parameters on the results (Lewandowska et al., 2004), i.e., through a global sensitivity analysis (Plischke et al., 2013). A global sensitivity indicator (delta) was used to define the importance of model parameters on the results, such that when delta equals 1, the results are highly sensitive to the parameter (Borgonovo, 2007). A more detailed description of this uncertainty analysis is provided in Appendix B (Sections B6 and B7).

5.3 Results and discussion

5.3.1 Portland cement clinker substitution with supplementary cementitious materials

Substituting PC clinker with SCMs (coal fly ash, blast furnace slag, rice husk ash, bagasse ash, calcined clay, limestone) can significantly reduce the climate change impact of cement production by 0.44 kg CO_2-eq. emissions per kg cement over the 30-year period from 2020 to 2050 (Figure 5-2), decreasing monotonically with decreasing clinker-to-cement ratio. The potential reduction in climate change impact achieved in 2050 (42%) relative to 2020 is consistent with the results of traditional LCA studies, which report a potential reduction in CO_2-eq. emissions footprint of 34-51% at clinker-to-cement ratios of 50% (Figure 5-2) (Miller, 2018; Rahla et al., 2019; Arrigoni et al., 2020). The wider range of CO_2-eq. emissions reduction potentials in the existing literature corresponds to different replacement ratios, with higher

clinker replacement ratios (up to 90%) resulting in higher impact reductions (~88%). This climate change impact reduction is mainly driven by avoided pyro-processing of limestone in the cement kiln. The contribution of SCM pre-treatment was found to be negligible, except in the case of calcined clay addition where pre-treatment requires significantly higher energy demand (Appendix B, Section B4.1).

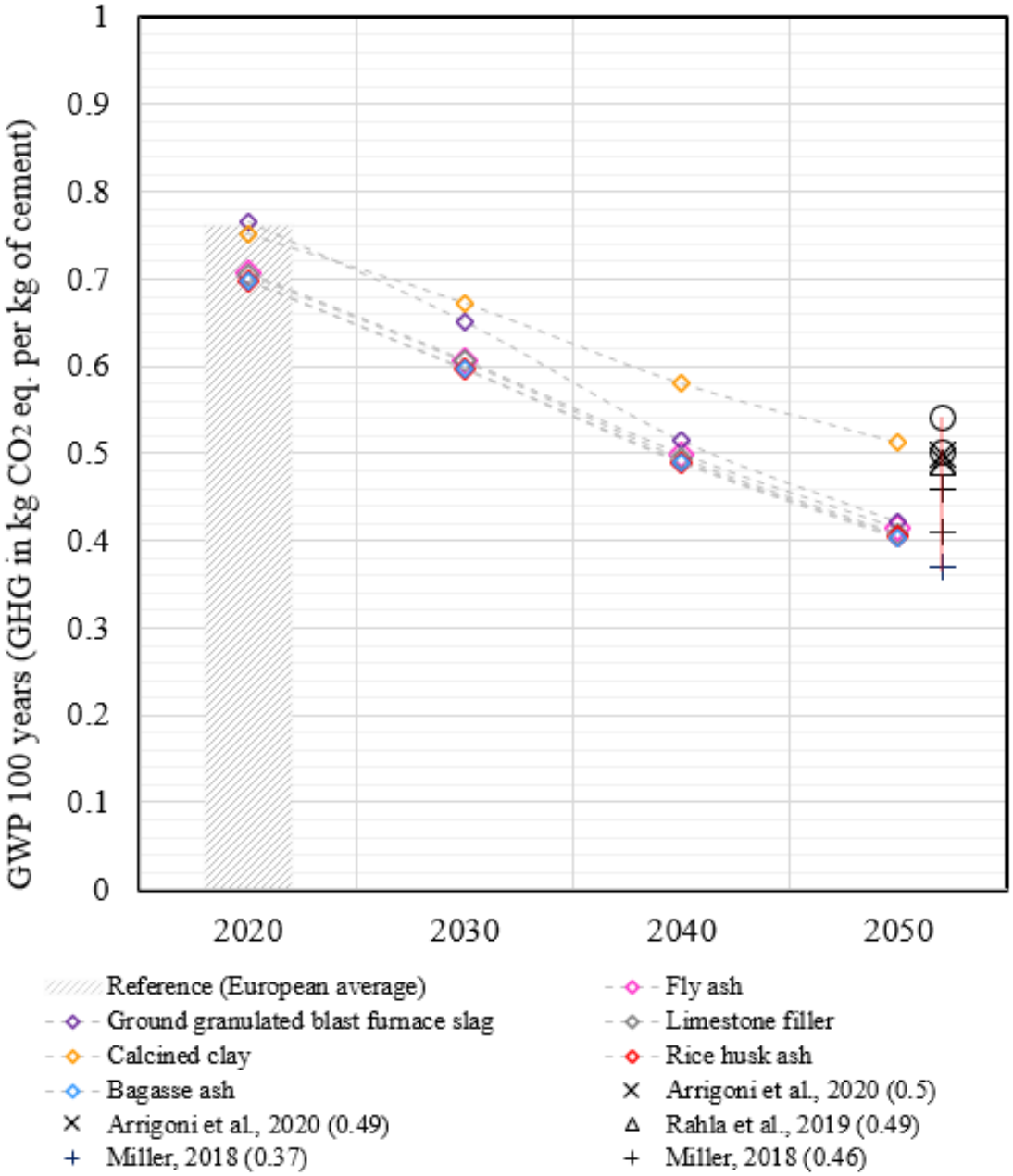


Figure 5-2: CO_2-eq. emissions of blended PCs with different SCMs, from 2020 to 2050, using the SSP2 RCP 2.6 background scenario, compared to reported values (for cements with a ~50% clinker replacement ratio) from traditional LCA studies in 2050 (Arrigoni et al., 2020; Miller, 2018; Miller et al., 2018; Rahla et al., 2019)

The climate change impacts of producing the different PC-SCM cements in the same year and clinker-to-cement ratio lie within 0.1 kg CO_2-eq. emissions per kg cement, which is a smaller difference than the decrease in CO_2-eq. emissions caused by decreasing the clinker-to-cement ratio from 0.8 in 2020 to 0.5 in 2050. Therefore, the PC clinker substitution level has a greater effect on CO_2-eq. emissions mitigation than the type of SCM used. However, blended cement with calcined clay has a noticeably higher climate change impact (0.52 kg CO_2-eq. emissions per kg cement at 2050) than the other SCMs analysed (coal fly ash, blast furnace slag, rice husk ash, bagasse ash, and limestone; 0.42-0.44 kg CO_2-eq. emissions per kg cement at 2050) mainly because of higher energy requirements (thermal treatment at ~600 °C) in calcined clay production.

5.3.2 Increasing alternative fuels utilisation

Utilisation of alternative fuels (biomass, RDF, TDF, hydrogen [NGSR, solar-powered electrolysis, wind-powered electrolysis]) can reduce CO_2-eq. emissions per kg cement by 25% (i.e., from 0.76 kg to 0.57 kg) between 2020 and 2050 (Figure 5-3). A minor reduction in CO_2-eq. emissions (i.e., about 4%) was observed from 2020 to 2030 relative to between 2030 to 2040 and 2040 to 2050 (i.e., about 12% and 10%, respectively), which is driven by greater changes in the projected European electricity mix during the latter years (Appendix B, Section B4.2).

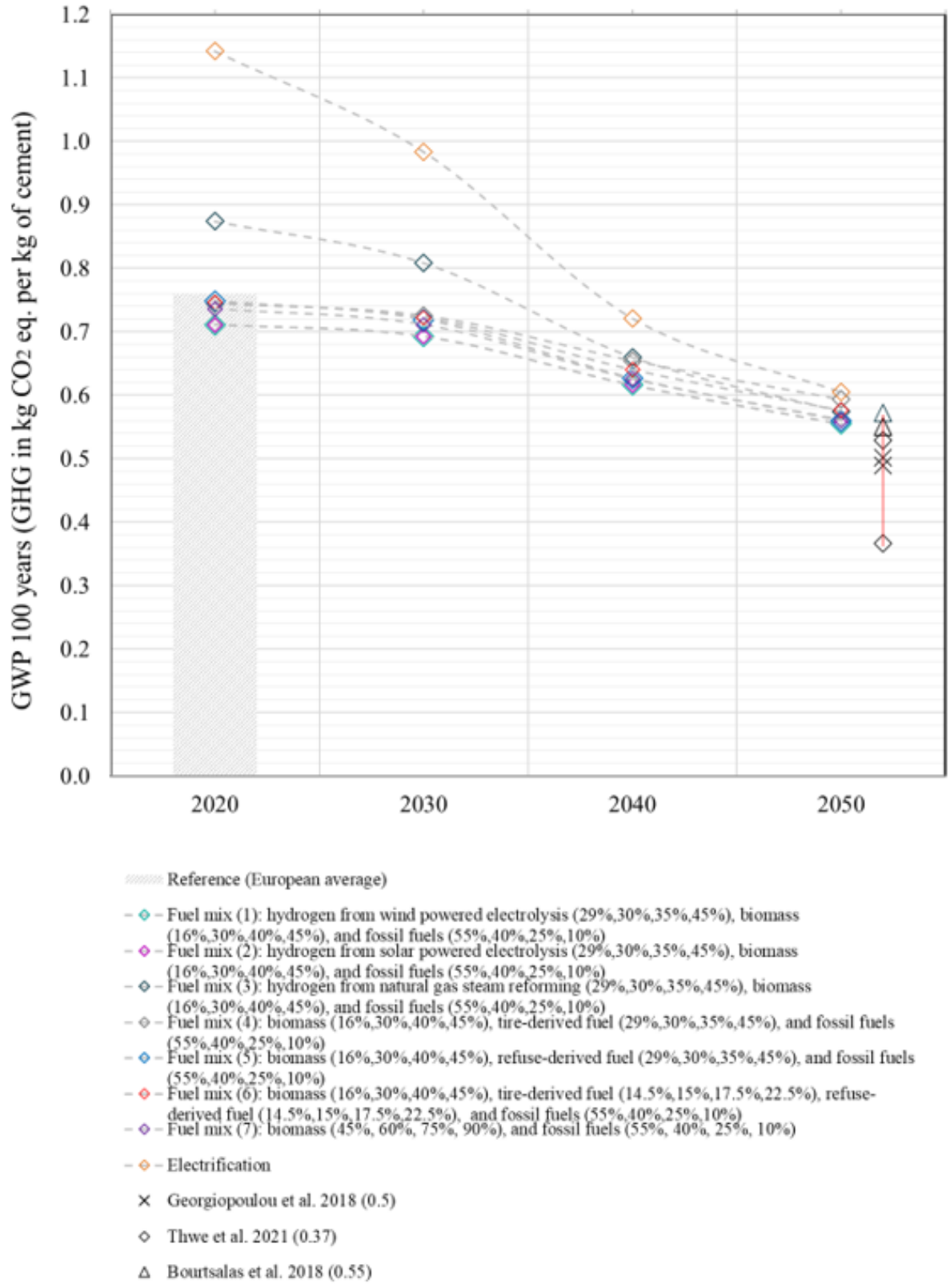


Figure 5-3: CO_2-eq. emissions from cement production with alternative fuels, from 2020 to 2050, using the SSP2 RCP 2.6 background scenario, compared to reported values from traditional LCA studies in 2050 (Georgiopoulou and Lyberatos, 2018; Thanos Bourtsalas et al., 2018; Thwe et al., 2021). The percentage fuel mix ratio of each individual fuel in 2020 to 2050 is displayed in the figure legend.

Changes in the electricity mix have a greater effect on the results for alternative fuels that require more electricity i.e., electrification and hydrogen production (through NGSR) (see also Appendix B, Section B5.2, Figure B11). For example, complete electrification in 2020 would have resulted in a significantly higher CO_2-eq. emissions (1.14 kg) than the reference value (0.76 kg) (Figure 5-3, orange diamonds), since electricity was generated mainly from fossil fuels (35%) at this time. Electrification only becomes a viable decarbonisation approach once the amount of fossil fuels used in electricity generation reduces, such that by 2050 electrified cement production has similar CO_2-eq. emissions to other alternative fuels (~0.6 kg CO_2-eq. emissions per kg cement, Figure 5-3). Therefore, the potential of electrification in cement production is controlled by the degree of decarbonisation the power generation achieves, causing a higher uncertainty regarding the final emission reduction. The calculated decarbonisation potential of electrification (~30% reduction) is roughly consistent with early results from research on solar energy for raw materials calcination, which showed a 48% potential reduction in climate change impacts where solar calcination was employed instead of conventional (petroleum coke based) calcination (Tomatis et al., 2020).

Figure 5-3 shows similar or slightly higher GHG emissions compared to those reported in the literature, mainly due to differences in the foreground unit process data used. For instance, the highest efficiency kiln type was defined with a thermal demand of 3,020 MJ per tonne clinker, which is substantially higher compared to the thermal demand assumed in the work of Georgiopoulou & Lyberatos (2018) of 1,800 MJ per ton clinker. Additional differences are explained by the different fuel mixes modelled here and in the literature. Thwe et al. (2021) investigated the climate change impact of natural gas as a single fuel during combustion, leading to an environmental impact of 0.36 kg CO_2 eq. emissions per kg cement.

The potential CO_2-eq. emissions reduction from the use of alternative fuels (Figure 5-3, 25% reduction between 2020 and 2050) is lower than that achieved through the SCMs scenario (Figure 5-2, 42% reduction between 2020 and 2050), which is consistent with process emissions being the major climate change hotspot in cement production. Overall, the alternative fuels scenario with the lowest GWP100 values combines hydrogen (providing 40% of the thermal demand, generated through wind power), biomass (providing 50% of the thermal demand), and fossil fuels (providing 10% of the thermal demand) (pink crosses, Figure 5-3), since this scenario utilises renewable energy sources with inherently low CO_2-eq. emissions.

5.3.3 Improving kiln configurations

Cement produced with higher efficiency kilns (i.e., 5% [3.68 MJ per kg PC clinker], 10% [3.39 MJ per kg clinker], and 20% [3.02 MJ per kg clinker] reduction in thermal energy demand) resulted in 4%, 9%, and 12% lower CO_2-eq. emissions in 2030, 2040, and 2050, respectively, relative to the year 2020 (Figure 5-4 (A)). The greatest reduction in CO_2-eq. emissions per kg cement (12%) was observed using the dry process (including a preheater and pre-calciner).

This result is slightly lower than the range of CO_2-eq. emissions reductions reported in the literature (15-20%) (Habert et al., 2010; Benhelal et al., 2013; Chunark, Hanaoka and Limmeechokchai, 2021). This discrepancy can be explained by differences in assumptions made e.g., the 20% reduction in CO_2-eq. emissions in Benhelal et al. (2013) resulted from a larger improvement in thermal efficiency, by switching from a wet kiln type (5.55 MJ per kg clinker) to a dry kiln configuration (3.00 MJ per kg clinker) than the improvements considered here between a semi-wet/semi-dry kiln (3.68 MJ per kg clinker) and a dry kiln with precalciner and preheater (3.02 MJ per kg clinker).

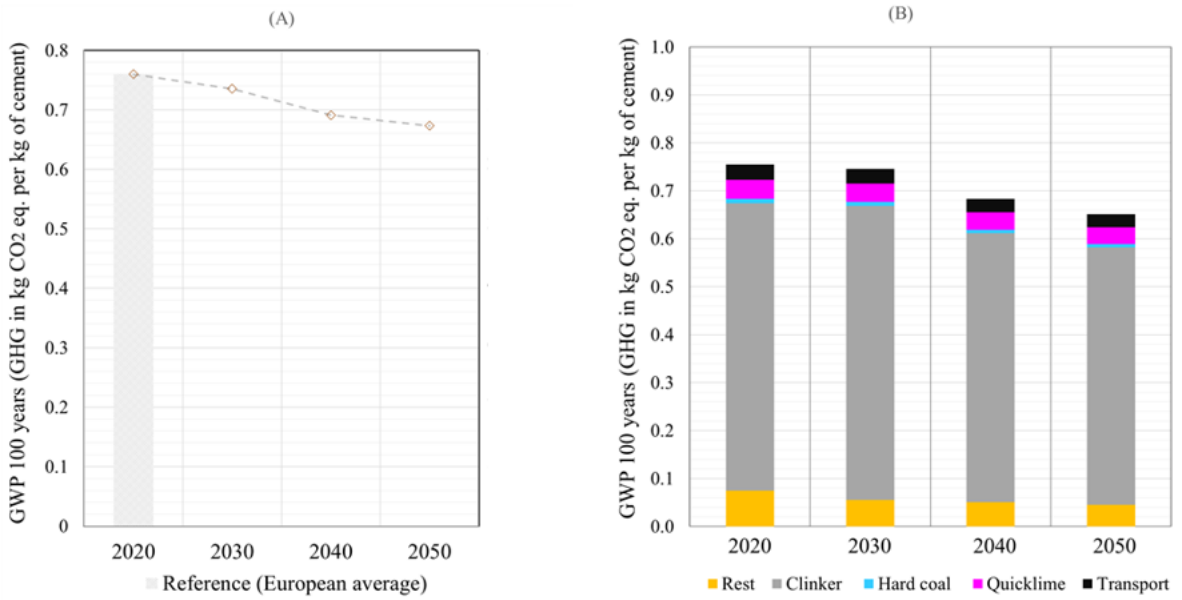


Figure 5-4: (a) CO_2-eq. emissions from cement production with several kiln configurations from 2020 (semi-wet/semi-dry) to 2050 (dry with preheater and precalciner), using the SSP2 RCP 2.6 background scenario. (b) Contribution analysis of the key contributing flows from 2020 to 2050 using different kiln configurations.

Introducing a kiln type with higher thermal efficiency has no direct effect on the liberation of CO_2 from clinker calcination reactions, which is the main hotspot of the cement production system. Similarly, higher thermal efficiency reduces fuel consumption but does not change the fuel type. Since this measure does not change the clinker-to-cement ratio nor the fuel type, the 4-12% CO_2-eq. emissions reduction shown in the results are attributable to the reduced fuel demand (Figure 5-4 (B)). In 2050, the climate change impact of electricity use is negligible due to decarbonisation of the grid shown by using alternative fuels, e.g., waste plastics.

5.3.4 Synergistic scenarios

Synergistic implementation of greater PC clinker substitution, higher alternative fuel use, and more efficient kiln types following the IEA and CEMBUREAU roadmaps and without CCS can result in a significantly larger reduction in CO_2-eq. emissions per kg cement (58% by 2050 with SSP2 RCP 2.6) compared to the implementation of individual decarbonisation measures (PC clinker substitution with SCMs, Section 5.3.1, up to 42%; increasing alternative fuel consumption, Section 5.3.2, up to 25%; improving kiln configuration, Section 5.3.3, up to 12%) (Figure 5-5).

Our analysis of the effect of allocation on the climate change impact of blended cement containing coal fly ash showed that it is moderately underestimated when no allocation was considered. Performing economic increased the climate change impact by ~20% by 2050 (Figure 5-5). The higher impact is attributed to the associated emissions linked to coal combustion at power plants. Reducing trends in climate change impact are observed for both no allocation and economic allocation approaches, with 53% and 44% reductions observed from 2020 to 2050, respectively. The low price of coal fly ash (€58 per tonne in 2020) compared to the price of electricity is responsible for the relatively small economic allocation factor (0.0693). This agrees with existing literature, that highlights the importance of including allocation for evaluating the climate change impact effect of SCMs in cement (Shobeiri et al., 2021), particularly at lower clinker-to-cement ratios (Seto, Churchill and Panesar, 2017).

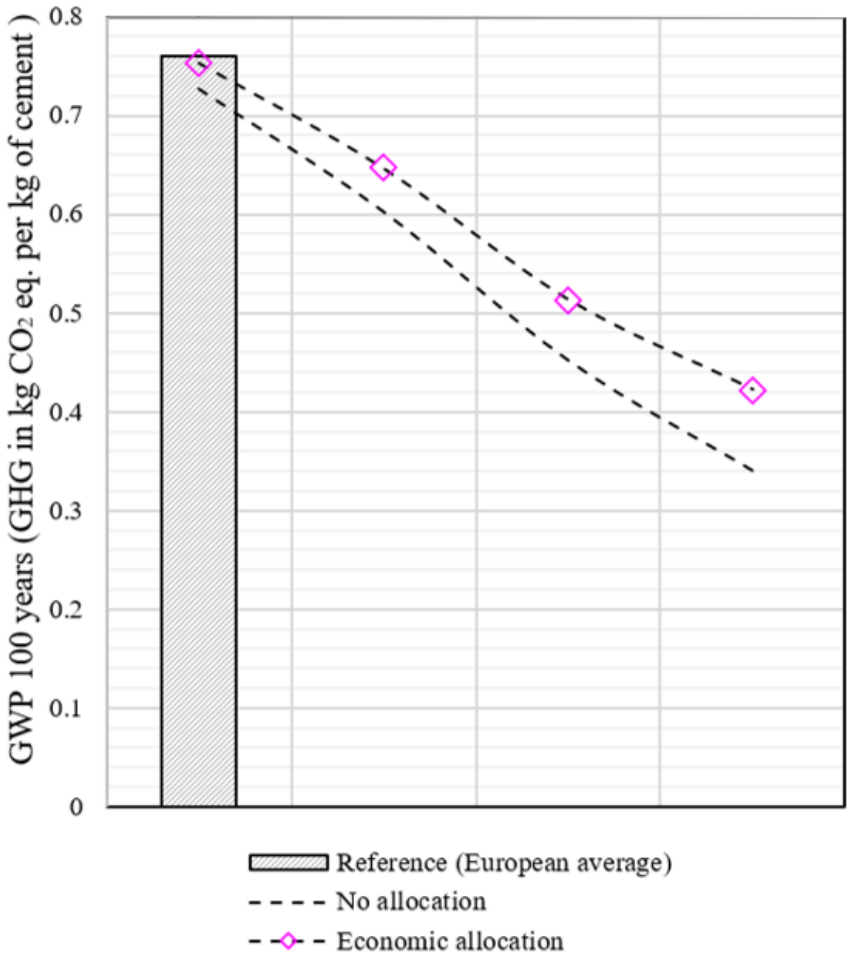


Figure 5-5. CO_2-eq. emissions of cement production with (synergistic) scenario 4 from 2020 to 2050, using SSP2 RCP 2.6, for blended PC with coal fly ash, and based on the IEA and CEMBUREAU roadmaps with no allocation and economic allocation approaches.

Figure 5-6 shows the climate change impacts of cement production with the synergistic scenarios including PC clinker substitution with coal fly ash (SCMs), increasing alternative fuels utilisation, and improving kiln configurations with (economic) and without allocation of impacts related to coal fly ash production, explicitly showing the contribution of biogenic carbon. As discussed above, no allocation has the lowest climate change impact (0.34 kg CO_2 eq. per kg cement in 2050) while economic allocation shows a higher impact (0.42 kg CO_2 eq. per kg cement in 2050) because of the inclusion of CO_2-eq. emissions associated with electricity generation. The case of no allocation with the reference fuel mix scenario (Figure 5-6A) would result in a ~17% higher impact (0.41 kg CO_2-eq. emissions per kg cement) by 2050, compared to the case where increased alternative fuels are used (Figure 5-6B). This results from the higher use of fossil fuels in the fuel kiln mix such as petroleum coke and hard coal instead of biomass. It leads to the difference between the amount of biogenic carbon stored in biomass between the two scenarios (0.25 kg CO_2 eq. emissions per kg cement and 0.67 kg

CO_2 eq. emissions per kg cement, respectively). Biomass fuel is often considered to have a CO_2-eq. emissions footprint of zero by equating the CO_2-eq. emissions released during the burning of biomass to the CO_2 emissions stored during photosynthesis (growing stage). However, various factors give biomass fuel a non-zero CO_2-eq. emissions footprint. One such factor is that processing is required to form the raw biomass (wood logs) into a fuel feedstock. This processing depends on the biomass source and accounts for both CO_2-eq. emitting and storing factors.

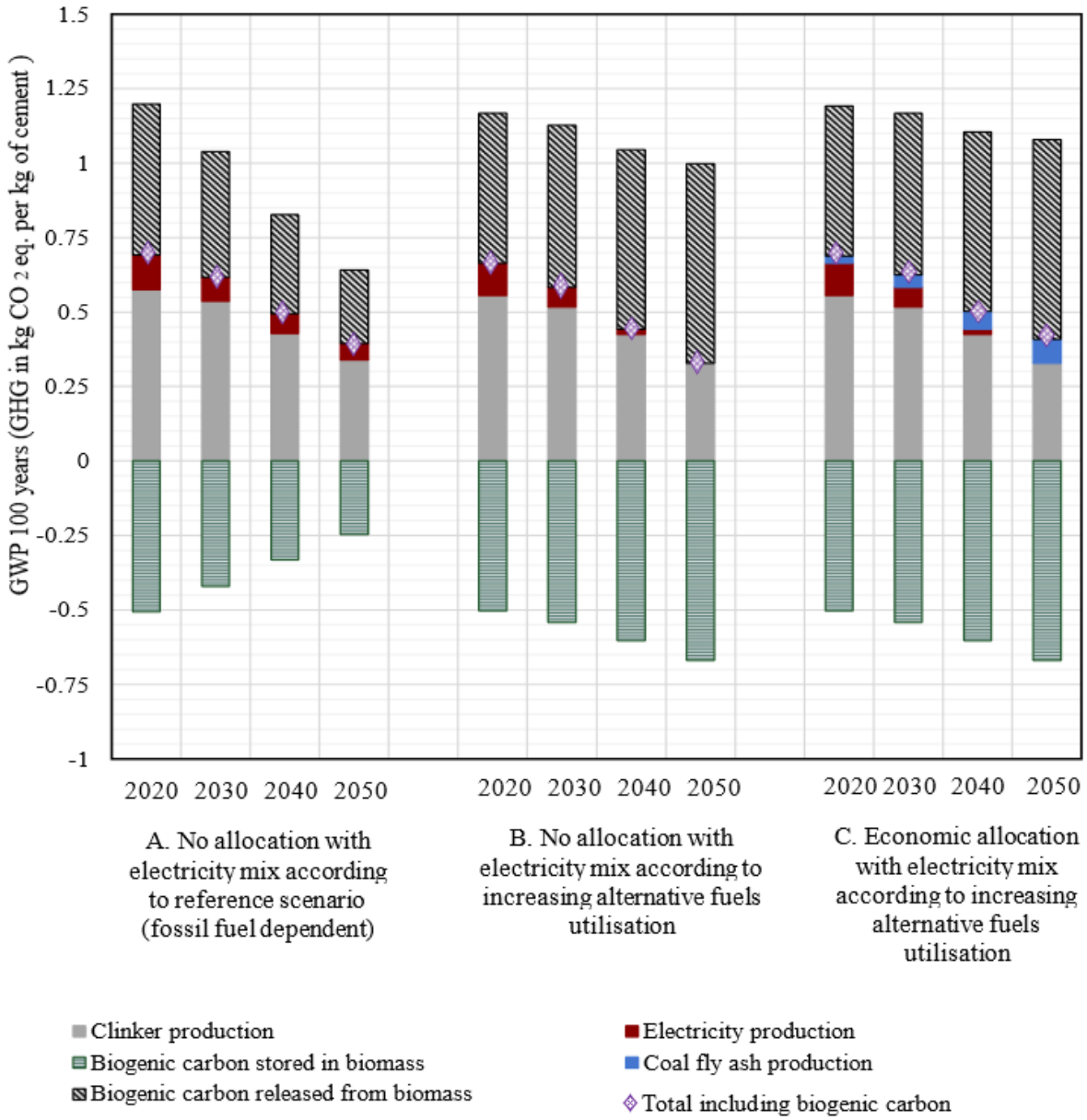


Figure 5-6: Contribution analysis for the CO_2-eq. emissions of cement production with synergistic scenarios including PC clinker substitution with coal fly ash, increasing alternative fuels utilisation, and improving kiln configurations from 2020 to 2050 (without CCS) with (economic) and without allocation, and including biogenic carbon.

The synergistic scenarios, according to the IEA and CEMBUREAU roadmaps and the GCCA roadmap, perform similarly: The former shows a slightly greater reduction in CO_2-eq. emissions per kg cement (58%) compared to the latter (53%) between 2020 and 2050 (Figure *5-7*). In both cases, the major contributor to the climate change impact is clinker production, which is responsible for 97% in both cases. The minor difference (~5%) between the two scenarios is attributable to the different fuel mixes in the roadmaps.

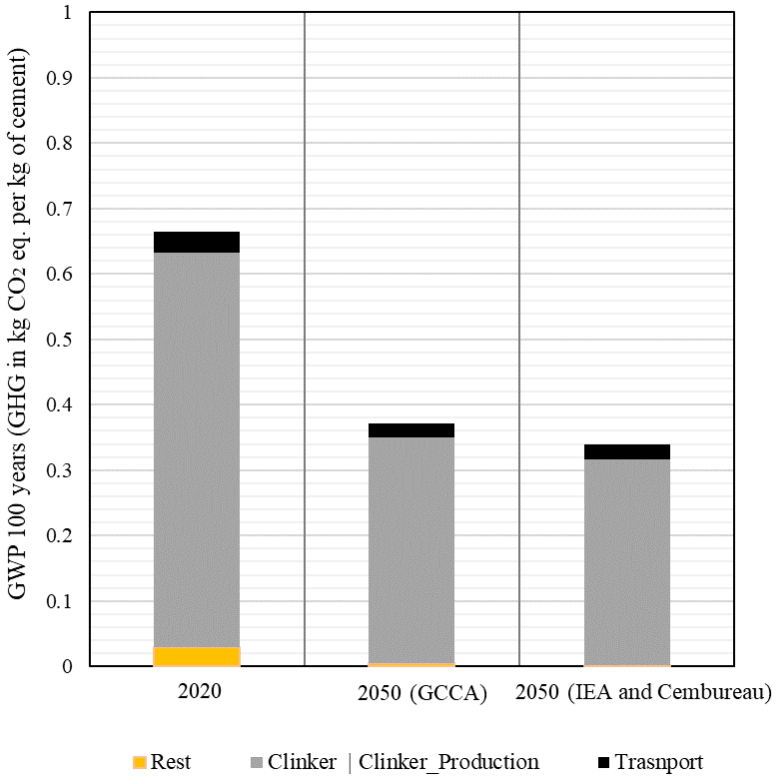


Figure 5-7: Comparison of synergistic scenarios developed based on the IEA and CEMBUREAU roadmaps, and the GCCA roadmap. CO_2-eq. emissions of cement production from a synergistic scenario incorporating PC clinker substitution with coal fly ash (SCMs), increasing alternative fuels utilisation, and improving kiln configurations from 2020 to 2050 (without CCS).

CCS can significantly reduce the climate change impact of cement production (55-88% reduction in GWP100 [0.19 kg CO_2-eq. emissions per kg cement] in 2050 compared to the value in 2020 [0.76 kg CO_2-eq. emissions per kg cement]) (Figure 5-8). By 2050, CCS was found to have a slightly higher CO_2-eq. emissions reduction potential than that reported in the literature (48-84%) (An et al., 2019; Miller et al., 2018; Volkart et al., 2013; Wang et al., 2022). This discrepancy was attributed to the additional combined effect of decarbonisation measures (i.e., PC clinker substitution with coal fly ash, increasing alternative fuels utilisation, and

improving kiln configurations) and the use of background scenarios that include decarbonisation of the electricity mix, which causes an additional reduction in CO_2-eq. emissions of 6-7% by 2050.

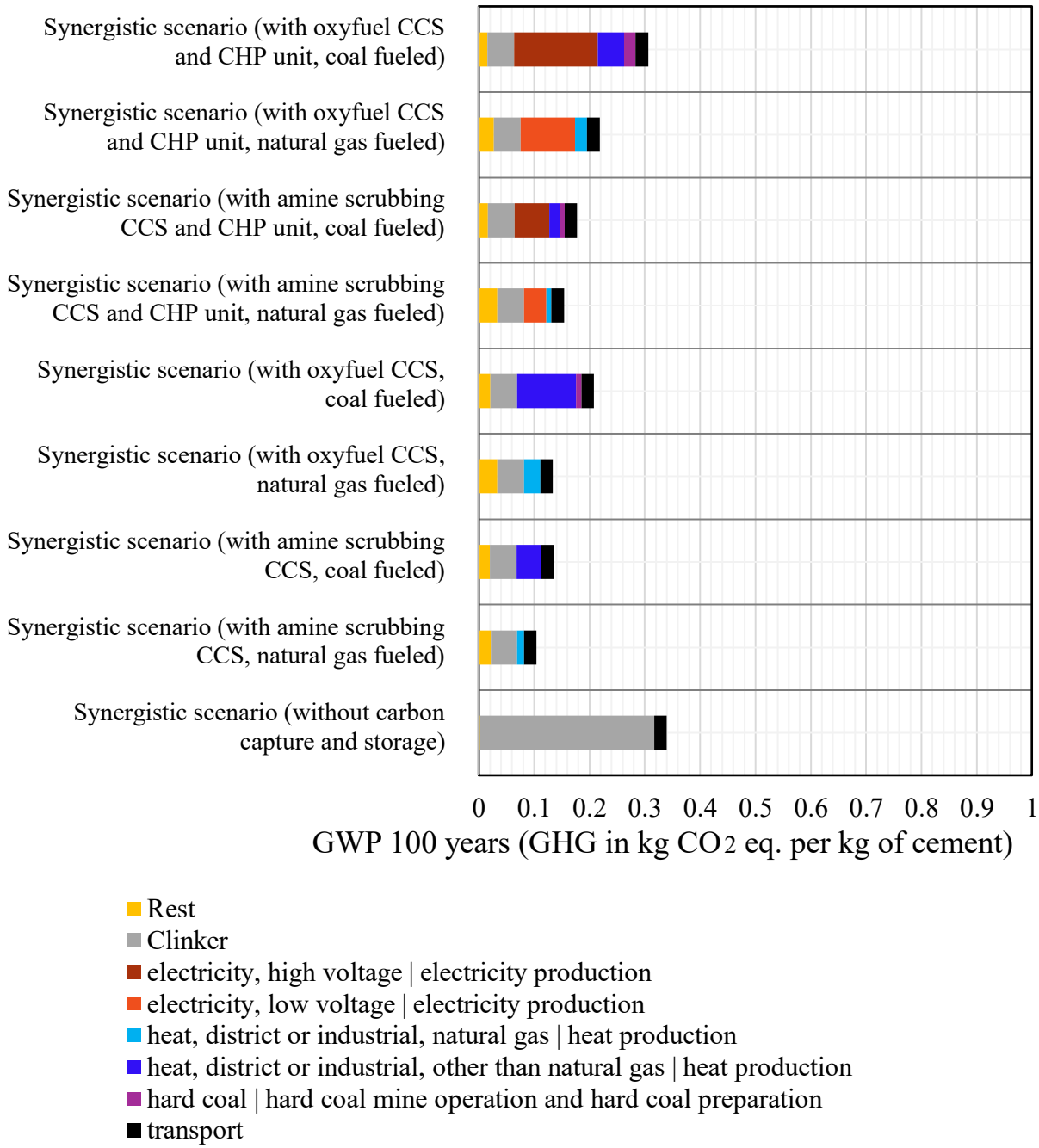


Figure 5-8: Contribution analysis showing CO_2-eq. emissions of cement production developed based on the IEA and CEMBUREAU roadmaps with synergistic scenarios including PC clinker substitution with SCMs, alternative fuel use, improved kiln configurations, and CCS technologies (i.e., amine scrubbing, oxy-fuel) in 2050, with and without combined heat and power unit (CHP).

Analysis of multiple CCS technologies (implemented in addition to PC clinker substitution, alternative fuel use, and kiln efficiency improvements) shows that natural gas-powered amine scrubbing without combined heat and power results in a lower climate change impact (0.09 kg CO_2-eq. emissions per kg cement) relative to oxyfuel CCS technology (0.16 kg CO_2-eq. emissions per kg cement) and synergistic scenarios without CCS (0.32 kg CO_2-eq. emissions per kg cement following the IEA/CEMBUREAU roadmaps) in 2050. The results show that the type of fuel used to provide heat for the CCS system has a lesser effect on the overall GWP100 of cement production than the type of CCS technology (Figure 5-8). The relatively minor influence of the fuel type for the operation of the CCS systems (typically there is a 5% difference between natural gas and hard coal scenarios) is consistent with the literature, which suggests that the use of natural gas results in slightly lower GWP100 than coal without a combined heat and power plant (An et al., 2019; García-Gusano et al., 2015b; Volkart et al., 2013; Wang et al., 2022). In the case of integrating a combined heat and power plant to satisfy the thermal demands of the CCS plant, this results in higher GWP100 values relative to direct use of natural gas and coal (Figure 5-8). This is because CO_2-eq. emissions arising from the generation of electricity for the CHP system are larger than those from the direct use of natural gas and coal. Amine scrubbing CCS technology results in a lower GWP100 value for cement production than oxyfuel technology due to the higher energy requirements, although both reduce CO_2-eq. emissions relative to cement production without CCS.

Therefore, the results show that CCS can significantly avoid CO_2-eq. emissions from cement production and the extent of this reduction is modified to a minor-moderate extent by the type of CCS technology (10-18% change) and its fuel type (5-10%). This is consistent with the previous work that studied the implementation of carbon capture technology (without CHP) in the power generation industry including background changes and found that the type of CCS technology is more influential (7%) than fuel type (3%) in reducing GHG emissions Volkart, Bauer and Boulet, 2013). The findings indicate that reducing the CO_2-eq. emissions from cement production to contribute to limiting the temperature rise to 2 °C by 2100 relative to pre-industrial levels (requiring 0.72 kg CO_2-eq. emissions per kg cement) is achievable without CCS, but with a substantial increase of alternative fuels and PC clinker substitution with SCMs i.e., 90% alternative fuel mix and 50% PC clinker substitution. Additional decarbonisation measures such as CCS are needed to reduce its GWP100 further, possibly down to 0.09 kg CO_2-eq. emissions per kg cement by 2050 using amine scrubbing CCS technology.

