IIIEE Theses 2020:17

To waste, or to resource?

A comparative Life Cycle Assessment of recovering bottom ash from waste incineration for use in road construction in Malmö.

Fynn Hauschke

Supervisor

Andrius Plepys

Thesis for the fulfilment of the
Master of Science in Environmental Management and Policy
Lund, Sweden, June 2020

© You may use the contents of the IIIEE publications for informational purposes only. You may not copy, lend, hire, transmit or redistribute these materials for commercial purposes or for compensation of any kind without written permission from IIIEE. When using IIIEE material you must include the following copyright notice: 'Copyright © Fynn Hauschke, IIIEE, Lund University. All rights reserved' in any copy that you make in a clearly visible position. You may not modify the materials without the permission of the author.

Published in 2020 by IIIEE, Lund University, P.O. Box 196, S-221 00 LUND, Sweden,
Tel: +46 – 46 222 02 00, Fax: +46 – 46 222 02 10, e-mail: iiiee@iiiee.lu.se.

ISSN 1401-9191

Acknowledgements

When I write this, a long journey of learning finally comes to an end. Almost 19 years spent in schools and universities, and there are many people to thank for this enormous privilege.

First and foremost, I have to thank my family. Mama, Jockel, Wiebke, Juli, Carlchen, Fritz Werner, Fine, Elsa, Lotta, Rudi, and Hugo. France, Spain, Portugal, Morocco, listening to my complaints, turning your eyes on me… - every one of you has supported me in their own special way. Mama, I can't start thanking you because otherwise there is no space for other people that are also (but not as) important. Papa, this time I would have had a beer with you to celebrate. Oma und Opa, auch Euch gilt mein ganz besonderer Dank. Wenn es sowas wie Vorbilder in meinem Leben gibt, dann seid Ihr es. Nicht von irgendwo habe ich meine Abschlussarbeit mit "der Müll" geschrieben.

Thanks to everyone at the Insti, my batch mates, and all the other batches that were in Lund during our time. For long nights, exhausting group works, exciting field trips, allowing me to ask questions, and a bit of complaining. I never imagined how incredible the life at the institute would be. I am looking forward to staying close and meeting along the way.

Thank you, Romy, Isak, and Rheanna for being great flat mates, connecting with our divine feminine, exploring the year 2179, oatmeal routines, and so much more.

Thank you for belaying me during the thesis period, Sherlock. For sharing little space, snacks, and just being a wonderful person. Climb on!

No thesis without your late wake up weather forecast and wildlife stories, Roohi. Emma, thank you for being my LCA buddy throughout the thesis. I am looking forward to my co-authorship for the first issue of "LCA for dummies: theoretical misconceptions and practical applications." This is for the prank war that never happened, Joonas. You're just too much of an angel.

Now there are many people that particularly contributed to the thesis project. Thank you Marjan for your helpful comments and interesting discussion on LCA. Without you, this project would not have been possible. My special thanks also to Raul and Sysav who have generously supported me all the way and the IIIEE that provided the laptop. Dr. Volker Wenning, many thanks for all the interesting insights and your valuable help. Thank you to my supervisor, Andrius Plepys, and all people that generously offered help and advice in Sweden, Denmark, and the Netherlands: Leif Nilsson, Jonas Nedenskov, Dorthe Lærke, Søren Dyhr-Jensen, Ole Hjelmar, Magnus Ahlcrona, Thomas Fruergaard Astrup, Martijn van Praagh, Abel Arkenbout, Peter Weywadt, and Magnus Svensson. ¡Sergio, muchas gracias por el buen café!

Steven, Philip, Murat, Leonidas, and Bea, thanks for your comments and reassuring words – I made it after all.

Thank you, Madeleine, and Miljöbron Skåne for making the cooperation with Sysav happen.

Finally, I would like to thank you, Frau Heyer. You have contributed a lot to the fact that I can write a master's thesis today.

Abstract

The low-carbon benefits and emissions savings from circular economy (CE) strategies are not yet well understood, yet are crucial for global development within Planetary Boundaries. Using the mineral fraction of waste incinerator bottom ash (MIBA) as construction material in road construction is one application of a CE strategy that could lead to environmental gains by substituting energy-intensive primary material and avoiding its alternative disposal in a landfill. The aim of this thesis is to improve the current understanding of, as well as quantify the resource efficiency and greenhouse gas (GHG) mitigation potential, from using MIBA as construction material, compared to virgin material. It contributes knowledge to local decision-making and to developing the CE concept more broadly. A Life Cycle Assessment (LCA) model to assess the environmental performance of a road built with MIBA, compared to a road built with virgin material, was developed and applied to a case study in Malmö, Sweden. Resource efficiency and GHG mitigation potential were assessed using a combination of two life cycle impact assessment methods: ILCD 2011 Midpoint+ and Cumulative Energy Demand. The results indicate less potential environmental impacts for the road scenario with MIBA in almost all impact categories. The analysis of the results shows that there can be some application contexts where using MIBA in lieu of primary material, creates larger benefits than in others. Important parameters were identified to be critical determinants for the environmental performance of the road with secondary material, including the transportation distance of materials, the types of substituted material, and specific properties of the secondary material. While the results are meant to support decision making, inherent limitations to the LCA methodology, must be considered when making decisions based on these results alone. Further research is needed to better account for resource-related impacts in a local context and to explore the effect of carbonation on the potential climate benefits of using MIBA.

Keywords: Life Cycle Assessment (LCA), waste incineration, bottom ash, secondary material, road construction

Executive Summary

Problem definition

It is widely known that the current economic development patterns are pushing humanity beyond multiple Planetary Boundaries, causing anthropogenic climate change, the alteration of biochemical flows, and the loss of biosphere integrity (Rockström et al., 2009; Steffen et al., 2015). A drastic change is needed to secure the availability of resources for future generations and to limit anthropogenic climate change to well below two degrees, compared to pre-industrial levels (IPCC, 2015, 2018). The concept of a circular economy (CE) is becoming increasingly popular among researchers, companies, and political decision-makers for its potential to enable a more sustainable, resource efficient, and low carbon human and economic development (e.g. Enkvist & Klevnäs, 2018; Hertwich et al., 2019; IRP, 2020). Secondary material use is an often-applied CE strategy that can reduce primary material demand and avoid the environmental impacts associated with waste management processes.

However, the low-carbon benefits from a CE are not yet well quantified (Enkvist & Klevnäs, 2018), and the emissions savings from material efficiency are still not well understood (Hertwich et al., 2019; IRP, 2020). Furthermore, it has been pointed out that secondary material use does not lead to a reduction of environmental impacts by default (e.g. Nußholz et al., 2019). Sourcing and preparing materials for their recovery involves additional activities, which in turn generate environmental impacts. Systematic assessments from a life cycle perspective are needed to measure material efficiency gains and life cycle emissions, to reveal possible trade-offs and/or synergies across product life cycles, and to design and evaluate CE policies (EC, 2020; IRP, 2020).