5.3.5 Effect of background scenarios

The SSP2 baseline scenario shows a smaller CO_2-eq. emissions reduction than the SSP2 RCP2.6 scenario (Figure 5-9). The two SSP2 background scenarios show similar CO_2-eq. emissions per kg cement values at year 2030 (~0.6 kg CO_2-eq. emissions per kg cement). However, by 2040, the influence of the background system becomes significant, such that the difference between the SSP2 baseline and RCP 2.6 scenarios are 6% in 2040 (0.46 vs. 0.43 kg CO_2-eq. emissions per kg cement, respectively) and 10% in 2050 (0.36 vs. 0.32 kg CO_2-eq. emissions per kg cement, respectively).

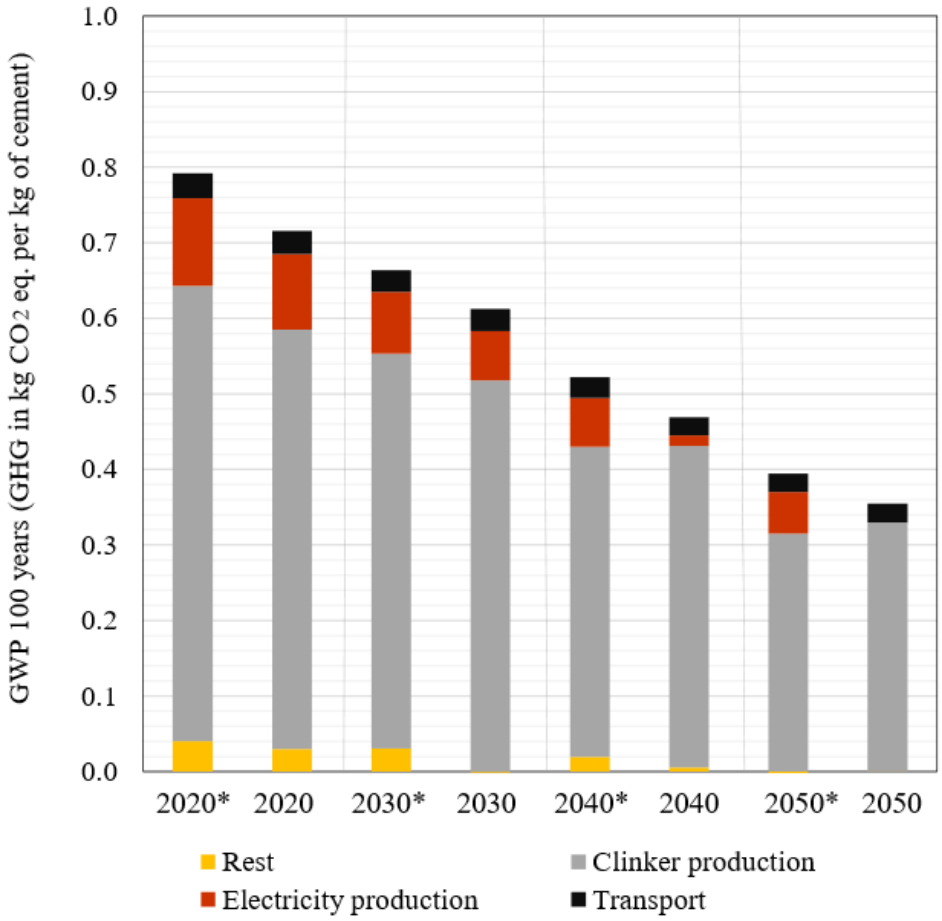


Figure 5-9. CO_2-eq. emissions of cement production from a synergistic scenario developed based on the IEA and CEMBUREAU roadmaps, incorporating PC clinker substitution with coal fly ash, increasing alternative fuels utilisation (fuel mix consists of hydrogen generated through wind power, biomass, and fossil fuels), and improving kiln configurations from 2020 to 2050 (without CCS), with SSP2 baseline (presented with an asterisk *) and SSP2 RCP 2.6 (no asterisk).

The difference in CO_2-eq. emissions per kg cement between the background scenarios is associated with the composition of the future electricity mix. For example, in the SSP2 RCP2.6 background scenario, the electricity mix contains more wind power than in the SSP2 baseline scenario, reducing its GWP100 to 0.34 kg CO_2 eq. emissions per kg rather than 0.39 kg CO_2 eq. emissions per kg, respectively. Thus, even though PC clinker substitution is a key driver of CO_2-eq. emissions, the results show that the electricity mix and thus changes in the background system are significant when SCM pre-treatment requires high amounts of electricity (Appendix B, Figure B3).

5.3.6 Challenges for global decarbonisation of cement production

A key challenge for global decarbonisation of cement production is the capital investment required to retrofit carbon capture and storage technology. The scale of this challenge depends on the individual company and country, according to local and regional opportunities and incentives. Local conditions are a driving force for controlling the rate and extent of decarbonisation, since variations in government regulations, electricity mixes, market sizes, raw material availabilities, and logistics influence the operational conditions in each production facility. Construction practices and acceptance, including standards, are additional parameters that differ from region to region and influence the adoption of novel and alternative composite Portland cements. These issues make global decarbonisation of cement production a complex task (IEA, 2018).

5.4 Conclusions

This chapter has shown that PC clinker substitution alone could reduce CO_2-eq. emissions per kg cement by 42% between 2020 and 2050, whereas alternative fuels (25%) and kiln types (12%) also reduce CO_2-eq. emissions but to lesser extents. Combining all three individual measures leads to a 58% reduction in CO_2-eq. emissions per kg cement by 2050 relative to its value in 2020 (i.e., 0.76 kg CO_2-eq. emissions per kg cement). This level of CO_2-eq. emissions reduction is sufficient to meet the targeted 24% reduction in line with the Paris Agreement and is achievable irrespective of the technology roadmap followed: IEA/CEMBUREAU or GCCA (minor difference ~5% in CO_2-eq. emissions). These results involve significant increases in

use of biomass fuels, and an area for future research is to investigate the effect of this on other impact categories, particularly land use.

CO_2-eq. emissions from cement production could be further reduced by combining individual decarbonisation measures with CCS. Implementing all three individual decarbonisation measures and CCS (amine scrubbing) together could lower CO_2-eq. emissions to as low as 0.09 kg CO_2-eq. emissions per kg cement in 2050. Cement production with amine scrubbing was found to have lower CO_2-eq. emissions than oxy-fuel technology regardless of fuel type, while the addition of a CHP plant for CCS led to an increase in CO_2-eq. emissions for both (amine scrubbing and oxyfuel) CCS technologies.

Accounting for future changes in the electricity mix in LCA studies of cement production, which were modelled here by changing the background database to reflect the general trend in decarbonising the electricity mix over time, is significant since it reduces calculated CO_2-eq. emissions per kg cement by ~10%. The importance of a prospective LCA approach through scenario analysis of cement-related technology developments and additional changes in the economic system, such as the energy transition, in assessing the future decarbonisation potential of conventional cement and concrete production with emerging technologies is highlighted.

Chapter 6. Environmental and mechanical performance of composite BR Portland cements

This section builds upon the previous ex-ante LCA presenting the environmental performance of novel cementitious material incorporating BR treated at a projected industrial scale. This chapter examines the influence of three BR treatment technologies, notably co-calcination, vitrification, and reductive smelting. Inventory data for co-calcination and reductive smelting were provided by project partners, while the vitrification technology was modelled using the upscaling methodology described in Chapter 4. Cement formulations incorporating processed BR were also assessed, using clinker production data from Chapter 5. Mortar-level assessments were performed to explore the influence of BR technology and cement mix formulations affect the climate change impact when mechanical properties are considered.

6.1 Background and motivation

BR utilisation in cementitious materials has received considerable attention as a management option, offering the potential to both reduce the stockpile and thus environmental risk of BR, and CO_2 emissions of conventional cement (Evans, 2016). Extensive research has focused on investigating the reactivity of modified BR as either a clinker in specialty cements or as an SCM in composite Portland cements (International Aluminium Institute, 2020c, 2020b, 2020a) at clinker replacement ratios of up to 50 wt.%.

The use of BR in cement and concrete has been explored as a sustainable alternative to traditional clinker, with most studies concluding that increasing unmodified BR content generally reduces compressive strength. For instance, replacing up to 25% of clinker with unmodified BR can reduce strength by up to 30%, with similar reductions observed across replacement levels of 5%, 10%, 15%, 20%, and 25% (Nikbin et al., 2016; Ulubeyli et al., 2017). At even higher BR levels (>30%) significant strength losses are consistently reported, supporting findings that strength declines when BR content exceeds 5% (Danner and Justnes, 2020; Ortega et al., 2019; Manfroi et al., 2014). Combining unmodified BR with supplementary materials, such as calcined BR, fly ash, clay, or limestone, has shown the potential to mitigate strength loss. For example, a mix with 20% BR, 20% clay, 5% limestone, and 55% Portland

cement retained 80% of the strength of pure cement after 28 days (Vladić Kancir and Serdar, 2022). While higher BR levels, such as 50% (combining both calcined and unmodified BR), resulted in a 36% strength reduction, the performance was still considered satisfactory for non-structural applications (Ribeiro et al., 2011). These studies suggest that higher BR replacement ratios are feasible when combined with SCMs (Ribeiro et al., 2011; Viyasun et al. 2021). At lower replacement levels (less than 5%), BR causes minimal strength loss or even slight improvements (Senff et al., 2014). Long-term studies (180 and 360 days) showed that roller-compacted concrete and self-compacting concrete with up to 50% BR can exhibit superior strength and elasticity over time (Yao et al., 2013;Liu and Poon., 2016; Tang et al., 2018) Variations in performance are attributed to differences in BR's physical properties, chemical composition, and alkalinity, which affect its role in cement hydration and strength development.

Despite the mixed results with unmodified BR, promising results have been observed when BR is modified (Ribeiro et al., 2011). Processed BR, especially when subjected to thermal treatments such as calcination, has demonstrated the potential to enhance performance. Studies have highlighted that calcination at 600°C, particularly in combination with materials like coal gangue, blast furnace slag, and Portland cement clinker, achieved compressive strengths between 37.6 MPa and 49.5 MPa, which are comparable to ordinary Portland cement (Zhang et al., 2009). Optimisation of calcination parameters further supported these findings, with BR calcined at 600°C for three hours, reaching 34.15 MPa, suitable for 32.5N class cement (Liu et al., 2011). However, the reactivity of BR decreases when calcination temperatures exceed 800°C, limiting clinker replacement to 12%, although compressive strengths still ranged from 42.2 MPa to 51.2 MPa (Danner and Justnes, 2020). Co-calcination with kaolinite at 750°C has further demonstrated improved performance, enabling up to 30% clinker replacement while achieving compressive strengths comparable to ordinary Portland cement. The formation of metakaolin and sodium aluminosilicate phases was found to enhance reactivity by binding free sodium (Danner & Justnes, 2020). Studies on vitrified BR also yielded impressive results, with alkali-activated mortars achieving strengths of 110 MPa at seven days and 131 MPa at 28 days (Giels et al., 2022).

While BR appears to be a promising material for cement, showing good results in compressive strength, it is also important to consider its environmental impact to understand its full potential for sustainable construction. Several studies have demonstrated that incorporating BR into

cement and concrete production significantly reduces environmental impacts, particularly in terms of greenhouse gas emissions. For instance, varying the (unmodified) BR replacement ratio between 0% and 25% in lightweight concrete production resulted in a maximum 26% reduction in climate change impact, decreasing from 556.8 kg CO_2 eq per m^3 to 409.9 kg CO_2 eq per m^3, along with notable reductions in emissions such as CO (32.5%), NOx (17.1%), Pb (31.8%), and SO_2 (22.4%) due to reduced clinker pyro processing (Nikbin et al., 2016). When (unmodified) BR was combined with silica fume (SF) at increasing replacement levels, up to 30% reductions in CO_2 emissions were observed (Ghalehnovi et al., 2022). Moreover, incorporating calcined BR at a 15% replacement ratio in composite cement has been shown to significantly reduce CO_2 emissions by 124.11 kg per ton of cement (Di Mare et al., 2021). Therefore, while BR incorporation can affect compressive strength, its potential to reduce environmental impact makes it a valuable material for future sustainable construction practices.

The integration of environmental and mechanical performance parameters has gained increasing attention, encouraging more comprehensive assessments of blended or composite cements. Damineli et al. (2010) introduced the concept of eco-efficiency, investigating binder and carbon intensity coefficients. These coefficients measure the binder required to achieve 1 MPa of compressive strength (kg binder per m^3 MPa) and the associated CO_2 emissions (kg CO_2 per m^3 MPa). These indicators have been applied to various construction materials, from mud bricks to recycled aggregate concrete, offering comparisons with traditional Portland cement (Scrivener et al., 2018; Zawal and Grabiec, 2022). For example, using recycled aggregates and SCMs like fly ash not only enhanced compressive strength but also reduced the binder and carbon intensity indicators (Zawal & Grabiec, 2022). Similarly, alternative materials like rammed earth and mud bricks showed lower carbon intensity compared to conventional Portland cement (Scrivener et al., 2018). Further advancements in sustainability indicators have targeted specific applications, such as evaluating self-consolidating concretes with or without steel fibers (Fantilli and Chiaia, 2013). Building on these assessments, Miranda de Souza et al. (2021) developed a desirability function tool to expand the environmental performance analysis by incorporating additional indicators such as total primary energy, raw material usage, recycled content, water consumption, and global warming potential. This more comprehensive approach allows for detailed evaluations of sustainability in construction materials. Additionally, Gettu et al. (2018) proposed integrating durability considerations, such as chloride diffusion and carbonation, alongside CO_2 emissions and energy consumption, into sustainability assessments. To further enhance the assessment of sustainable concrete, Cordoba

et al. employed both Damineli et al. (2010) and Gettu et al. (2018) sustainability parameters, to analyse the compressive strength and durability of concrete with and without SCMs. Their study utilised multiple functional units to capture variations in concrete mix composition (Miller et al., 2015; Panesar et al., 2017). Moreover, indices combining embodied environmental impacts with mechanical properties, such as compressive strength, have been used to optimise the mix design of self-compacting concrete (Long et al., 2015).

Despite these advances, research on the assessment of modified BR as a cementitious material remains limited, particularly in terms of its environmental evaluation. Most studies either focus on mechanical or environmental performance separately, likely due to a lack of interdisciplinary collaboration. While aspects such as the influence of various processing technologies on the overall BR performance are not considered. Many methods, such as calcination and vitrification, have only been tested at laboratory or pilot scale, where energy use and emissions can differ considerably from industrial applications. Bridging this gap is necessary not only to assess the environmental benefits of BR incorporation but also to determine if it can meet or exceed the mechanical properties of conventional cement for large-scale use.

This chapter addresses a critical gap in the literature by evaluating the climate change contribution of emerging BR processing technologies (calcination, vitrification, sintering and smelting) at an industrial scale for their use as cementitious materials (SCM). While these technologies have been primarily tested at the laboratory and pilot scales, their climate change impact at a larger scale remains underexplored. Further analysis includes integrating both environmental and mechanical performance by adopting an eco-efficiency perspective and evaluating key indicators such as binder intensity (kg of binder per MPa) and carbon intensity (kg CO_2 per MPa). By assessing the environmental performance across multiple life stages; from BR processing (calcination, vitrification, and reductive smelting) to cement blending and mortar production, this chapter provides a holistic understanding of the trade-offs and synergies between climate change impact and mechanical strength. The findings contribute to the development of more sustainable cementitious materials and offer strategies for optimising BR incorporation in cement production, improving both environmental and mechanical performances.

6.2 Methodology

6.2.1 Description of Bauxite residue technologies

The development of calcination, vitrification, and smelting technologies for BR presents a promising solution for producing high-reactivity SCMs, capable of replacing up to 30% of Portland cement and serving as alternatives to conventional SCMs like fly ash and GGBFS, which are facing a supply shortage. constraints. While co-calcination offers a simpler method for SCM production, vitrification and reductive smelting provide greater flexibility by allowing material adjustments during processing, enabling the creation of specialty cements tailored to industry needs. Further research into these technologies could expand BR's potential in cement production, addressing performance requirements and environmental objectives.

Figure 6-1 presents a process flow diagram outlining the industrial-scale processing steps for the three BR treatment technologies: co-calcination with kaolin clay, vitrification, and reductive smelting.

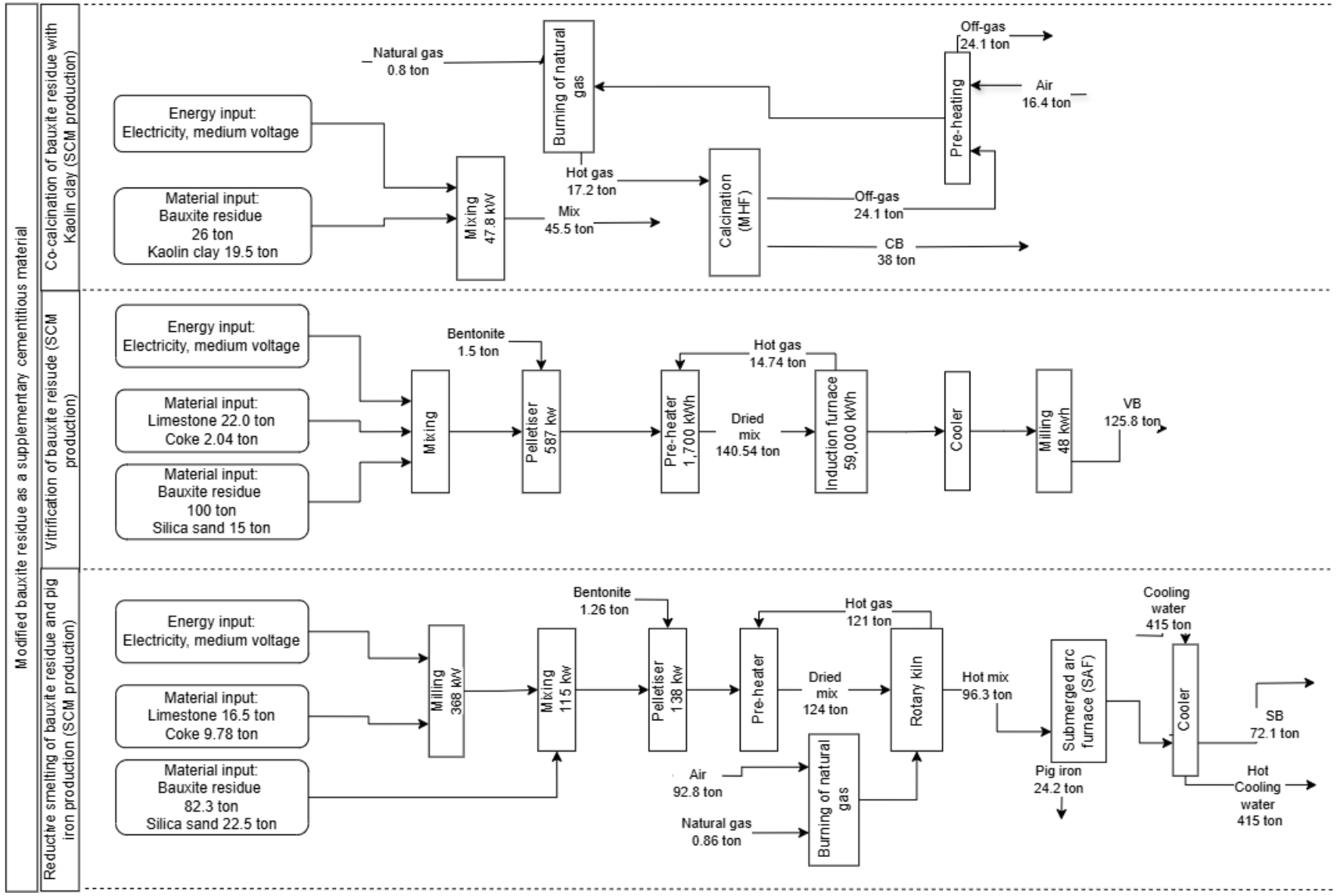


Figure 6-1: Process flow diagram illustrating the processing technologies for producing modified BR suitable as an SCM

6.2.1.1 Co-calcination of BR

In the calcination process, 57 wt.% dry BR is mixed with 43 wt.% dry kaolin clay, both classified as fine materials with particle sizes ranging from 2 to 6 μm, so no additional milling is required. The mixture is calcined at 700°C in a Multiple Hearth Furnace (MHF), to provide temperature control and efficient handling of dusty materials. The process incorporates heat recovery from off-gases, preheating the combustion air for improved energy efficiency. After calcination, the material is cooled and ready for use as an SCM, replacing up to 30% of OPC. Dust emissions from the furnace are filtered and reintroduced into the system, ensuring efficient material use.

6.2.1.2 Vitrification

In the vitrification process, 74.7 wt.% dry BR is mixed with 11.2 wt.% silica, 12.2 wt.% milled limestone, and 1.9 wt.% milled coke. The mixture is pelletised, and the pellets are transferred to a direct-fired rotary kiln with a Lepol preheater to reduce dust and increase mechanical strength before entering the kiln. Inside the kiln, the material is heated to 1200-1350°C in a slightly reducing atmosphere. The process aims to produce a vitrified material with a targeted composition of 18% CaO and 20% SiO_2, promoting the formation of an alkaline glassy phase. This material can replace up to 30% of Portland clinker in cement as an SCM. Off-gases are filtered, and heat recovery is utilised to dry BR before the vitrification process, improving overall energy efficiency.

6.2.1.3 Reductive smelting

In the reductive smelting process, 62.8 wt.% dry BR is mixed with 12.6 wt.% limestone, 17.1 wt.% silica, and 7.5 wt.% coke. The mixture is pelletised, and the pellets are first transferred to a Lepol preheater, where the product is dried and hardened, exiting at 250°C. The preheated material is then transferred to a rotary kiln, where it is further heated to 900°C, allowing for iron pre-reduction. Finally, the preheated material is transferred to a submerged arc furnace (SAF), where it is smelted at 1580°C, resulting in the separation of pig iron and reactive, glassy slag. The slag is air-cooled and milled to a fine particle size, similar to GGBFS, making it suitable for use as a SCM, which can replace up to 30% of OPC.

For a more detailed technical overview of each process, refer to Chapter 2, Section 2.6.1.

6.2.2 Life cycle assessment

6.2.2.1 Goal and scope

The aim of this chapter was to evaluate the climate change impacts of emerging technologies i.e., calcination, vitrification, and smelting, for processing BR into a cementitious material (SCM). The analysis focused on two key indicators: the climate change, quantified as CO_2-equivalent emissions, due to its dominant contribution to the carbon footprint of cement production; and cumulative energy demand (CED) used to assess the embodied energy of BR-incorporated cementitious materials. The assessment was performed at an industrial scale using process modelling.

To reflect the different stages along the cementitious value chain, three functional units were applied: (i) 1 kg of processed BR suitable for use as an SCM at the factory gate, (ii) 1 kg of composite cement with BR-incorporated SCM, and (iii) 1 kg binder per cubic meter of mortar. This staged approach enabled a more consistent comparison from BR treatment stage to its integration in final cementitious products. A cradle-to-gate system boundary was adopted to focus the analysis on production and formulation stages, excluding downstream use and end-of-life phases.

The system boundary encompasses foreground activities including raw material extraction and preparation for BR processing technologies, the individual unit operations within each technology (calcination, vitrification, and smelting), raw material extraction and preparation for clinker production, cement and mortar formulation, and transportation (as seen in Figure 6-2). End-of-life processes such as casting, construction, use, deconstruction/demolition of the structure, and recycling and final disposal of concrete were excluded due to their high uncertainty and lack of representative data. In addition, these stages are highly dependent on context-specific factors (e.g., building design, service life, and regional recycling practices), which are beyond the scope of this assessment. Emissions from the transportation of materials

from production/extraction plants to the ready-mix concrete plant or casting site were also not included.

Europe (average European markets) was selected as the geographical scope due to the high availability of data and significant potential for BR utilisation in this region. This selection aligns with the Global Cement and Concrete Association (GCCA) tool (cement production database), ensuring consistency with the inventory used. An average European market was selected to maintain a generic scope, rather than focusing on company or site-specific data.

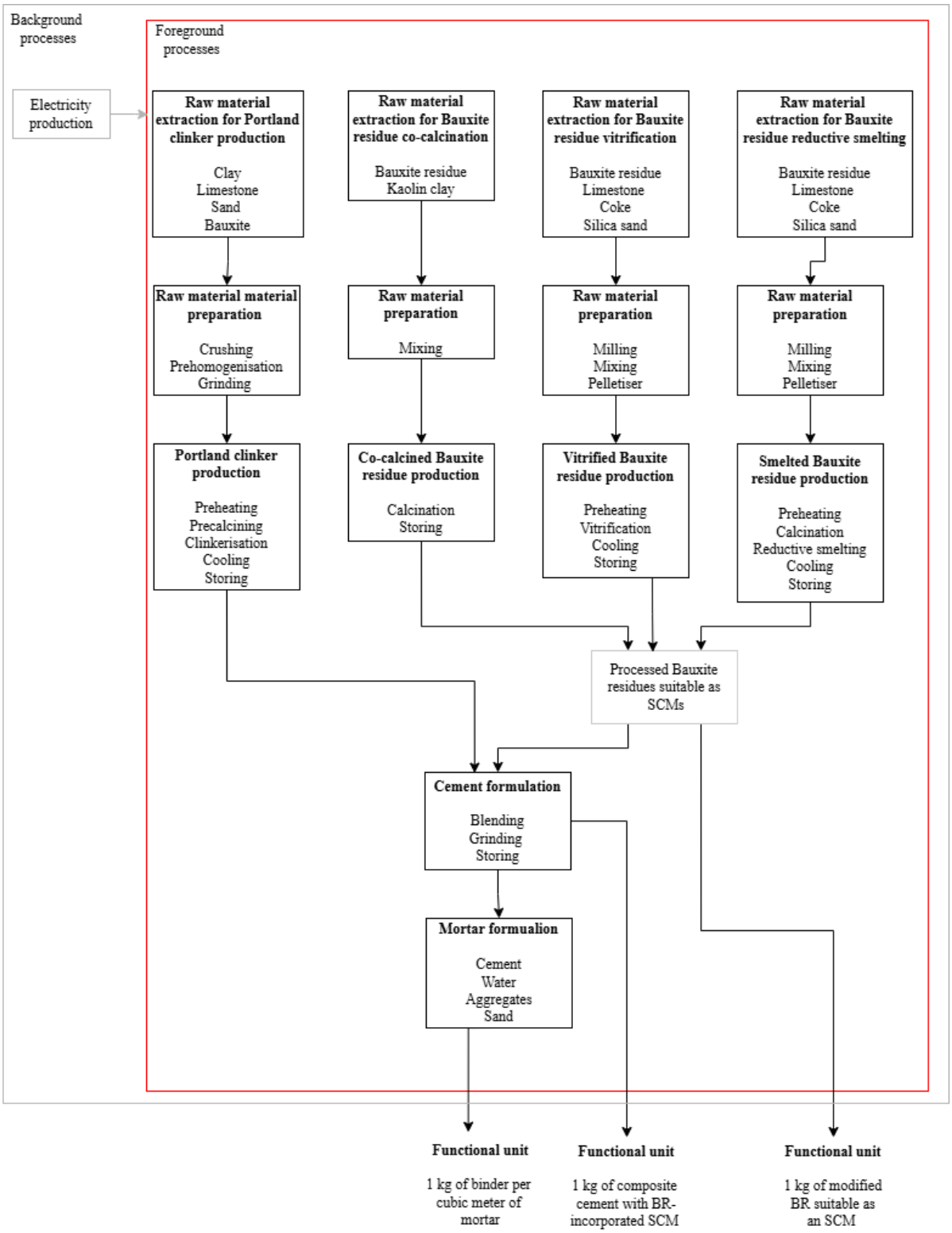


Figure 6-2: System boundary outlining the main life cycle stages for processing BR into suitable SCMs using co-calcination, vitrification, and smelting. It includes raw material extraction, BR treatment, Portland clinker, cement and mortar production.

6.2.2.2 Life cycle inventory

Foreground data including the material and energy flows for the BR technologies; co-calcination and reductive smelting at industrial conditions, had been calculated using the HSC software based on process modelling and were provided by project partners (Konlechner et al., 2023). For the vitrification technology, the industrial scale inventory was according to the process model generated in Chapter 4, in consultation with the technology developers. The background data, such as energy source materials and generated emissions, were modelled based on the ecoinvent database version 3.6.1. Material and energy balances for BR-incorporated cementitious materials are detailed in Appendix C; co-calcination (Tables C-1); vitrification (Table C-2); and reductive smelting (Table C-3). While compressive strength data for mortar mixtures are provided in Table C-5 to C-8.

Cement formulations incorporating BR-based SCMs were modelled using clinker production data according to the methodology described in Chapter 5. Input data for the inventory analysis were obtained from the GCCA environmental product declaration tool (WBCSD, 2016), which relies on data from the Getting the Numbers Right (GNR) database (Dauriat et al., 2019). The GCCA data cover raw material extraction and preparation, transportation, clinker production, and cement formulation. The open-source LCA software Activity Browser (v2.8) was used with the ecoinvent database (v3.9.1 cut-off system) as the background database.

6.2.2.2.1 Composite cement formulation

In this chapter, three BR-incorporated composite cements were examined, each utilising approximately 30 wt.% of BR as an SCM. The specific mix formulations for the BR-incorporated composite cements, including co-calcined (CBR), vitrified (VBR), limestone vitrified (LVBR), and low-iron smelted (SBR) bauxite, are detailed in Table 6-1. Additionally, the mix formulation of OPC (CEM I 42.5 N, according to EN 197–1) is provided as a reference.

Table 6-1: Formulation of BR-incorporated composite cements. CBR is co-calcined processed bauxite/kaolinitic clay. VBR is vitrified processed bauxite. LVBR is limestone vitrified processed bauxite. GGBR is low iron smelted processed bauxite.

	Composite cement composition (wt.%)				
Composite cement	**Portland clinker**	**Processed BR**	**Gypsum**	**Ground limestone**	**GGBFS**
CEM II/B-B with CBR	65	30	5	0	0
CEM II/C (B-LL) with CBR	50	45	5	0	0
CEM II/B-B with VBR	64.6	29	6.4	0	0
CEM II/C (B-LL) with VBR	50.3	29	5.7	15	0
CEM II/B-B with SBR	66.5	30	3.5	0	0
CEM II/B	68.4	0	3.6	22.4	5.6
CEM I	95	0	5	0	0

6.2.2.2.2 Allocation

A cut-off allocation approach was employed to manage the use of BR, classified as a waste material, focusing solely on the impacts associated with its treatment into a cementitious material. This approach excludes the environmental burdens of primary production, such as alumina production, meaning raw BR is introduced into the treatment process as a burden-free material.

If BR was to gain economic value through commercial-scale use by 2050, it would shift from a waste material to an industrial by-product, necessitating the allocation of upstream emissions. To account for this, an economic allocation was performed for untreated BR to assess any upstream burdens from alumina production. Due to the high price of aluminium, even with a 200% increase in the price of BR, the impact would still be less than 0.01. The results of this analysis are presented in Appendix C, Section C.2, Table C-4. Additionally, economic allocation was applied to the iron-free smelting process, which produces both pig iron and smelted BR-derived cementitious materials. This allocation divides environmental impacts based on the economic ratio between the main product (pig iron) and the by-products (e.g., smelted BR and CA cement) (Shobeiri et al., 2021).

The economic allocation factor, c_e, is calculated as:

$$c_e = \frac{(u_p \cdot m)by-product}{(u_p \cdot m)by-product + (u_p \cdot m)main-product} \quad \text{(Eq. 6-1)}$$

Where c_e is the economic allocation factor, m is the mass quantity, and u_p is the unit price for the main product (pig iron) and the by-product (BR product), respectively.

Reductive smelting technology is a multi-functional systems that produce both a BR-incorporated cementitious material (SBR) and pig iron. Economic allocation was performed based on estimated market prices (e.g., SBR, pig iron) to distribute CO_2 emissions between the production of pig iron and the BR-derived cementitious materials (Table 6-2).

Table 6-2. Estimated market prices for product/by-product flows according to the pre-feasibility study (Efstathiou et al., 2022)

Flow	**Iron-free BR-derived SCM**	c_e
	Market price (€/t)/(€/kWh)	
Calcium aluminate slag	240	0.47
Pig iron	1000	0.53

6.2.2.2.3 Mortar formulation

Further analysis was conducted on CEM II/B-B with CBR, CEM II/B-B VBR and CEM II/B-B with SBR, and CEM II/B-LL with VBR, incorporating environmental performance and mechanical indicators, with a focus on the compressive strength of mortars. These cement types were chosen due to their larger market share, as SCMs are commonly used to replace conventional cement in general applications, offering a great potential for emissions reduction. Table 6-3 presents the mix compositions of conventional cements used for mortar production, serving as reference formulations.

Table 6-3: Composition of conventional cements where limestone (L), fly ash (V), and blast furnace slag (S)

	Composite cement composition (wt.%)				
Composite cement	**Portland clinker**	**Fly ash**	**GGBFS**	**Ground limestone**	**Gypsum**
CEM II/A-V	80	15	0	0	5
CEM II/A-L	75	0	0	20	5
CEM II/A-S	75	0	20	0	5
CEM II/B-V	68	27	0	0	5
CEM II/B-L	65	0	0	30	5
CEM II/B-S	65	0	30	0	5

The mortar samples were prepared according to EN 196–1, as seen in Table 6-4 (British Standards Institution., 2005). The compressive was tested according to EN 196–1 at 28 days. Further details on the mix design samples and compressive strength can be found in Appendix C, Section C.3, Table C5 to Table C8.

Table 6-4: Mortar composition according to EN 196–1

Component	**Mortar composition**
Cement (binder)	450 g
Water	225 g (water to binder ratio of 0.5)
Standard CEN sand	1350

6.2.2.4 Life cycle impact assessment

The life cycle impact assessment was conducted using the global warming potential (GWP100) indicator from the ReCiPe 2016 midpoint method (hierarchist perspective), and the cumulative energy demand. These indicators were selected due to their high relevance to cement production, and because they reflect the major environmental hotspots (energy use) associated with BR treatment technologies, as identified in prior studies. The open-source Activity Browser software was used to compile the LCI.

6.2.2.3 Environmental sustainability indicators

To provide a more comprehensive assessment of sustainability, indices proposed by Damineli et al. (2010) have been used to measure the binder and CO_2 intensity of mortars, respectively. These indicators were calculated according to the compressive strength of mortar samples at 28 days according to equations [6-2] and [6-3].

The binder (*bi*) intensity index used here is defined as the binder content needed to achieve a compressive strength of 1 MPa (kg binder per m^3 MPa mortar), which reflects the efficiency of binder use. A lower binder intensity index would indicate greater efficiency, as higher compressive strengths are achieved with less binder material.

$$B_i = \frac{b}{cs_{28}} \quad \text{(Eq. 6-2)}$$

The carbon intensity index determines the amount of CO_2 released during the binder production of 1 MPa compressive strength (kg CO_2 eq. per m^3 MPa). In this study, c_i is determined using p as the compressive strength (c_s) at 28 days.

$$C_i = \frac{Em_{co_2}}{cs_{28}} \quad \text{(Eq. 6-3)}$$

6.3 Results and discussion

6.3.1 Processing technology

Figure 6-3a presents the environmental impacts of processed BR compared to commercially available SCMs and conventional PC clinker production. Reductive smelting and vitrification, led to the highest climate change impacts, with CO_2-eq. emissions of 0.47 kg per kg of smelted (SCM) BR and 0.45 kg per kg of vitrified BR, respectively. For both methods, the dominant contributor to emissions stemmed from direct CO_2 release during raw material processing, accounting for 62% in reductive smelting and 57% in vitrification. Heat demand is the second-largest contributor, accounting for 27% in smelting and 34% in vitrification. By comparison,

calcined clay performs significantly better, with an impact of 0.28 kg CO_2-eq. per kg, driven primarily by CO_2 emissions during calcination (73%), released during fossil fuel (petroleum coke) combustion. Co-calcination of BR results in even lower emissions at 0.20 kg CO_2-eq. per kg, with the primary contributions associated with upstream kaolin clay processing (58%) and thermal demands (42%).

Figure 6-3b reports the cumulative energy demand (CED) across these processes, which depends not only on the energy intensity of the process technology and the energy source, but also on the raw materials used. Smelting and vitrification resulted in the highest CED due to significant heat requirements, with operating temperatures of 1300 °C for vitrification and 1550 °C for smelting. Reductive smelting shows the highest energy demand at 4.7 MJ-eq., where 54% of the demand comes from processing petroleum coke used as a raw feed and 41.2% from natural gas for heat production. Similarly, in vitrification, 64% of the energy demand is for thermal treatment and 22.8% for petroleum coke processing, though less petroleum coke is used compared to smelting. Calcination processes showed similar energy demands (~2.5 MJ-eq. per kg) due to comparable operating conditions (700 °C). The key difference lies in the energy sources; natural gas for BR calcination and petroleum coke for clay calcination. Lower cumulative energy demands are observed for SCMs like GGBFS, fly ash, and silica fume, which avoid heat treatment as a treatment technology.

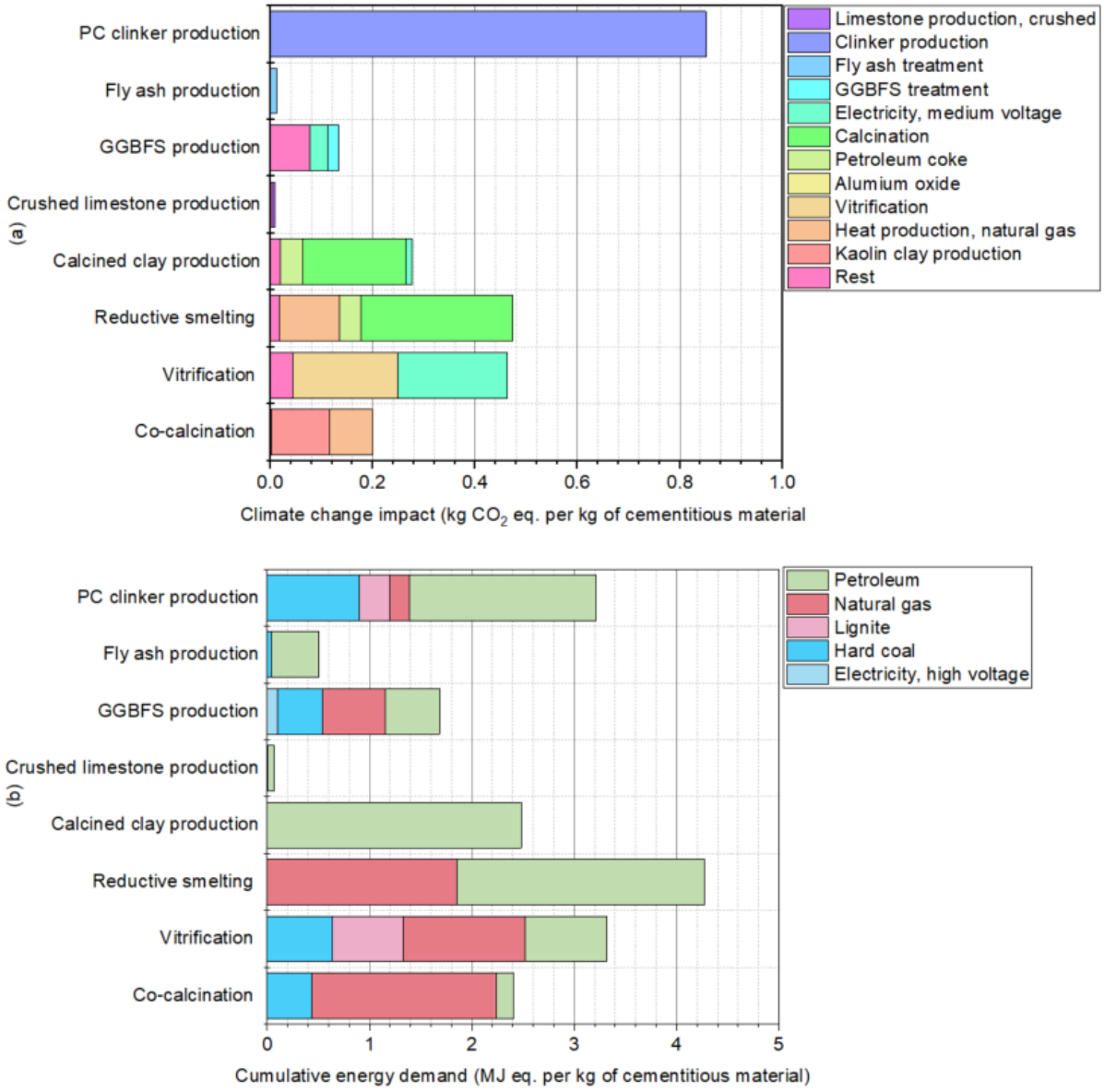


Figure 6-3: Split up of (a) climate change (kg CO_2 emissions per kg of cementitious material) and (b) cumulative energy demand (MJ per kg of cementitious material)

6.3.2 Composite cement

The OPC (CEM I) showed the highest environmental burden, emitting 0.83 kg CO_2 eq. (Figure 6-4a), which is consistent with previous studies (Gettu et al., 2019; Miller, 2018; Nikbin et al., 2016). This is largely due to its high clinker-to-cement ratio (95wt.%), as clinker production is the most carbon-intensive phase of cement manufacturing. CEM II/B, used as the reference with a 32% clinker replacement, resulted in a climate change impact of 0.68 kg CO_2 eq., primarily driven by clinker production, while the impact from limestone is minimal. Similar climate performances were observed in other CEM II/B-B variants with slightly varying replacement ratios. CEM II/B-B with CBR has the lowest impact at 0.60 kg CO_2 eq. due to its lower clinker content. However, for other composite blends (i.e., CEM II/B-B with SBR and CEM II/B-B with VBR), the marginally higher replacement ratios are counterbalanced by greater environmental impacts from binder (BR) processing, resulting in comparable emissions (~0.68 kg CO_2 eq.). The lowest climate change impacts were seen in composite cements with higher clinker replacement, such as CEM II/B-LL with VBR (0.56 kg CO_2 eq.) and CEM II/B-LL with CBR (0.52 kg CO_2 eq.). These composite cements outperform the conventional CEM II/B by 17.6% and 23.5%, respectively, though the slightly higher emissions in CEM II/B-LL with VBR are attributed to the direct emissions from the vitrification process (from coke and limestone decomposition).

In terms of CED, CEM I exhibited the highest demand at 4 MJ eq., which is expected due to the energy-intensive nature of clinker production, requiring high temperatures of 1450°C (Figure 6-4b). The high energy demand in clinker production was driven by the use of fossil fuels such as coal, lignite, and petroleum coke. Similar energy demands were observed in CEM II/B-B with SBR due to the additional energy required for BR, where smelting is operating at 1550°C. In composite cements like CEM II/B-B with B-LL and CEM II/B-B with CBR, which include a 30% clinker replacement, the CED remained consistent with the reference at 3.4 MJ eq. However, increasing the clinker replacement to 45% in BR co-calcination reduced the energy demand (to 3.3 MJ eq.) slightly due to the additional energy demands from BR co-calcination and kaolin clay production. The lowest CED was observed in CEM II/B-LL with a 45% clinker replacement, resulting in 2.8 MJ eq., primarily due to the use of limestone as the replacement material, which had a minimal energy demand.

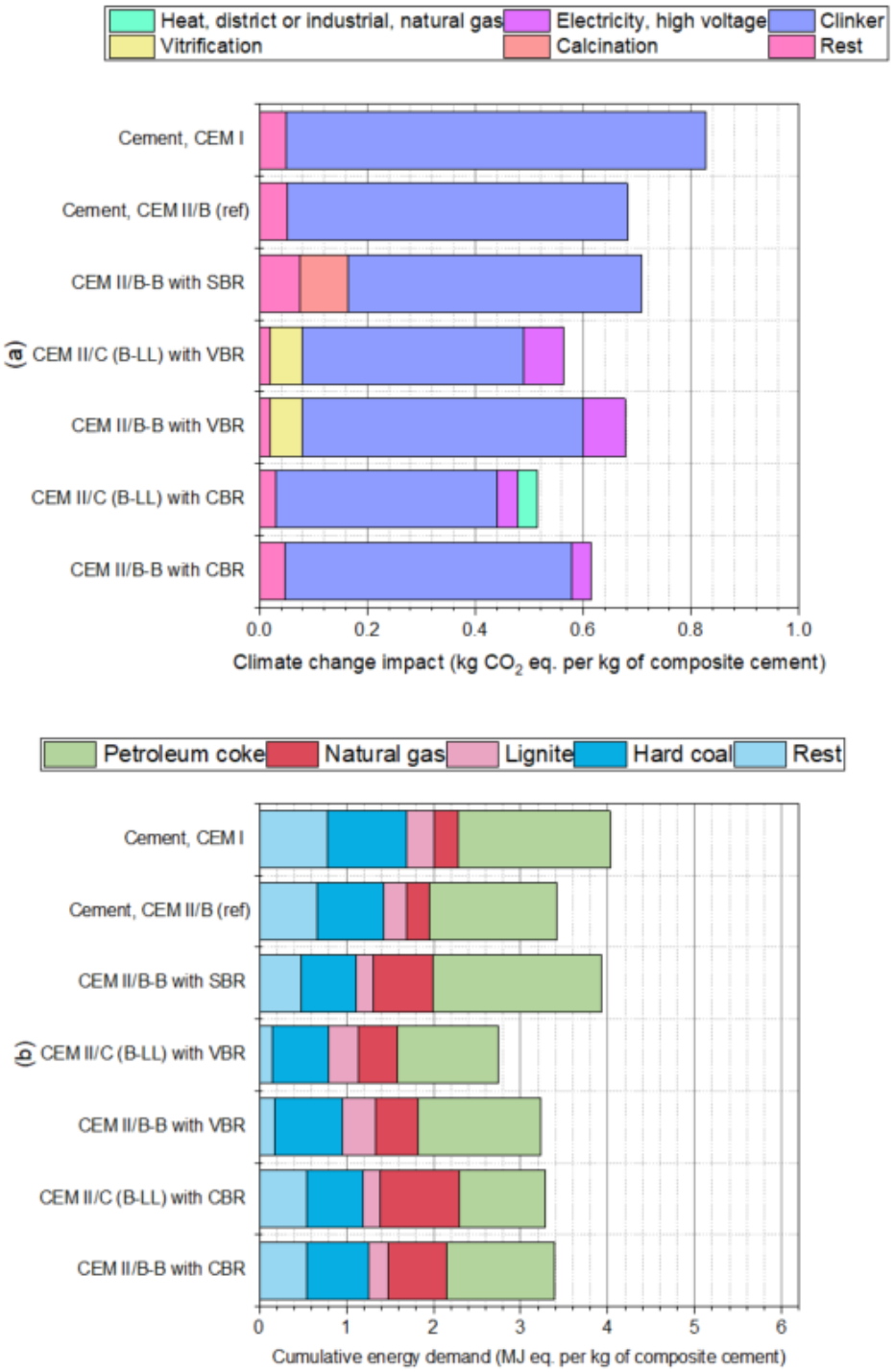


Figure 6-4: Split up of (a) CO_2-eq. emissions and (b) energy demand per kg of composite cements with BR-incorporated SCMs

6.3.3 Eco-efficiency indicators

In this study, the binder content of 450 kg per mortar prism was kept constant, with any variation in the binder intensity index to be directly attributed to differences in compressive

strength. The highest binder intensity was observed in mortar containing CEM II/B-B with SBR, resulting in a binder index of 14.5 kg binder per unit of mortar MPa, due to its low compressive strength of 31 MPa (Figure 6-5a). Mortars incorporating co-calcined BR demonstrated a clear reduction in binder intensity as kaolin clay content increased. As kaolin increased from 0% to 30%, the binder intensity reduced by 26%, reflecting a significant improvement in strength; hence, in co-calcination, the binder intensity index can be improved by increasing kaolin content without sacrificing compressive strength. In terms of VBR, the fineness was identified as a key factor, with a fineness of 4300 cm^3/g yielding the lowest binder intensity at 7.38 kg per unit of mortar MPa. The inclusion of limestone further increased the compressive strength, highlighting the importance of optimising both fineness and material composition to enhance mortar performance.

In terms of carbon intensity index, mortars incorporating SBR, VBR and CEM II/B-L exhibited the highest carbon intensity indexes at 12.3, 11.6 and 12.4 kg CO_2 eq. per unit of mortar MPa, primarily due to their lower compressive strengths and carbon-intensive production technologies (Figure 6-5b). Conversely, mortars containing co-calcined BR demonstrated a significantly lower carbon intensity index, with a clear decreasing trend as compressive strength improved. The lowest carbon intensity index of 5.5 kg CO_2 eq. per unit of mortar MPa was observed for a 55 MPa mortar. This trend was similarly observed in conventional cements, where CEM II/B and CEM II/A mortars also displayed a reduction in carbon intensity as compressive strength increased. However, conventional cements generally exhibited higher carbon intensities. Notably, CEM II/B-VBR_{3700} and CEM II/B-SBR, despite achieving comparable compressive strengths to several BR-incorporated mortars such as $LVBR_{4800}$, $LVBR_{3700}$, CBR with 90% BR and 10% K, CBR with 80% BR and 20% K, and VBR_{4800}, underperformed in terms of carbon intensity, with values reaching 11.6 and 12.3 kg CO_2 eq. per unit of mortar MPa. This highlights the potential of BR-incorporated mortars as viable alternatives for applications requiring compressive strengths exceeding 42.5 MPa. Mortars with VBR and limestone showed a higher consistency in compressive strength (ranging from 46 to 52 MPa), while VBR without limestone exhibited greater variability, ranging from 32 to 62 MPa. Four mortars achieved carbon intensities below 6.25 kg CO_2 eq. per unit of mortar MPa: CBR with 80% BR and 20% K, CBR with 70% BR and 30% K, $LVBR_{4800}$, and VBR_{4300} while maintaining compressive strengths between 52 and 62 MPa. VBR_{4300}, in particular, offers significant potential for reducing carbon emissions while maintaining satisfactory compressive

strength, making it a viable option for both high-strength applications and environmentally sustainable designs.

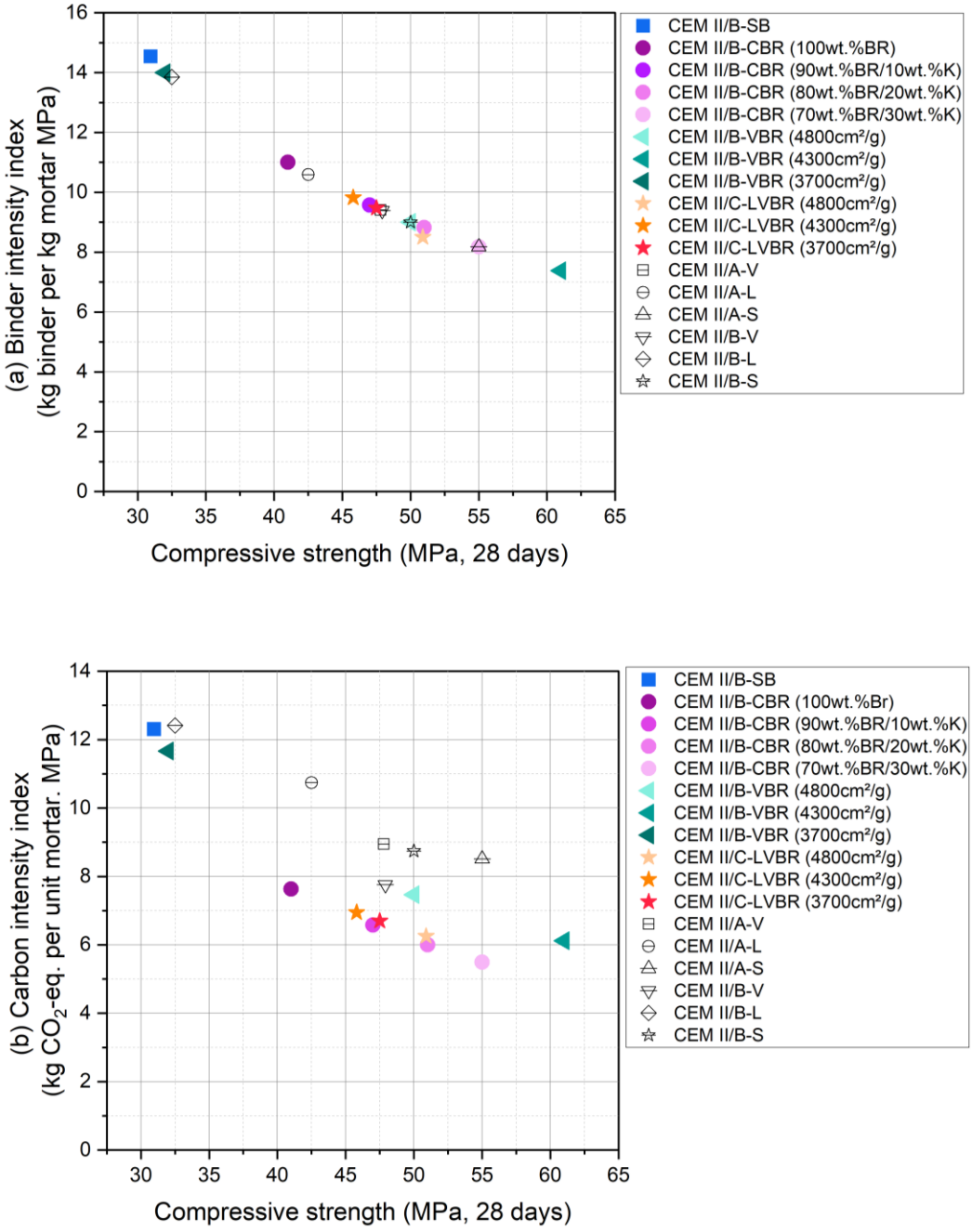


Figure 6-5: Split up of (a) binder intensity (bi) versus 28-day compressive strength for mortar mixes with composite cements and (b) carbon intensity (ci) versus 28-day compressive strength for mortar mixes with composite cements. SBR is represented in blue, while CBR, VBR, and LVBR are shown using colour gradients: pink for CBR, green for VBR, and orange for LVBR. The colour gradients indicate variations in the SCM: for CBR, the pink gradient reflects changes in raw feed composition affecting CBR behaviour, while for VBR and LVBR, the green and orange gradients represent variations in material fineness for VBR and LVBR, respectively. Conventional materials are depicted with black outlines (unfilled markers).

6.4 Conclusions

The results from this chapter indicated that several key factors such as total energy requirements, energy sources, raw material composition, and clinker replacement ratio play a significant role in influencing the environmental performance of cementitious materials. At the processing technology level, the production of SCMs through vitrification was shown to have higher climate change impacts (0.45 kg CO_2-eq.) and cumulative energy demand (3.5 MJ-eq.) compared to other methods, such as BR calcination. However, when analysing cement blends, the use of VBR in conjunction with limestone (LVBR) demonstrated a 25% and 5% reduction in climate change impact compared to composite cements containing 29% VBR (CEM II/B-B with VBR) and 30% calcined BR (CEM II/B-B with CBR), respectively. This highlights the synergistic effects of BR-derived SCMs, when combined with additional materials such as limestone.

In mortar production, with a constant binder content, significant variations in compressive strength were observed, ranging from 30 MPa to 60 MPa at replacement ratios of 30% to 45%. Despite vitrification underperforming at both the technology and cement blending stages compared to other alternatives, mortars containing 29% VBR in composite cements outperformed others by achieving lower climate change impacts and higher compressive strengths. This was maintained at a constant climate change impact of 0.64 kg CO_2-eq. per kg of CEM II/B-VBR, showing the importance of additional processing steps, such as milling, to optimise fineness and enhance mechanical properties. Similarly, mortars with CEM II/B-LL containing CBR with varying kaolin content in the raw mix exhibited the synergistic influence of kaolin clay, which further enhanced mechanical performance. Although fineness optimisation had a limited effect on energy demand and climate impact, it significantly improved the mechanical properties.

Therefore, following a more strategic use of BR materials by optimising parameters such as fineness and raw material composition, it can lead to further reduction in the environmental impact of cement production without compromising mechanical strength. Future research should focus on the scalability and long-term durability of BR-incorporated cementitious products to support broader adoption in construction.

Chapter 7. Conclusions

This thesis presented a comprehensive environmental assessment of BR valorisation in novel cementitious materials, focusing on emerging processing technologies and their integration within the cement sector. Using forward-looking LCA approaches, this work addressed the environmental trade-offs, upscaling challenges, and performance of BR-derived material compared to conventional cement types. The research provides insights into the influence of scale effects when assessing emerging technologies and highlights the importance of preliminary LCAs in ensuring comparability and consistency with conventional technologies while accounting for future changes.

Although academic and industrial interest in these areas is growing, integrating environmental and mechanical considerations during the early stages of technology development remains limited. This often results in separate efforts to optimise environmental performance, mechanical properties, durability, and economics, rather than addressing them holistically to inform early-stage decisions. Typically, such optimisation occurs at later stages when the technology has matured, leaving little room for flexibility for significant adjustments. This is primarily due to the challenge of data uncertainty or even the complete lack of data, for instance market price for novel materials, at early development stages.

Chapter 2 began with an extensive literature review of current BR disposal and management practices, identifying the application of BR in construction materials as the most promising path, given its vast availability and the cement industry's need for alternative raw materials due to shortages of currently conventional SCMs like fly ash and GGBFS. Additionally, the review explored how emerging thermal treatment technologies could enhance BR's reactivity as a cementitious material. Despite promising results, little progress has been made in scaling these technologies, with limited data available on their economic and environmental performance.

In response to this gap, Chapter 3 presented the traditional LCA framework and advancements in LCA approaches for assessing the environmental performance of emerging BR technologies and novel cementitious materials. The chapter emphasised the need to expand traditional LCA methods as an iterative design tools, allowing environmental impacts to guide both process development and assessment.

The environmental performance of novel vitrified BR-derived SCMs was assessed in Chapter 4, revealing the potential environmental impacts when implemented in cement. This chapter also explored the challenges of upscaling technologies from the laboratory to the industrial scale, highlighting the sensitivity of the system to various upscaling approaches. The analysis demonstrated that upscaling to pilot and industrial scale significantly reduces the overall environmental impact by approximately 83% and 94%, respectively, compared to the laboratory scale. This indicates how industrial-scale models can offer a more accurate representation of environmental impacts. Additionally, the results show that when appropriate parameters are selected, proxy modelling can perform comparably to regression analysis and more complex process modelling. This supports the use of simpler, less time-intensive approaches in early-stage assessments where data may be limited. While such simplifications reduce complexity and are valuable in early-stage assessments, they also introduce uncertainties that must be carefully considered. This is particularly important given that system hotspots can vary significantly depending on operating and process parameters. Therefore, further research is needed to evaluate the sensitivity of different upscaling approaches to ensure their broader applicability and to establish whether consistent trends can be generalised across technologies.

Building on this, Chapter 5 analysed the future environmental performance of conventional cement technologies by implementing various decarbonisation measures such as increasing Portland cement (PC) clinker substitution, increasing the use of alternative fuels, improving kiln efficiency, and implementing carbon capture and storage (CCS). The findings suggest that while CCS could significantly reduce emissions, increasing the PC clinker substitution with SCMs is the most practical and readily applicable approach, as it alone can reduce CO_2-eq. emissions per kg of cement by 42% between 2020 and 2050. This highlights the market demand for viable, widely available SCMs and the potential role of BR-derived materials in meeting this need. Clinker substitution with SCMs is a practical strategy that can be readily integrated into existing cement infrastructure, enabling its short- to medium-term feasibility. This is important when introducing emerging technologies into a mature market i.e., cement industry. In this thesis, cement production was modelled through 2050 to reflect an anticipated decarbonised scenario, allowing for a more realistic evaluation of BR valorisation at commercial scale. This forward-looking baseline provides a transferable framework for assessing the future performance of emerging technologies across other sectors.

Finally, Chapter 6 posed a critical question regarding the holistic assessment of novel BR cementitious materials. It investigated how combining environmental and mechanical performance indicators transforms the overall evaluation of emerging materials. The analysis revealed that while hotspots like thermal treatment are significant from a technological perspective, their environmental impact is often minor when viewed in the context of the material's broader benefits, such as enabling higher clinker replacement. The study could expand by emphasising how these combined assessments could influence decision-making processes within the cement industry and policy frameworks. This thesis provides a clearer path to commercialisation and industry adoption by showing how a combined assessment of environmental and mechanical properties can promote the use of BR materials.

The main conclusions directly drawn from this thesis are as follows:

- Upscaling BR-derived cementitious technologies to commercial scale significantly improves their environmental performance across 18 indicators, with reductions of up to 94% observed when transitioning from lab-scale to industrial applications.
- Thermal processing technologies such as vitrification are highly sensitive to fuel type, energy input, and regional parameters. The environmental performance of such technologies depends heavily on site-specific factors such as electricity mix, fuel availability, and furnace configuration.
- The decarbonisation of background systems, particularly electricity plays a critical role in evaluating environmental advantages. A 32% reduction in climate impact was observed for BR processing when assuming future grid decarbonisation and 10% for cement production, highlighting the importance of modelling realistic future scenarios not only for the foreground system but for background activities as well.
- Clinker substitution remains the most impactful decarbonisation strategy in the cement sector. Partial replacement of Portland clinker with SCMs can reduced the climate change impact by 42%, highlighting the need for practical, scalable SCM alternatives.
- Incorporating additional decarbonisation measures such as increased alternative fuel use, kiln efficiency improvements, and carbon capture and storage (CCS), could reduce the total cradle-to-gate climate change impact of cement production by up to 88% by 2050.

- The integration of environmental and mechanical performance indicators enables a more comprehensive evaluation of modified BR materials. Both CBR and VBR, when incorporated into mortars, exceeded the performance of conventional cement types. While CBR had a lower climate change impact (by 55%) and CED (by 27%) as a technology, the higher processing intensity of VBR was reflected in better material reactivity, showing that mechanical performance benefits at the application level can outweigh environmental burdens at the processing stage.
- Additional performance factors such as clinker replacement potential, material fineness, and synergy with other SCMs have a significant influence on overall climate performance. These parameters should be considered in early-stage optimisation.
- A combined assessment approach revealed that even where processing conditions vary, the environmental benefits of clinker substitution dominate. For instance, fineness had a minor influence on climate change impact but significantly enhanced mechanical strength, showing how small optimisations can yield better material performance.
- The valorisation of BR as a cementitious material supports circular economy objectives and addresses growing shortages in conventional SCMs such as fly ash and GGBFS, strengthening both environmental and supply resilience.
- The forward-looking LCA framework developed in this thesis is flexible and scalable, allowing for adaptation across geographic regions, energy systems, and BR processing routes, and can be applied to other emerging SCMs.
- Establishing a synergistic circular value chain, where unutilised BR is recovered and valorised offers environmental benefits for both the cement and alumina industries, encouraging cross-sector collaboration in industrial symbiosis.

7.1 Recommendations for future work

In Chapter 4, the upscaling of BR vitrification using various approaches highlighted the significant influence of modelling assumptions, such as the fuel mix, on environmental performance. A key future step for refining these preliminary LCA insights would be to assess the environmental impact at an industrial scale, helping to determine how closely real-world commercial operations align with the estimated models. Additionally, incorporating case studies or pilot projects of successful lab-to-industrial transitions could provide practical insights into the challenges and opportunities of upscaling, further streamlining preliminary

environmental assessments. Another important consideration is the variability in BR quality and composition across different geographical locations. The mineralogy, moisture content, and contaminant levels of BR can significantly influence both process and environmental performance. As such, region-specific optimisation or pre-treatment strategies may be required to ensure consistent product quality when up scaling BR processing technologies.

The results presented in Chapter 6 showed that the primary hotspot in novel composite cements with BR-incorporated SCMs was the production of PC clinker. Therefore, the decarbonisation scenarios explored in Chapter 5, i.e., increased use of alternative fuels, improved thermal efficiencies, and the implementation of CCS technologies, should be considered in future analyses to obtain a more realistic assessment of environmental impacts. While this thesis applied a general European context, it is important to recognise that regional differences in raw material availability, fuel types, and BR composition can significantly affect environmental outcomes. Future studies should therefore incorporate location-specific sourcing scenarios for both cement production and BR processing to ensure that assessments reflect real-world conditions. When combined with mechanical performance indicators, a more comprehensive understanding of how these novel materials would perform under future decarbonised scenarios could be gained.

Expanding the sustainability indices beyond climate change to include additional environmental impact categories, such as resource depletion, water usage, and toxicity impacts, would enable a more detailed comparison between conventional and BR-incorporated cementitious materials. This would provide a better understanding of the trade-offs involved, guiding the optimisation and resilience of these novel materials for future applications.

Furthermore, additional parameters should be incorporated alongside environmental impacts, particularly durability aspects; i.e., chloride diffusion, carbonation, and energy consumption are examples which could enhance the comparability between alternatives. The embodied energy in concrete is a key factor, as clinker production remains highly energy-intensive and predominantly dependant on fossil fuels. Thus, it would be necessary to explore how the shift toward decarbonisation in cement production, especially with the electrification of the industry and a decarbonised energy grid, would affect these indices.

In addition to environmental assessments, further work should focus on optimising the mix design of novel cementitious materials for specific applications. Adjusting mix proportions could achieve different strength classes. For instance, in the case of mortar with CEM II/B-B with VBR, if the objective was to attain a specific strength class, a higher replacement ratio of clinker could be possible without compromising mechanical performance. This could reduce climate change impact due to higher clinker substitution. Moreover, optimising structure design could indicate how utilising these higher strength materials could reduce overall material usage.

A key area for future research is the economic viability of these novel materials. Conducting a techno-economic analysis would account for the individual production and operational costs of each emerging BR technology and assess whether these materials are competitive with current alternatives. Apart from identifying and minimising the environmental impacts, exploring potential cost-saving measures during the upscaling process would be necessary to facilitate the market adoption of BR-incorporated cementitious materials. Additionally, an in-depth analysis of the long-term cost benefits for manufacturers, especially considering the expected shortage of conventional SCMs and rising prices, could highlight the potential for utilising BR into cementitious materials (i.e., SCMs).

Another important aspect is integrating social concerns through a social life cycle assessment to assess the social implications of using BR in the cement industry. Key considerations could include public perception of cementitious materials containing BR waste and potential issues about the colour of BR-derived materials, which may range from red to black, and how this affects their desirability and adoption.

Regulations need to adapt to facilitate the development and commercialisation of novel BR cementitious materials. Although these materials showed promising environmental performance compared to conventional materials, their large-scale adoption will depend on current regulatory frameworks. Governments would need to update building codes, harmonise testing and certification standards to reflect the capabilities of these novel BR-incorporated materials. Performance-based standards, which focus on durability and safety rather than prescriptive material compositions, would promote the adoption of novel cementitious materials as long as they meet the required performance criteria. Furthermore, carbon regulations, such as carbon taxes and credits, could incentivise the use of low-carbon materials.

Finally, to support informed decision-making, future research should adopt multicriteria decision analysis (MCDA) to evaluate emerging technologies based on a combination of factors such as environmental performance, mechanical and durability properties, costs, and social impacts. Although this thesis examined five emerging BR technologies which aimed at developing low-carbon cements, decarbonising the cement industry will require a range of mitigation measures that could consider various emerging SCMs (influenced by regional availability). MCDA could be used to identify which novel materials have the greatest potential for long-term viability by prioritising technologies that are not only technically and environmentally feasible but also economically and socially viable.

Chapter 8. References

Adrianto, L.R., van der Hulst, M.K., Tokaya, J.P., Arvidsson, R., Blanco, C.F., Caldeira, C., Guillén-Gonsálbez, G., Sala, S., Steubing, B., Buyle, M., Kaddoura, M., Navarre, N.H., Pedneault, J., Pizzol, M., Salieri, B., van Harmelen, T., Hauck, M., 2021. How can LCA include prospective elements to assess emerging technologies and system transitions? The 76th LCA Discussion Forum on Life Cycle Assessment, 19 November 2020, in: International Journal of Life Cycle Assessment. Springer Science and Business Media Deutschland GmbH, pp. 1541–1544. https://doi.org/10.1007/s11367-021-01934-w

Aïtcin, P.-C., 2000. Cements of yesterday and today: Concrete of tomorrow. Cem Concr Res 30, 1349–1359. https://doi.org/https://doi.org/10.1016/S0008-8846(00)00365-3

Alberici, S., De Beer, J., Van Der Hoorn, I., Staats, M., 2017. Fly ash and blast furnace slag for cement manufacturing.

Alberts, G., Lotti, G., Dienst, S., Onderzoek, L., 2015. From the Sugar Platform to biofuels and biochemicals Ilaria Marsili Libelli Mecoil Diagnosi Meccaniche Srl from the Sugar Platform to biofuels and biochemicals Final report for the European Commission Directorate-General Energy Consorzio per la Ricerca e la Dimostrazione sulle Energie Rinnovabili (RE-CORD). https://doi.org/10.13140/RG.2.1.2060.9127

Alfaro, J.F., Sharp, B.E., Miller, S.A., 2010. Developing LCA techniques for emerging systems: Game theory, agent modeling as prediction tools, in: Proceedings of the 2010 IEEE International Symposium on Sustainable Systems and Technology, ISSST 2010. https://doi.org/10.1109/ISSST.2010.5507728

Almeida, J., Ribeiro, A.B., Silva, A.S., Faria, P., 2020. Overview of mining residues incorporation in construction materials and barriers for full-scale application. Journal of Building Engineering. https://doi.org/10.1016/j.jobe.2020.101215

An, J., Middleton, R.S., Li, Y., 2019. Environmental performance analysis of cement production with CO2 capture and storage technology in a life-cycle perspective. Sustainability (Switzerland) 11. https://doi.org/10.3390/su11092626

Andreas Ehrenberg, 2023. The steel production transformation process in Europe: New slag types will substitute granulated blast furnace slag. ZKG cement lime gypsum.

Arfelis Espinosa, S., Bala, A., Fullana-I-Palmer, P., 2022. Life cycle assessment as a tool for evaluating chemical processes at industrial scale: a review. Green Chemistry. https://doi.org/10.1039/d2gc02089k

Arrigoni, A., Panesar, D.K., Duhamel, M., Opher, T., Saxe, S., Posen, I.D., MacLean, H.L., 2020. Life cycle greenhouse gas emissions of concrete containing supplementary cementitious materials: cut-off vs. substitution. J Clean Prod 263. https://doi.org/10.1016/j.jclepro.2020.121465

Arvidsson, R., Kushnir, D., Sandén, A.S., Molander, S., 2014. Prospective Life Cycle Assessment of Graphene Production by Ultrasonication and Chemical Reduction. https://doi.org/10.1021/es405338k

Arvidsson, R., Molander, S., 2017. Prospective Life Cycle Assessment of Epitaxial Graphene Production at Different Manufacturing Scales and Maturity. J Ind Ecol 21, 1153–1164. https://doi.org/10.1111/jiec.12526

Arvidsson, R., Nguyen, D., Svanströ, M., 2015. Life Cycle Assessment of Cellulose Nanofibrils Production by Mechanical Treatment and Two Different Pretreatment Processes. https://doi.org/10.1021/acs.est.5b00888

Arvidsson, R., Sandén, B.A., Svanström, M., 2023. Prospective, Anticipatory and Ex-Ante-What's the Difference? Sorting Out Concepts for Time-Related LCA.

Arvidsson, R., Svanström, M., Sandén, B.A., Thonemann, N., Steubing, B., Cucurachi, S., 2024. Terminology for future-oriented life cycle assessment: review and

recommendations. International Journal of Life Cycle Assessment. https://doi.org/10.1007/s11367-023-02265-8

Arvidsson, R., Tillman, A.M., Sandén, B.A., Janssen, M., Nordelöf, A., Kushnir, D., Molander, S., 2018. Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. J Ind Ecol. https://doi.org/10.1111/jiec.12690

Aryapratama, R., Janssen, M., 2017. Prospective life cycle assessment of bio-based adipic acid production from forest residues. J Clean Prod 164, 434–443. https://doi.org/10.1016/j.jclepro.2017.06.222

Aspen Technology, Inc., 2013. Aspen plus. Aspen technology, Inc., USA.

Azapagic, A., Stamford, L., Youds, L., Barteczko-Hibbert, C., 2016. Towards sustainable production and consumption: A novel DEcision-Support Framework IntegRating Economic, Environmental and Social Sustainability (DESIRES). Comput Chem Eng 91, 93–103. https://doi.org/10.1016/j.compchemeng.2016.03.017

Balgobin, T., Evrard, D., 2020. A framework for modelling emerging processes' upscaling from an environmental perspective, in: Procedia CIRP. Elsevier B.V., pp. 154–158. https://doi.org/10.1016/j.procir.2020.01.055

Balomenos, E., Davris, P., Sakkas, K.M., Georgopoulos, C., Makrigiannis, I., 2024. Bauxite Residue Valorisation through Reductive Smelting: Coproduction of Pig Iron and Precursor for Inorganic Polymer Fire Resistant Building Materials. OAlib 11, 1–15. https://doi.org/10.4236/oalib.1111029

Battiston, S., Fiameni, S., Fasolin, S., Barison, S., Armelao, L., 2023. Life cycle environmental impact assessment of lab-scale preparation of porous alumina pellets as substrate for hydrogen separation metal layer-based membranes. International Journal of Life Cycle Assessment 28, 1117–1131. https://doi.org/10.1007/s11367-023-02179-5

Bauer, N., Calvin, K., Emmerling, J., Fricko, O., Fujimori, S., Hilaire, J., Eom, J., Krey, V., Kriegler, E., Mouratiadou, I., Sytze de Boer, H., van den Berg, M., Carrara, S., Daioglou,

V., Drouet, L., Edmonds, J.E., Gernaat, D., Havlik, P., Johnson, N., Klein, D., Kyle, P., Marangoni, G., Masui, T., Pietzcker, R.C., Strubegger, M., Wise, M., Riahi, K., van Vuuren, D.P., 2017. Shared Socio-Economic Pathways of the Energy Sector – Quantifying the Narratives. Global Environmental Change 42, 316–330. https://doi.org/10.1016/j.gloenvcha.2016.07.006

Benhelal, E., Zahedi, G., Shamsaei, E., Bahadori, A., 2013. Global strategies and potentials to curb CO2 emissions in cement industry. J Clean Prod. https://doi.org/10.1016/j.jclepro.2012.10.049

Bergek, A., Jacobsson, S., Carlsson, B., Lindmark, S., Rickne, A., 2008. Analyzing the functional dynamics of technological innovation systems: A scheme of analysis. Res Policy 37, 407–429. https://doi.org/10.1016/j.respol.2007.12.003

Bergerson, J., Cucurachi, S., Seager, T.P., 2020. Bringing a life cycle perspective to emerging technology development. J Ind Ecol. https://doi.org/10.1111/jiec.12990

Bergerson, J.A., Brandt, A., Cresko, J., Carbajales-Dale, M., MacLean, H.L., Matthews, H.S., McCoy, S., McManus, M., Miller, S.A., Morrow, W.R., Posen, I.D., Seager, T., Skone, T., Sleep, S., 2019. Life cycle assessment of emerging technologies: Evaluation techniques at different stages of market and technical maturity. J Ind Ecol 24, 11–25. https://doi.org/10.1111/jiec.12954

Bergesen, J.D., Suh, S., 2016. A framework for technological learning in the supply chain: A case study on CdTe photovoltaics. Appl Energy 169, 721–728. https://doi.org/10.1016/j.apenergy.2016.02.013

BINDER, 2024. Drying and heating chambers [WWW Document].