In this context, the mineral fraction of waste incinerator bottom ash (MIBA) could be used as secondary construction aggregates in road construction (Arm et al., 2017). In Sweden, one million tons of bottom ash are produced annually that could reduce the demand for energy-intensive primary materials and avoid its alternative disposal in a landfill (Blasenbauer et al., 2020). However, strict limits on the total content and leaching of heavy metals currently impede its use outside landfills (Van Praagh et al., 2018). While the majority of research regarding the environmental assessments of MIBA utilization so far has focused on the material properties such as the content and leaching of toxic substances, more assessments with extended system boundaries in the form of Life Cycle Assessments (LCA) are needed to understand its environmental performance (Silva et al., 2019).

The aim of this thesis is therefore to improve the current understanding of, as well as quantify the resource efficiency and greenhouse gas (GHG) mitigation potential, from using MIBA as construction material, compared to virgin material. The thesis thereby contributes to better understanding the environmental consequences of utilizing incineration residues in road construction, as well as helping to address uncertainties around the benefits of CE strategies more broadly. Further, by applying the model to a real case, the thesis also provides case-specific evidence for making more informed decisions regarding the use of MIBA in Malmö.

This was achieved through two key steps: 1) by developing an LCA model to assess the resource use and environmental impacts from utilizing MIBA in a road subbase, compared to primary raw material; and 2) by applying the model to a hypothetical case of reusing MIBA in road construction in the Malmö harbor area in Sweden. The following specific research questions were used to guide the research:

- RQ1: What is a suitable LCA model to assess the environmental performance in terms of resource efficiency and GHG mitigation potential of a road using MIBA, compared to a conventional road, in the specific case of the Malmö harbor area?
- RQ2: What are the environmental impacts in terms of selected impact categories from using post-incineration products for road construction, compared to virgin raw materials, based on the developed model?
- RQ3: What are the major parameters that influence the environmental performance of MIBA as secondary material, compared to virgin materials?

Research design, materials and methods

A single case study was chosen as a research strategy and in order to identify and organize relevant data. The case is the hypothetical recovery of MIBA for its use in road construction in the Malmö harbor area as envisaged by Sysav AB. Sysav is a municipality owned company managing and treating industrial waste and the household waste collected in its owner municipalities throughout the southern area of the region of Skåne, Sweden. It operates an incineration plant in the harbor area and seeks to promote the secondary use of MIBA in order to save scarce landfill capacities.

Qualitative data to describe the case study was mainly collected in the form of interviews with case study informants and was analyzed by developing a detailed case description. Quantitative data was collected by means of data questionnaires, interviews, and from LCA databases. This was analyzed through an LCA performed with the SimaPro v3.5 software.

The developed LCA model compares two variations of the same road. Scenario (A), refers to a road built with primary aggregates, crushed rock, in the subbase layer. Scenario (B) refers to a road in which the primary raw material is substituted by MIBA from the Sysav waste incineration facility. An attributional modelling approach with extended system boundaries was applied in order to include the alternative disposal of MIBA in a landfill.

Due to a focus on the comparison, only life cycle stages which are different between the two scenarios were considered in the system boundary. Identical life cycle stages such as the use, maintenance, and EoL were excluded. A further delimitation was the exclusion of impacts from the total contents and potential leaching of toxic substances from MIBA in all life cycle stages.

Resource efficiency and GHG mitigation potential were assessed by using a combination of two life cycle impact assessment (LCIA) methods: the ILCD 2011 Midpoint+ method and the Cumulative Energy Demand (CED). Different contribution and sensitivity analyses were conducted to identify parameters that influence the environmental performance of MIBA as secondary material, compared to virgin materials, in the specific case.

Results of the Life Cycle Assessment

Figure A and B display the main LCA results. Figure A shows a comparison of the normalized impact scores of scenario A and scenario B for impact categories included in the ILCD 2011 Midpoint+ method. The results are expressed in "Person Equivalents" (PE) and show the relative importance of the impacts by relating the characterization results to the flow of emissions or resources of an average European person in the year 2010. Figure B shows a comparison of the CED for scenario A and scenario B.

The assessment shows significantly less potential environmental impacts for the road scenario with MIBA, across all categories except water resource depletion (WRD). Even in a sensitivity analysis that considered a best-case scenario in terms of transportation to a landfill in scenario A, and a mass allocation approach for partitioning the environmental burdens of the sorting process in scenario B, a road with secondary material performs better across almost all impact categories. The GHG mitigation potential of scenario B compared to scenario A amounts to a difference of 386,145 kg-CO2 eq: a 281% improvement. The CED for scenario B, is 3.763 TJ less than for scenario A. With regard to the materials production stage for both scenarios, the assessment shows that, in the specific case, the energy demand for producing secondary material is only one third that of the primary material.

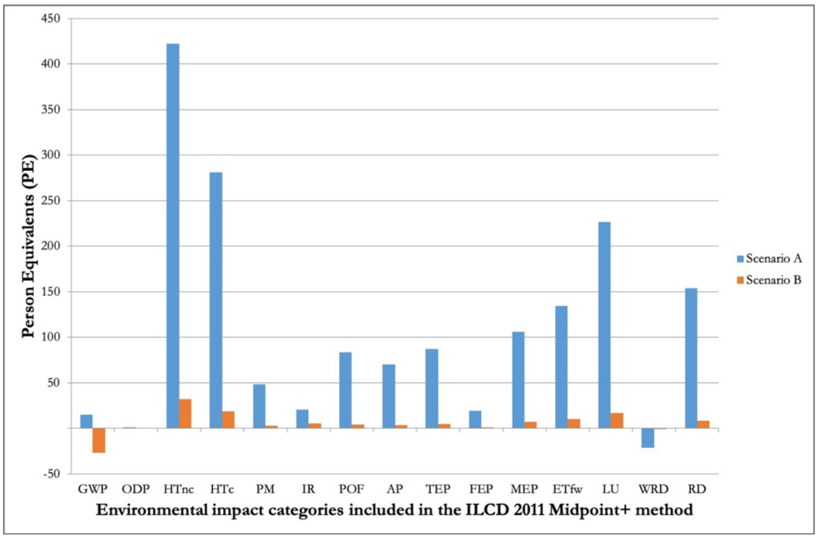
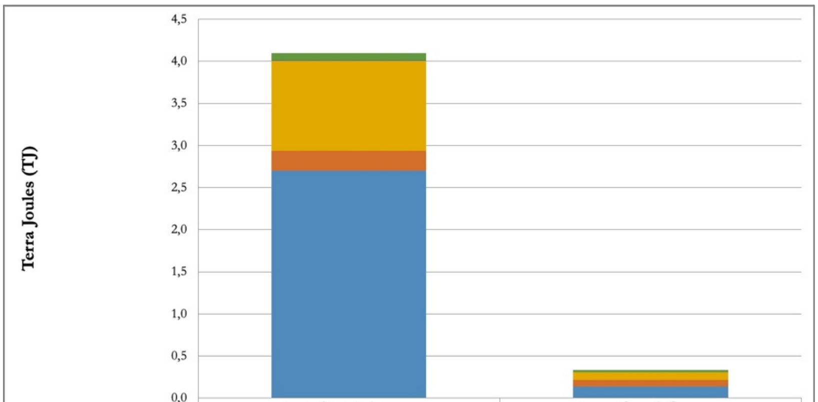


Figure A: Comparison of the normalized LCIA results of scenario A and scenario B.