Binnemans, K., Jones, P.T., Blanpain, B., Van Gerven, T., Pontikes, Y., 2015. Towards zero-waste valorisation of rare-earth-containing industrial process residues: A critical review. J Clean Prod. https://doi.org/10.1016/j.jclepro.2015.02.089

Bisinella, V., Christensen, T.H., Astrup, T.F., 2021. Future scenarios and life cycle assessment: systematic review and recommendations. International Journal of Life Cycle Assessment. https://doi.org/10.1007/s11367-021-01954-6

Bjerge, L.M., Brevik, P., 2014. CO2 capture in the cement industry, Norcem CO2 capture project (Norway), in: Energy Procedia. Elsevier Ltd, pp. 6455–6463. https://doi.org/10.1016/j.egypro.2014.11.680

Boesch, M.E., Hellweg, S., 2010. Identifying improvement potentials in cement production with life cycle assessment. Environ Sci Technol 44, 9143–9149. https://doi.org/10.1021/es100771k

Boily R., 2012. Twenty Cases of Red Hazard, an Inventory of Ecological Problems Caused by Bauxite Residue from Alumina Production. Laval, Quebec, Canada.

Borgonovo, E., 2007. A new uncertainty importance measure. Reliab Eng Syst Saf 92, 771–784. https://doi.org/10.1016/j.ress.2006.04.015

Borra, C.R., Blanpain, B., Pontikes, Y., Binnemans, K., Van Gerven, T., 2016. Smelting of Bauxite Residue (Red Mud) in View of Iron and Selective Rare Earths Recovery. Journal of Sustainable Metallurgy 2, 28–37. https://doi.org/10.1007/s40831-015-0026-4

Bray, A.W., Stewart, D.I., Courtney, R., Rout, S.P., Humphreys, P.N., Mayes, W.M., Burke, I.T., 2018. Sustained Bauxite Residue Rehabilitation with Gypsum and Organic Matter 16 years after Initial Treatment. Environ Sci Technol 52, 152–161. https://doi.org/10.1021/acs.est.7b03568

British Standards Institution., 2005. Methods of testing cement: part 1 : determination of strength. British Standard.

Brough, D., Jouhara, H., 2020. The aluminium industry: A review on state-of-the-art technologies, environmental impacts, and possibilities for waste heat recovery. International Journal of Thermofluids. https://doi.org/10.1016/j.ijft.2019.100007

Bruce, H., 2023. Plans to expand mud waste pool could lead to 'largest environmental catastrophe ever in Ireland.' Extra.ie.

Bruhn, S., Sacchi, R., Cimpan, C., Birkved, M., 2023. Ten questions concerning prospective LCA for decision support for the built environment. Build Environ 242. https://doi.org/10.1016/j.buildenv.2023.110535

Bullerjahn, F., Mehringskötter, M., 2021. Synthetic granulated blast furnace-like slag from bauxite residue smelting and its use in multi-component Portland composite cement. J Clean Prod 329, 129667. https://doi.org/https://doi.org/10.1016/j.jclepro.2021.129667

Buyle, M., Audenaert, A., Billen, P., Boonen, K., Van Passel, S., 2019. The future of ex-ante LCA? Lessons learned and practical recommendations. Sustainability (Switzerland). https://doi.org/10.3390/su11195456

Buyle, M., Maes, B., Van Passel, S., Boonen, K., Vercalsteren, A., Audenaert, A., 2021. Ex-ante LCA of emerging carbon steel slag treatment technologies: Fast forwarding lab observations to industrial-scale production. J Clean Prod 313. https://doi.org/10.1016/j.jclepro.2021.127921

Caduff, M., Huijbregts, M.A.J., Althaus, H.J., Hendriks, A.J., 2011. Power-law relationships for estimating mass, fuel consumption and costs of energy conversion equipments. Environ Sci Technol 45, 751–754. https://doi.org/10.1021/es103095k

Caduff, M., Huijbregts, M.A.J., Koehler, A., Althaus, H.-J., Hellweg, S., 2014. Scaling Relationships in Life Cycle Assessment. J Ind Ecol 18, 393–406. https://doi.org/https://doi.org/10.1111/jiec.12122

Carbon Clean, 2020. LafargeHolcim and Carbon Clean to develop large scale CCUS plant [WWW Document].

Celik, I., Song, Z., Cimaroli, A.J., Yan, Y., Heben, M.J., Apul, D., 2016. Life Cycle Assessment (LCA) of perovskite PV cells projected from lab to fab. Solar Energy Materials and Solar Cells 156, 157–169. https://doi.org/10.1016/j.solmat.2016.04.037

Cembureau, 2020. Reaching climate neutrality along the cement and concrete value chain by 2050 Cementing the European Green Deal.

CEMENTOS LA UNION S.A. (2024). Environmental Product Declaration CEM II B-L 32.5 N. www.environdec.com

CEMMAC cement. (2017a). CEM II/A-S 42,5R Portland-slag cement.

CEMMAC cement. (2017b). CEM II/B-S 42,5N Portland-slag cement.

Centre, J.R., Studies, I. for P.T., Kourti, I., Delgado Sancho, L., Schorcht, F., Roudier, S., Scalet, B., 2013. Best available techniques (BAT) reference document for the production of cement, lime, and magnesium oxide – Industrial Emissions Directive 2010/75/EU (integrated pollution prevention and control). Publications Office. https://doi.org/doi/10.2788/12850

Cetinkaya, E., Dincer, I., Naterer, G.F., 2012. Life cycle assessment of various hydrogen production methods. Int J Hydrogen Energy 37, 2071–2080. https://doi.org/10.1016/j.ijhydene.2011.10.064

Chen, C., Habert, G., Bouzidi, Y., Jullien, A., 2010. Environmental impact of cement production: detail of the different processes and cement plant variability evaluation. J Clean Prod 18, 478–485. https://doi.org/10.1016/J.JCLEPRO.2009.12.014

Chen, C., Yang, S., 2021. The energy demand and environmental impacts of oxy-fuel combustion vs. post-combustion capture in China. Energy Strategy Reviews 38. https://doi.org/10.1016/j.esr.2021.100701

Chunark, P., Hanaoka, T., Limmeechokchai, B., 2021. Shared socioeconomic pathways and long-term GHG mitigation towards 2050 in Thailand cement industry. Cleaner and Responsible Consumption 2, 100006. https://doi.org/10.1016/j.clrc.2020.100006

Clift, R., Doig, A., Finnveden, G., 2000. The application of Life Cycle Assessment to Integrated Solid Waste Management. Part 1 - Methodology. Process Safety and Environmental Protection 78, 279–287. https://doi.org/10.1205/095758200530790

CM Group, 2020. An assessment of global megatrends and regional and market sector growth outlook for aluminium demand.

Colelli, G., Chacartegui, R., Ortiz, C., Carro, A., Arena, A.P., Verda, V., 2022. Life cycle and environmental assessment of calcium looping (CaL) in solar thermochemical energy storage. Energy Convers Manag 257. https://doi.org/10.1016/j.enconman.2022.115428

Conhyea, M., & Goodary, R. (2021). Effect of Fly Ash Additive on Mechanical Properties of Concrete. *International Journal of Civil Infrastructure*. https://doi.org/10.11159/ijci.2021.009

Cooling, D.J., 2007. Improving the Sustainability of Residue Management Practices-Alcoa World Alumina Australia, in: In R Jewell & AB Fourie (Eds), Paste 2007: Proceedings of the Tenth International Seminar on Paste and Thickened Tailings. Australian Centre for Geomechanics, Perth, pp. 3–16.

Cooper, D.R., Gutowski, T.G., 2020. Prospective Environmental Analyses of Emerging Technology: A Critique, a Proposed Methodology, and a Case Study on Incremental Sheet Forming. J Ind Ecol 24, 38–51. https://doi.org/10.1111/jiec.12748

Cooper, J., Noon, M., Jones, C., Kahn, E., Arbuckle, P., 2013. Big data in life cycle assessment. J Ind Ecol 17, 796–799. https://doi.org/10.1111/jiec.12069

Cossutta, M., Mckechnie, J., Pickering, S.J., 2017. Comparative LCA of different graphene production routes. https://doi.org/DOI https://doi.org/10.1039/C7GC02444D

Council of European Union, 1999. Council directive 1999/31/EC of 26 April 1999 on the landfill of waste. Official Journal of the European Communities.

Cox, B., Mutel, C.L., Bauer, C., Mendoza Beltran, A., Van Vuuren, D.P., 2018. Uncertain Environmental Footprint of Current and Future Battery Electric Vehicles. https://doi.org/10.1021/acs.est.8b00261

CTL, 2003. Life cycle inventory of slag cement manufacturing process.

Cucurachi, S., Borgonovo, E., Heijungs, R., 2015. A Protocol for the Global Sensitivity Analysis of Impact Assessment Models in Life Cycle Assessment. Risk Analysis 36, n/a-n/a. https://doi.org/10.1111/risa.12443

Cucurachi, S., Van Der Giesen, C., Guinée, J., 2018. Ex-ante LCA of Emerging Technologies, in: Procedia CIRP. Elsevier B.V., pp. 463–468. https://doi.org/10.1016/j.procir.2017.11.005

Cuéllar-Franca, R.M., García-Gutiérrez, P., Taylor, S.F.R., Hardacre, C., Azapagic, A., 2016. A novel methodology for assessing the environmental sustainability of ionic liquids used for CO2 capture. Faraday Discuss. https://doi.org/10.1039/c6fd00054a

Curran, M.A., 2017. Overview of goal and scope definition in life cycle assessment, in: Goal and Scope Definition in Life Cycle Assessment. Springer, pp. 1–62.

Curran, M.A., Mann, M., Norris, G., 2005. The international workshop on electricity data for life cycle inventories, in: Journal of Cleaner Production. pp. 853–862. https://doi.org/10.1016/j.jclepro.2002.03.001

Cusano, G., Rodrigo Gonzalo, M., Farrell, F., Remus, R., Roudier, S., Delgado Sancho, L., 2017. Best Available Techniques (BAT) Reference Document for the Non-Ferrous Metals Industries - Industrial Emissions Directive 2010/75/EU (Integrated Pollution Prevention and Control).

Damineli, B.L., Kemeid, F.M., Aguiar, P.S., John, V.M., 2010. Measuring the eco-efficiency of cement use. Cem Concr Compos 32, 555–562. https://doi.org/10.1016/J.CEMCONCOMP.2010.07.009

Danner, T., Justnes, H., 2020. Bauxite Residue as Supplementary Cementitious Material – Efforts to Reduce the Amount of Soluble Sodium. Nordic Concrete Research 62, 1–20. https://doi.org/10.2478/ncr-2020-0001

Dauriat, A., Bitar, J., Liao, X., Delerce-Mauris, C., 2019. GCCA INDUSTRY EPD TOOL FOR CEMENT AND CONCRETE (V3.0) LCA Model, International version.

DECC, 2015. Industry decarbonisation & energy efficiency roadmaps to 2050: Cement.

Dequéant, M.L., Glynn, E., Gaudenz, K., Wahl, M., Chen, J., Mushegian, A., Pourquié, O., 2006. A complex oscillating network of signaling genes underlies the mouse segmentation clock. Science (1979) 314, 1595–1598. https://doi.org/10.1126/science.1133141

DESPATCH, 2024. Thermal processing technology [WWW Document].

Di Mare, M., Nattrodt Monteiro, V., Brial, V., Ouellet-Plamondon, C.M., Fortin, S., Tsesmelis, K., Montini, M., Rosani, D., 2021. A calculator for valorizing bauxite residue in the cement industry. Cleaner Materials 1. https://doi.org/10.1016/j.clema.2021.100009

Driver, J.G., Bernard, E., Patrizio, P., Fennell, P.S., Scrivener, K., Myers, R.J., 2024. Global decarbonization potential of CO_2 mineralization in concrete materials. Proceedings of the National Academy of Sciences 121. https://doi.org/10.1073/pnas.2313475121

Dufour, J., Serrano, D.P., Gá Lvez C, J.L., Gonzá Lez C, A., Soria, E., Fierro, J.L.G., 2011. Life cycle assessment of alternatives for hydrogen production from renewable and fossil sources. https://doi.org/10.1016/j.ijhydene.2011.09.135

Dziok, T., Bury, M., Bytnar, K., Burmistrz, P., 2021. Possibility of using alternative fuels in Polish power plants in the context of mercury emissions. Waste Management 126, 578–584. https://doi.org/10.1016/j.wasman.2021.03.053

Ecoinvent v.3.9.1, 2007a. Dataset Documentation "clinker production-Europe without Switzerland-clinker."

Ecoinvent v.3.9.1, 2007b. Dataset Documentation "ceramic tile production-RoW-ceramic tile."

Ekvall, T., 2019. Attributional and Consequential Life Cycle Assessment. https://doi.org/10.5772/intechopen.89202

Ellis, L.D., Badel, A.F., Chiang, M.L., Park, R.J.Y., Chiang, Y.M., 2020. Toward electrochemical synthesis of cement—An electrolyzer-based process for decarbonating CaCO3 while producing useful gas streams. Proc Natl Acad Sci U S A 117, 12584–12591. https://doi.org/10.1073/pnas.1821673116

EN 15804, 2019. Sustainability of construction works - Environmental product declarations - Core rules for the product category of construction products. Brussels: European Committee for Standardization (CEN).

Energy Agency, I., 2021. Net Zero by 2050 - A Roadmap for the Global Energy Sector.

Engin, T., Ari, V., 2005. Energy auditing and recovery for dry type cement rotary kiln systems - A case study. Energy Convers Manag 46, 551–562. https://doi.org/10.1016/j.enconman.2004.04.007

Erakca, M., Baumann, M., Helbig, C., Weil, M., 2024. Systematic review of scale-up methods for prospective life cycle assessment of emerging technologies. J Clean Prod. https://doi.org/10.1016/j.jclepro.2024.142161

European Aluminium (EA), 2015. Bauxite Residue Management: Best Practice.

European Commission, 2024. Innovation fund projects [WWW Document].

European Commission, 2014. Commission Decision of 18 December 2014 amending Decision 2000/ 532/ EC on the list of waste pursuant to Directive 2008/ 98/ EC of the European Parliament and of the Council - (2014/ 955/ EU). Official Journal of the European Union L370, 44–86.

European Commission, 2010. "International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance."

European Commission, 2007. Best available techniques (BAT) reference document in the ceramic manufacturing industry CER BREF).

European Parliament, Council of the European Union, 2024. Regulation (EU) 2024/1781 of the European Parliament and of the Council of 13 June 2024 establishing a framework for the setting of ecodesign requirements for sustainable products, amending Directive (EU) 2020/1828 and Regulation (EU) 2023/1542 and repealing Directive 2009/125/EC. Official Journal of the European Union 1–42.

European Union, 2008. Directive 2008/98/EC of the European parliament and of thecouncil on waste and repealing certain directives.

Evans, K., 2016. The History, Challenges, and New Developments in the Management and Use of Bauxite Residue. Journal of Sustainable Metallurgy 2, 316–331. https://doi.org/10.1007/s40831-016-0060-x

Evans, K., 2015. Successes and challenges in the management and use of bauxite residue.

Ewais, E.M.M., Khalil, N.M., Amin, M.S., Ahmed, Y.M.Z., Barakat, M.A., 2009. Utilization of aluminum sludge and aluminum slag (dross) for the manufacture of calcium aluminate cement. Ceram Int 35, 3381–3388. https://doi.org/10.1016/j.ceramint.2009.06.008

Fantilli, A.P., Chiaia, B., 2013. Eco-mechanical performances of cement-based materials: An application to self-consolidating concrete. Constr Build Mater 40, 189–196. https://doi.org/10.1016/j.conbuildmat.2012.09.075

Fantini, M., Spinelli, M., Magli, F., Consonni, S., 2020. Calcium Looping technology demonstration in industrial environment: The CLEANKER project and status of the CLEANKER pilot plant, in: E3S Web of Conferences. EDP Sciences. https://doi.org/10.1051/e3sconf/202019708006

Favier, A., De Wolf, C.;, Scrivener, K.;, Habert, G., 2018. A sustainable future for the European Cement and Concrete Industry Technology assessment for full decarbonisation of the industry by 2050. https://doi.org/10.3929/ethz-b-000301843

Feng, L., Yao, W., Zheng, K., Cui, N., Xie, N., 2022. Synergistically Using Bauxite Residue (Red Mud) and Other Solid Wastes to Manufacture Eco-Friendly Cementitious Materials. Buildings. https://doi.org/10.3390/buildings12020117

Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in Life Cycle Assessment. J Environ Manage. https://doi.org/10.1016/j.jenvman.2009.06.018

García-Gusano, D., Cabal, H., Lechón, Y., 2015. Long-term behaviour of CO2 emissions from cement production in Spain: Scenario analysis using an energy optimisation model. J Clean Prod 99, 101–111. https://doi.org/10.1016/j.jclepro.2015.03.027

Gartner, E., Hirao, H., 2015. A review of alternative approaches to the reduction of CO2 emissions associated with the manufacture of the binder phase in concrete. Cem Concr Res. https://doi.org/10.1016/j.cemconres.2015.04.012

Gavankar, S., Anderson, S., Keller, A.A., 2015. Critical Components of Uncertainty Communication in Life Cycle Assessments of Emerging Technologies: Nanotechnology as a Case Study. J Ind Ecol 19, 468–479. https://doi.org/10.1111/jiec.12183

GCCA, 2021. The GCCA 2050 Cement and Concrete Industry Roadmap for Net Zero Concrete, concrete roadmap full document GCCA Concrete Future-Roadmap to Net Zero.

Gear, M., Sadhukhan, J., Thorpe, R., Clift, R., Seville, J., Keast, M., 2018. A life cycle assessment data analysis toolkit for the design of novel processes – A case study for a thermal cracking process for mixed plastic waste. J Clean Prod 180, 735–747. https://doi.org/10.1016/j.jclepro.2018.01.015

Georgiades, M., Shah, I.H., Steubing, B., Cheeseman, C., Myers, R.J., 2023. Prospective life cycle assessment of European cement production. Resour Conserv Recycl 194. https://doi.org/10.1016/j.resconrec.2023.106998

Georgiopoulou, M., Lyberatos, G., 2018. Life cycle assessment of the use of alternative fuels in cement kilns: A case study. J Environ Manage 216, 224–234. https://doi.org/10.1016/j.jenvman.2017.07.017

Gerber, L., Gassner, M., Maréchal, F., 2011. Systematic integration of LCA in process systems design: Application to combined fuel and electricity production from lignocellulosic biomass. Comput Chem Eng 35, 1265–1280. https://doi.org/10.1016/j.compchemeng.2010.11.012

Gettu, R., Patel, A., Rathi, V., Prakasan, S., Basavaraj, A.S., Palaniappan, S., Maity, S., 2019. Influence of supplementary cementitious materials on the sustainability parameters of cements and concretes in the Indian context. Materials and Structures/Materiaux et Constructions 52. https://doi.org/10.1617/s11527-019-1321-5

Gettu, R., Pillai, R.G., Santhanam, M., Basavaraj, A.S., Rathnarajan, S., Dhanya, B.S., 2018. Sustainability-based decision support framework for choosing concrete mixture proportions. Mater Struct 51, 165. https://doi.org/10.1617/s11527-018-1291-z

Ghalehnovi, M., Roshan, N., Hakak, E., Shamsabadi, E.A., de Brito, J., 2019. Effect of red mud (bauxite residue) as cement replacement on the properties of self-compacting concrete incorporating various fillers. J Clean Prod 240. https://doi.org/10.1016/j.jclepro.2019.118213

Ghalehnovi, M., Roshan, N., Taghizadeh, A., Asadi Shamsabadi, E., Ali Hadigheh, S., de Brito, J., 2022. Production of Environmentally Friendly Concrete Incorporating Bauxite Residue and Silica Fume. Journal of Materials in Civil Engineering 34. https://doi.org/10.1061/(asce)mt.1943-5533.0004060

Gibon, T., Wood, R., Arvesen, A., Bergesen, J.D., Suh, S., Hertwich, E.G., 2015. A Methodology for Integrated, Multiregional Life Cycle Assessment Scenarios under Large-Scale Technological Change. https://doi.org/10.1021/acs.est.5b01558

Giels, M., 2023. Novel Sustainable Binders incorporating High Volumes of Bauxite Residue. KU Leuven.

Giels, M., Hertel, T., Gijbels, K., Schroeyers, W., Pontikes, Y., 2022. High performance mortars from vitrified bauxite residue; the quest for the optimal chemistry and processing conditions. Cem Concr Res 155, 106739. https://doi.org/10.1016/J.CEMCONRES.2022.106739

Giels, M., Hertel, T., Pontikes, Y., 2023. Transformation of Bauxite Residue into a Reactive Supplementary Cementitious Material. The 16th International Congress on the Chemistry of Cement 2023 (ICCC2023) "Further Reduction of CO2 -Emissions and Circularity in the Cement and Concrete Industry."

Gnansounou, E., Daurlat, A., Panichelli, L., Villegas, J.D., 2008. Accounting_for_indirect_land-use_changes_in_GHG_ba.

Goedkoop, M., Heijungs, R., Huijbregts, M., Schryver, A., Struijs, J., Zelm, R., 2008. ReCiPE 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level.

Gomes, H.I., Mayes, W.M., Rogerson, M., Stewart, D.I., Burked, I.T., 2016. Alkaline residues and the environment: A review of impacts, management practices and opportunities. J Clean Prod. https://doi.org/10.1016/j.jclepro.2015.09.111

González-García, S., Argiz, L., Míguez, P., Gullón, B., 2018. Exploring the production of bio-succinic acid from apple pomace using an environmental approach. Chemical Engineering Journal 350, 982–991. https://doi.org/10.1016/j.cej.2018.06.052

Grzesik, K., Malinowski, M., 2016. Life cycle assessment of refuse-derived fuel production from mixed municipal waste. Energy Sources, Part A: Recovery, Utilization and Environmental Effects 38, 3150–3157. https://doi.org/10.1080/15567036.2015.1136976

Guinee, J.B., 2002. Handbook on life cycle assessment operational guide to the ISO standards. Int J Life Cycle Assess 7, 311–313. https://doi.org/10.1007/BF02978897

Guinée, J.B., Cucurachi, S., Henriksson, P.J.G., Heijungs, R., 2018. Digesting the alphabet soup of LCA. International Journal of Life Cycle Assessment. https://doi.org/10.1007/s11367-018-1478-0

Guinée, J.B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T., Rydberg, T., 2011. Life cycle assessment: Past, present, and future. Environ Sci Technol 45, 90–96. https://doi.org/10.1021/es101316v

Gursel, A.P., 2014. Life-Cycle Assessment of Concrete: Decision-Support Tool and Case Study Application.

Habert, G., Billard, C., Rossi, P., Chen, C., Roussel, N., 2010. Cement production technology improvement compared to factor 4 objectives. Cem Concr Res 40, 820–826. https://doi.org/10.1016/j.cemconres.2009.09.031

Habert, G., D'Espinose De Lacaillerie, J.B., Roussel, N., 2011. An environmental evaluation of geopolymer based concrete production: Reviewing current research trends. J Clean Prod 19, 1229–1238. https://doi.org/10.1016/j.jclepro.2011.03.012

Habert, G., Miller, S.A., John, V.M., Provis, J.L., Favier, A., Horvath, A., Scrivener, K.L., 2020. Environmental impacts and decarbonization strategies in the cement and concrete industries. Nat Rev Earth Environ. https://doi.org/10.1038/s43017-020-0093-3

Harris, T.M., Eranki, P.L., Landis, A.E., 2019. Life cycle assessment of proposed space elevator designs. Acta Astronaut 161, 465–474. https://doi.org/10.1016/j.actaastro.2019.02.028

Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I., 2018. Life Cycle Assessment.

Heijungs, R., Huijbregts, M.A.J., 2004. A Review of Approaches to Treat Uncertainty in LCA.

Henzler, K., Maier, S.D., Jäger, M., Horn, R., 2020. SDG-based sustainability assessment methodology for innovations in the field of urban surfaces. Sustainability (Switzerland) 12. https://doi.org/10.3390/su12114466

Hertel, T., Blanpain, B., Pontikes, Y., 2016. A Proposal for a 100 % Use of Bauxite Residue Towards Inorganic Polymer Mortar. Journal of Sustainable Metallurgy 2, 394–404. https://doi.org/10.1007/s40831-016-0080-6

Hertel, T., Giels, M., Muhammad, A., Pontikes, Y., 2023. Reducing the clinker factor in vitrified bauxite residue-containing ternary blended cements. The 16th International Congress on the Chemistry of Cement 2023 (ICCC2023) "Further Reduction of CO2 - Emissions and Circularity in the Cement and Concrete Industry."

Hertel, T., Pontikes, Y., 2020. Geopolymers, inorganic polymers, alkali-activated materials, and hybrid binders from bauxite residue (red mud) – Putting things in perspective. J Clean Prod. https://doi.org/10.1016/j.jclepro.2020.120610

Hertel, T., Van den Bulck, A., Onisei, S., Sivakumar, P.P., Pontikes, Y., 2021. Boosting the use of bauxite residue (red mud) in cement - Production of an Fe-rich calciumsulfoaluminate-ferrite clinker and characterisation of the hydration. Cem Concr Res 145. https://doi.org/10.1016/j.cemconres.2021.106463

Hetherington, Alexandra C, Borrion, A.L., Griffiths, O.G., McManus, M.C., 2014. Use of LCA as a development tool within early research: challenges and issues across different sectors. Int J Life Cycle Assess 19, 130–143. https://doi.org/10.1007/s11367-013-0627-8

Hetherington, Alexandra C., Borrion, A.L., Griffiths, O.G., McManus, M.C., 2014. Use of LCA as a development tool within early research: Challenges and issues across different sectors. International Journal of Life Cycle Assessment 19, 130–143. https://doi.org/10.1007/s11367-013-0627-8

Hills, T., Florin, N., Fennell, P.S., 2016. Decarbonising the cement sector: A bottom-up model for optimising carbon capture application in the UK. J Clean Prod 139, 1351–1361. https://doi.org/10.1016/j.jclepro.2016.08.129

Hischier, R., Salieri, B., Pini, M., 2017. Most important factors of variability and uncertainty in an LCA study of nanomaterials – Findings from a case study with nano titanium dioxide. NanoImpact 7, 17–26. https://doi.org/10.1016/j.impact.2017.05.001

HOBART, 2024. Mixers [WWW Document].

Holcim, 2020. LafargeHolcim expands carbon capture projects with government funding from the US and Germany [WWW Document].

Hoque, N., Biswas, W., Mazhar, I., Howard, I., 2020. Life cycle sustainability assessment of alternative energy sources for the Western Australian transport sector. Sustainability (Switzerland) 12. https://doi.org/10.3390/su12145565

Hospido, A., Davis, J., Berlin, J., Sonesson, U., 2010. A review of methodological issues affecting LCA of novel food products. International Journal of Life Cycle Assessment. https://doi.org/10.1007/s11367-009-0130-4

Hren, R., Naumoska, K., Jug, U., Čuček, L., Likozar, B., Novak, U., Vujanović, A., 2023. Life Cycle Assessment of Pilot-Scale Bio-Refining of Invasive Japanese Knotweed Alien Plant towards Bio-Based Bioactive Compounds. Processes 11. https://doi.org/10.3390/pr11051393

Huang, X., Ai, N., Li, L., Jiang, Q., Wang, Q., Ren, J., Wang, J., 2022. Simulation of CO2 Capture Process in Flue Gas from Oxy-Fuel Combustion Plant and Effects of Properties of Absorbent. Separations 9. https://doi.org/10.3390/separations9040095

Huijbregts, M.A.J., Breedveld, L., Huppes, G., de Koning, A., van Oers, L., Suh, S., 2003. Normalisation figures for environmental life-cycle assessment: The Netherlands

(1997/1998), Western Europe (1995) and the world (1990 and 1995). J Clean Prod 11, 737–748. https://doi.org/https://doi.org/10.1016/S0959-6526(02)00132-4

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. International Journal of Life Cycle Assessment 22, 138–147. https://doi.org/10.1007/s11367-016-1246-y

Hung, C.R., Ellingsen, L.A.W., Majeau-Bettez, G., 2020. LiSET: A Framework for Early-Stage Life Cycle Screening of Emerging Technologies. J Ind Ecol 24, 26–37. https://doi.org/10.1111/jiec.12807

Husillos Rodríguez, N., Martínez-Ramírez, S., Blanco-Varela, M.T., Donatello, S., Guillem, M., Puig, J., Fos, C., Larrotcha, E., Flores, J., 2013. The effect of using thermally dried sewage sludge as an alternative fuel on Portland cement clinker production. J Clean Prod. https://doi.org/10.1016/j.jclepro.2013.02.026

ICF International, 2008. Cement Sector Trends in Beneficial Use of Alternative Fuels and Raw Materials Cement Sector Trends in Beneficial Use of Alternative Fuels and Raw Materials.

IEA, 2018. Technology Roadmap - Low-Carbon Transition in the Cement Industry.

IEA, (International Energy Agency), 2021. Net Zero by 2050 - A Roadmap for the Global Energy Sector.

Igos, E., Benetto, E., Meyer, R., Baustert, P., Othoniel, B., 2019. How to treat uncertainties in life cycle assessment studies? International Journal of Life Cycle Assessment 24, 794–807. https://doi.org/10.1007/s11367-018-1477-1

INDUTHERM, 2024. Induction technology [WWW Document].

International Aluminium Association, 2022. Sustainable Bauxite Residue Management Guidance.

International Aluminium Institute, 2020a. Opportunities for use of bauxite residue in Supplementary Cementitious Materials.

International Aluminium Institute, 2020b. Opportunities for use of bauxite residue in special cements.

International Aluminium Institute, 2020c. Opportunities for using bauxite residue in Portland Cement clinker production.

International Aluminium Institute, 2018. Bauxite residue management [WWW Document].

International Association of Oil & Gas producers, 2022. CCUS projects in Europe.

IPPC Bureau, E., Economy, C., Leadership Unit, I., – Growth, D.B., Research Centre, J., Commission, E., 2022. Best Available Techniques (BAT) Reference Document for the Smitheries and Foundries Industry.

Irish Cement. (2017). Portland Limestone Cement CEM II/A-L 42,5N.

ISO 14025, 2006. ISO 14025:2006, Environmental labels and declarations - Type III environmental declarations - Principles and procedures. Geneva: International Organization for Standardization.

ISO: 14040, 2006. Environmental management: Life cycle assessment: Principles and framework.

ISO: 14044, 2006. Environmental management: Life cycle assessment: Requirements and guidelines.

ISO EN 197-1, 2011. Standards Publication Cement Part 1: Composition, specifications, and conformity criteria for common cements.

Joyce, P.J, Björklund, A., 2019. Using Life Cycle Thinking to Assess the Sustainability Benefits of Complex Valorization Pathways for Bauxite Residue 5, 69–84. https://doi.org/10.1007/s40831-019-00209-x

Jittin, V., Minnu, S.N., Bahurudeen, A., 2021. Potential of sugarcane bagasse ash as supplementary cementitious material and comparison with currently used rice husk ash. Constr Build Mater. https://doi.org/10.1016/j.conbuildmat.2020.121679

Joel Jaeger, 2023. These 10 Countries Are Phasing Out Coal the Fastest [WWW Document]. World resources institute.

Joyce, P.J., Björklund, A., 2019. Using Life Cycle Thinking to Assess the Sustainability Benefits of Complex Valorization Pathways for Bauxite Residue. Journal of Sustainable Metallurgy 5, 69–84. https://doi.org/10.1007/s40831-019-00209-x

Joyce, P.J., Hertel, T., Goronovski, A., Tkaczyk, A.H., Pontikes, Y., Björklund, A., 2018. Identifying hotspots of environmental impact in the development of novel inorganic polymer paving blocks from bauxite residue. Resour Conserv Recycl 138, 87–98. https://doi.org/10.1016/j.resconrec.2018.07.006

Joyce, P. James., 2018. Environmental Considerations in the Zero-waste Valorisation of Bauxite Residue: A Life Cycle Perspective.

Juenger, M., Provis, J., Elsen, J., Matthes, W., Hooton, D., Duchesne, J., Courard, L., He, H., Michel, F., Snellings, R., de Belie, N., 2012. Supplementary Cementitious Materials for Concrete: Characterization Needs. MRS Proceedings 1488. https://doi.org/10.1557/opl.2012.1536

Keller, H., Rettenmaier, N., Reinhardt, G.A., 2015. Integrated life cycle sustainability assessment - A practical approach applied to biorefineries. Appl Energy 154, 1072–1081. https://doi.org/10.1016/j.apenergy.2015.01.095

Khairul, M.A., Zanganeh, J., Moghtaderi, B., 2019. The composition, recycling, and utilisation of Bayer red mud. Resour Conserv Recycl. https://doi.org/10.1016/j.resconrec.2018.11.006

Khanna, V., Bakshi, B.R., Lee, L.J., 2008. Carbon nanofiber production: Life cycle energy consumption and environmental impact. J Ind Ecol 12, 394–410. https://doi.org/10.1111/j.1530-9290.2008.00052.x

Kheirinik, M., Ahmed, S., Rahmanian, N., 2021. Comparative techno-economic analysis of carbon capture processes: Pre-combustion, post-combustion, and oxy-fuel combustion operations. Sustainability (Switzerland) 13. https://doi.org/10.3390/su132413567

Khurana, S., Banerjee, R., Gaitonde, U., 2002. Energy balance and cogeneration for a cement plant.

Kim, Y., Lee, Y., Kim, M., Park, H., 2019. Preparation of high porosity bricks by utilizing red mud and mine tailing. J Clean Prod 207, 490–497. https://doi.org/10.1016/j.jclepro.2018.10.044

Kirschen, M., Risonarta, V., Pfeifer, H., 2009. Energy efficiency and the influence of gas burners to the energy related carbon dioxide emissions of electric arc furnaces in steel industry. Energy 34, 1065–1072. https://doi.org/10.1016/j.energy.2009.04.015

Klauber, C., Gräfe, M., Power, G., 2011. Bauxite residue issues: II. options for residue utilization. Hydrometallurgy 108, 11–32. https://doi.org/10.1016/j.hydromet.2011.02.007

Konlechner, D., Rubio, G.M.D.M., Sparis, D., Flessoura, G., Panias, D., Preveniou, A., Kladis, A., Davris, P., Balomenos, E., Peys, A., Hertel, T., Giels, M., Pontikes, Y., Benard, P., Ferey, F., 2023. Flowsheets That Can Turn BR Into Building Materials and Save CO2.

Koshy, N., Dondrob, K., Hu, L., Wen, Q., Meegoda, J.N., 2019. Synthesis and characterization of geopolymers derived from coal gangue, fly ash and red mud. Constr Build Mater 206, 287–296. https://doi.org/10.1016/j.conbuildmat.2019.02.076

Kuramochi, T., Ramírez, A., Turkenburg, W., Faaij, A., 2012. Comparative assessment of CO2 capture technologies for carbon-intensive industrial processes. Prog Energy Combust Sci. https://doi.org/10.1016/j.pecs.2011.05.001

Kvande, H., 2010. Production of primary aluminium. Fundamentals of Aluminium Metallurgy 49–69. https://doi.org/10.1533/9780857090256.1.49

Lacirignola, M., Blanc, P., Girard, R., Pérez-López, P., Blanc, I., 2017. LCA of emerging technologies: addressing high uncertainty on inputs' variability when performing global sensitivity analysis. Science of the Total Environment 578, 268–280. https://doi.org/10.1016/j.scitotenv.2016.10.066

LEILAC, 2017. Low Emission Intensity Lime and Cement project.

LEILAC 2., 2021. LEILAC 2-Scaling Up Low-Carbon Solutions.

Lewandowska, A., Fohynowicz, Z., Podlesny, A., 2004. Data Quality Assurance LCA Methodology Comparative LCA of Industrial Objects Part 1: LCA Data Quality Assurance-Sensitivity Analysis and Pedigree Matrix. https://doi.org/10.1065/Ica2004.03.152.2

Liu, R.X., Poon, C.S., 2016. Utilization of red mud derived from bauxite in self-compacting concrete. J Clean Prod 112, 384–391. https://doi.org/10.1016/j.jclepro.2015.09.049

Liu, S., Guan, X., Zhang, S., Dou, Z., Feng, C., Zhang, H., Luo, S., 2017. Sintered bayer red mud based ceramic bricks: Microstructure evolution and alkalis immobilization mechanism. Ceram Int 43, 13004–13008. https://doi.org/10.1016/j.ceramint.2017.07.036

Liu, X., Zhang, N., Sun, H., Zhang, J., Li, L., 2011. Structural investigation relating to the cementitious activity of bauxite residue - red mud. Cem Concr Res 41, 847–853. https://doi.org/10.1016/j.cemconres.2011.04.004

Lloyd, S., Ries, R., 2007. Characterizing, Propagating, and Analyzing Uncertainty in Life-Cycle Assessment: A Survey of Quantitative Approaches. J Ind Ecol 11, 161–179. https://doi.org/10.1162/jiec.2007.1136

Long, G., Gao, Y., Xie, Y., 2015. Designing more sustainable and greener self-compacting concrete. Constr Build Mater 84, 301–306. https://doi.org/https://doi.org/10.1016/j.conbuildmat.2015.02.072

Lyew-Ayee, P.A., Persaud, S., Evans, K., Tapp, R., 2011. From red to green – a regulator and company partnership in bauxite residue remediation, in: Proceedings of 29th International ICSOBA Conference. Paper 40, Travaux 36, Goa, India, pp. 137–145.

Lyu, F., Hu, Y., Wang, L., Sun, W., 2021. Dealkalization processes of bauxite residue: A comprehensive review. J Hazard Mater. https://doi.org/10.1016/j.jhazmat.2020.123671

Ma, J., Harstvedt, J.D., Dunaway, D., Bian, L., Jaradat, R., 2018. An exploratory investigation of Additively Manufactured Product life cycle sustainability assessment. J Clean Prod 192, 55–70. https://doi.org/10.1016/j.jclepro.2018.04.249

Maes, B., Audenaert, A., Craeye, B., Buyle, M., 2022. Ex-ante LCA on an emerging electro-mass separation technology: The importance of the background system, in: E3S Web of Conferences. EDP Sciences. https://doi.org/10.1051/e3sconf/202234904008

Maes, B., Buyle, M., Audenaert, A., Craeye, B., 2021. Enhanced fly ash use in concrete: Ex-ante LCA on an emerging electro-mass separation technology. Clean Eng Technol 2, 100076. https://doi.org/10.1016/j.clet.2021.100076

Mainardis, M., Buttazzoni, M., Gievers, F., Vance, C., Magnolo, F., Murphy, F., Goi, D., 2021. Life cycle assessment of sewage sludge pretreatment for biogas production: From laboratory tests to full-scale applicability. J Clean Prod 322. https://doi.org/10.1016/j.jclepro.2021.129056

Manfroi, E.P., Cheriaf, M., Rocha, J.C., 2014. Microstructure, mineralogy, and environmental evaluation of cementitious composites produced with red mud waste. Constr Build Mater 67, 29–36. https://doi.org/https://doi.org/10.1016/j.conbuildmat.2013.10.031

Maraghechi, H., Avet, F., Wong, H., Kamyab, H., Scrivener, K., 2018. Performance of Limestone Calcined Clay Cement (LC3) with various kaolinite contents with respect to chloride transport. Materials and Structures/Materiaux et Constructions 51. https://doi.org/10.1617/s11527-018-1255-3

Marini, C., Blanc, I., 2014. Towards Prospective Life Cycle Assessment: How to Identify Key Parameters Inducing Most Uncertainties in the Future? Application to Photovoltaic Systems Installed in Spain, in: Murgante, B., Misra, S., Rocha, A.M.A.C., Torre, C., Rocha, J.G., Falcão, M.I., Taniar, D., Apduhan, B.O., Gervasi, O. (Eds.), Computational Science and Its Applications -- ICCSA 2014. Springer International Publishing, Cham, pp. 691–706.

Material Economics, 2019. Industrial Transformation 2050 Pathways to Net-Zero Emissions from EU Heavy Industry.

McKinsey Global Institute, 2017. Reinventing construction.

McManus, M.C., Taylor, C.M., 2015. The changing nature of life cycle assessment. Biomass Bioenergy 82, 13–26. https://doi.org/https://doi.org/10.1016/j.biombioe.2015.04.024

MEKA, 2024. MEKA Planetary mixers [WWW Document].

Mendoza Beltran, A., Cox, B., Mutel, C., van Vuuren, D.P., Font Vivanco, D., Deetman, S., Edelenbosch, O.Y., Guinée, J., Tukker, A., 2020. When the Background Matters: Using Scenarios from Integrated Assessment Models in Prospective Life Cycle Assessment. J Ind Ecol 24, 64–79. https://doi.org/10.1111/jiec.12825

Mikulčić, H., Klemeš, J.J., Vujanović, M., Urbaniec, K., Duić, N., 2016. Reducing greenhouse gasses emissions by fostering the deployment of alternative raw materials and energy

sources in the cleaner cement manufacturing process. J Clean Prod 136, 119–132. https://doi.org/10.1016/j.jclepro.2016.04.145

Miller, S.A., 2018. Supplementary cementitious materials to mitigate greenhouse gas emissions from concrete: can there be too much of a good thing? J Clean Prod 178, 587–598. https://doi.org/10.1016/j.jclepro.2018.01.008

Miller, S.A., Habert, G., Myers, R.J., Harvey, J.T., 2021. Achieving net zero greenhouse gas emissions in the cement industry via value chain mitigation strategies. One Earth. https://doi.org/10.1016/j.oneear.2021.09.011

Miller, S.A., Horvath, A., Monteiro, P.J.M., Ostertag, C.P., 2015. Greenhouse gas emissions from concrete can be reduced by using mix proportions, geometric aspects, and age as design factors. Environmental Research Letters 10. https://doi.org/10.1088/1748-9326/10/11/114017

Miller, S.A., John, V.M., Pacca, S.A., Horvath, A., 2018. Carbon dioxide reduction potential in the global cement industry by 2050. Cem Concr Res 114, 115–124. https://doi.org/10.1016/j.cemconres.2017.08.026

Miller, S.A., Keoleian, G.A., 2015. Framework for analyzing transformative technologies in life cycle assessment. Environ Sci Technol 49, 3067–3075. https://doi.org/10.1021/es505217a

Miranda de Souza, A., Soares de Lima, G.E., Nalon, G.H., Salgado Lopes, M.M., Luís de Oliveira Júnior, A., Rodrigues Lopes, G.J., José de Andrade Olivier, M., Pedroti, L.G., Lopes Ribeiro, J.C., Franco de Carvalho, J.M., 2021. Application of the desirability function for the development of new composite eco-efficiency indicators for concrete. Journal of Building Engineering 40. https://doi.org/10.1016/j.jobe.2021.102374

Mokrzycki, E., Uliasz-Bocheńczyk, A., Sarna, M., 2003. Use of alternative fuels in the Polish cement industry. Appl Energy 74, 101–111. https://doi.org/10.1016/S0306-2619(02)00136-8

Moni, M., 2020. A framework for life cycle assessment (LCA) of emerging technologies at low technology readiness levels.

Moni, S.M., Mahmud, R., High, K., Carbajales-Dale, M., 2020. Life cycle assessment of emerging technologies: A review. J Ind Ecol 24, 52–63. https://doi.org/10.1111/jiec.12965

Morsali, S., Yildirim, F., 2024. Environmental impact assessment of red mud utilization in concrete production: a life cycle assessment study. Environ Dev Sustain 26, 12219–12238. https://doi.org/10.1007/s10668-023-03767-z

Mukiza, E., Zhang, L.L., Liu, X., Zhang, N., 2019. Utilization of red mud in road base and subgrade materials: A review. Resour Conserv Recycl. https://doi.org/10.1016/j.resconrec.2018.10.031

Murray, H.H., 1981. Alumina from Non-bauxite Sources. Natural Resources Forum, Blackwell Publishing 5(1), 85–89.

Ness, B., Urbel-Piirsalu, E., Anderberg, S., Olsson, L., 2007. Categorising tools for sustainability assessment. Ecological Economics. https://doi.org/10.1016/j.ecolecon.2006.07.023

Nikbin, I.M., Aliaghazadeh, M., Sh Charkhtab, Fathollahpour, A., 2016. Environmental impacts and mechanical properties of lightweight concrete containing bauxite residue (red mud). J Clean Prod 172, 2683–2694. https://doi.org/10.1016/j.jclepro.2017.11.143

Nuccetelli, C., Pontikes, Y., Leonardi, F., Trevisi, R., 2015. New perspectives and issues arising from the introduction of (NORM) residues in building materials: A critical assessment on the radiological behaviour. Constr Build Mater 82, 323–331. https://doi.org/10.1016/j.conbuildmat.2015.01.069

OHORONGO cement. (2023). CEM II B-V 42.5N, Portland composite cement.

O'Neill, B.C., Kriegler, E., Ebi, K.L., Kemp-Benedict, E., Riahi, K., Rothman, D.S., van Ruijven, B.J., van Vuuren, D.P., Birkmann, J., Kok, K., Levy, M., Solecki, W., 2017. The roads ahead: Narratives for shared socioeconomic pathways describing world futures in the 21st century. Global Environmental Change 42, 169–180. https://doi.org/10.1016/j.gloenvcha.2015.01.004

O'Neill, B.C., Kriegler, E., Riahi, K., Ebi, K.L., Hallegatte, S., Carter, T.R., Mathur, R., van Vuuren, D.P., 2014. A new scenario framework for climate change research: The concept of shared socioeconomic pathways. Clim Change 122, 387–400. https://doi.org/10.1007/s10584-013-0905-2

Ortega, J.M., Cabeza, M., Tenza-Abril, A.J., Real-Herraiz, T., Climent, M.Á., Sánchez, I., 2019. Effects of Red Mud Addition in the Microstructure, Durability and Mechanical Performance of Cement Mortars. Applied Sciences 9. https://doi.org/10.3390/app9050984

Ortiz Ortega, E., Hosseinian, H., Berenice, I., Meza, A., María, ·, Rosales López, J., Rodríguez, A., · V., Hosseini, S., 2022. Material Characterization Techniques and Applications.

OTTOJUNKER, 2024. Coreless induction furnace [WWW Document].

Pacheco-Torgal, F., Cabeza, L.F., Labrincha, J.A., de Magalhães, A., 2013. Eco-efficient construction and building materials: Life cycle assessment (LCA), eco-labelling and case studies.

Padilla-Rivera, A., Hannouf, M., Assefa, G., Gates, I., 2023. A systematic literature review on current application of life cycle sustainability assessment: A focus on economic dimension and emerging technologies. Environ Impact Assess Rev. https://doi.org/10.1016/j.eiar.2023.107268

Pallas, G., Vijver, M.G., Peijnenburg, W.J.G.M., Guinée, J., 2020a. Life cycle assessment of emerging technologies at the lab scale: The case of nanowire-based solar cells. J Ind Ecol 24, 193–204. https://doi.org/10.1111/jiec.12855

Pallas, G., Vijver, M.G., Peijnenburg, W.J.G.M., Guinée, J., 2020b. Ex ante life cycle assessment of GaAs/Si nanowire–based tandem solar cells: a benchmark for industrialization. International Journal of Life Cycle Assessment 25, 1767–1782. https://doi.org/10.1007/s11367-020-01791-z

Palmero, P., Tulliani, J.-M., Occhicone, A., Vukčevi´vukčevi´c, M., Boskovi´cboskovi´c, I., Mingione, S., Ferone, C., 2022. Alkali-Activated Red Mud and Construction and Demolition Waste-Based Components: Characterization and Environmental Assessment. https://doi.org/10.3390/ma

Pamenter, S., Myers, R.J., 2022. Decarbonizing the cementitious materials cycle: A whole systems review of measures to decarbonize the cement supply chain in the UK and European contexts. J Ind Ecol 25, 359–376. https://doi.org/10.1111/jiec.13105

Panesar, D.K., Seto, K.E., Churchill, C.J., 2017. Impact of the selection of functional unit on the life cycle assessment of green concrete. Int J Life Cycle Assess 22, 1969–1986. https://doi.org/10.1007/s11367-017-1284-0

Paramasivam, V., Senthil, V., Rajam Ramasamy, N., 2011. Decision making in equipment selection: An integrated approach with digraph and matrix approach, AHP and ANP. International Journal of Advanced Manufacturing Technology 54, 1233–1244. https://doi.org/10.1007/s00170-010-2997-4

Patangia, J., Saravanan, T.J., Kabeer, K.I.S.A., Bisht, K., 2023. Study on the utilization of red mud (bauxite waste) as a supplementary cementitious material: Pathway to attaining sustainable development goals. Constr Build Mater 375, 131005. https://doi.org/https://doi.org/10.1016/j.conbuildmat.2023.131005

PBL Netherlands Environmental Assessment Agency. (2014). Integrated assessment of global environmental change with model description and policy applications: IMAGE 3.0.

Pehnt, M., 2006. Dynamic life cycle assessment (LCA) of renewable energy technologies. Renew Energy 31, 55–71. https://doi.org/https://doi.org/10.1016/j.renene.2005.03.002

Perez-Cortes, P., Escalante-Garcia, J.I., 2020. Design and optimization of alkaline binders of limestone-metakaolin – A comparison of strength, microstructure, and sustainability with portland cement and geopolymers. J Clean Prod 273. https://doi.org/10.1016/j.jclepro.2020.123118

Pérez-Villarejo, L., Corpas-Iglesias, F.A., Martínez-Martínez, S., Artiaga, R., Pascual-Cosp, J., 2012. Manufacturing new ceramic materials from clay and red mud derived from the aluminium industry. Constr Build Mater 35, 656–665. https://doi.org/10.1016/j.conbuildmat.2012.04.133

Pesonen, H.L., Ekvall, T., Fleischer, G., Huppes, G., Jahn, C., Klos, Z.S., Rebitzer, G., Sonnemann, G.W., Tintinelli, A., Weidema, B.P., Wenzel, H., 2000. Framework for scenario development in LCA. International Journal of Life Cycle Assessment. https://doi.org/10.1007/BF02978555

Petek Gursel, A., Masanet, E., Horvath, A., Stadel, A., 2014. Life-cycle inventory analysis of concrete production: A critical review. Cem Concr Compos 51, 38–48. https://doi.org/10.1016/j.cemconcomp.2014.03.005

Peys, A., Hertel, T., Snellings, R., 2022. Co-Calcination of Bauxite Residue with Kaolinite in Pursuit of a Robust and High-Quality Supplementary Cementitious Material. Front Mater 9. https://doi.org/10.3389/fmats.2022.913151

Pfeifer, Herbert, Kirschen, Marcus, Magnesita, R., Pfeifer, H, Kirschen, M, Simoes, J.P., 2005. Thermodynamic analysis of electrical energy demand Thermodynamic analysis of EAF electrical energy demand.

Piccinno, F., Hischier, R., Seeger, S., Som, C., 2018. Predicting the environmental impact of a future nanocellulose production at industrial scale: Application of the life cycle assessment scale-up framework. J Clean Prod 174, 283–295. https://doi.org/https://doi.org/10.1016/j.jclepro.2017.10.226

Piccinno, F., Hischier, R., Seeger, S., Som, C., 2016. From laboratory to industrial scale: a scale-up framework for chemical processes in life cycle assessment studies. J Clean Prod 135, 1085–1097. https://doi.org/10.1016/j.jclepro.2016.06.164

Plischke, E., Borgonovo, E., Smith, C.L., 2013. Global sensitivity measures from given data. Eur J Oper Res 226, 536–550. https://doi.org/10.1016/j.ejor.2012.11.047

Pontikes, Y., Angelopoulos, G.N., 2013. Bauxite residue in cement and cementitious applications: Current status and a possible way forward. Resour Conserv Recycl. https://doi.org/10.1016/j.resconrec.2013.01.005

Power G., Gräfe M., Klauber C., 2011. Bauxite residue issues: I. Current management, disposal, and storage practices. Hydrometallurgy 108, 33–45. https://doi.org/https://doi.org/10.1016/j.hydromet.2011.02.006

Prado, V., Heijungs, R., 2018. Implementation of stochastic multi attribute analysis (SMAA) in comparative environmental assessments. Environmental Modelling and Software 109, 223–231. https://doi.org/10.1016/j.envsoft.2018.08.021

Prado-Lopez, V., Wender, B.A., Seager, T.P., Laurin, L., Chester, M., Arslan, E., 2016. Tradeoff Evaluation Improves Comparative Life Cycle Assessment: A Photovoltaic Case Study. J Ind Ecol 20, 710–718. https://doi.org/10.1111/jiec.12292

Priecel, P., Lopez-Sanchez, J.A., 2019. Advantages and Limitations of Microwave Reactors: From Chemical Synthesis to the Catalytic Valorization of Biobased Chemicals. ACS Sustain Chem Eng 7, 3–21. https://doi.org/10.1021/acssuschemeng.8b03286

Profile, S., Mortimer, N.D., Sylvester-Bradley, R., 2008. Understanding and managing uncertainties to improve biofuel GHG emissions calculations.

Putrevu, M., Thiyagarajan, J.S., Pasla, D., Kabeer, K.I.S.A., Bisht, K., 2021. Valorization of Red Mud Waste for Cleaner Production of Construction Materials. J Hazard Toxic Radioact Waste 25. https://doi.org/10.1061/(asce)hz.2153-5515.0000629

Rahla, K.M., Mateus, R., Bragança, L., 2019. Comparative sustainability assessment of binary blended concretes using Supplementary Cementitious Materials (SCMs) and Ordinary Portland Cement (OPC). J Clean Prod 220, 445–459. https://doi.org/10.1016/J.JCLEPRO.2019.02.010

Ravikumar, D., Seager, T.P., Cucurachi, S., Prado, V., Mutel, C., 2018. Novel Method of Sensitivity Analysis Improves the Prioritization of Research in Anticipatory Life Cycle Assessment of Emerging Technologies. https://doi.org/10.1021/acs.est.7b04517

Rebello, T.A., Zulcão, R., Calmon, J.L., Gonçalves, R.F., 2019. Comparative life cycle assessment of ornamental stone processing waste recycling, sand, clay, and limestone filler. Waste Management and Research 37, 186–195. https://doi.org/10.1177/0734242X18819976

Ren, J., Toniolo, S., 2018. Life cycle sustainability decision-support framework for ranking of hydrogen production pathways under uncertainties: An interval multi-criteria decision-making approach. J Clean Prod 175, 222–236. https://doi.org/10.1016/j.jclepro.2017.12.070

Riahi, K., van Vuuren, D.P., Kriegler, E., Edmonds, J., O'Neill, B.C., Fujimori, S., Bauer, N., Calvin, K., Dellink, R., Fricko, O., Lutz, W., Popp, A., Cuaresma, J.C., KC, S., Leimbach, M., Jiang, L., Kram, T., Rao, S., Emmerling, J., Ebi, K., Hasegawa, T., Havlik, P., Humpenöder, F., Da Silva, L.A., Smith, S., Stehfest, E., Bosetti, V., Eom, J., Gernaat, D., Masui, T., Rogelj, J., Strefler, J., Drouet, L., Krey, V., Luderer, G., Harmsen, M., Takahashi, K., Baumstark, L., Doelman, J.C., Kainuma, M., Klimont, Z., Marangoni, G., Lotze-Campen, H., Obersteiner, M., Tabeau, A., Tavoni, M., 2017. The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. Global Environmental Change 42, 153–168. https://doi.org/10.1016/j.gloenvcha.2016.05.009

Ribeiro, D.V., Labrincha, J.A., Morelli, M.R., 2011. Potential use of natural red mud as pozzolan for Portland cement. Materials Research 14, 60–66. https://doi.org/10.1590/S1516-14392011005000001

Ribeiro, I., Matos, F., Jacinto, C., Salman, H., Cardeal, G., Carvalho, H., Godina, R., Peças, P., 2020. Framework for life cycle sustainability assessment of additive manufacturing. Sustainability (Switzerland) 12. https://doi.org/10.3390/su12030929

Roes, A.L., Patel, M.K., 2011. Ex-ante environmental assessments of novel technologies – Improved caprolactam catalysis and hydrogen storage. J Clean Prod 19, 1659–1667. https://doi.org/https://doi.org/10.1016/j.jclepro.2011.05.010

Rosental, M., Fröhlich, T., Liebich, A., 2020. Life Cycle Assessment of Carbon Capture and Utilization for the Production of Large Volume Organic Chemicals. Frontiers in Climate 2. https://doi.org/10.3389/fclim.2020.586199

Sacchi, R., Terlouw, T., Siala, K., Dirnaichner, A., Bauer, C., Cox, B., Mutel, C., Daioglou, V., Luderer, G., 2022. PRospective EnvironMental Impact asSEment (premise): A streamlined approach to producing databases for prospective life cycle assessment using integrated assessment models. Renewable and Sustainable Energy Reviews 160, 112311. https://doi.org/https://doi.org/10.1016/j.rser.2022.112311

Salas, D.A., Ramirez, A.D., Ulloa, N., Baykara, H., Boero, A.J., 2018. Life cycle assessment of geopolymer concrete. Constr Build Mater 190, 170–177. https://doi.org/10.1016/j.conbuildmat.2018.09.123

Salim, M.U., Mosaberpanah, M.A., Danish, A., Ahmad, N., Khalid, R.A., Moro, C., 2023. Role of bauxite residue as a binding material and its effect on engineering properties of cementitious Composites: A review. Constr Build Mater 409, 133844. https://doi.org/https://doi.org/10.1016/j.conbuildmat.2023.133844

Salvi Malacarne, C., Rubens Cardoso da Silva, M., Danieli, S., Gonçalves Maciel, V., Paula Kirchheim, A., 2021. Environmental and technical assessment to support sustainable strategies for limestone calcined clay cement production in Brazil. Constr Build Mater 310. https://doi.org/10.1016/j.conbuildmat.2021.125261

Sandén, B., Hillman, K., Karlström, M., Tillman, A.-M., 2005. LCA of emerging technologies: a methodological framework. pp. 37–41.

Sandén, B.A., Karlström, M., 2007. Positive and negative feedback in consequential life-cycle assessment. J Clean Prod 15, 1469–1481. https://doi.org/https://doi.org/10.1016/j.jclepro.2006.03.005

Schneider, M., Hoenig, V., Ruppert, J., Rickert, J., 2023. The cement plant of tomorrow. Cem Concr Res 173. https://doi.org/10.1016/j.cemconres.2023.107290

Scientific Applications International Corporation (SAIC), 2006. Life-cycle Assessment: Principles and Practice.

Scrivener, K.L., John, V.M., Gartner, E.M., 2018. Eco-efficient cements: Potential economically viable solutions for a low-CO2 cement-based materials industry. Cem Concr Res 114, 2–26. https://doi.org/10.1016/j.cemconres.2018.03.015

Senff, L., Modolo, R.C.E., Silva, A.S., Ferreira, V.M., Hotza, D., Labrincha, J.A., 2014. Influence of red mud addition on rheological behavior and hardened properties of mortars. Constr Build Mater 65, 84–91. https://doi.org/https://doi.org/10.1016/j.conbuildmat.2014.04.104

Seto, K.E., Churchill, C.J., Panesar, D.K., 2017. Influence of fly ash allocation approaches on the life cycle assessment of cement-based materials. J Clean Prod 157, 65–75. https://doi.org/10.1016/j.jclepro.2017.04.093

Sgarbossa, A., Boschiero, M., Pierobon, F., Cavalli, R., Zanetti, M., 2020. Comparative life cycle assessment of bioenergy production from differentwood pellet supply chains. Forests 11, 1–16. https://doi.org/10.3390/f11111127

Shibasaki, M., Albrecht, S., Kupfer, T., 2007. Small Scale and Large Scale Plants-Effect on Life Cycle Assessment.

Shibasaki, M., Warburg, N., Eyerer, P., 2006. Upscaling effect and Life Cycle Assessment.

Shobeiri, V., Bennett, B., Xie, T., Visintin, P., 2021. A comprehensive assessment of the global warming potential of geopolymer concrete. J Clean Prod 297. https://doi.org/10.1016/j.jclepro.2021.126669

Simon, B., Bachtin, K., Kiliç, A., Amor, B., Weil, M., 2016. Proposal of a framework for scale-up life cycle inventory: A case of nanofibers for lithium iron phosphate cathode applications. Integr Environ Assess Manag 12, 465–477. https://doi.org/10.1002/ieam.1788

Snellings, R., Suraneni, P., Skibsted, J., 2023. Future and emerging supplementary cementitious materials. Cem Concr Res 171, 107199. https://doi.org/https://doi.org/10.1016/j.cemconres.2023.107199

Sohn, J., Kalbar, P., Goldstein, B., Birkved, M., 2020. Defining Temporally Dynamic Life Cycle Assessment: A Review. Integr Environ Assess Manag. https://doi.org/10.1002/ieam.4235

Sousa, I., Eisenhard, J.L., Wallace, D., 2001. Approximate Life-Cycle Assessment of Product Concepts Using Learning Systems.

Spielmann, M., Scholz, R.W., Tietje, O., De Haan, P., 2005. Scenario modelling in prospective LCA of transport systems: Application of formative scenario analysis. International Journal of Life Cycle Assessment 10, 325–335. https://doi.org/10.1065/lca2004.10.188

Stehfest, E., Vuuren, D., Kram, T., Bouwman, Alexander, Alkemade, R., Bakkenes, M., Biemans, H., Bouwman, A, Elzen, M., Janse, J., Lucas, P., van Minnen, J., Müller, M., Prins, A., 2014. Integrated Assessment of Global Environmental Change with IMAGE 3.0. Model description and policy applications.

Sterpellone, A., 2020. Circular economy potential of inorganic polymers from bauxite residues and other slags: An LCA study.

Steubing, B., de Koning, D., 2021. Making the use of scenarios in LCA easier: the superstructure approach. International Journal of Life Cycle Assessment. https://doi.org/10.1007/s11367-021-01974-2

Steubing, B., de Koning, D., Haas, A., Mutel, C.L., 2020. The Activity Browser — An open source LCA software building on top of the brightway framework. Software Impacts 3, 100012. https://doi.org/10.1016/j.simpa.2019.100012

Steubing, B., Wernet, G., Reinhard, J., Bauer, C., Moreno-Ruiz, E., 2016. The ecoinvent database version 3 (part II): analyzing LCA results and comparison to version 2. International Journal of Life Cycle Assessment 21, 1269–1281. https://doi.org/10.1007/s11367-016-1109-6

Sutar, H., Mishra, S.C., Sahoo, S.K., Chakraverty, A.P., Maharana, H.S., 2014. Progress of red mud utilization: an overview. Am. Chem. Sci. J. 4, 255 – 279.

Tang, W.C., Wang, Z., Liu, Y., Cui, H.Z., 2018. Influence of red mud on fresh and hardened properties of self-compacting concrete. Constr Build Mater 178, 288–300. https://doi.org/10.1016/j.conbuildmat.2018.05.171

Tecchio, P., Freni, P., De Benedetti, B., Fenouillot, F., 2016. Ex-ante Life Cycle Assessment approach developed for a case study on bio-based polybutylene succinate. J Clean Prod 112, 316–325. https://doi.org/10.1016/j.jclepro.2015.07.090

Bourtsalas, A.C., Zhang, J., Castaldi, M.J., Themelis, N.J., 2018. Use of non-recycled plastics and paper as alternative fuel in cement production. J Clean Prod 181, 8–16. https://doi.org/10.1016/j.jclepro.2018.01.214

The Business Research Company, 2019. Cement and concrete global market opportunities and strategies to 2022.

Thonemann, N., Schulte, A., 2019. From Laboratory to Industrial Scale: A Prospective LCA for Electrochemical Reduction of CO2 to Formic Acid. Environ Sci Technol 53, 12320–12329. https://doi.org/10.1021/acs.est.9b02944

Thonemann, N., Schulte, A., Maga, D., 2020. How to conduct prospective life cycle assessment for emerging technologies? A systematic review and methodological guidance. Sustainability (Switzerland). https://doi.org/10.3390/su12031192

Thwe, E., Khatiwada, D., Gasparatos, A., 2021. Life cycle assessment of a cement plant in Naypyitaw, Myanmar. Cleaner Environmental Systems 2, 100007. https://doi.org/10.1016/j.cesys.2020.100007

Tilak, P., El-Halwagi, M.M., 2018. Process integration of Calcium Looping with industrial plants for monetizing CO2 into value-added products. Carbon Resources Conversion 1, 191–199. https://doi.org/10.1016/j.crcon.2018.07.004

Tillman, A.-M., Ekvall, T., Baumann, H., Rydbergl, T., 1994. Choice of system boundaries in life cycle assessment.

Tomatis, M., Jeswani, H.K., Stamford, L., Azapagic, A., 2020. Assessing the environmental sustainability of an emerging energy technology: Solar thermal calcination for cement production. Science of the Total Environment 742. https://doi.org/10.1016/j.scitotenv.2020.140510

Toniolo, N., Rincón, A., Avadhut, Y.S., Hartmann, M., Bernardo, E., Boccaccini, A.R., 2018. Novel geopolymers incorporating red mud and waste glass cullet. Mater Lett 219, 152–154. https://doi.org/10.1016/j.matlet.2018.02.061

Tsakiridis, P.E., Agatzini-Leonardou, S., Oustadakis, P., 2004. Red mud addition in the raw meal for the production of Portland cement clinker. J Hazard Mater 116, 103–110. https://doi.org/10.1016/j.jhazmat.2004.08.002

Tsalidis, G.A., Korevaar, G., 2022. Environmental assessments of scales: The effect of ex-ante and ex-post data on life cycle assessment of wood torrefaction. Resour Conserv Recycl 176. https://doi.org/10.1016/j.resconrec.2021.105906

Tsang, M.P., Li, D., Garner, K.L., Keller, A.A., Suh, S., Sonnemann, G.W., 2017. Modeling human health characterization factors for indoor nanomaterial emissions in life cycle assessment: A case-study of titanium dioxide. Environ Sci Nano 4, 1705–1721. https://doi.org/10.1039/c7en00251c

Tsoy, N., Steubing, B., van der Giesen, C., Guinée, J., 2020. Upscaling methods used in ex ante life cycle assessment of emerging technologies: a review. International Journal of Life Cycle Assessment. https://doi.org/10.1007/s11367-020-01796-8

Tufvesson, L.M., Tufvesson, P., Woodley, J.M., Börjesson, P., 2013. Life cycle assessment in green chemistry: Overview of key parameters and methodological concerns. International Journal of Life Cycle Assessment 18, 431–444. https://doi.org/10.1007/s11367-012-0500-1

Tun, T.Z., Bonnet, S., Gheewala, S.H., 2020. Life cycle assessment of Portland cement production in Myanmar. International Journal of Life Cycle Assessment 25, 2106–2121. https://doi.org/10.1007/s11367-020-01818-5

Turner, B.T., Plevin, R.J., O'hare, M., Org, E., 2007. UC Berkeley Recent Work Title Creating Markets for Green Biofuels: Measuring and improving environmental performance Publication Date.

Ulubeyli, S., Kazaz, A., Arslan, V., 2017. Construction and Demolition Waste Recycling Plants Revisited: Management Issues, in: Procedia Engineering. Elsevier Ltd, pp. 1190–1197. https://doi.org/10.1016/j.proeng.2017.02.139

UN DESA, 2015. World Population Prospects: The 2015 Revision.

UNFCCC, 2015. Adoption of the Paris Agreement - Paris Agreement.

UNION PROCESS, 2024. Attrition mills [WWW Document].

Valente, A., Iribarren, D., Dufour, J., 2019. Life cycle sustainability assessment of hydrogen from biomass gasification: A comparison with conventional hydrogen. Int J Hydrogen Energy 44, 21193–21203. https://doi.org/10.1016/j.ijhydene.2019.01.105

Valsasina, L., Pizzol, M., Smetana, S., Georget, E., Mathys, A., Heinz, V., 2017. Life cycle assessment of emerging technologies: The case of milk ultra-high pressure homogenisation. J Clean Prod 142, 2209–2217. https://doi.org/10.1016/j.jclepro.2016.11.059

Van Den Heede, P., De Belie, N., 2012. Environmental impact and life cycle assessment (LCA) of traditional and "green" concretes: Literature review and theoretical calculations. Cem Concr Compos 34, 431–442. https://doi.org/10.1016/j.cemconcomp.2012.01.004

Van der Giesen, C., Cucurachi, S., Guinée, J., Kramer, G.J., Tukker, A., 2020. A critical view on the current application of LCA for new technologies and recommendations for improved practice. J Clean Prod 259. https://doi.org/10.1016/j.jclepro.2020.120904

Van Der Giesen, C., Kleijn, R., Kramer, G.J., Guinée, J., 2013. Towards application of life cycle sustainability analysis. Revue de Metallurgie. Cahiers D'Informations Techniques 110, 29–36. https://doi.org/10.1051/metal/2013058

Van der Hulst, M.K., Huijbregts, M.A.J., van Loon, N., Theelen, M., Kootstra, L., Bergesen, J.D., Hauck, M., 2020. A systematic approach to assess the environmental impact of emerging technologies: A case study for the GHG footprint of CIGS solar photovoltaic laminate. J Ind Ecol 24, 1234–1249. https://doi.org/10.1111/jiec.13027

Van Oers, L., Guinée, J., 2016. The abiotic depletion potential: Background, updates, and future. Resources. https://doi.org/10.3390/resources5010016

Van Oss. Hendrik G., 2016. 2014 Minerals Yearbook: Cement, United States Geological Survey data release.

Van Oss, H.G., 2007. MINERAL COMMODITY SUMMARIES 2007.

Van Ruijven, B.J., Levy, M.A., Agrawal, A., Biermann, F., Birkmann, J., Carter, T.R., Ebi, K.L., Garschagen, M., Jones, B., Jones, R., Kemp-Benedict, E., Kok, M., Kok, K., Lemos, M.C., Lucas, P.L., Orlove, B., Pachauri, S., Parris, T.M., Patwardhan, A., Petersen, A., Preston, B.L., Ribot, J., Rothman, D.S., Schweizer, V.J., 2014. Enhancing the relevance of shared socioeconomic pathways for climate change impacts, adaptation, and vulnerability research. Clim Change 122, 481–494. https://doi.org/10.1007/s10584-013-0931-0

Van Vuuren, D.P., Carter, T.R., 2014. Climate and socio-economic scenarios for climate change research and assessment: Reconciling the new with the ol. Clim Change 122, 415–429. https://doi.org/10.1007/s10584-013-0974-2

van Vuuren, D. P., Stehfest, E., Gernaat, D. E. H. J., Doelman, J. C., van den Berg, M., Harmsen, M., de Boer, H. S., Bouwman, L. F., Daioglou, V., Edelenbosch, O. Y., Girod, B., Kram, T., Lassaletta, L., Lucas, P. L., van Meijl, H., Müller, C., van Ruijven, B. J., van der Sluis, S., & Tabeau, A. (2017). Energy, land-use, and greenhouse gas emissions trajectories under a green growth paradigm. *Global Environmental Change*, *42*, 237–250. https://doi.org/10.1016/j.gloenvcha.2016.05.008

Vaughan, J., Peng, H., Seneviratne, D., Hodge, H., Hawker, W., Hayes, P., Staker, W., 2019. The Sandy Desilication Product Process Concept. JOM 71, 2928–2935. https://doi.org/10.1007/s11837-019-03617-2

Venkatesh, C., Chand, M.S.R., Nerella, R., 2019. A state of the art on red mud as a substitutional cementitious material. Annales de Chimie: Science des Materiaux 43, 99–106. https://doi.org/10.18280/acsm.430206

Villares, M., Işildar, A., Mendoza Beltran, A., Guinee, J., 2016. Applying an ex-ante life cycle perspective to metal recovery from e-waste using bioleaching. J Clean Prod 129, 315–328. https://doi.org/10.1016/j.jclepro.2016.04.066

Villares, M., Işıldar, A., van der Giesen, C., Guinée, J., 2017. Does ex ante application enhance the usefulness of LCA? A case study on an emerging technology for metal recovery from

e-waste. Int J Life Cycle Assess 22, 1618–1633. https://doi.org/10.1007/s11367-017-1270-6

Viyasun, K., Anuradha, R., Thangapandi, K., Santhosh Kumar, D., Sivakrishna, A., Gobinath, R., 2021. Investigation on performance of red mud based concrete. Mater Today Proc 39, 796–799. https://doi.org/https://doi.org/10.1016/j.matpr.2020.09.637

Vladić Kancir, I., Serdar, M., 2022. Contribution to Understanding of Synergy between Red Mud and Common Supplementary Cementitious Materials. Materials 15. https://doi.org/10.3390/ma15051968

Volkart, K., Bauer, C., Boulet, C., 2013. Life cycle assessment of carbon capture and storage in power generation and industry in Europe. International Journal of Greenhouse Gas Control 16, 91–106. https://doi.org/10.1016/j.ijggc.2013.03.003

Walczak, K.A., Hutchins, M.J., Dornfeld, D., 2014. Energy system design to maximize net energy production considering uncertainty in scale-up: A case study in artificial photosynthesis, in: Procedia CIRP. Elsevier B.V., pp. 306–312. https://doi.org/10.1016/j.procir.2014.06.032

Wang, J., Liu, X., Zhang, Z., Liu, Y., 2024. Synergistic utilization, critical mechanisms, and environmental suitability of bauxite residue (red mud) based multi-solid wastes cementitious materials and special concrete. J Environ Manage. https://doi.org/10.1016/j.jenvman.2024.121255

Wang, Y., Pan, Z., Zhang, W., Borhani, T.N., Li, R., Zhang, Z., 2022. Life cycle assessment of combustion-based electricity generation technologies integrated with carbon capture and storage: A review. Environ Res 207, 112219. https://doi.org/10.1016/J.ENVRES.2021.112219

Wardenaar, T., van Ruijven, T., Beltran, A.M., Vad, K., Guinée, J., Heijungs, R., 2012. Differences between LCA for analysis and LCA for policy: a case study on the consequences of allocation choices in bio-energy policies. Int J Life Cycle Assess 17, 1059–1067. https://doi.org/10.1007/s11367-012-0431-x

WBCSD, 2016. The Cement Sustainability Initiative (CSI) Cement Industry Energy and CO 2 Performance Getting the Numbers Right (GNR).