	Scenario A	Scenario B
Renewable, water	0,0824	0,0252
Renewable, wind, solar, geothe	0,0099	0,0042
Renewable, biomass	1,0694	0,09
Non-renewable, biomass	0,0007	4,16E-05
Non-renewable, nuclear	0,2348	0,0759
Non renewable, fossil	2,6999	0,1386

Figure B: Comparison of the CED for scenario A and scenario B.

Conclusions and recommendations

Several important parameters were identified that critically impact the environmental performance of the road with secondary material, compared to the same road with primary material, in the specific case. The transportation distance of different materials was shown to be the largest contributor to environmental loads in both scenarios. Choosing biodiesel for transportation led to a tradeoff between the global warming potential (GWP) and resource related impact categories. The results further corroborate the importance of considering the alternative disposal scenario in assessments of secondary materials. The high energy demand of landfill processes and potentially long transportation distances to alternative disposal sites, can be important contributors to the environmental burden of the product system including primary material.

A further important consideration impacting the environmental performance of a road with MIBA relative to a conventional road scenario, is the type of primary material that is substituted. While crushed rock is used in Sweden, the improvements achieved by using MIBA in other countries like Denmark, where the use of natural gravel as construction aggregate is common, can be less significant. Finally, properties specific to the investigated material, i.e. carbonation and the ability to sequestrate carbon, were most influential in determining the climate benefits that could potentially be achieved by using secondary material instead of crushed rock.

The empirical findings are mostly in line with previous LCA studies on MIBA utilization from a product perspective (Birgisdóttir et al., 2006; Geng et al., 2010; Olsson et al., 2006). This thesis goes beyond earlier studies by also including resource related impact categories and by applying the CED to analyze the primary energy demand of the investigated product systems. It also highlights the importance of the carbonation process for the GHGs mitigation potential that was not part of earlier studies.

By including novel aspects, new areas that need consideration were identified. The characterization of resource use into impact categories has received less attention in reviewed studies. In a local context, however, the depletion of certain resources including bio-productive land, can be of particular importance. Methodological development is needed to more adequately account for such impacts. Further exploration on the impacts of carbonation on the GHG mitigation potential could be of interest as well.

For actors that seek to make environmentally sound decisions on whether to use primary or secondary material, be it planning departments of public authorities or private companies, this study shows that such a decision is an act of balancing trade-offs and carefully considering the specific circumstances. Using secondary material is not environmentally preferable per se. However, the results indicate that there could be some application contexts where using MIBA in lieu of primary material, creates larger benefits than in others. For example, the Malmö Harbor case appears to be one such application context.

LCA can provide useful insights for decision makers. Considering LCA results in addition to local risk assessments seems particularly important when considering secondary materials, as they might provide benefits that are not captured by local studies with narrow system boundaries. However, this study also highlights potential limitations of LCAs due to limited impact coverage, subjective assumptions, methodological shortcomings, and general data quality. What questions an LCA really answers and which conclusions its limitations actually allow, needs careful consideration.

Table of Contents

ACKNOWLEDGEMENTS .... **I**

ABSTRACT .... **II**

EXECUTIVE SUMMARY .... **III**

LIST OF FIGURES .... **IX**

LIST OF TABLES .... **XI**

ABBREVIATIONS .... **XII**

1 INTRODUCTION **1**

1.1 BACKGROUND AND PROBLEM DEFINITION 2
1.2 AIM AND RESEARCH QUESTIONS 6
1.3 SCOPE 6
1.4 ETHICAL CONSIDERATIONS 7
1.5 AUDIENCE 8
1.6 DISPOSITION 8

2 RESEARCH DESIGN, MATERIALS, AND METHODS **9**

2.1 CASE STUDY RESEARCH 9
2.2 CASE STUDY QUALITY 10
2.3 INTERVIEWS 10
2.4 DATA QUESTIONNAIRES, DOCUMENTS, AND SITE VISITS 11

3 LITERATURE REVIEW **12**

3.1 ENVIRONMENTAL AND LIFE CYCLE ASSESSMENT 12
3.1.1 Goal and scope definition *13*
3.1.2 Inventory analysis *16*
3.1.3 Impact assessment *16*
3.1.4 Interpretation *21*
3.2 BOTTOM ASH UTILIZATION IN ROAD CONSTRUCTION 21
3.2.1 Residues from waste incineration *21*
3.2.2 Treatment of incinerator bottom ash *22*
3.2.3 MIBA in road construction *23*
3.2.4 Alternative treatment of MIBA: landfilling *25*
3.2.5 Alternative material: primary raw material *27*
3.3 REUSE OF RESIDUES IN ROADS IN ENVIRONMENTAL AND LIFE CYCLE ASSESSMENT 27
3.3.1 Different assessment approaches *27*
3.3.2 Methodological choices *29*
3.3.3 Impact assessment results *32*
3.4 SUMMARY 34

4 THE CASE OF USING INCINERATION RESIDUES IN ROAD CONSTRUCTION IN MALMÖ **35**

4.1 CASE SPECIFIC CONTEXT: BOTTOM ASH UTILIZATION AND SYSAV AB 35
4.2 THE LIFE CYCLE ASSESSMENT MODEL 36
4.2.1 Goal and scope *36*
4.2.2 Inventory analysis *39*
4.2.3 Impact assessment *45*
4.2.4 Interpretation *47*

5 LIFE CYCLE ASSESSMENT RESULTS **48**

5.1 LIFE CYCLE IMPACT ASSESSMENT OF SCENARIO A 48

5.1.1 ILCD 2011 Midpoint+ ... *48*
5.1.2 Sensitivity analysis ... *50*
5.1.3 Cumulative Energy Demand ... *50*
5.2 LIFE CYCLE IMPACT ASSESSMENT OF SCENARIO B ... 51
5.2.1 ILCD 2011 Midpoint+ ... *51*
5.2.2 Sensitivity analysis ... *52*
5.2.3 Cumulative Energy Demand ... *53*
5.3 COMPARISON BETWEEN SCENARIO A AND B ... 54
5.3.1 ILCD 2011 Midpoint+ ... *54*
5.3.2 Sensitivity analysis ... *56*
5.3.3 Cumulative Energy Demand ... *57*

6 DISCUSSION ... 59

6.1 RESULTS IN THE CONTEXT OF PREVIOUS KNOWLEDGE ... 59
6.1.1 Transportation ... *59*
6.1.2 Fuel choice and land use ... *60*
6.1.3 Type of substituted material ... *61*
6.1.4 Energy use and the importance of considering the alternative disposal ... *61*
6.1.5 Climate benefits and carbonation ... *62*
6.2 REFLECTIONS AND LIMITATIONS ... 63
6.2.1 Critical assumptions ... *63*
6.2.2 Method-related limitations ... *64*
6.2.3 Limited impact coverage ... *65*
6.2.4 Inventory data quality ... *66*
6.2.5 Generalizability ... *67*
6.2.6 Should bottom ash be used in road construction in Malmö? – Some further reflections ... *68*