Weidema B.P, Bauer C., Hischier R., Mutel C., Nemecek T., Reinhard J., Vadenbo C.O., Wernet G., 2013. Overview and methodology: Data quality guideline for the ecoinvent database version 3.

Wender, B.A., Foley, R.W., Hottle, T.A., Sadowski, J., Prado-Lopez, V., Eisenberg, D.A., Laurin, L., Seager, T.P., 2014a. Anticipatory life-cycle assessment for responsible research and innovation. J Responsible Innov 1, 200–207. https://doi.org/10.1080/23299460.2014.920121

Wender, B.A., Foley, R.W., Prado-Lopez, V., Ravikumar, D., Eisenberg, D.A., Hottle, T.A., Sadowski, J., Flanagan, W.P., Fisher, A., Laurin, L., Bates, M.E., Linkov, I., Seager, T.P., Fraser, M.P., Guston, D.H., 2014b. Illustrating anticipatory life cycle assessment for emerging photovoltaic technologies. Environ Sci Technol 48, 10531–10538. https://doi.org/10.1021/es5016923

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. International Journal of Life Cycle Assessment 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Wernet, G., Papadokonstantakis, S., Hellweg, S., Hungerbühler, K., 2009. Bridging data gaps in environmental assessments: Modeling impacts of fine and basic chemical production. Green Chemistry 11, 1826–1831. https://doi.org/10.1039/b905558d

Weyand, S., Wittich, C., Schebek, L., 2019. Environmental Performance of Emerging Photovoltaic Technologies: Assessment of the Status Quo and Future Prospects Based on a Meta-Analysis of Life-Cycle Assessment Studies. Energies (Basel) 12. https://doi.org/10.3390/en12224228

Whittington, B.I., Fletcher, B.L., Talbot, C., 1998. The effect of reaction conditions on the composition ž / of desilication product DSP formed under simulated Bayer conditions, Hydrometallurgy.

Willan, M., Ghataora, G., 2015. Management of bauxite residue in a temperate climate using mud-farming techniques, in: Proceedings of the 18th International Seminar on Paste and Thickened Tailings. Australian Centre for Geomechanics, Perth, pp. 209–222. https://doi.org/10.36487/acg_rep/1504_14_willan

World resources institute, 2024. G7 Leaders Agree to Shut Down Coal Plants [WWW Document]. World resources institute .

Wunderlich, J., Armstrong, K., Buchner, G.A., Styring, P., Schomäcker, R., 2021. Integration of techno-economic and life cycle assessment: Defining and applying integration types for chemical technology development. J Clean Prod 287, 125021. https://doi.org/10.1016/J.JCLEPRO.2020.125021

Yang, J., Zhang, D., Hou, J., He, B., Xiao, B., 2008. Preparation of glass-ceramics from red mud in the aluminium industries. Ceram Int 34, 125–130. https://doi.org/10.1016/j.ceramint.2006.08.013

Yao, X., Wang, W., Liu, M., Yao, Y., Wu, S., 2019. Synergistic use of industrial solid waste mixtures to prepare ready-to-use lightweight porous concrete. J Clean Prod. https://doi.org/10.1016/j.jclepro.2018.11.252

Yao, Y., Hu, M., Di Maio, F., Cucurachi, S., 2020. Life cycle assessment of 3D printing geo-polymer concrete: An ex-ante study. J Ind Ecol 24, 116–127. https://doi.org/10.1111/jiec.12930

Yao, Y., Li, Y., Liu, X., Jiang, S., Feng, C., Rafanan, E., 2013. Characterization on a cementitious material composed of red mud and coal industry byproducts. Constr Build Mater 47, 496–501. https://doi.org/https://doi.org/10.1016/j.conbuildmat.2013.05.030

Yingtao, Z., 2021. Life Cycle Assessment of Concrete from Bauxite Residue in China A case study based on “applying the zero-waste concepts” to the Chinese Aluminium industry.

Yu, B., Liu, Q., Gu, X., 2018. Data quality and uncertainty assessment methodology for pavement LCA. International Journal of Pavement Engineering 19, 519–525. https://doi.org/10.1080/10298436.2016.1176166

Zaimes, G., Soratana, K., Harden, C., Landis, A., Khanna, V., 2015. Biofuels via Fast Pyrolysis of Perennial Grasses: A Life Cycle Evaluation of Energy Consumption and Greenhouse Gas Emissions. Environ Sci Technol 49. https://doi.org/10.1021/acs.est.5b00129

Zawal, D., Grabiec, A.M., 2022. Influence of selected mineral additives on properties of recycled aggregate concrete (RAC) considering eco-efficiency coefficients. Case Studies in Construction Materials 17, e01405. https://doi.org/10.1016/j.cscm.2022.e01405

Zhang, N., Sun, H., Liu, X., Zhang, J., 2009. Early-age characteristics of red mud-coal gangue cementitious material. J Hazard Mater 167, 927–932. https://doi.org/10.1016/j.jhazmat.2009.01.086

Zhang, R., Zheng, S., Ma, S., Zhang, Y., 2011. Recovery of alumina and alkali in Bayer red mud by the formation of andradite-grossular hydrogarnet in hydrothermal process. J Hazard Mater 189, 827–835. https://doi.org/10.1016/j.jhazmat.2011.03.004

Appendix A

Appendix A presents the detailed LCIs for upscaling emerging vitrification technology, which is used to modify BR into an SCM (covered in Chapter 4). The LCIs consider three scales: laboratory, pilot, and industrial. Following this, the appendix includes the LCIs for the industrial scale upscaling approaches such as process modelling, regression analysis, and proxy technologies. A detailed discussion of furnace technology is also included, which outlines the key considerations in the vitrification process. Additionally, Appendix A provides quantitative and tabulated data on the environmental performance during upscaling at each scale, as demonstrated in Figure 4-5. Finally, it presents the environmental performance of different industrial upscaling approaches, visualised in Figure *4-6* and Figure *4-7*.

A.1 Life cycle inventory

The LCI for the unit process datasets used for the different scales (i.e., laboratory, pilot, and industrial) and industrial upscaling approaches (i.e., process modelling, regression analysis, and proxy technology), are presented in tables A1-A6.

A.1.1 Laboratory scale

Table A1: LCA data for the BR vitrification at the laboratory scale

Lab Scale						
Vitrification						
Material input	**Quantity**	**Unit**	**Material output**	**Quantity**	**Unit**	**Source**
Bauxite residue	1.75	kg	VBR	1.79	kg	
Limestone	0.18	kg	CO_2	0.18	kg	
Coke	0.04	kg	Vapour	0.13	kg	
Silica sand	0.13	kg				

Lab Scale						
Vitrification						
Energy input	**Quantity**	**Unit**				
Mixer	1.90	kWh				(HOBART, 2024)
Dryer 1	3.40	kWh				(BINDER, 2024)
Vitrification	29	kWh				(INDUTHERM, 2024)
Dryer 2	3.40	kWh				(BINDER, 2024)
Mill	8.80	kWh				(UNION PROCESS, 2024)

A.1.2 Pilot

Table A2: LCA data for the BR vitrification at the pilot scale

Pilot Scale						
Vitrification						
Material input	**Quantity**	**Unit**	**Material output**	**Quantity**	**Unit**	**Source**
Bauxite residue	100	kg	VBR	102.06	kg	
Limestone	10.43	kg	CO_2	10.36	kg	
Coke	2.04	kg	Vapour	7.45	kg	
Silica sand	7.40	kg				
Energy input	**Quantity**	**Unit**				
Dryer	38.40	kWh				(DESPATCH, 2024)
Mixer	2.50	kWh				(MEKA, 2024)
Vitrification	300	kWh				(OTTOJUNKER, 2024)

Pilot Scale						
Vitrification						
Energy input	**Quantity**	**Unit**				
Mill	65.90	kWh				(UNION PROCESS, 2024)

A.1.3 Industrial scale

A.1.3.1 Process modelling

Table A3 presents the data inventory for the upscaling of BR vitrification to industrial scale with process modelling.

Table A3: LCA data for the BR vitrification at the industrial scale

Process modelling					
Vitrification					
Material input	**Quantity**	**Unit**	**Material output**	**Quantity**	**Unit**
Bauxite residue	100	tonne	VBR	125.80	tonne
Limestone	22	tonne	CO_2	25.88	tonne
Coke	2.04	tonne	Vapour	9.18	tonne
Silica sand	15	tonne			
Bentonite (binder)	1.50	tonne			
Energy input	**Quantity**	**Unit**			
Mixer	-	kWh			
Pelletiser	587	kwh			
Preheater	1700	kWh			
Vitrification	59000	kWh			
Mill	48	kWh			

A.1.3.2 Regression analysis

Existing literature and industrial data sheets for unit operations (i.e., dryer, mixer, pelletiser, furnace, cooler, and mill) were used to establish correlations between material input and energy consumption. These correlations were graphically represented. Regression analysis was applied to estimate energy consumption data based on material inputs, by fitting both linear and power relationships.

Table A4 presents the data inventory for upscaling BR vitrification to an industrial scale with power and linear regression analysis.

Table A4: LCA data for the BR vitrification at the industrial scale

Power/Linear					
Vitrification					
Material input	**Quantity**	**Unit**	**Material output**	**Quantity**	**Unit**
Bauxite residue	100	tonne	VBR	125.80	tonne
Limestone	22	tonne	CO_2	25.88	tonne
Coke	2.04	tonne	Vapour	9.18	tonne
Silica sand	15	tonne			
Bentonite (binder)	1.5	tonne			
Energy input	**Quantity**	**Unit**			
Dryer 1	186/210	kWh			
Mixer	130/132	kWh			
Pelletisation	355/363	kwh			
Vitrification	68445/71347	kWh			
Cooler	25/27	kWh			
Mill	75/82	kWh			

A.1.3.1 Proxy technology

Table A5 presents the data inventory for upscaling BR vitrification to an industrial scale using clinker production as a proxy technology. Two scenarios are considered: scenario 1, clinker production modelled according to Centre et al., 2013 and scenario 2, modelled based on ecoinvent v3.9.1 "clinker production - Europe without Switzerland (Ecoinvent v.3.9.1, 2007a).

Table A5: LCA data for the BR vitrification at the industrial scale

Clinker production Scenario 1/ Scenario 2					
Vitrification					
Material input	**Quantity**	**Unit**	**Material output**	**Quantity**	**Unit**
Bauxite residue	100	tonne	VBR	125.80	tonne
Limestone	22	tonne	CO_2	25.88	tonne
Coke	2.04	tonne	Vapour	9.18	tonne
Silica sand	15	tonne			
Bentonite (binder)	1.5	tonne			
Energy input	**Quantity**	**Unit**			
Grinder	16688/16688	kWh			
Furnace	74215/ 115304	kWh			

Table A6 presents the data inventory for upscaling BR vitrification to an industrial scale using clinker production as a proxy technology. Two scenarios are considered: scenario 3, ceramic tile production modelled according to European Commission. (2007) and scenario 4, modelled based on ecoinvent v.3.9.1 "ceramic tile production - RoW - ceramic tile" (Ecoinvent v.3.9.1, 2007b).

Table A6: LCA data for the BR vitrification at the industrial scale

Ceramic production Scenario 3/ Scenario 4					
Vitrification					
Material input	**Quantity**	**Unit**	**Material output**	**Quantity**	**Unit**
Bauxite residue	100	tonne	VBR	125.80	tonne
Limestone	22	tonne	CO_2	25.88	tonne
Coke	2.04	tonne	Vapour	9.18	tonne
Silica sand	15	tonne			
Bentonite (binder)	1.50	tonne			
Energy input	**Quantity**	**Unit**			
Grinder	17055/16688	kWh			
Pelletiser	3904/0	kWh			
Preheater	22448/0	kWh			
Furnace	120022/ 234219	kWh			

A.1.3.1.2 Proxy Selection

A thorough evaluation of technologies with similar operating conditions, such as technology type (vitrification), material properties, and firing temperature, was conducted to select appropriate proxies. The selected proxies were used as reference technologies to estimate the energy demand and associated emissions from energy production. Clinker production and ceramic vitrification were identified as suitable proxies. Relevant processing stages were identified, and energy consumption data were collected on a product basis (per kilogram of vitrified clay or clinker). This addressed the variability in energy demands across different furnaces and equipment types, each with distinct residence times. Modelling considerations for proxy technologies i.e., clinker production and ceramic tile production are presented in Table A7-A8.

Table A7. Modelling assumptions and data sources for proxy scenarios

Scenario no	Proxy material	Data source	Assumptions or specific conditions
Scenario 1	Clinker production	Best Available Techniques (BAT) reference document for the production of cement, lime and magnesium oxide	• Dry process: five stage preheater, precalciner, grate cooler, vertical raw mill • Fuel: 100 % petroleum coke • Heat consumption: 3 300 kJ/kg clinker • Pet coke (net) calorific value (NCV): 33 500 kJ/kg fuel • Furnace operating temperature: 1450 °C
Scenario 2	Clinker production	ecoinvent 3.9.1 – 'clinker production - Europe without Switzerland'	• The dataset describes the production of Portland cement clinker in the EU without Switzerland • Furnace operating temperature: 1450 °C
Scenario 3	Ceramic tile production	Best Available Techniques (BAT) Reference document for the ceramic manufacturing industry	• Fuel composition: 98.1% natural gas, 1.1% fuel oil, 0.8% heavy fuel oil • Furnace operating temperature: 1150 °C
Scenario 4	Ceramic tile production	ecoinvent 3.9.1 – 'ceramic tile production - RoW - ceramic tile'	• Technology: The infrastructure data corresponds to large-scale production in Europe, using gas-fired kiln technology • Ceramic tiles are produced by slurry processing. • Furnace operating temperature: 1150 °C

Table A8: Unit processes and data inventory for proxy scenarios

		Thermal energy		**Electricity**		**Reference**
Scenario 1	Total energy consumption	3.3	MJ/kg clinker	96	kWh/t	Centre et al., 2013
Scenario 2	Total electricity consumption			0.058	kWh/kg clinker	Ecoinvent v.3.9.1, 2007a
	Hard coal consumption	0.0354	kg/kg clinker			
	Heavy fuel oil consumption	0.0255	kg/kg clinker			
	Light fuel oil consumption	0.000374	kg/kg clinker			
	natural gas consumption	0.000175	m^3/kg clinker			
	Petroleum coke consumption	0.00391	kg/kg clinker			
	Diesel consumption	0.0134	MJ/kg clinker			
Scenario 3	Feed preparation, dry grinding			15.28	kWh/t	European Commission, 2007
	Shaping, pressing			27.78	kWh/t	
	Drying	0.55	GJ/t	6.94	kWh/t	
	Firing	3.35	GJ/t	23.61	kWh/t	
Scenario 4	Total energy consumption	5.5765	MJ/kg	0.313	kWh/kg	Ecoinvent v.3.9.1, 2007b

A.2 Ground granulated blast furnace slag (GGBFS)

Ground granulated blast furnace slag (GGBFS) was modelled using the cut-off approach. Although GGBFS meets all the conditions set by the European Union Directive (European Union, 2008) to be classified as an industrial by-product rather than a waste, it is generated during pig iron production. The cut-off approach was used to assure consistency with the processing of BR (classified as waste). The inventory for GGBFS production includes the (i) quenching/granulation, (ii) dewatering/ drying, (iii) crushing, and (iv) grinding of molten blast furnace slag.

A.3 Limitations in upscaling methodology

Limitations in upscaling emerging technologies depend on the upscaling approach (process modelling, regression analysis, and proxy technology).

Using Aspen Plus presents challenges due to the lack of components (waste flows, emerging materials) such as bauxite residue in its database, leading to missing thermodynamic properties (e.g., heat capacities). Hence, average data from existing literature must be used. In process simulation, bauxite residue was modelled based on its oxide composition, according to Hertel et al. (2016). This shows the need for further development of representative datasets to enhance the accuracy and comprehensiveness of emerging technology simulations. Similar constraints have been reported in prior ex-ante studies, such as Cuéllar-Franca et al. (2016). Another limitation of Aspen Plus is the inability to capture phase transitions and the enthalpy changes during the conversion of a solid to a partially melted slag (as in VBR). Hence, thermodynamic calculations were used to estimate the heat transfer required during phase changes and phase formation (Giels, 2023).

Limitations in proxy technologies involve adapting established processes to novel applications, assuming analogous performance based on existing data (e.g., efficiencies, yields, and material flows) (Buyle et al., 2019). Proxy technologies may not capture process synergies, such as heat integration and recovery or carbon capture and storage techniques, designed to minimise energy demands and associated process emissions.

Regression analysis has limitations in capturing non-linear relationships between uncertain process parameters such as efficiencies and yields (Buyle et al., 2019). These parameters are linked not only to scale but also to technical performance. Additional limitations include the inability to consider process dynamics, thermodynamic changes, and process synergies.

A.5 Furnace technology

A.4.1 Semi-quantitative discussion on furnace technology selection

Selecting suitable industrial equipment is crucial for process system productivity, efficiency, and safety (Paramasivam et al., 2011). The equipment selection process aims to achieve desired quality levels, cost-effectiveness, and reliability ensuring sustainable industrial operations. In furnace selection, close collaboration with industrial experts and technology developers is essential. In the present study, a generic electric-powered furnace model was employed for BR vitrification. Possible electric-powered furnace examples include induction furnaces, electric arc furnaces, microwaving and submerged arc furnaces.

Despite being electric-powered, electric arc and submerged arc furnaces led to a partial reduction of Fe, resulting in the co-production of metallic Fe and an Fe-depleted VBR, which is unsuitable as an SCM. Induction furnaces, however, can produce a VBR with the desired Fe content. Therefore, selecting the appropriate furnace requires a thorough assessment of factors such as product (VBR) quality, properties (i.e., reactivity) and processing parameters (cost, and efficiency) for each furnace type.

A rotary kiln furnace is another viable option that can ensure the desired Fe content. However, as a fuel-powered technology, its environmental performance particularly the climate change impact, depends on the fuel mix composition. As discussed in Section 3.2, the decarbonisation of the electricity grid plays a critical role in the environmental performance of the vitrification process. Hence, fuel mix selection for the rotary kiln should consider future trends and opportunities for decarbonisation utilising alternative fuels such as waste materials, biomass,

hydrogen, etc. In the cement industry, the use of alternative materials varies regionally, with a global average replacement of ~6%. Common waste-derived fuels include tyres, oils, plastics, and textiles, while increasing demand for biomass, such as waste wood and sawdust, plays a significant role in reducing CO_2 emissions.

Plasma technology is a promising alternative, generating a hot beam of ionised plasma that reaches temperatures from 1500°C to over 7000°C. However, plasma technology has not yet been implemented in industrial-scale cement manufacturing, with only a few conceptual applications and methods explored (Schneider et al., 2023).

Selecting an appropriate furnace technology also involves evaluating scalability and practicality. The rotary kiln furnace is well-established in various industries, particularly in cement production and may appear more suitable compared to the induction furnace in certain applications.

The thermal efficiency and scalability of industrial, electric-powered furnaces vary by type, leading to differences in energy demands. Microwaving shows an efficiency range of 55% to 70% with some cases reaching up to 82% (Priecel and Lopez-Sanchez, 2019). Electric arc furnaces achieve efficiency rates between 36% and 93% (Pfeifer et al., 2005; Kirschen et al., 2009). Induction furnaces typically operate at 20% to 50% efficiency but can attain 75% with best operating practices (IPPC Bureau et al., 2022). Shaft furnaces, used in glass, ceramic, automotive and aviation industries, exhibit efficiency rates of 40% to 50% for non-ferrous applications, while reverberatory furnaces range from 15% to 40% (Cusano et al., 2017). These wide efficiency ranges highlight the critical importance of considering energy efficiency factors when selecting kiln types.

A.5 Results

A.5.1 Effect of technology scale

Table A9 presents the effects of upscaling from the laboratory to pilot and industrial scale. The industrial scale results represent the average values based on the seven upscaling scenarios explored.

Table A9: ReCiPe 2016 midpoint impact results of laboratory, pilot and industrial (mean) scale vitrification technology

Impact categories	**Units**	**Laboratory**	**Pilot**	**Industrial (mean)**
Terrestrial acidification potential (TAP)	kg SO_2-eq.	0.035	0.005	0.001
Global warming potential (GWP100) [a]	kg CO_2-eq.	10.092	1.688	0.559
Freshwater ecotoxicity potential (FETP)	kg 1,4-D, C, B-eq.	1.095	0.060	0.007
Marine ecotoxicity potential (METP)	kg 1,4-D, C, B-eq.	1.389	0.079	0.010
Terrestrial ecotoxicity potential (TETP)	kg 1,4-D, C, B-eq.	52.696	2.361	0.414
Fossil fuel potential (FFP)	kg oil-eq.	2.436	0.418	0.103
Freshwater eutrophication potential (FEP)	kg P-eq.	8.22×10^{-3}	1.29×10^{-3}	1.59×10^{-4}
Marine eutrophication potential (MEP)	kg N-eq.	6.08×10^{-4}	9.96×10^{-5}	1.24×10^{-5}
Human toxicity potential (HTPc)	kg 1,4-D, C, B-eq.	0.632	0.101	0.014
Human toxicity potential (HTPnc)	kg 1,4-D, C, B-eq.	16.722	1.747	0.236
Ionising radiation potential (IRP)	kg Co-60-eq.	5.096	0.843	0.071
Agricultural land occupation (LOP)	m^2 a crop-eq.	0.285	0.043	0.005
Material resources: metals/minerals | surplus ore potential (SOP)	kg Cu-eq.	0.079	0.010	0.002
Ozone depletion potential (ODPinfinite)[b]	kg CFC-11-eq.	4.08×10^{-6}	6.60×10^{-7}	8.00×10^{-8}
Particulate matter formation potential (PMFP)[c]	kg PM2.5-eq.	0.014	0.002	4.13×10^{-4}
Photochemical oxidant formation potential: humans (HOFP)	kg NO_x-eq.	0.016	0.003	5.94×10^{-4}

Impact categories	**Units**	**Laboratory**	**Pilot**	**Industrial (mean)**
Photochemical oxidant formation potential: ecosystems (EOFP)	kg NO_x-eq.	0.017	0.003	6.31×10^{-4}
Water consumption potential (WCP)	m^3	0.153	0.018	0.002

[a] Reported using global warming potential over a 100-year horizon (GWP100).

[b] Measured in kg CFC-11-eq.; Chlorofluorocarbons (CFCs)

[c] Measured in kg PM2.5-eq.; Particulate matter (PM).

A.5.2 Effect of upscaling approach

Table A10 presents the impacts of industrial upscaling scenarios, which include; process simulation (M3.1.1), linear regression (M3.2.1), power regression (M3.2.2), clinker production modelled based on Centre et al. (2013), (M3.3.1), clinker production modelled based on Ecoinvent v.3.9.1. (2007a), (M3.3.2), ceramic vitrification modelled based on European Commission. 2007 M3.3.1), ceramic vitrification modelled based on Ecoinvent v.3.9.1. (2007b), (M3.3.2).

Table A10: ReCiPe 2016 midpoint impact results of individual upscaling scenarios for industrial scale vitrification technology

Impact categories	**Units**	**M3.1.1**	**M3.2.1**	**M3.2.2**	**M3.3.1**	**M3.3.2**	**M3.3.3**	**M3.3.4**
Terrestrial acidification potential (TAP)	kg SO_2-eq.	8.20×10^{-4}	7.50×10^{-4}	7.80×10^{-4}	1.38×10^{-3}	2.79×10^{-3}	3.30×10^{-3}	6.60×10^{-4}
Global warming potential (GWP100) [a]	kg CO_2-eq.	0.472	0.448	0.458	0.480	0.709	0.563	0.785
Freshwater ecotoxicity potential (FETP)	kg 1,4-D, C, B-eq.	0.009	0.008	0.009	0.005	0.010	0.003	0.007
Marine ecotoxicity potential (METP)	kg 1,4-D, C, B-eq.	0.012	0.011	0.012	0.006	0.014	0.004	0.010
Terrestrial ecotoxicity potential (TETP)	kg 1,4-D, C, B-eq.	0.403	0.370	0.383	0.620	0.442	0.286	0.393
Fossil fuel potential (FFP)	kg oil-eq.	0.076	0.070	0.073	0.073	0.110	0.125	0.192
Freshwater eutrophication potential (FEP)	kg P-eq.	2.00×10^{-4}	1.80×10^{-4}	1.90×10^{-4}	1.00×10^{-3}	2.70×10^{-3}	5.00×10^{-4}	1.30×10^{-4}
Marine eutrophication potential (MEP)	kg N-eq.	1.61×10^{-5}	1.46×10^{-5}	1.52×10^{-5}	8.97×10^{-6}	1.76×10^{-5}	3.97×10^{-6}	1.01×10^{-5}
Human toxicity potential (HTPc)	kg 1,4-D, C, B-eq.	0.015	0.014	0.014	0.009	0.021	0.008	0.015
Human toxicity potential (HTPnc)	kg 1,4-D, C, B-eq.	0.273	0.247	0.258	0.172	0.427	0.084	0.195

Impact categories	**Units**	**M3.1.1**	**M3.2.1**	**M3.2.2**	**M3.3.1**	**M3.3.2**	**M3.3.3**	**M3.3.4**
Ionising radiation potential (IRP)	kg Co-60-eq.	0.129	0.117	0.122	0.015	0.027	0.016	0.069
Agricultural land occupation (LOP)	m^2 a crop-eq.	0.007	0.007	0.007	0.003	0.005	0.002	0.005
Material resources: metals/minerals | surplus ore potential (SOP)	kg Cu-eq.	0.002	0.002	0.002	0.001	0.001	0.002	0.004
Ozone depletion potential (ODPinfinite)[b]	kg CFC-11-eq.	1.06×10^{-7}	9.61×10^{-8}	1.00×10^{-7}	5.00×10^{-8}	9.14×10^{-8}	4.44×10^{-8}	9.53×10^{-8}
Particulate matter formation potential (PMFP)[c]	kg PM2.5-eq.	3.30×10^{-4}	3.00×10^{-4}	3.10×10^{-4}	5.10×10^{-4}	9.80×10^{-4}	1.70×10^{-4}	3.00×10^{-4}
Photochemical oxidant formation potential: humans (HOFP)	kg NO_x-eq.	4.60×10^{-4}	4.20×10^{-4}	4.40×10^{-4}	‘6.00×10^{-4}	1.00×10^{-3}	3.90×10^{-4}	6.40×10^{-4}
Photochemical oxidant formation potential: ecosystems (EOFP)	kg NO_x-eq.	4.80×10^{-4}	4.40×10^{-4}	4.60×10^{-4}	‘6.40×10^{-4}	1.00×10^{-3}	4.40×10^{-4}	7.10×10^{-4}
Water consumption potential (WCP)	m^3	0.003	0.002	0.003	0.001	8.60×10^{-4}	6.60×10^{-4}	0.002

[a] Reported using global warming potential over a 100-year horizon (GWP100).

[b] Measured in kg CFC-11-eq.; Chlorofluorocarbons (CFCs)

[c] Measured in kg PM2.5-eq.; Particulate matter (PM).

Table A11 reports the process contributions in each upscaling scenario, including process simulation, regression analysis, and proxy technology.

Table A11: Process contributions in terms of kg CO_2-eq. emissions of VBR processed at industrial scale according to upscaling scenarios; process simulation, regression analysis, and proxy technology. "Other" represents the impact of upstream background activities such as hard coal preparation, and petroleum/natural gas production

Climate change (kg CO_2-eq. emissions)	**M3.1.1**	**M3.2.1**	**M3.2.2**	**M3.3.1**	**M3.3.2**	**M3.3.3**	**M3.3.4**
Other	0.05	0.05	0.05	0.07	0.07	0.10	0.13
Furnace (electricity consumption)	0.21	0.19	0.20	0.03	0.05	0.03	0.12
Furnace (heat consumption, natural gas)						0.23	0.33
Furnace (heat consumption, other than natural gas)				0.18	0.38		
Furnace (direct process emissions)	0.21	0.21	0.21	0.21	0.21	0.21	0.21

Appendix B

Appendix B discusses the foreground parameter selection for scenario development related to cement decarbonisation (presented in Chapter 5) and provides the LCIs associated with increasing PC clinker substitution with SCMs, increasing alternative fuels consumption, improving kiln type configurations, and implementing carbon capture and storage technologies. Further, it includes details on allocation approaches considered for commercially available SCMs (i.e., fly ash). Additionally, results from a Monte Carlo analysis are provided, examining the influence of uncertainties in clinker content influence the climate change impact, while the global sensitivity analysis identifies the most influential parameters.

B.1 Parameter selection for scenario development

Table B1 presents the parameters and their values that are used in our foreground scenarios that model changes in PC clinker-to-cement ratios, increasing alternative fuel utilization, improving kiln configuration, and carbon capture and storage technology. Table B2 shows the foreground scenarios and the options modelled for those scenarios.

Table B1: Foreground scenarios and corresponding parameter values.

	Scenario	**PC clinker substitution with SCMs**	**Increase in alternative fuel utilisation**	**Improving kiln configuration**	**Implementation of carbon capture and storage**
	Parameter	**Clinker-to-cement ratio**	**Individual fuel mix ratio**	**Thermal energy**	**Carbon capture efficiency**
Year (values)	**2020**	20%	55% fossil fuels, of which 32% is petroleum coke and 16% is coal 45% alternative fuels, of which 16% is biomass and 29% are other alternative fuels in minor quantities	Semi-wet/semi-dry kiln process (3.87 MJ per kg PC clinker)	n/a
	2030	30%	40% fossil fuels, of which 18% is petroleum coke and 16% is coal 60% alternative fuels, of which 30% is biomass and 30% are other alternative fuels based on fuel mix scenarios as seen in Table B2	Dry kiln with preheater (3.68 MJ per kg PC clinker)	n/a
	2040	40%	25% fossil fuels, of which 16% is petroleum coke and 8% is coal 75% alternative fuels, of which 45% is biomass and 30% are other alternative fuels based on fuel mix scenarios as seen in Table B2	Dry kiln with preheater and pre-calciner (3.39 MJ per kg PC clinker)	n/a
	2050	50%	10% fossil fuels, of which 0% is petroleum coke and 14% is coal 90% alternative fuels, of which 50% is biomass and 40% is other alternative fuels based on fuel mix scenarios as seen in Table B2	Dry kiln with preheater and pre-calciner (3.02 MJ per kg PC clinker)	Commercially applicable in large scale operation

	Scenario	**PC clinker substitution with SCMs**	**Increase in alternative fuel utilisation**	**Improving kiln configuration**	**Implementation of carbon capture and storage**
	Parameter	**Clinker-to-cement ratio**	**Individual fuel mix ratio**	**Thermal energy**	**Carbon capture efficiency**
		Selected based on various future estimations (Energy Agency, 2021; GCCA, 2021; IEA, 2018	Direct fuel mix design expectations (Cembureau, 2020)	Selected based on future estimations in thermal energy intensity of clinker (Energy Agency, 2021; GCCA, 2021; IEA, 2018)	

Table B2: Foreground scenarios and parameter-specific options.

Foreground Scenario	**Parameter**	**Options modelled**
PC clinker substitution with SCMs	SCM type	Coal fly ash, blast furnace slag, rice husk ash, bagasse ash, calcined clay, limestone
Increase in alternative fuel utilisation	Individual fuel mix ratio	1. Hydrogen (providing 40% of the thermal demand, generated through wind power), biomass (providing 50% of the thermal demand), and fossil fuels (providing 10% of the thermal demand). 2. Hydrogen (providing 40% of the thermal demand, generated through solar power), biomass (providing 50% of the thermal demand), and fossil fuels (providing 10% of 86 the thermal demand).

Foreground Scenario	**Parameter**	**Options modelled**
		3. Hydrogen (providing 40% of the thermal demand, generated through natural gas steam reforming (NGSR) power), biomass (providing 50% of the thermal demand), 89 and fossil fuels (providing 10% of the thermal demand). 4. Biomass (providing 50% of the thermal demand), TDF (providing 40% of the thermal demand), and fossil fuels (providing 10% of the thermal demand). 5. Biomass (providing 50% of the thermal demand), RDF (providing 40% of the thermal demand), and fossil fuels (providing 10% of the thermal demand). 6. Biomass (providing 50% of the thermal demand), RDF (providing 20% of the thermal demand), RDF (providing 20% of the thermal demand), and fossil fuels (providing 10% of the thermal demand). 7. Biomass (providing 90% of the thermal demand), and fossil fuels (providing 10% of the thermal demand).
Implementation of carbon capture and storage	Carbon capture efficiency	1. Oxy-fuel CCS (natural gas-powered) 2. Oxy-fuel CCS (coal-powered) 3. Post-combustion CCS (i.e., amine scrubbing) (natural gas-powered) 4. Post-combustion CCS (i.e., amine scrubbing) (coal-powered) 5. Oxy-fuel CCS with CHP (natural gas-powered) 6. Oxy-fuel CCS with CHP (coal-powered) 7. Post-combustion CCS with CHP (i.e., amine scrubbing) (natural gas-powered) 8. Post-combustion CCS with CHP (i.e., amine scrubbing) (coal-powered)

B.2 Life cycle inventory

The LCI data for the unit process datasets used in the SCMs, alternative fuels, kiln configurations, and CCS foreground scenarios are shown here.

B.2.1 Supplementary cementitious materials

Table B2: LCI data for ground granulated blast furnace slag production and preparation.

Ground Granulated Blast-furnace Slag		
Name	**Value**	**Unit**
Economic inflow		
Ground Granulated Blast-furnace Slag	1	kg
Electricity, medium voltage	0.095	kWh
Economic outflow		
Ground Granulated Blast-furnace Slag (treated)	1	kg
Environmental resources		
Environmental emissions		
(Petek Gursel et al., 2014)		

Included activities start: This activity starts with the reception of raw materials (i.e., by-product of iron and steelmaking in a blast-furnace).

Included activities end: This activity ends with the ground granulated blast-furnace slag treated in the cement facility. Packaging is not included, and no production waste is assumed.

Table B3: LCI data for coal fly ash production and preparation.

Fly ash (FA)		
Name	**Value**	**Unit**
Economic inflow		
Fly ash	1	kg
Electricity, medium voltage	0.007	kWh
Economic outflow		
Fly ash (treated)	1	kg
Environmental resources		
Environmental emissions		
(Petek Gursel et al., 2014)		

Included activities start: This activity starts with the reception of raw materials (i.e., coal fly ash is a by-product of coal-fired power plants).

Included activities end: This activity ends with the coal fly ash treated in the cement facility. Packaging is not included, and no production waste is assumed.

Table B4: LCI data for rice husk ash production and preparation.

Rice husk ash (RHA)		
Name	Value	Unit
Economic inflow		
Rice husk ash	1	kg
Electricity, medium voltage	2.09	kWh
Economic outflow		
Rice husk ash (treated)	1	kg
Environmental resources		
Environmental emissions		
(Jittin et al., 2021)		

Included activities start: This activity starts with the reception of raw materials (i.e., agricultural waste from rice milling).

Included activities end: This activity ends with the rice husk ash treated in the cement facility. Packaging is not included, and no production waste is assumed.

Table B5: LCI data for bagasse ash production and preparation.

Bagasse ash (BA)		
Name	**Value**	**Unit**
Economic inflow		
Bagasse ash	1	kg
Electricity, medium voltage	0.25	kWh
Economic outflow		
Bagasse ash (treated)	1	kg
Environmental resources		
Environmental emissions		
(Jittin et al., 2021)		

Included activities start: This activity starts with the reception of raw materials (i.e., agricultural waste from sugar manufacturing industry).

Included activities end: This activity ends with the bagasse ash treated in the cement facility. Packaging is not included, and no production waste is assumed.

Table B6: LCI data for calcined clay production.

Calcined clay (CC)		
Name	**Value**	**Unit**
Economic inflow		
Clay	1.14	kg
Electricity, medium voltage	0.08	kWh
Transport, freight, lorry 16 - 32 metric t, EURO6	0.57	tkm
Petroleum coke	0.03	kg
Economic outflow		
Calcined clay	1	kg
Environmental resources		

Calcined clay (CC)		
Name	**Value**	**Unit**
Environmental emissions		
Carbon dioxide, fossil	0.1	kg
Methane, fossil	3.14E-06	kg
Water	1.45E-04	m^3
(Salvi Malacarne et al., 2021)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the clay transported and calcined in the cement facility. Packaging is not included, and no production waste is assumed.

B.2.2 Alternative fuels

Table B7: LCI data for hydrogen production through solar power.

Hydrogen production (Electrolysis through solar)		
Name	**Value**	**Unit**
Economic inflow		
water, deionised	1.0	kg
electrode, negative, Ni	1.1e-06	kg
heat, central or small-scale, natural gas	0.754	MJ
electricity, high voltage	0.00488	kWh
potassium hydroxide	3.9e-07	kg
transport, freight, lorry 16-32 metric t, EURO6	0.05	tkm
Economic outflow		
Hydrogen	1	Nm^3
Environmental resources		
Environmental emissions		
(Dufour et al., 2011)		

Included activities start: This activity starts with the reception of raw materials for the electrolysis process.