CONCLUSION ... 69

BIBLIOGRAPHY ... 71

APPENDIX ... 79

APPENDIX I: ROAD LAYOUT PROVIDED BY THE CITY OF MALMÖ ... 79
APPENDIX II: LIST OF INTERVIEWEES ... 79
APPENDIX III: SELECTED INTERVIEW RESULTS ... 81
APPENDIX IV: DATA QUESTIONNAIRE ... 83
APPENDIX V: DATA DESCRIPTION AND TABLES SCENARIO A ... 83
APPENDIX VI: DATA DESCRIPTION AND TABLES SCENARIO B ... 85
APPENDIX VII: LCA RESULTS, GRAPHS, AND FIGURES SCENARIO A ... 89
APPENDIX VIII: LCA RESULTS, GRAPHS, AND FIGURES SCENARIO B ... 93
APPENDIX IX: LCA RESULTS, GRAPHS, AND FIGURES COMPARISON SCENARIO A AND B ... 97

List of Figures

Figure 1: Illustration of the role of waste incineration with energy recovery with subsequent scrap metal extraction, and utilization of bottom ash in a CE. 3

Figure 2: Illustration of the difference between IBA and MIBA. 4

Figure 3: The life cycle assessment framework proposed by the International Organization for Standardization. 12

Figure 4: Schematic representation of the environmental mechanisms from inventory to endpoints.. 17

Figure 5: Different solid residues remaining from the incineration process. 21

Figure 6: General structure of a road.. 24

Figure 7: Categorization of environmental assessment methods regarding the reuse of by-products in road construction in Sweden proposed by Roth & Eklund (2003). 28

Figure 8: Map of the Malmö harbor area. 35

Figure 9: Simplified layout of the two considered road scenarios. 37

Figure 10: System boundaries considered in scenario A. 40

Figure 11: System boundaries considered in scenario B. 42

Figure 12: Simplified illustration of the sorting process and its outputs........................... 43

Figure 13: Normalized LCIA results for scenario A grouped into processes related to the alternative disposal of MIBA and processes related to the production of crushed rock. 48

Figure 14: Normalized LCIA results of scenario A grouped into contributions of transport processes, the crushed rock production process, and processes related to the alternative disposal of MIBA. 49

Figure 15: Comparison of the normalized LCIA results of scenario A considering two transportation scenarios for MIBA to its final disposal. 50

Figure 16: The CED for scenario A. 51

Figure 17: Normalized LCIA results for scenario B grouped into processes related to the treatment of MIBA and transportation processes. 52

Figure 18: Comparison of the normalized LCIA results for scenario B using economic allocation and mass allocation for partitioning the environmental burdens of the sorting process. 53

Figure 19: The CED for scenario B. 54

Figure 20: Comparison of the normalized LCIA results of scenario A and scenario B. . 55

Figure 21: Comparison of the normalized LCIA results for scenarios A1, A2, B1, and B2. 57

Figure 22: Comparison of the CED for scenario A and scenario B.................................. 58

Figure 23: Road layout provided by the City of Malmö. 79

Figure 24: Network diagram of scenario A. 89

Figure 25: Normalized impact assessment results for scenario A without grouping. 90

Figure 26: Characterization results for scenario A including two groups of processes. . 90

Figure 27: Characterization results for scenario A including three groups of processes.. 91

Figure 28: Characterization results for Scenario A for a comparison between the current (baseline) transportation scenario and a hypothetical future transportation scenario. 91

Figure 29: Characterization results for the CED of scenario A including three groups of processes...92

Figure 30: The CED of scenario A divided into three process groups. 92

Figure 31: Network diagram of scenario B. 93

Figure 32: Normalized LCIA results for scenario B without grouping. 94

Figure 33: Characterization results for scenario B including two groups of processes....94

Figure 34: Characterization results of scenario B for a comparison between economic allocation and mass allocation for partitioning the environmental burdens of the sorting process. 95

Figure 35: Characterization results for the CED of scenario B including two groups of processes...95

Figure 36: The CED of scenario B divided into two process groups................................96

Figure 37: Comparison of the characterization results for scenario A and B...................98

Figure 38: Comparison of the characterization results for scenarios A1, A2, B1, and B2. 98

Figure 39: Comparison of the characterization results of the CED for scenario A and B. 99

List of Tables

Table 1: Systematized framework of resource efficiency indicators at life cycle level presented by Huysman et al. (2015)....19

Table 2: Material intensity of 1 km of road for scenario A and scenario B....38

Table 3: Allocation factors used for the economic allocation applied to the multifunctional sorting process based on 100 t of sorted IBA....45

Table 4: List of impact categories of the ILCD 2011 Midpoint+ method used for the impact assessment. 46

Table 5: Characterization results for scenario A, scenario B, and the difference between the two scenarios. 55

Table 6: Characterization results (in kg-CO2 eq) for the impact category GWP for scenario A and scenario B. 56

Table 7: Different scenarios tested in the sensitivity analysis. 56

Table 8: Characterization results for the GWP impact category (in kg-CO2 eq) for different scenarios. 57

Table 9: Examples of assumptions made to develop the LCA model. 63

Table 10: List of people that were consulted in the course of the study. 79

Table 11: Summary of literature and interview results with regard to potential differences in the life cycles of a road with and without bottom ash. 81

Table 12: Questionnaire used to collect data from Sysav and AB Sydsten. 83

Table 13: Considered database processes and inventory data for the production of crushed rock in scenario A. 84

Table 14: Average amount of bottom ash transported from the incineration plant in Malmö to different landfill sites for final disposal during 2018 and 2019. 84

Table 15: Transportation considered in scenario A. 84

Table 16: Calculation of the energy demand of the Sysav sorting plant per kilogram of IBA treated. 85

Table 17: Ecoinvent datasets included in the sorting process. 86

Table 18: Fuel consumption of construction vehicles at the bottom ash sorting facility and the resulting energy demand from diesel fuel. 86

Table 19: Transportation considered in scenario B. 87

Table 20: Expected amount of material received by the Sysav sorting facility, the sorting efficiency, and the expected amount of material sorted out. 87

Table 21: Calculation of MIBA management cost used for the economic allocation. 88

Table 22: Characterization and normalization results for scenario A and B. 97

Abbreviations

AoP – Area of protection

ALCA – Attributional life cycle assessment

CE – Circular economy

CED – Cumulative energy demand

CLCA – Consequential life cycle assessment

EoL – End-of-life

EPD – Environmental product declaration

Fe metals – Ferrous metals

GHG – Greenhouse gas

GWP – Global warming potential

ILCD – International Reference Life Cycle Data System

IBA – Incinerator bottom ash

IPCC – Intergovernmental Panel on Climate Change

LCA – Life cycle assessment

LCIA – Life cycle impact assessment

MIBA – Mineral fraction incinerator bottom ash

MSW – Municipal solid waste

NFe metals – Non-ferrous metals

PE – Person equivalents

1 Introduction

Our current production and consumption patterns are pushing humanity beyond multiple Planetary Boundaries and are destabilizing the critical processes that regulate the Earth's System. Anthropogenic climate change, the alteration of biochemical flows, and the loss of biosphere integrity are all symptoms of unsustainable human "development" (Rockström et al., 2009; Steffen et al., 2015). A drastic change is needed to secure the availability of resources for future generations and to limit anthropogenic climate change to well below two degrees compared to pre-industrial levels (IPCC, 2015, 2018). However, over the coming four decades, the global consumption of materials – including minerals, metals, biomass, and fossil fuels – is projected to double, while the generation of waste is expected to increase by 70% (OECD, 2019). To change this destructive trajectory, there is a clear need for a more sustainable, resource efficient, and low carbon human and economic development.