Included activities end: This activity ends with the hydrogen transported in the cement facility. Packaging is not included, and no production waste is assumed.

Table B8: LCI data for hydrogen production through wind power.

Hydrogen production (Electrolysis through wind)		
Name	**Value**	**Unit**
Economic inflow		
water, deionised	1.0	kg
electrode, negative, Ni	1.1e-06	kg
heat, central or small-scale, natural gas	0.754	MJ
electricity, high voltage	0.00488	kWh
potassium hydroxide	3.9e-07	kg
transport, freight, lorry 16-32 metric t, EURO6	0.05	tkm
Economic outflow		
Hydrogen	1	Nm^3
Environmental resources		
Environmental emissions		
(Dufour et al., 2011)		

Included activities start: This activity starts with the reception of raw materials for the electrolysis process.

Included activities end: This activity ends with the hydrogen transported in the cement facility. Packaging is not included, and no production waste is assumed.

Table B9: LCI data for hydrogen production through natural gas steam reforming.

Hydrogen production (Natural gas steam reforming -NGSR)		
Name	**Value**	**Unit**
Economic inflow		
concrete, normal	1.2612e-05	m^3
aluminium, primary, ingot	3.3e-05	kg
cast iron	4.9e-05	kg
hard coal	0.13249	kg
limestone, unprocessed	0.016	kg
natural gas, unprocessed, at extraction	0.0043	m^3
heavy fuel oil	0.0088	kg
tap water	19.7762	kg
steel, low-alloyed	0.0040	kg
electricity, medium voltage	46.017	kWh
iron scrap, sorted, pressed	0.0112	m^3
iron ore, crude ore, 46% Fe	0.0103	kg
transport, freight, lorry 16-32 metric t, EURO6	0.05	tkm
Economic outflow		
Hydrogen	1	kg
Environmental resources		
Environmental emissions		
Carbon dioxide, fossil	9.3587	kg
Carbon monoxide, fossil	8.7574	kg
Nitrogen oxides	0.00404	kg
NMVOC, non-methane volatile organic compounds, unspecified origin	0.0053	kg
Particulates, > 2.5 um, and < 10um	0.0016	kg
Sulphur oxides	0.00457	kg

Hydrogen production (Natural gas steam reforming -NGSR)		
Name	**Value**	**Unit**
Environmental emissions		
Benzene	0.00027	kg
Methane, fossil	0.0598	kg
(Cetinkaya et al., 2012)		

Included activities start: This activity starts with the reception of raw materials for natural gas steam reforming.

Included activities end: This activity ends with the hydrogen transported in the cement facility. Packaging is not included, and no production waste is assumed.

Table B10: LCI data for sewage sludge treatment.

Sewage sludge		
Name	**Value**	**Unit**
Economic inflow		
electricity, medium voltage	1.6	kWh
sewage sludge	1.0	m^3
polyacrylamide	0.22	kg
Economic outflow		
De-watering of sewage sludge	0.11	t
Environmental resources		
Environmental emissions		
(Dufour et al., 2011)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the treated sewage sludge in the cement facility. Packaging is not included, and no production waste is assumed.

Table B11: LCI data for tire-derived fuel production.

Tire-derived fuel (TDF)		
Name	**Value**	**Unit**
Economic inflow		
used tires	1000.0	kg
electricity, low voltage	47.2222	kWh
tap water	150.0	kg
lubricating oil	0.011	kg
steel, low-alloyed	0.23	kg
transport, freight, lorry 16-32 metric t, EURO6	0.090935	tkm
Economic outflow		
Grinded tires	966	kg
Environmental resources		
Environmental emissions		
(Petek Gursel et al., 2014)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the transported and processed tire-derived fuel in the cement facility. Packaging is not included, and no production waste is assumed.

Table B12: LCI data for refused-derived fuel production.

Refused-derived fuel (RDF)		
Name	**Value**	**Unit**
Economic inflow		
transport, freight, lorry 16-32 metric t, EURO6	0.090935	tkm
light fraction-high calorific	348.9	kg
electricity, medium voltage	101.17	kWh
Economic outflow		
Refused-derived fuel	318.5	kg
Environmental resources		

Refused-derived fuel (RDF)		
Name	**Value**	**Unit**
Environmental emissions		
(Grzesik & Malinowski, 2016)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the transported and processed refused-derived fuel in the cement facility. Packaging is not included, and no production waste is assumed.

Table B13: Average composition of MSW in Krakow, Poland

Material fraction	**Percentage (%)**
Bio-waste	29.22
Wood	0.99
Paper and cardboard	20.48
Plastics	14.59
Glass	8.53
Textiles	2.93
Metals	2.25
Hazardous waste	0.72
Composites	4.6
Inert waste	6.85
Fine fraction (under 10mm)	8.84
(Sieja et al., 2011; Grzesik & Malinowski, 2016)	

The production of wood pellets was modelled from whole tree logs from forest thinning operations. This includes forest operations (starting at the point of extraction), transportation, and processing into wood pellets.

Table B14: LCI data for biomass (wood pellet) production.

Biomass (wood)		
Wood pellet production		
Name	**Value**	**Unit**
Economic inflow		
diesel	12.83	kg
lubricating oil	0.549	kg
waste wood, untreated	1000.0	kg
transport, freight, lorry 16-32 metric t, EURO6	50.0	tkm
Economic outflow		
log	1	t
Environmental resources		
Environmental emissions		
Wet sawdust production		
Name	**Value**	**Unit**
Economic inflow		
diesel	29.0	Kg
electricity, low voltage	30.7	kWh
log	1	t
Economic outflow		
Wet sawdust	1	t
Environmental resources		
Dry sawdust production		
Name	**Value**	**Unit**
Economic inflow		
electricity, medium voltage	55.8	kWh
heat, for reuse in municipal waste incineration only	1321.8	MJ
waste wood, untreated	300.0	kg

Biomass (wood)		
Wood pellet production		
Name	**Value**	**Unit**
Economic inflow		
wet sawdust	1.8	t
Economic outflow		
Dry sawdust	1	t
Environmental resources		
Conditioned sawdust		
Name	**Value**	**Unit**
Economic inflow		
Dry sawdust	0.97	t
electricity, medium voltage	8.54	kWh
Economic outflow		
Conditioned sawdust	1	t
Environmental resources		
Pellet production		
Name	**Value**	**Unit**
Economic inflow		**Economic inflow**
Conditioned sawdust	1.0	t
lubricating oil	0.16	kg
transport, freight, lorry 16-32 metric t, EURO6	300.0	tkm
EUR-flat pallet	0.66	unit
electricity, medium voltage	104.16	kWh
Economic outflow		
Packed pellet	1	t
Environmental resources		
(Sgarbossa et al., 2020)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the packed pellet (final biomass form) transported to the cement facility. Packaging is not included, and no production waste is assumed.

Although the effects of land use changes were not quantitively modelled in this study, future work should consider them since they can cause significant impacts in categories such as climate change ozone depletion, photochemical ozone creation, and human toxicity (Sgarbossa et al., 2020). Consideration of these land use change impacts is important since cement industry roadmaps model significant increases in biomass fuel use. Associated land transformation to biofuel crops could significantly affect food security (through competition with land use for agriculture and pasture), biodiversity, and soil and water quality (Turner et al., 2007;Gnansounou et al., 2008). Additionally, changes in land use can vary the soil carbon stock and thus cause climate change impacts (Dequéant et al., 2006). Nitrous oxide emissions from soils caused by microbial activity activated by fertilizer application can lead to significant human toxicity impacts (Profile et al., 2008). Expanding bio-crop activities to meet future biomass demand would correspond to extensive use of agricultural substances (i.e., fertilizers), which could lead to higher nitrous oxide emissions and human health considerations.

B.2.3 Kiln type configuration (thermal efficiency)

The kiln configurations studied are presented ranging at low, average, and high thermal demand (Table B15).

Table B15: Kiln configuration with energy efficiencies based on GNR data (GNR, 2014). The data in bold text indicate the used data for this work.

	Low (GJ/ t clinker)	Average (GJ/ t clinker)	High (GJ/ t clinker)
Semi-wet/Semi-dry (default)	3.78	**3.90 (default)**	5.36
Dry with preheater without precalciner	3.49	**3.68**	3.97
Dry with preheater and precalciner	**3.02**	**3.39**	4.06

B.2.4 Carbon capture and storage

The LCI data for the carbon capture infrastructure and operation are presented in Table B16-20.

Table B16: Life cycle inventory for post-combustion carbon capture (with natural gas).

Post Combustion Carbon capture (natural gas)		
Infrastructure	**Value**	**Unit**
Chromium steel	3.17e-05	kg
Concrete	1e-05	m^3
limestone	0.0141	kg
Operation		
market for ammonia, liquid	0.00204	kg
ethanolamine production	0.00264	kg
market for heat, district, or industrial, natural gas	0.378	MJ
Environmental resources		
(Chen & Yang, 2021)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the construction and operation of post combustion carbon capture (natural gas powered). Packaging is not included, and no production waste is assumed.

Table B17: Life cycle inventory for post-combustion carbon capture (with coal).

Post Combustion carbon capture (coal)		
Infrastructure	**Value**	**Unit**
Chromium steel	3.17e-05	kg
Concrete	1e-05	m^3
limestone	0.0141	kg
Operation		
market for ammonia, liquid	0.00204	kg

Post Combustion carbon capture (coal)		
Operation		
ethanolamine production	0.00264	kg
heat production, at hard coal industrial furnace 1-10MW	0.378	MJ
Environmental resources		
(Chen & Yang, 2021)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the construction and operation of post combustion carbon capture (coal powered). Packaging is not included, and no production waste is assumed.

Table B18: Life cycle inventory for oxyfuel carbon capture (with natural gas).

Oxyfuel Carbon capture (natural gas)		
Infrastructure	**Value**	**Unit**
aluminium	6.25e-06	kg
Steel unalloyed	2.38e-05	kg
Chromium steel	1.4e-05	kg
Concrete	1.56e-05	m^3
Diesel	0.00117	MJ
Oil	6.31e-06	kg
Operation		
market for heat, district, or industrial, natural gas	0.922	MJ
(Chen & Yang, 2021)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the construction and operation of oxyfuel carbon capture (natural gas powered). Packaging is not included, and no production waste is assumed.

Table B19: Life cycle inventory for oxyfuel carbon capture (with coal).

Oxyfuel Carbon capture (coal)		
Infrastructure	**Value**	**Unit**
aluminium	6.25e-06	kg
Steel unalloyed	2.38e-05	kg
Chromium steel	1.4e-05	kg
Concrete	1.56e-05	m^3
Diesel	0.00117	MJ
Oil	6.31e-06	kg
Operation		
heat production, at hard coal industrial furnace 1-10MW	0.922	MJ
(Chen & Yang, 2021)		

Included activities start: This activity starts with the reception of raw materials.

Included activities end: This activity ends with the construction and operation of oxyfuel carbon capture (coal powered). Packaging is not included, and no production waste is assumed. The incorporation of a CHP unit is modelled by changing the source of electricity and heat production used to satisfy the energy demands of the system, market for heat, district or industrial, natural gas is replaced by heat and power co-generation, natural gas, 160 kW electrical (An et al., 2019). The liquefaction process of captured CO_2 and its transportation to an underground storage facility was not included in this study.

Implementing carbon capture and storage in cement and power generation plants has been found to reduce climate change, ozone depletion, and abiotic resource depletion impacts, but significantly worsens human toxicity, freshwater eutrophication, and ecotoxicity impacts (García-Gusano et al., 2015). Similarly, introducing a CHP unit in cement and power plants contributes significant climate change, human toxicity, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, and marine eutrophication (Volkart et al., 2013). This is notably because CCS and CHP increase the cumulative energy demand of the system; depending on the energy source (i.e., coal-powered, or natural gas-powered) the environmental impacts can significantly vary (Rosental et al., 2020). Assessing these trade-offs between impact categories using prospective LCA is important but outside the scope of this study.

B.2.5 Allocation

Emission allocation was included for the case of fly ash as it is considered to meet all the conditions set by European Union Directive (European Union, 2008) which classifies fly ash as an industrial by-product and not as a waste. By-product classification requires emission allocation of the process emissions associated with the production of the original product (i.e., electricity). Studies on the effect of different allocation type have shown that the climate change impact of concrete is being underestimated when no allocation is considered (Chen et al., 2010; Habert et al., 2011; Seto et al., 2017; Van Den Heede & De Belie, 2012). In this study, two allocation types were performed: no allocation (assuming fly ash as a waste product), and economic allocation (by-product). The allocation methodology was used according to the principles proposed by Chen et al. (2010) and (Shobeiri et al., 2021)
Two allocation types were considered: no allocation and economic allocation according to forecasted data for fly ash market price.

Economic allocation associates the environmental impacts of a process to the main product and by-product considering the market price of each product (Equation B1).

$$C_e = \frac{(\$\times m)_{by-product}}{(\$\times m)_{main\ product} + (\$\times m)_{by-product}} \quad \text{(B1)}$$

Where C_e represented the economic allocation coefficient, and ($×m) represents the unit price of the material product i.e., by-product (fly ash) and main product (electricity). Price values were obtained from the study of (Shobeiri et al., 2021), where fly ash and national electricity market forecasting reports were considered.

B.3 Uncertainty analysis

The uncertainty associated with the mass of PC clinker was classified using data quality indicators in the pedigree matrix (Weidema B.P et al., 2013). A scoring system is applied which assesses judgements on integrity, reliability, temporal correlation, geographical, and technological uncertainty in the data (Table B20).

Table B20: Data quality indicators in pedigree matrix.

Data quality indicators	**Pedigree values**
Reliability	(1) Verified data based on measurements
Completeness	(2) Representative relevant data from > 50% sites, over an adequate period
Temporal correlation	(1) Data less than 3 years old
Geographical correlation	(1) Data from area under study
Further technological correlation	(2) Data from processes and materials under study, different enterprise

The probability density functions (PDFs) were generated assuming a 10% PC clinker mass uncertainty (0.68-0.84 kg), and according to the pedigree indicator scores based on a log-normal distribution (Figure B1-B2).

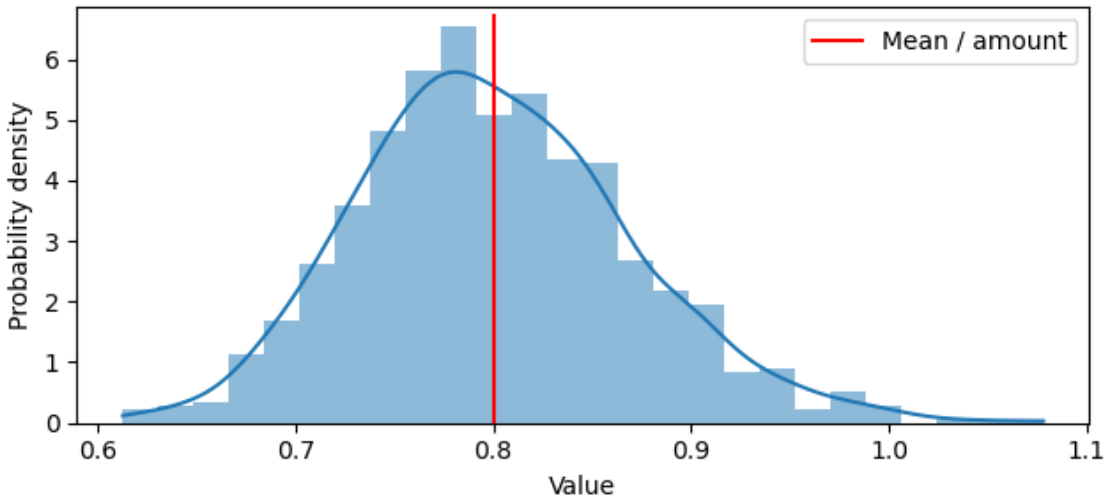


Figure B1: Probability density function for cement production assuming a 20% coal fly ash (SCM) substitution.

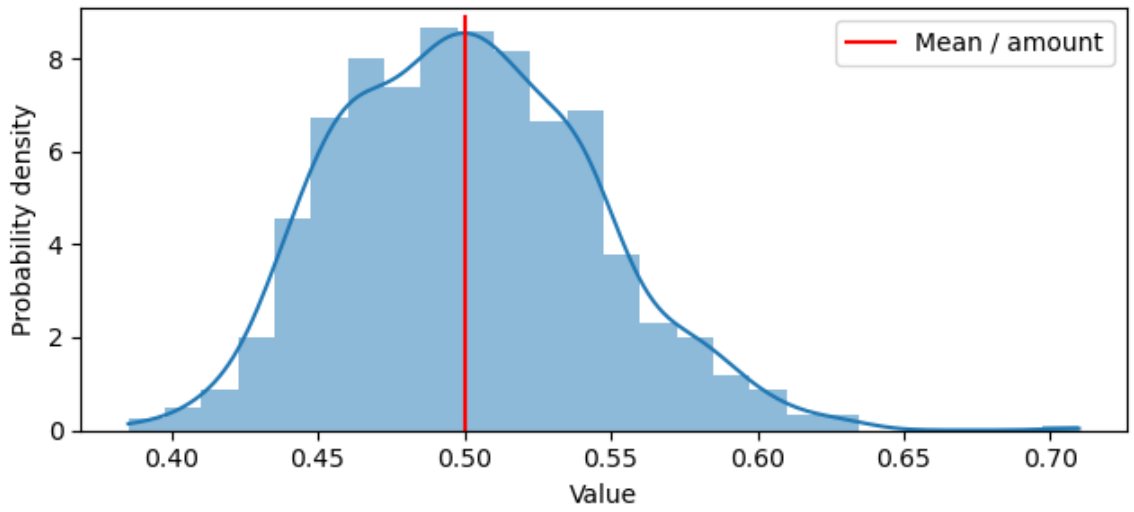


Figure B2: Probability density function for cement production assuming a 50% coal fly ash (SCM) substitution.

B.4 Contribution analysis results based on SSP2 RCP

The process contribution analysis of cement production with SCMs (Figure B3), and alternative fuels (Figure B4) are presented based on the SSP2 RCP 2.6 background scenario.

B.4.1 PC clinker substitution with SCMs

In 2050, the higher clinker replacement ratio reduces the impact of clinker production with a relative impact increase from 82% in 2020 to 88% in 2050.

The contribution analysis for this ground granulated blast furnace slag (Figure B3) shows that electricity production, which is responsible for 10% of the CO_2-eq. emissions in 2020, is almost completely diminished to 1.4% by 2050 because of the SSP2 RCP 2.6 scenario.

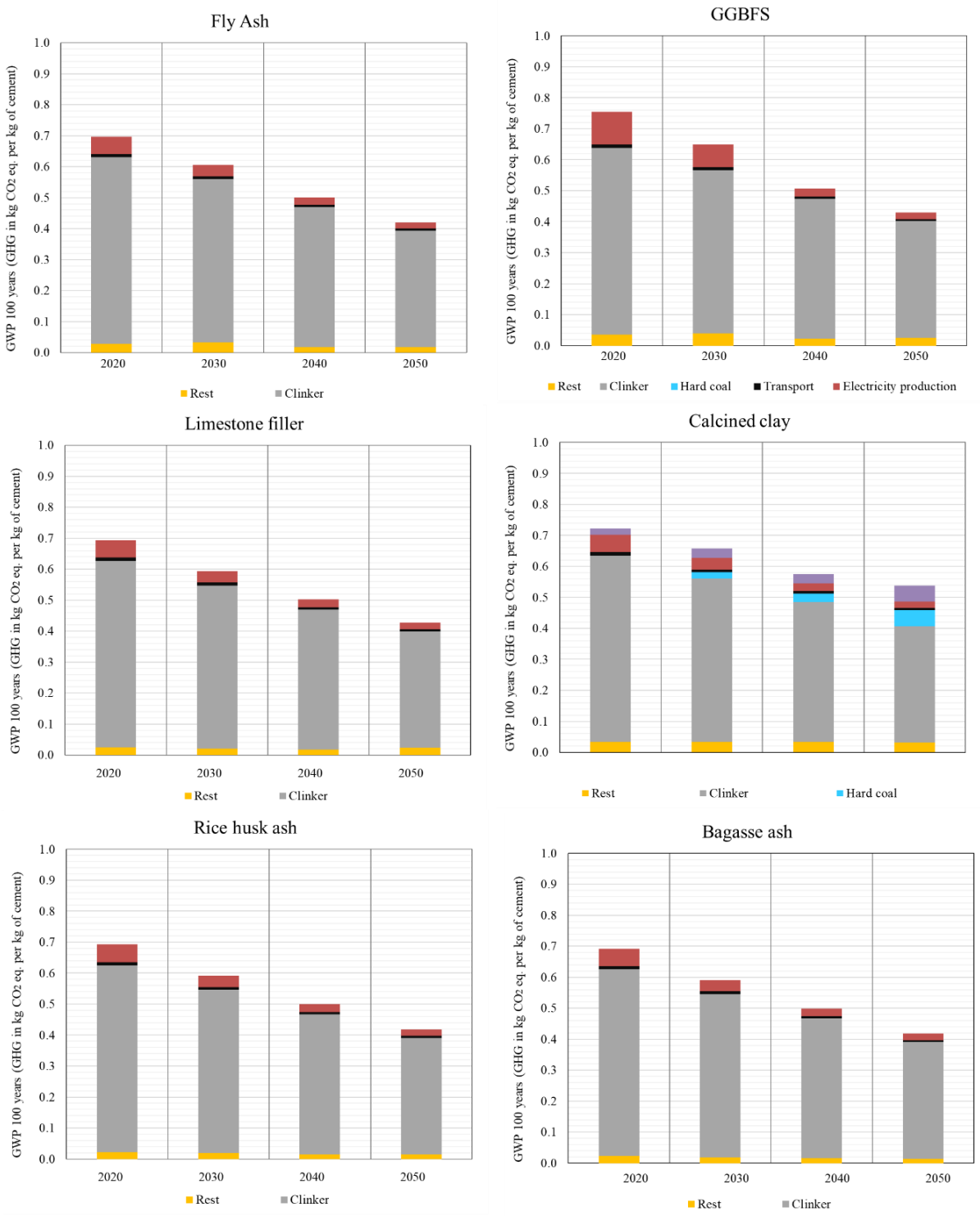


Figure B3: Contribution analysis of the key contributing flows from 2020 to 2050, with increasing SCMs-to clinker ratio (with SSP2 RCP2.6 background).

B.4.2 Increasing alternative fuels utilisation

The reduction in GWP100 caused by increasing alternative fuel consumption is mainly due to changes in electricity generation (Figure B4). In 2050, electricity is generated from renewable sources and lower-carbon technologies (i.e., the sequential generation of electricity and thermal energy from a single energy source in a heat and power cogeneration system) that have lower climate change impact. Therefore, the results show that the variability between different alternative fuel mixes becomes smaller with time, and by 2050 the variability is minor.

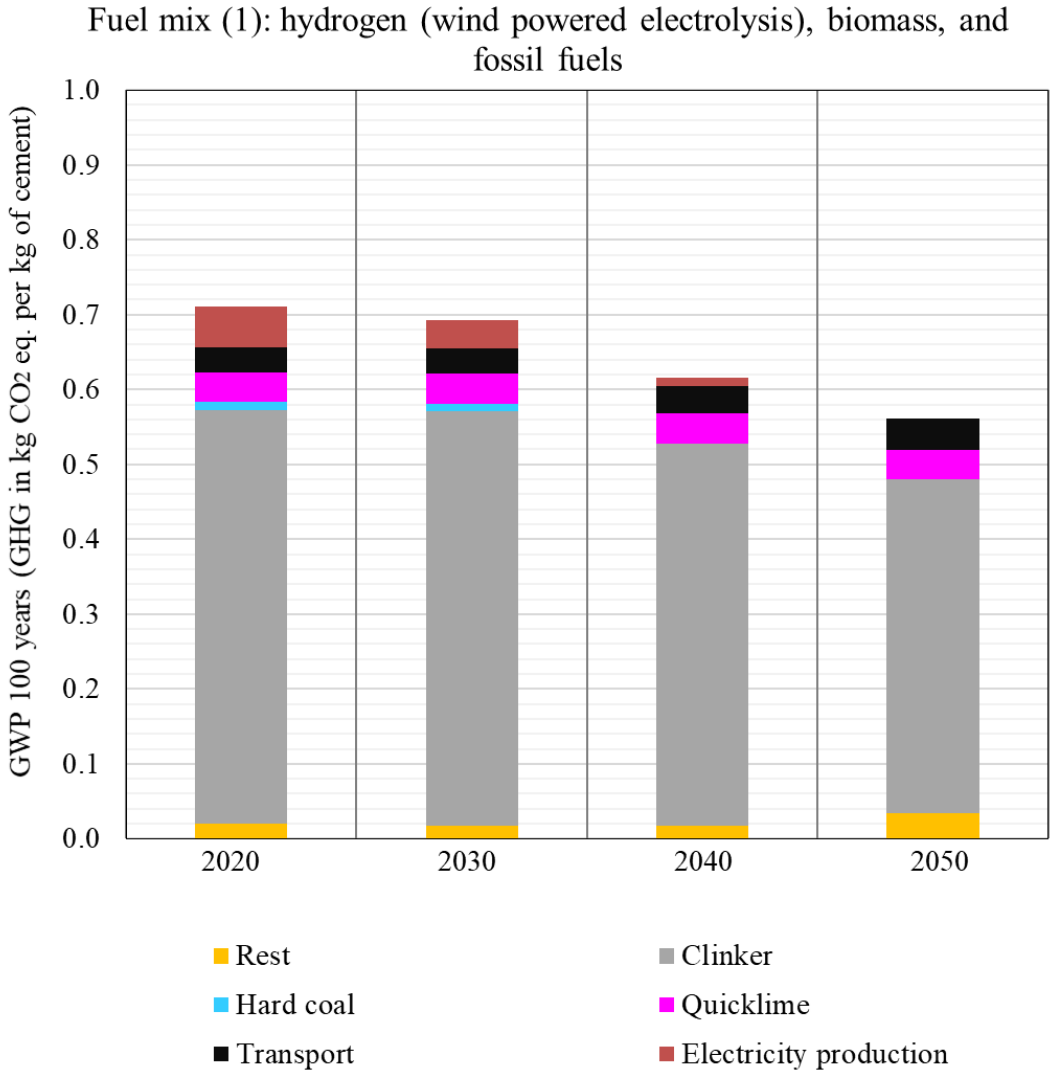


Figure B4: Contribution analysis of the key contributing flows from 2020 to 2050, with increasing alternative fuel consumption according to fuel mix (1).

Similarly, to electrification, decreasing the reliance of electricity generation on fossil fuels significantly influences the results for the hydrogen (natural gas steam reforming, NGSR)

scenario. Initially, the use of hydrogen produced from NGSR causes a higher climate change impact (0.87 kg CO_2-eq. per kg cement) than reference cement production (0.76 kg CO_2-eq. per kg cement), although by 2050 a significantly lower climate change impact of 0.57 kg CO_2-eq. is determined (Figure B5).

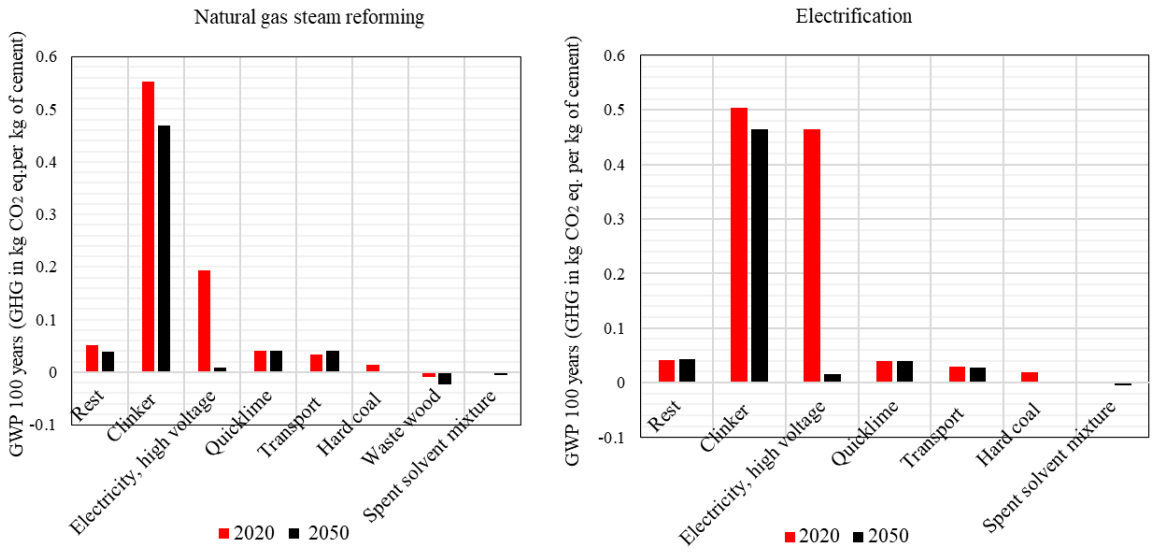


Figure B5: CO_2-eq. emissions from process contributions for hydrogen production with natural gas (NGSR) and electrification in 2020 and 2050.

B.4.3 Synergistic scenario without CCS, following the IEA and CEMBUREAU roadmaps

The application of a synergistic scenario led to an initial 24% impact reduction between 2020 to 2030 (Figure B6). This reduction is mainly attributed to the reduced impact caused by clinker production and electricity generation as well as the complete elimination of hard coal as a fuel used in background processes. An additional 20% decline is observed in CO_2 emissions by 2040, which can be clearly seen as the result of a further impact minimization of electricity generation. By 2050 a further 14% reduction was observed to 0.32 kg CO_2 eq., with the complete elimination of electricity generation impact as well as the negative emissions associated to use of waste solvent, a total of 58% impact reduction was attained.

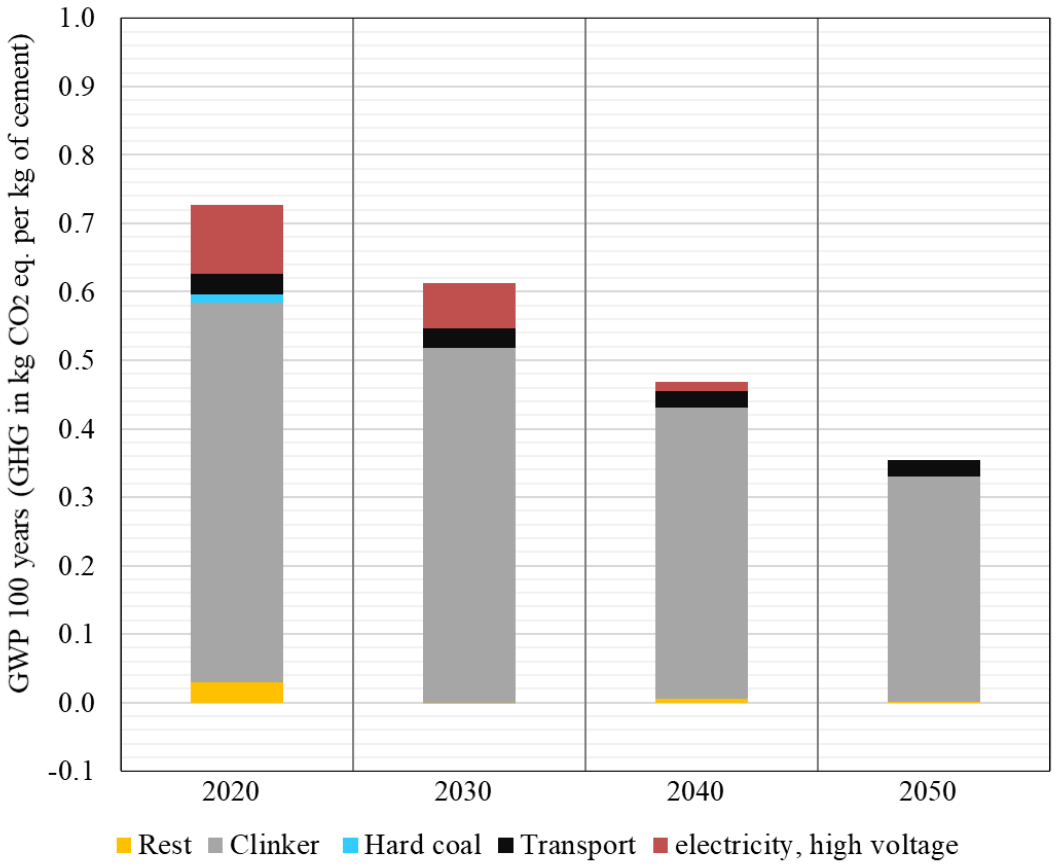


Figure B6. Contribution analysis of cement production from a synergistic scenario incorporating PC clinker substitution with coal fly ash (SCMs), increasing alternative fuels utilisation (fuel mix consists of hydrogen generated through wind power, biomass, and fossil fuels), and improving kiln configurations from 2020 to 2050.

PC clinker substitution was the most influential single decarbonisation measure contributing to the results for the synergistic scenarios, consistent with our results for this measure alone (Section 5.3.1). Therefore, a sensitivity analysis on the SCMs within the synergistic scenario was performed (Figure B7). Blended cement with calcined clay remains with the highest climate change impact of the PC-SCM cements modelled here because of the higher energy demands during SCM pre-treatment, with 0.43 kg CO_2-eq. emissions compared to the other SCMs ranging between 0.32-0.33 kg CO_2-eq. emissions in 2050 following the SSP2 RCP 2.6 scenario. Blended cement with calcined clay (with synergistic effect) becomes comparable to the impact of the individual effect of PC clinker substitution with SCMs in (i.e., Section 5.3.1), where a climate change impact range of 0.42-0.44 kg CO_2-eq. per kg cement in 2050 (for blended cement with coal fly ash, or ground granulated blast furnace slag, or rice husk ash, or bagasse ash, or limestone).

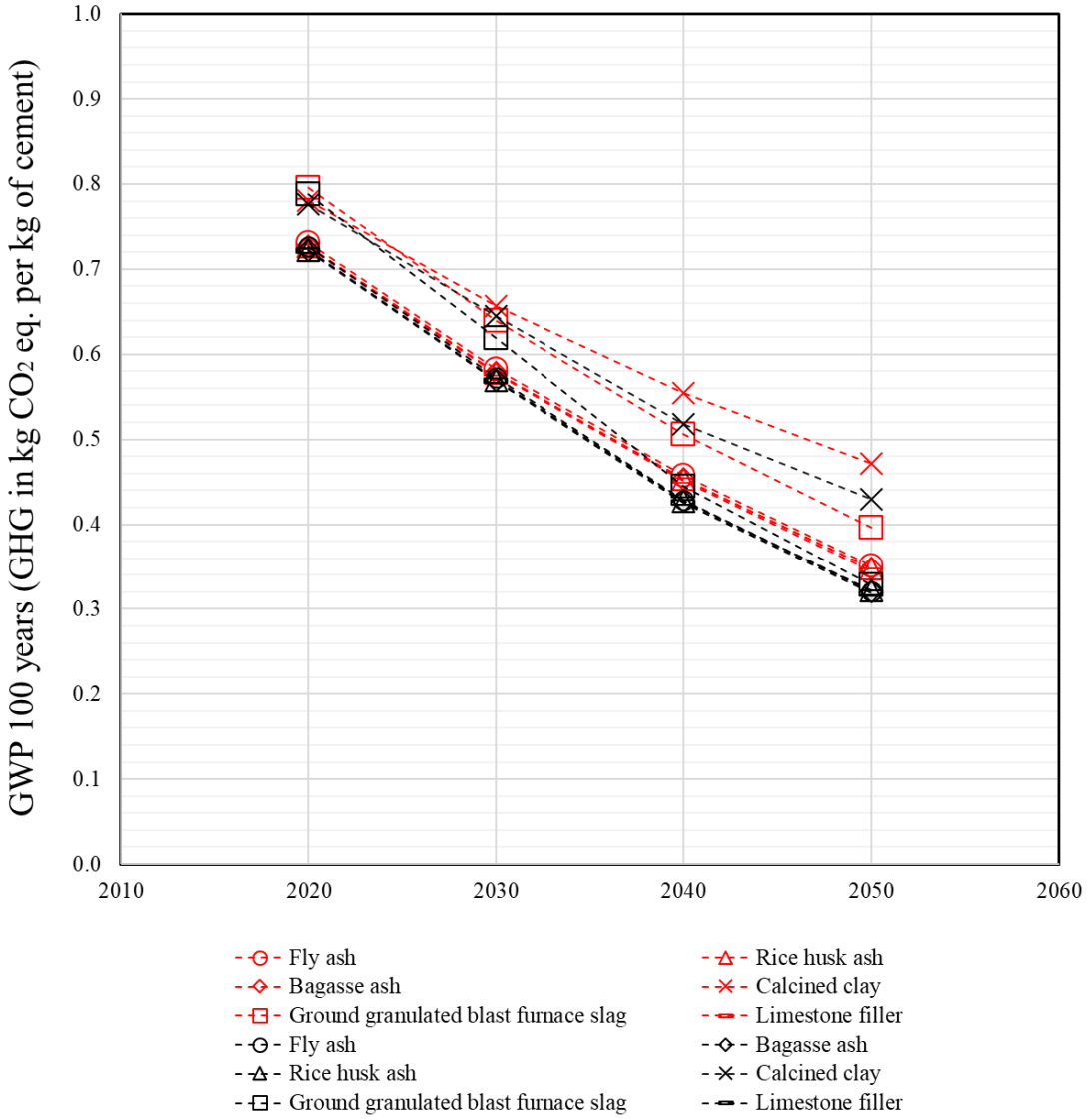


Figure B7: CO_2-eq. emissions of cement production with synergistic scenario from 2020 to 2050, with SSP2 Baseline (red) and SSP2 RCP 2.6 (black) for all SCMs studied, and all based on the IEA and Cembureau roadmaps.