In this context, the idea of a circular economy (CE) is becoming increasingly popular among researchers, companies, and political decision-makers (Schroeder et al., 2019). Although the concept has varying interpretations, it is generally agreed that at its core it seeks to retain the value of resources in the economic system as long as possible and to minimize waste. As such, a CE is understood as an umbrella concept that groups different strategies for resource life-extension (Blomsma & Brennan, 2017). Its principles offer a range of strategic options to facilitate resource efficiency and provide an effective policy framework to transform the unsustainable use of materials (IRP, 2020). From a product design and business model perspective, possible strategic options to move towards a CE model include, for example, "closing", "slowing", and "narrowing" resource loops (Bocken et al., 2016). Applied to the waste and resource management sector, the concept contributes to defining what are "meaningful and actionable waste and resource management practices" (Blomsma & Brennan, 2017, p. 611). A concrete operationalization of the concept is found in the Waste Management Hierarchy laid down in the EU Waste Framework Directive (Council Directive 2008/98/EC).

An economy based on CE principles, rather than a linear take-make-dispose logic, is generally viewed as a more sustainable approach to production and consumption, and supports the ultimate goal of decoupling economic growth from environmental consequences (Ghisellini et al., 2016). It is acknowledged that a CE provides development opportunities in line with the principles of sustainable development (Kirchherr et al., 2017) and policy makers and research particularly highlight its potential in terms of a low-carbon development (e.g. Enkvist & Klevnäs, 2018; Hertwich et al., 2019; IRP, 2020). It is argued that while addressing energy efficiency and low carbon energy are important, increasing the material efficiency in general and reducing the demand for primary materials particularly, is indispensable to reach international and European climate targets (Enkvist & Klevnäs, 2018).

As much as 50% of the total greenhouse gas (GHG) emissions and more than 90% of water stress and biodiversity loss result from the extraction and processing of resources (EC, 2020). The share of emissions from materials production in the global carbon footprint increased from 15% to 23% between 1995 and 2015. This is equivalent to the emissions from forestry, land use change, and agriculture combined. The majority (80%) of emissions resulting from solid materials[1] production are associated with their use in manufactured goods and construction (IRP, 2020). It stands to reason that a more efficient use of these materials can decrease the

[1] Including construction material, wood, metals, and plastics. Excluding food, fuel, and chemicals.

demand for energy-intensive primary materials and through this, offers significant potential to mitigate GHGs (Hertwich et al., 2019).

However, not only is the global economy – with only an estimated 9% circularity – far from being "circular" (de Wit et al., 2019), but the low-carbon benefits from a CE are not yet well quantified (Enkvist & Klevnäs, 2018), and the emissions savings from material efficiency are still not well understood (Hertwich et al., 2019; IRP, 2020). Indeed, each anthropogenic material cycle is unique, and more work is needed to quantify and assess the diversity of material uses and circumstances, if informed decisions are to be made by actors seeking to improve the environmental, social and/or economic performance of material cycles.

The use of residual material or by-products in new products, is an often-applied CE strategy that can reduce primary material demand and avoid the environmental impacts associated with waste management processes. These materials are referred to as secondary materials. However, secondary material use does not lead to a reduction of environmental impacts by default. Sourcing and preparing them for their recovery involves additional activities, which in turn generate environmental impacts. Environmental regulations and technical requirements can demand substantial upgrading processes, while other context dependent issues like transportation, can further contribute to the environmental burden of secondary materials (Nußholz, 2020). A systematic assessment of the environmental consequences is therefore needed in the specific case. Life Cycle assessment (LCA) is a suitable tool to measure material efficiency gains and life cycle emissions, to reveal possible trade-offs and/or synergies across product life cycles, and to design and evaluate CE policies (EC, 2020; IRP, 2020).

This thesis aims at quantifying and understanding the resource efficiency and GHG mitigation potential from secondary resource use for a specific case and in a specific context. A comparative environmental assessment based on LCA methodology is applied to a case study on the recovery of waste incineration residues as construction material for its use in road constructions in Malmö, Sweden. The thesis is located at the nexus of waste management and construction. "Reuse" is an umbrella term used in academic literature referring to the reuse and recovery of products and materials (Nußholz, 2020). In the terminology specific to the field of waste management, the term "recovery", as defined in the EU Waste Framework Directive, would be applied in this specific case (Council Directive 2008/98/EC).[2] Throughout the thesis, the term reuse will be applied.

1.1 Background and problem definition

In 2018, every Swede produced 466 kg of household waste. Of this, 34.3% were recycled, 15.5% went to biological treatment, 49.7% to energy recovery, and only 0.7% was landfilled (Avfall Sverige, 2019a). With almost 50% of the treated waste, incineration with energy recovery constitutes the backbone of the Swedish waste management system. 34 municipal solid waste (MSW) incinerators amount to a total capacity of 5.7 million tons per year or 591 kg per capita (Blasenbauer et al., 2020; Wilts et al., 2017). Sweden is therefore, the only European country where incineration capacities exceed the amount of mixed municipal waste produced (Wilts et al., 2017). In this thesis, waste incineration with energy recovery refers to incineration that is defined as "other recovery" under EU legislation, based on the "R1" formula. It defines whether

[2] In the EU Waste Framework Directive, recovery is defined as "any operation the principal result of which is waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfil a particular function, or waste being prepared to fulfil that function, in the plant or in the wider economy" (Council Directive 2008/98/EC, Art. 3, 15). Specific recovery operations are further defined in Annex II.

incinerators can be classified as "recovery operations" as opposed to "disposal operations" based on the energy generation efficiency (EC, 2011).

While the ultimate goal of a CE is to design-out waste (EMF, 2013), there is still a need for safe disposal options (Brunner & Rechberger, 2015). Waste incineration with energy recovery (hereafter called "incineration"), provides one disposal option that allows for the use of waste as a resource by recovering energy in the form of heat and/or electricity, as well as by facilitating material recycling of scrap metals from incineration residues (Van Caneghem et al., 2019). As such, incineration is considered an important complementary part of sustainable resource management and a CE (Brunner & Rechberger, 2015; Kral et al., 2019).

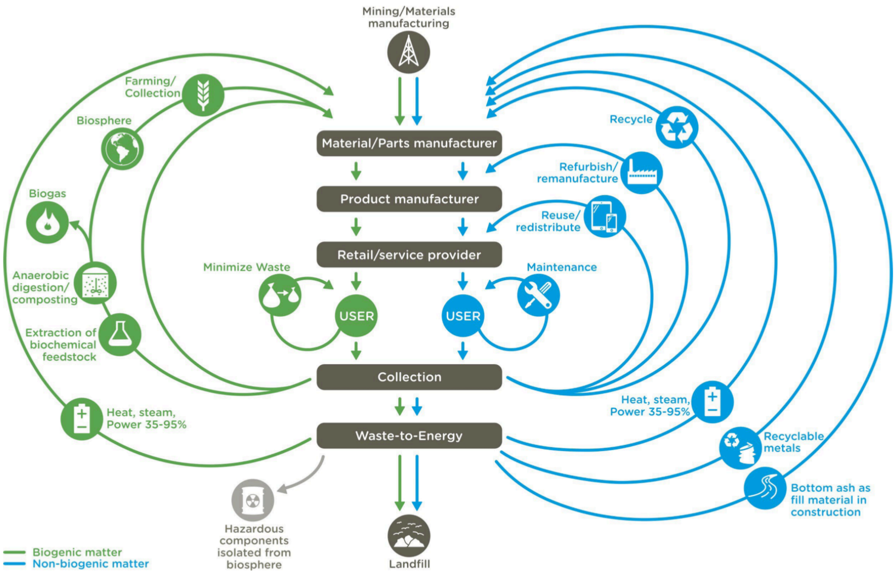


Figure 1: Illustration of the role of waste incineration with energy recovery with subsequent scrap metal extraction, and utilization of bottom ash in a CE.

Source: Kahle et al., 2015, p.4.

Incineration, however, is not a final disposal option as it produces different solid residues (Hykš & Hjelmar, 2018). These include *incinerator bottom ash (IBA)*, which is the remaining part of the burned waste collected at the bottom of incineration chambers, as well as fly ash and air pollution control residues that are both filtered out from flue gases. IBA is the largest fraction of incineration residues, ranging from 150 to 300 kg per ton of treated waste (Astrup et al., 2016).

Currently, within the EU, including Norway and Switzerland, close to 500 incineration plants generate about 17.6 Million tons of IBA annually (Blasenbauer et al., 2020). Due to the long life-time of facilities and infrastructural lock-in effects (Corvellec et al., 2013), energy recovery is likely to remain an essential part of a CE for the coming decades. This results in the need to manage 15-20 million tons of IBA annually, in the medium-term future (Hykš & Hjelmar, 2018). In Sweden, one million tons of IBA are produced annually (Blasenbauer et al., 2020).

Three general management scenarios exist for IBA after it has been pre-treated: landfilling, use as construction material in landfills, or its use in the civil engineering sector. This thesis considers two of these scenarios: landfilling of bottom ash in a special compartment of a sanitary landfill, and its use as secondary construction material in the civil engineering sector. While there are different potential applications for IBA in civil engineering, the thesis is concerned with the most prominent application: the use of pre-treated IBA as secondary material in the subbases layer of road constructions (Lynn et al., 2018).

In the following, IBA refers to the unprocessed bottom ash. The term MIBA ("mineral fraction incinerator bottom ash") is introduced to describe IBA after its treatment by means of sorting and aging. Sorting includes several processes such as crushing, extraction of oversize materials, ferrous (Fe) metals, non-ferrous (NF) metals, and unburned organic material. Aging means that the sorted bottom ash is stockpiled for a period of usually 2-6 months in order to reduce its acidity and immobilize contaminants. The end product (MIBA) thus refers to a largely inert mineral fraction extracted from IBA (see Figure 2).

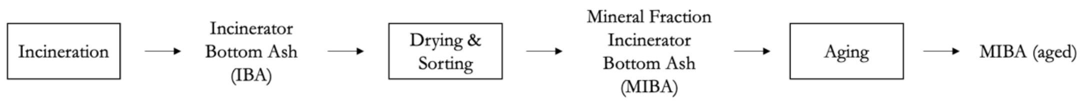


Figure 2: Illustration of the difference between IBA and MIBA. Throughout the thesis, the term "MIBA" refers to the aged MIBA as shown in this figure.

CE principles suggest the use of MIBA as secondary construction material instead of landfilling. Potential benefits result from saving scarce landfill space, reducing the demand for virgin materials in road construction, and avoiding emissions from both the operation of the landfill and the extraction of primary material (e.g. Balaguera et al., 2018; Silva et al., 2019). Blasenbauer et al. (2020) estimate that using MIBA could lead to a reduction of the required volume of landfills for non-hazardous waste in the EU by 7-8 vol% or 5 million m^3.

Meanwhile, vast quantities of virgin materials such as crushed rock and natural gravel are extracted for their use in road construction. In Sweden, 86 million tons of aggregates were consumed by the construction sector in 2016. This corresponds to almost one metric ton per capita. More than half (56%) is used for road constructions. Although bedrock used in the production of crushed rock is still an abundant resource in Sweden, the production of aggregates from natural gravel has long been under strain due to its depletion, as well as concerns regarding impacts on the quality of groundwater (SGU, 2017).

MIBA can replace some of these virgin materials. Considering the entire life cycle of a road, different studies indicate that the production of the construction material and its transportation constitutes the most important life cycle stage in terms of environmental impacts (Hammervold, 2014; Mroueh et al., 2001; Trunzo et al., 2019). The use of secondary materials yields overall better environmental results in terms of the climate impact and other impact categories and is therefore recommended as a strategy to improve the environmental performance of roads (Hertwich et al., 2019; Moretti et al., 2018; Simion et al., 2013).

Nevertheless, the use of MIBA outside landfills in Sweden, other than in test roads and small scale pilot projects, is nonexistent (Avfall Sverige, 2019b; Blasenbauer et al., 2020). This is also the case with other secondary materials used for the subbase layers of roads in Sweden (Respondent 4, personal communication, December 12, 2019). While there are numerous barriers to the more widespread use of secondary materials in general (e.g. Kirchherr et al., 2018; Milios, 2016; Nußholz et al., 2019), and particularly for MIBA utilization in road construction (e.g. Blasenbauer et al., 2020; Kahle et al., 2015), it is the low legal limits on the total content and leaching of heavy metals in Sweden that is held to impede its utilization (Van Praagh et al., 2018).

The content of salts and trace metals such as lead, cadmium, and zinc in MIBA, is higher than in primary aggregates, and is considered the main potential risk to human health and the environment (Astrup et al., 2016; Lynn et al., 2018). However, if the specific application conditions in the subbase layer of roads are considered, i.e. under an impermeable cover, studies conducted in Sweden indicate that the leaching of salts and pollutants from MIBA should not be a limiting variable for its use in roads (Arm et al., 2017; Avfall Sverige, 2015; Van Praagh et al., 2018). Further studies corroborate that the use of MIBA in road subbases can fulfill environmental requirements that are based on the actual risk posed by its use in road subbases (Hykš & Hjelmar, 2018) (also see Lynn et al., 2018; and Silva et al., 2019).