B.4.4 Synergistic scenario with CCS, following the IEA and CEMBUREAU roadmaps

Some studies show amine scrubbing as the preferable option to oxyfuel, while some other studies report oxyfuel as the technology with the lowest CO_2 emissions even though it has a higher energy requirement (An et al., 2019; Chen & Yang, 2021; Wunderlich et al., 2021). The results show that with the same capture efficiency (85%) amine scrubbing performs better in comparison to oxyfuel technology (i.e., amine scrubbing (0.09 kg CO_2-eq. per kg cement)

relative to oxyfuel CCS technology (0.14 kg CO_2-eq. per kg cement) natural-gas powered, Figure 8). This study did not account for changes in the CO_2 purity in the flue gas stream which could alter the operating conditions of the flue gas processing system. Energy demands for the additional units i.e.; air separation unit was included even though in the existing literature there are inconsistencies with the additional energy requirements not always included. (DECC, 2015; Huang et al., 2022; Kheirinik et al., 2021; Material Economics, 2019; Pamenter & Myers, 2021).

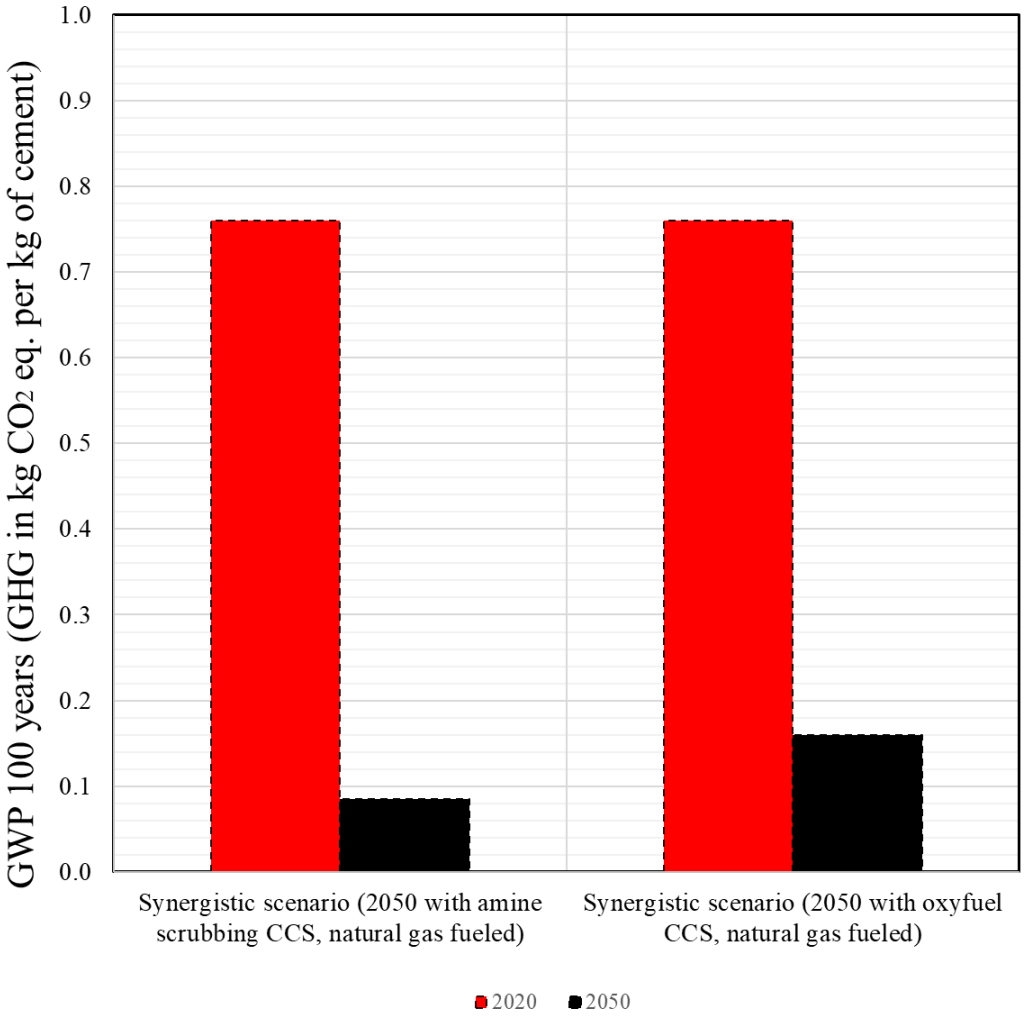


Figure B8: CO_2-eq. emissions of cement production with synergistic scenarios including PC clinker substitution with SCMs, alternative fuel use, improved kiln configurations, and natural gas-powered CCS technologies (i.e., amine scrubbing, oxy-fuel).

B.5 LCA results based on SSP2 baseline scenario

The LCA results of cement production with SCMs (Figure B9), alternative fuels (Figure B10 and Figure B12) and kiln efficiency (Figure B13) are presented based on the SSP2 baseline scenario.

B.5.1 PC clinker substitution with SCMs

Substituting PC clinker with SCMs such as coal fly ash, ground granulated blast furnace slag, rice husk ash, bagasse ash, calcined clay, and limestone reduces the climate change impact of cement production by 0.3 kg GHG (40%) per kg cement over a 30-year period (Figure B9). Similarly, to SSP2 RCP2.6, a monotonic reduction is observed throughout the time period.

A larger variability is observed between the environmental impact of SCMs, in comparison to the RCP2.6. The change in variability is associated with the reduced decarbonisation of electricity generation technologies in the SSP2 baseline scenario.

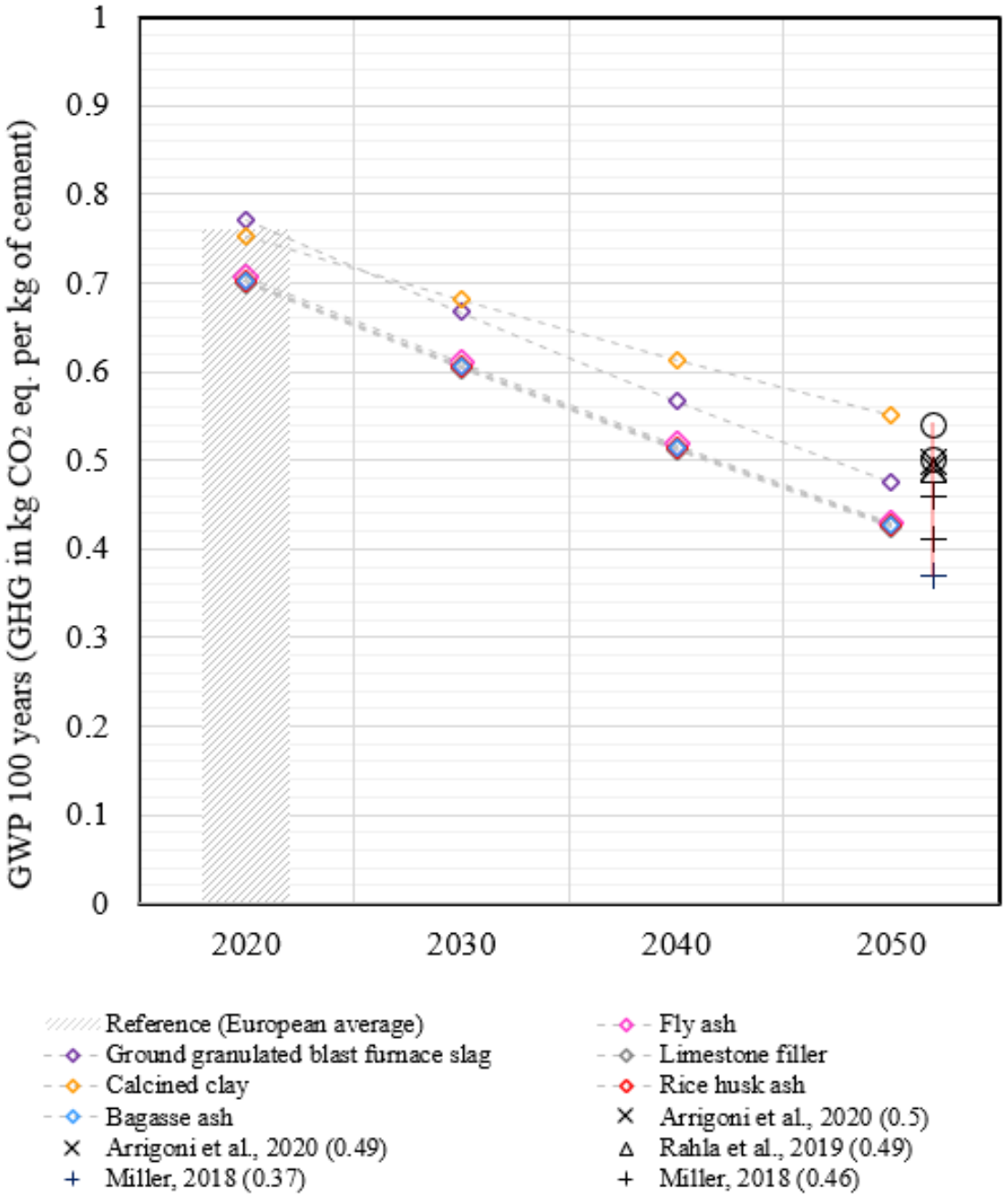


Figure B9: CO_2-eq. emissions from cement production with substitution of PC clinker for SCMs, from 2020 to 2050. GWP100 of cement production using prospective LCA in comparison with existing literature (traditional LCA) in 2050.

B.5.2 Increasing alternative fuels utilisation

Switching the fuel mix to higher ratios of alternative fuels such as biomass, RDF, TDF, hydrogen (NGSR, solar-powered electrolysis, wind-powered electrolysis), can reduce the GWP100 of cement production by 22% between 2020 and 2050, under the SSP2 baseline scenario (Figure B10).

The partial decarbonisation of the electricity mix is responsible for the impact reduction observed. However, the lower decarbonisation is responsible for the larger variability observed in the alternative fuels. The decarbonisation of electricity generation technologies is highly important in the electrification scenario. Under the SSP2 RCP 2.6 and by 2050 electrification becomes a favourable option, while under the SSP2 baseline electrification results in 0.88 kg CO_2-eq. emissions, which is considerably larger than the reference impact of PC (0.76 kg CO_2-eq. emissions).

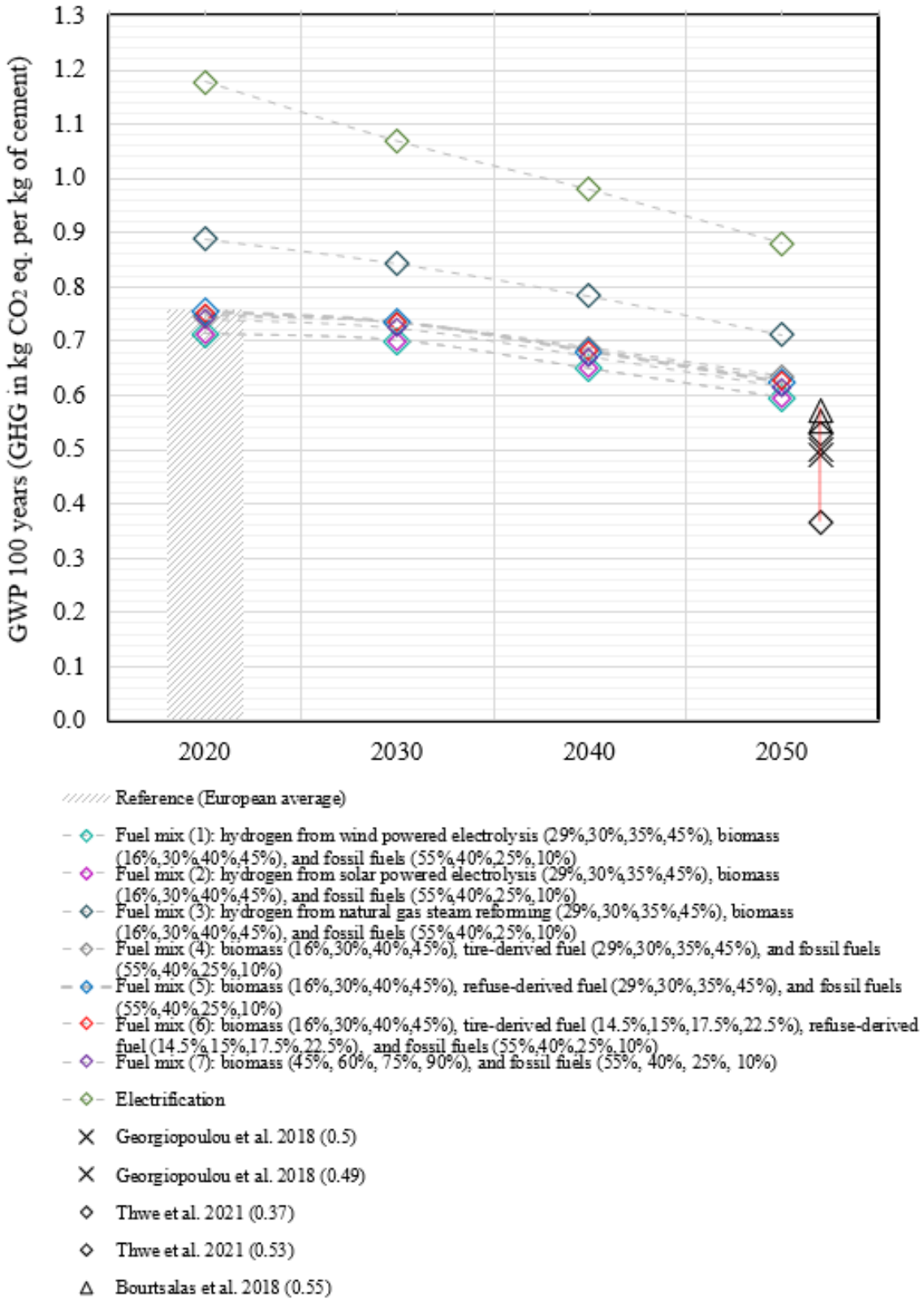


Figure B10: CO_2-eq. emissions from cement production with alternative fuels, from 2020 to 2050. GWP100 of cement production using prospective LCA in comparison with existing literature (traditional LCA) in 2050.

B.5.2.1 Electrification

Electrification was highly sensitive to the background scenarios. The SSP2 baseline scenario yields a higher impact (0.88 kg CO_2 eq. emissions) than the reference impact (0.76 kg CO_2 eq. emissions) of PC, showing electrification as an unfavourable option for decarbonisation of cement. The large impact was the result of the dependency of the electricity generation technologies on fossil fuel sources, in contrast to the significant reduction observed under the SSP2 RCP 2.6 scenario. In the latter scenario, a larger impact reduction was achieved (0.6 kg CO_2-eq. emissions), with electrification being comparable to the other alternative fuels (Figure B11).

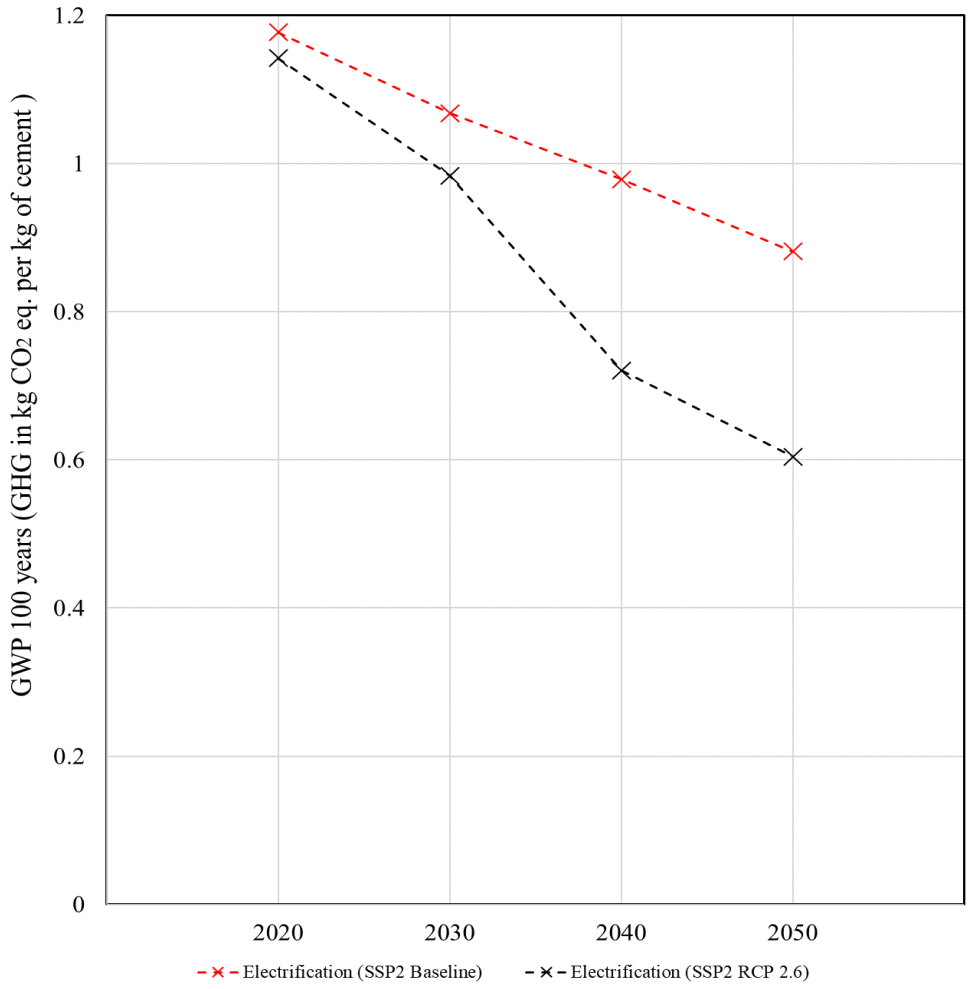


Figure B11: CO_2-eq. emissions from electrification of the cement kiln for 2020 and the prospective climate change impacts for 2030, 2040, and 2050.

B.5.3 Improving kiln configurations

The GWP100 of cement, produced under higher efficiency kiln configurations and SSP2 baseline scenario result in an impact reduction of 8.4% by 2050 (Figure B12). The largest reduction in GWP100 was observed in the dry process, including a precalciner and a pre-heater prior of the kiln.

The impact reduction is mainly due to the partial decarbonisation of the electricity mix from fossil fuels. Modifying the kiln configuration has no effect in the fuel mix nor the clinker-to-clinker ratio, the two major contributing stages of cement process, hence the lowest GHG emission reduction was observed compared to clinker replacement and alternative fuels.

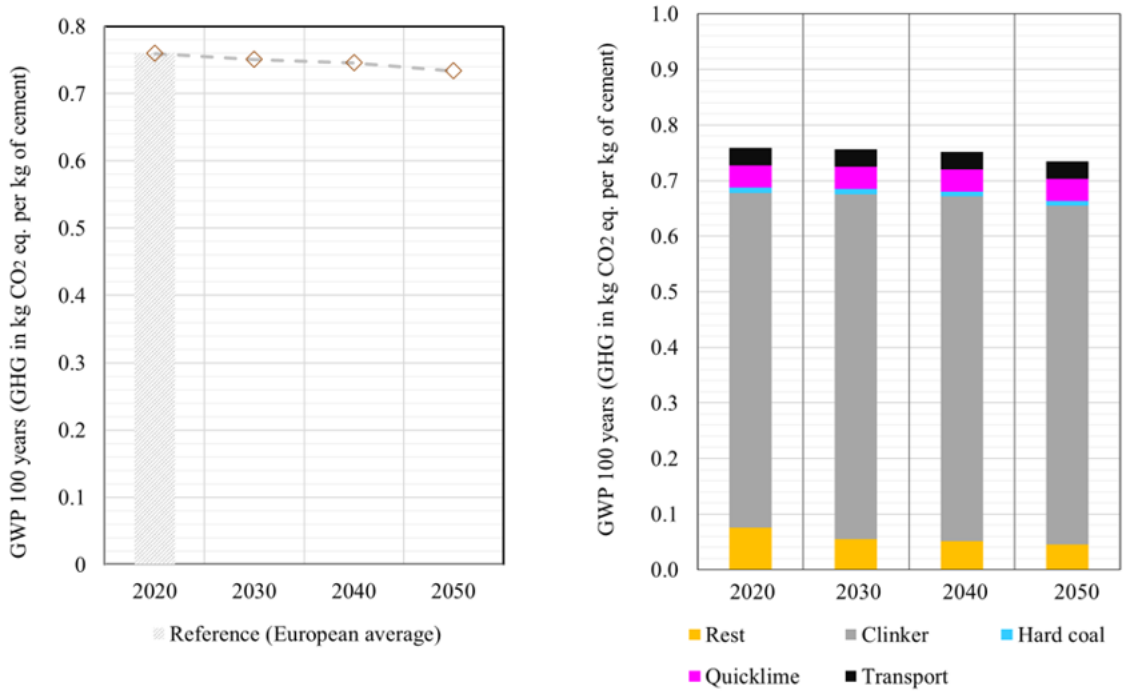


Figure B12: CO_2-eq. emissions from cement production different kiln configurations (kiln efficiencies, from 2020 to 2050.

B.6 Monte Carlo analysis

Given the importance of the clinker-to-cement ratio on the LCA results, the uncertainty in the mass of PC clinker at years 2020 and 2050 was assessed, taking coal fly ash as the SCM substituting clinker. For the Monte Carlo analysis, coal fly ash is selected as it is found to give the lowest environmental impact and is currently commercially used.

The Monte Carlo analysis for cement production in 2020 resulted in a low variability (4-6%) from the initial impact (0.72 kg CO_2-eq. emissions), with values ranging between 0.72-0.73 kg CO_2-eq. emissions as the most probable (37% probability) (Figure B13). This impact range is stable, indicating that in 2020 the effect of PC clinker mass uncertainty is relatively low. The small influence of clinker uncertainty in 2020 can be attributed to the several parameters which can equally affect the impact, as seen by the global sensitivity analysis (Figure B14).

However, in 2050 the PC clinker mass uncertainty has a more significant effect leading to a wider impact variability 0.3-0.51 kg CO_2-eq. emissions, which represents a –5 to +55% change from the initial impact score (0.32 kg CO_2-eq. emissions). The most probable value (14% probability) lies in the range of 0.38-0.39 kg CO_2-eq., which is ~20% higher than the initial impact. The wide range of variability caused by clinker mass uncertainty indicates a higher sensitivity. In 2050, clinker production is found to be the main parameter (highest delta value, 0.613) affecting the overall climate change impact. This analysis shows the importance of the clinker-to-cement ratio in calculations of future climate change impacts of cement production.

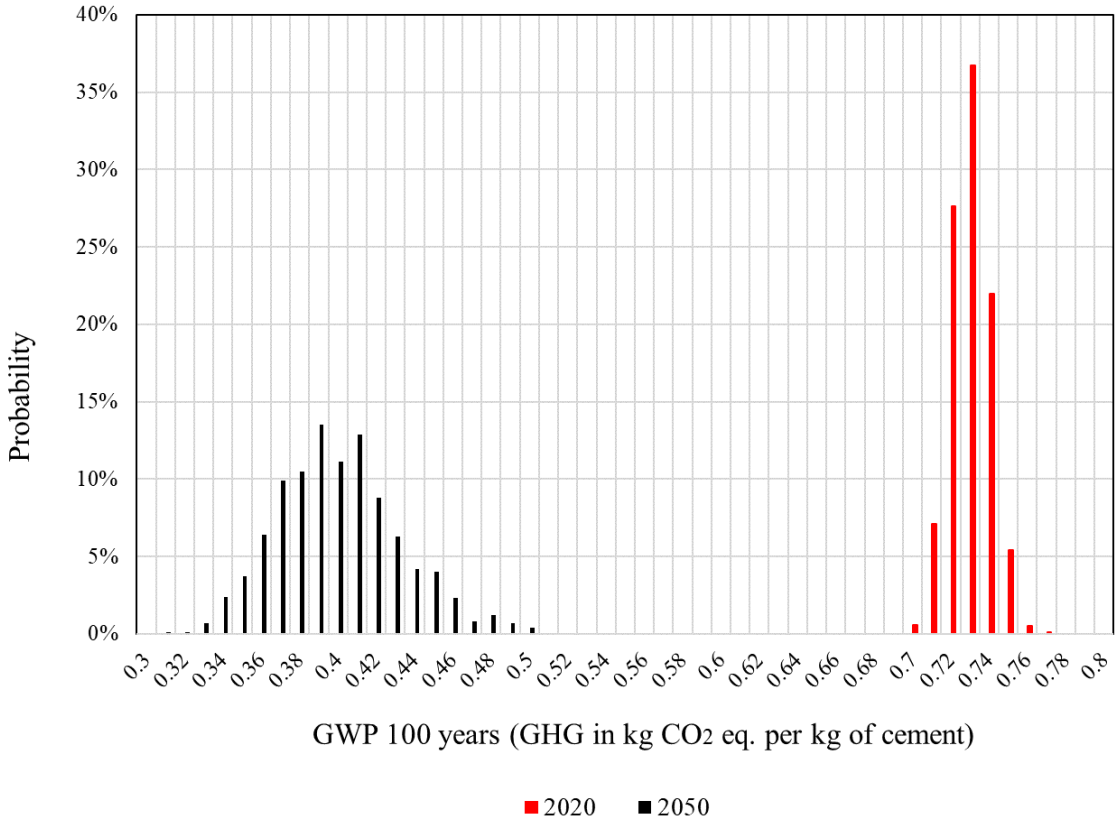


Figure B13: Monte Carlo results on cement production in 2020 and 2050, including SCMs substitution (coal fly ash), alternative fuels (hydrogen and biomass) and kiln configurations (from semi-wet/semi-dry process to dry process with pre-calciner and preheater.

This outcome concludes that current uncertainty in mass of clinker causes a relatively low impact fluctuation, indicating an insensitivity of impact to change. Inherently future modelling leads to an additional uncertainty, which is associated to the higher influence of clinker as a parameter hence higher instability and wider impact variability is observed for 2050.

B.7 Global sensitivity analysis

Monte Carlo was performed on a synergistic scenario taking coal fly ash (SCM) substitution, hydrogen (wind) and biomass as the primary fuels and kiln configuration improvements (semi wet/semi dry process to dry process with pre calciner and preheater).

The global sensitivity analysis generated the major influencing parameters such as clinker, electricity mixes municipal solid waste generation, and transport (Figure B14). The major

parameter influencing the impact in 2050 was found to be the clinker-to-cement ratio, consistent with the results discussed in Section B5 above.

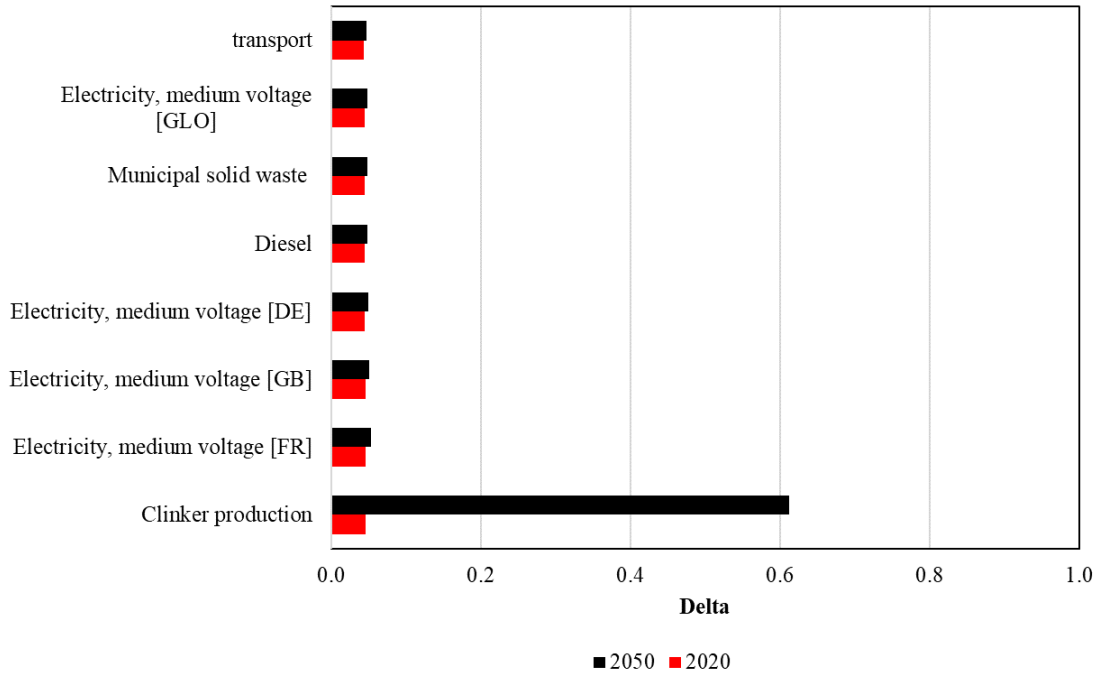


Figure B14: Global sensitivity analysis with major contributing flows of cement production with coal fly ash (SCM) replacement, fuelled by hydrogen (wind-powered) and biomass, and semi-wet/semi dry process in 2020 and dry process with precalciner and preheater in 2050.

B.8 Global decarbonisation approaches in cement industry

Global decarbonisation of cement production is limited by several factors including variation in local/national government regulations, logistics, electricity mixes, market demand, and raw material availability (IEA, 2018). However, certain measures have been regarded as necessary to decarbonise cement production. They are:

- Energy efficiency improvements, which involve the deployment of state-of-the-art technologies like kiln configurations (e.g., rotary kiln type [wet to dry type including a preheater, and a precalciner], and grinder types [high-pressure grinding rolls and vertical roller mills]). These state-of-the-art technologies can be deployed in new cement plants or retrofitted in existing ones (GCCA, 2021).

- Switching to alternative fuels, which promotes the consumption of fuels such us biomass and waste materials that would otherwise be landfilled and incinerated. By making use of waste materials, CO_2-eq. emissions from incineration are prevented and fossil-fuel dependency (including associated mining and processing) is reduced (Cembureau, 2020).

- Reducing the clinker to cement ratio, which aims to minimise the use of clinker per tonne of cement, e.g., by using supplementary cementitious materials like calcined clay (GCCA, 2021).

- Using emerging and innovative technologies. These technologies vary and include exhaust heat recovery, which utilizes waste heat from kiln flue gases to preheat raw materials prior entering the kiln, and carbon capture and storage and utilization technology (IEA, 2018).

- Alternative raw materials, which aim to minimise the amount of limestone feedstock for clinker production by replacement with other materials. These other materials include materials such as processed construction and demolition wastes, industrial ashes, and other industrial by-products (IEA, 2018).

Appendix C

Appendix C presents the LCIs for industrial-scale process models of BR processing technologies namely, calcination, vitrification, and reductive smelting. These processes are used to produce modified BR suitable for up to 30% PC clinker replacement as an SCM in composite cements. Additionally, the appendix provides details on the economic allocation of untreated BR, as well as mix designs and compressive strength results for composite cements incorporating BR-based SCMs.

C.1 Life cycle inventory

The LCI data for the industrial-scale BR processing technologies are shown here (as seen in Table C-1-. Material and energy flows for the co-calcination and reductive smelting technologies were provided by project partners (Konlechner et al., 2023). For vitrification, the industrial-scale inventory was developed using process modelling according to the upscaling approach described in Chapter 4. Process modelling was selected in consultation with project partners and technology developers as the most appropriate approach for representing the vitrification technology at scale.

Table C-1. LCI data for co-calcination of BR with Kaolin clay for SCM production

Co-calcined BR with kaolin clay		
Name	**Value**	**Unit**
Input flows		
Unmodified BR	26,000	kg
Kaolin clay	19,500	kg
Heat production, natural gas	40,736	MJ
Electricity consumption mixing	47.8	kwh
Transport, freight, sea, container ship	83	t-km
Output flows		
Co-calcined BR (SCM)	38,000	kg
Emissions to the environment		
Total particulate matter (PM)	660	kg

(Konlechner et al., 2023)

Included activities start: This activity starts with the reception of raw materials (e.g., unmodified BR (emission-free) from alumina production).

Included activities end: This activity ends with the modified BR suitable as an SCM. Packaging is not included, and no production waste is assumed.

Table C-2. LCI data for vitrification of BR for SCM production according to process modelling described in Chapter 4

Vitrified BR (binary blend)		
Name	**Value**	**Unit**
Input flows		
Unmodified BR (dry)	100,000	kg
Coke	2,040	kg
Limestone	22,000	kg
Silica	15,000	kg
Bentonite	1,500	kg
Pelletiser	587	kWh
Preheater	1,700	kWh
Vitrification	59,000	kWh
Milling	48	kWh
Transport, freight, sea, container ship	83	t-km
Output flows		
VBR (SCM)	125,800	
Emissions to the environment		
Carbon dioxide (CO_2)	25,880	kg
Vapour	9,180	kg
(Konlechner et al., 2023)		

Included activities start: This activity starts with the reception of raw materials (i.e., unmodified BR (emission-free) from aluminium production).

Included activities end: This activity ends with the modified BR suitable as an SCM. Packaging is not included, and no production waste is assumed.

Table C-3. LCI data for reductive smelting of BR for SCM production

Smelted BR		
Name	**Value**	**Unit**
Input flows		
Unmodified BR (wet, 25 wt% moisture)	100,000	kg
Unmodified BR (dry)	82,300	kg
Coke	9,780	kg
Limestone	16,500	kg
Silica	22,500	kg
Bentonite	1,623	kg
Cooling air	385000	kg
Heat production, natural gas	327,410	MJ
Transport, freight, sea, container ship	83	t-km
Output flows		
Pig Iron	24,200	kg
Smelted BR (SCM)	72,100	kg
Emissions to the environment		
Carbon dioxide (CO_2)	46417	kg

An economic allocation factor (0.47) was applied to BR to allocate the processing emission of smelted BR slag and pig iron accordingly.

Included activities start: This activity starts with the reception of raw materials (i.e., BR (emission-free) from aluminium production).

Included activities end: This activity ends with the modified BR suitable as an SCM. Packaging is not included, and no production waste is assumed.

C.2 Economic allocation for untreated BR

The unmodified BR is a waste by-product from aluminium production. Table C-4 presents the mass quantity and market price of untreated BR and aluminium. Although BR is currently classified as a waste and requires no allocation, if it were commercially used on an industrial scale as a cementitious feedstock, it would be reclassified as a by-product, requiring allocation of upstream emissions from alumina production. BR currently has a negative price of €10 per

ton due to disposal costs. Even with a 200% price increase to €10 per ton, the economic allocation factor remains negligible, with minimal impact from alumina production, given the significantly higher price of aluminium.

Table C-4. Parameters and economic allocation factor for untreated BR and aluminium

Parameter	**Quantity**	**Price**	**Revenue**	**Economic allocation factor**
Unit	t/year	euro/t	euro/year	
Revenues				
BR	93900	10	939000	0.0089
Aluminium	46950	2233	104839350	0.9911
Total revenues			105778350	

C.3 Mix design and compressive strength

Table C-5 shows the compressive strength of mortars made with composite cement with CBR, with varying BR/kaolin content in the raw mix. The BR content ranges from 100% to 70%, while the kaolin content increases correspondingly from 0% to 30%.

Table C-5: Compressive strength of mortars incorporating CEM II/B-B with CBR, with varying BR and kaolin clay content in the raw mix (data obtained from (Peys et al., 2022))

	Compressive strength (MPa)
Mortar mix design	**28 days**
BR100wt.%	41
BR90wt.%/K10wt.%	47
BR80wt.%/K20wt.%	51
BR70wt.%/K30wt.%	55

Table C-6 presents the compressive strength of mortars made with CEM II/B-B and CEM II/B-LL combined with VBR. Three fineness levels were evaluated for each composite cement; 4800, 4300, and 3700 cm^3/g, to examine the effect of surface area on environmental performance.

Table C-6: Compressive strength of mortars incorporating CEM II/B-B with VBR and CEM II/B-LL with VBR, with varying fineness. Three VBR fineness were explored according to Blaine method; 4800, 4300, and 3700 cm^3/g (data obtained from (Giels et al., 2022))

	Compressive strength (MPa)
Mortar mix design	**28 days**
$LVBRC_{4800}$	50.9
$LVBRC_{4300}$	45.8
$LVBRC_{3700}$	47.5
VBR_{4800}	50.0
VBR_{4300}	61.0
VBR_{3700}	32.0

Table C-7 presents the compressive strength of a mortar made with CEM II/B-B with SBR.

Table C-7: Compressive strength of mortars incorporating CEM II/B-B with SBR.

	Compressive strength (MPa)
Mortar mix design	**28 days**
SBR	30.95

Table C-8 presents the compressive strength of a mortar made with conventional cement types.

Table C-8: Compressive strength of mortars incorporating conventional cement types

	Compressive strength (MPa)	
Mortar mix design	**28 days**	
CEM II/A-V	47.8	(Conhyea & Goodary, 2021)
CEM II/A-L	42.5	(Irish Cement, 2017)
CEM II/A-S	55	(CEMMAC cement, 2017a)
CEM II/B-V	48	(OHORONGO cement, 2023)
CEM II/B-L	32.5	(CEMENTOS LA UNION S.A., 2024)
CEM II/B-S	50	(CEMMAC cement, 2017b)