There is no uniform regulation at the EU level regarding the use of MIBA in road construction. Multiple EU countries have introduced regulatory quality criteria and guidelines for its use as unbound aggregate (Blasenbauer et al., 2020). Regulations vary to a great extend between countries as well as in some cases between regions within Member States, such as Germany (BDI, 2020; Onkelbach, 2016). In most cases, these quality criteria include limits on the total content of substances and on leaching accompanied by conditions on its use. In some cases, there are multiple sets of limit values corresponding to different more or less restrictive use conditions (Hykš & Hjelmar, 2018). The situation in Sweden is very different from some other countries like the Netherlands, Denmark, or France, where the utilization rate of MIBA outside landfills is high, 100%, 99%, and 80% respectively (Blasenbauer et al., 2020).

The heterogenous policy landscape regarding the use of MIBA as unbound aggregate in road constructions, demonstrates a disagreement regarding the actual and perceived environmental and health risk posed by the MIBA (e.g. leaching of heavy metals), versus the manner in which its potential benefits from saving of natural resources and GHG emissions are valued. At the same time, legal standards are based on environmental assessments that do not account for the potential benefits resulting from its use. It further shows that the CE concept provides general guidelines for resource strategies that contribute to sustainable development; however, an assessment of concrete cases is necessary to support environmentally sound decisions.

In this context, a life cycle approach to the assessment is suggested by academia and EU legislation, in order to serve as a decision-making tool for circular waste management strategies (Council Directive 2008/98/EC); the use of secondary materials in road construction (Roth & Eklund, 2003; Schwab et al., 2014); and to assess the utilization of MIBA in road construction in particular (Birgisdóttir, 2005; Santero et al., 2010; Silva et al., 2019; Toller, 2008).

However, the majority of research regarding the environmental assessments of MIBA utilization so far has focused on the material properties like the total content and leaching of heavy metals and salts (Silva et al., 2019). This is also influenced by the legal requirements regarding its use in most countries. Few comprehensive LCA studies have been done on the use of MIBA as secondary material. These either examine different waste management strategies (e.g. Allegrini et al., 2015; Birgisdóttir et al., 2007; Toller et al., 2009) or specific road-construction scenarios that include the use of MIBA. In depth assessments of road construction scenarios comparing the same road built with and without MIBA were conducted by three scholars (Birgisdóttir et al., 2006; Geng et al., 2010; Olsson et al., 2006).

Nevertheless, LCA studies that assess the use of secondary materials in road construction in general, and MIBA utilization in particular are scarce. Santero et al. (2010) point out that due to the special characteristics of roads and different methodological choices, studies are hardly comparable, and that drawing general conclusions on, for example, whether the use of secondary material is preferable from an environmental perspective, is impossible. As a result, conclusions are limited to the case that is investigated in a given study. Furthermore, "there is still a lack of studies that address the overall environmental impacts of the reuse of mineral industrial wastes" (Schwab et al., 2014, p. 1885), and not enough emphasis has been given to conducting LCAs of MIBA utilization in order to understand the environmental performance (Silva et al., 2019). To improve the basis for decision making with regard to the use of secondary materials in roads, further studies with expanded system boundaries, i.e. considering potential environmental benefits or burdens from avoiding primary resources or the alternative disposal of residues, are therefore needed (Roth & Eklund, 2003).

Against this background, this thesis analyses the resource use and environmental impacts for a road built with MIBA compared to a road built with virgin raw material based on LCA methodology. The analysis focuses on the potential of MIBA as a secondary resource in terms of resource efficiency and GHG mitigation, as well as looking at the parameters that determine the environmental performance of the circular strategy compared to the linear approach.

1.2 Aim and research questions

The aim of this thesis is to improve the current understanding of, as well as quantify the resource efficiency and greenhouse gas (GHG) mitigation potential, from using MIBA as construction material, compared to virgin material. It contributes to knowledge generation at three levels:

1. It provides a case study of an environmental assessment for a specific CE strategy. By doing so, the thesis seeks to contribute to addressing the current lack of understanding regarding the potential low carbon benefits, and emission savings from material efficiency, of CE strategies more broadly.
2. The assessment of reusing MIBA with extended system boundaries, addresses a weak spot in current scientific literature, and contributes to a better understanding of the environmental consequences of utilizing incineration residues in road construction.
3. By conducting the research as a specific case study, the thesis also provides scientific evidence for local practitioners and political decision makers to make more informed (policy) decisions.

The thesis is composed of two major tasks: 1) the development of an LCA model that is suitable to assess the resource use and environmental impacts from utilizing MIBA in a road subbase, compared to primary raw material; and 2) the application of the model to a specific case. The following specific research questions are used to guide the research:

RQ1: What is a suitable LCA model to assess the environmental performance in terms of resource efficiency and GHG mitigation potential of a road using MIBA, compared to a conventional road, in the specific case of the Malmö harbor area?

RQ2: What are the environmental impacts in terms of selected impact categories from using post-incineration products for road construction, compared to virgin raw materials, based on the developed model?

RQ3: What are the major parameters that influence the environmental performance of MIBA as secondary material, compared to virgin materials?

1.3 Scope

The thesis focuses on a single case study for which an LCA model was developed. The case is the hypothetical reuse of MIBA in the Malmö harbor area. Bottom ash is to be sourced from and treated at, the waste incineration plant operated by Sysav AB (hereafter called "Sysav") which is located in the same area, and is compared to virgin materials for the building of a planned (so far hypothetical) road near to the source of the secondary material. A specific type of road suitable for the case study area was selected together with representatives of the City of Malmö. The case and its context are elaborated in detail in Section 4.1.

The comparative LCA considers two scenarios for the same road. In one scenario, the road is built with primary aggregate, crushed rock, in the subbase layer. In the second scenario, the primary raw material is substituted by MIBA in the form of unbound aggregate from the Sysav waste incineration facility. The climate impacts and resource efficiency are evaluated based on impact assessment methods included in the SimaPro software that was used to conduct the LCA.

An important delimitation of the LCA model is the limited impact coverage. Although the potential leaching of heavy metals is considered to be a major contributor to the potential environmental impacts from a road built with MIBA in the subbase, it is deliberately excluded from the scope. The main reasons for this include:

- the underdeveloped toxicology part of LCA and difficulties related to the inclusion of leaching into an LCA model as well as the subsequent characterization step;
- a lack of time and background data to develop a sufficiently accurate model for leaching and including it in the LCA methodology; and
- significant inherent uncertainties when comparing and weighing global (e.g. climate impacts) vs. local impacts (e.g. toxic impacts), as well as impacts that occur in the short-term future vs. impacts that occur beyond the first hundred years.

A detailed description of the scope of the developed LCA model is provided in Section 4.2.

1.4 Ethical considerations

The project was conducted in collaboration with several organizations. The partner organization that commissioned the study is Sysav Utveckling AB. The company finances and conducts projects with the aim to develop and improve existing and new methods and technologies in the field of waste management. It is a subsidiary company of Sysav AB, a municipality owned company managing and treating industrial waste and the household waste collected in its owner municipalities throughout the southern area of the region of Skåne, Sweden.

The research is conducted as part of a collaboration between Sysav, Swerock AB, and the RISE Research Institute of Sweden AB (hereafter called "RISE"). Swerock is a subsidiary company of Peab AB, the third largest construction company in Sweden. Swerock supplies construction materials and machines and operates a recycling plant for construction and demolition waste in the Malmö harbor area. RISE is a state-owned research institute, that collaborates with universities, industry, and the public sector to perform industry research and innovation as well as certification. The joint interest of the organizations is the potential use of secondary materials in road construction.

The research was supported financially by Sysav Utveckling AB. The company provided a compensation in the form of a lump sum payed for every credit point, in total 20,000 SEK. Sysav further payed any travel expenses incurred by research activities, and payed for the cost of the software used to conduct the LCA, 2550 EUR. In addition, Sysav Utveckling AB funded and arranged for meetings with a technical expert from RISE, in order to provide guidance and discuss the LCA model. The cost for the consultancy was 20,000 SEK.

Participation in interviews for this thesis was entirely voluntary. They were conducted with the informed consent of the participants. This was achieved verbally by briefing participants on all relevant aspects of the project and the intended use of the data prior to the interview. There is no reason to believe that any participant may suffer any disadvantage or damage from their participation in the study. Interviews with experts were anonymized, although this was not requested by any respondent. Quantitative data was handled with Excel and SimaPro on a computer owned by Lund University. Confidentiality was discussed with Sysav, and a non-disclosure agreement was found to not be necessary.

1.5 Audience

The thesis is predominantly aimed at generating knowledge to support informed decision making. By describing in detail, the potential impacts and benefits of using MIBA in the specific case of Malmö, it enables multiple actors to make more informed decisions regarding the use of secondary materials in road construction. Particularly, the producers and individual users of MIBA as well as environmental planners, benefit from a broader information base. Users include, for example, companies organized in the Copenhagen Malmö Port that develops the North Harbor area and Swerock AB. The Real Estate and Street Office of the City of Malmö, which is concerned with city planning, also expressed particular interest due to the potential climate benefits of using secondary materials. For producers of bottom ash like Sysav, information regarding potential benefits of using MIBA could provide arguments in their quest to find alternative uses for bottom ash. The thesis is also of interest for researchers in the field of construction materials and waste management. Although few similar studies have been conducted previously, new aspects such as the sequestration of carbon during aging have been included in this study. Finally, the thesis also provides a case study of secondary resource use that strengthens the empirical basis for the assessment of CE strategies.

1.6 Disposition

Following Chapter 1 which introduces the thesis, research questions and aim, Section 2 presents the overall methodological approach of the thesis. Case study research is introduced as a means to identify relevant data (2.1), and the methods used to collect qualitative and quantitative data for the LCA are described (2.2 and 2.3). Section 3 presents a review of literature regarding methodological considerations of relevance, background information on the studied product system, and previous research regarding the use of MIBA in roads. Chapter 3.1 begins by elaborating on the LCA approach in general, as well as on the assessment of GHG mitigation and resource efficiency. Following this, Chapter 3.2 presents the different components of the product system within scope, and their implications on LCA methodology. This includes subsections on residues from waste incineration (3.2.1), the treatment of IBA (3.2.2), road construction (3.2.3), landfilling (3.2.4), and primary construction aggregates (3.2.5). The literature review closes in Chapter 3.3, with a synthesis of previous research regarding the environmental assessment of road constructions and the use of MIBA in road construction. This section is complemented with findings from expert interviews. Section 4 addresses RQ1 by providing a detailed description of the case and the LCA model developed to assess it. Section 5 then presents the results of the life cycle impact assessment, answering RQ2. Results are presented separately for both road scenarios (5.1 and 5.2) and are then compared in Section 5.3. The discussion in Chapter 6, further compares the results of this study to previous research, identifies important variables that influence the environmental performance of MIBA as secondary construction material compared to primary raw material (RQ3), and highlights important methodological limitations. Conclusions, including future research opportunities, are presented in the final Section.

2 Research design, materials, and methods

2.1 Case study research

A single case study is chosen as a research strategy that links the collected empirical data to the research questions posed in the beginning. Rather than a methodological choice, a case study can be regarded as "a choice of what is to be studied" (Stake, 2005, p. 445) and as "an umbrella term for a family of research methods having in common the decision to focus on inquiry around an instance" (Adelman et al., 1997 in Blaikie & Priest, 2019). Yin (2009) defines a case study as an "empirical inquiry that investigates a contemporary phenomenon within its real-life context" (p. 13). The subject of this study is the environmental impacts and benefits of secondary resource use. The particular instance, i.e. the unit of analysis or "the case", is the hypothetical reuse of MIBA in a road construction in the Malmö harbor area as envisaged by Sysav. As such, the unit of analysis is not the company, Sysav, but the phenomena of applying a CE strategy in a specific (real-life) context. Sysav is part of this context. The case study is mainly of descriptive and explanatory nature (Yin, 2009).

A case study approach is deemed suitable based on several reasons. The underlying case constitutes a contemporary event and there is little control over behavioral events (Yin, 2009). It allows for the investigation of contextual conditions, which are particularly pertinent to the phenomenon. The case is further regarded as an extreme case (Yin, 2009): there is pressing demand for an alternative solution for the disposal of large quantities of incineration residues from the Sysav incineration plant. While at the same time, there is demand for construction material in the immediate vicinity of the incinerator. However, the use of incineration residues to substitute virgin material in road construction in Sweden is restricted by law.

According to Flyvbjerg (2006), such atypical cases are rich in information because they "activate more actors and more basic mechanisms in the situation studied" (p. 229). For these cases, from an action-oriented perspective, it is usually more important to conduct an in-depth investigation of the underlying causes and consequences of a problem rather than describing its symptoms and the frequency of their occurrence (Blaikie & Priest, 2019; Flyvbjerg, 2006). Finally, the case was also selected due to the significant practical importance and because access to data was warranted by the partner organizations.

The case study approach served to identify relevant data and as "a mode of organizing data in terms of some chosen unit" (Goode and Hatt, 1952, p. 339 in Blaikie & Priest, 2019). In this context, a particular strength of case studies is that they allow for the inclusion of multiple sources of evidence. In the thesis, different methods were used to collect data. Data types include primary (e.g. interview summaries), secondary (e.g. material intensities measured and provided by partner organizations), and tertiary (e.g. journal articles) in both, qualitative and quantitative form.

Qualitative data to describe the case study was mainly collected in the form of interviews and was analyzed by developing a detailed case description. Quantitative data was collected by means of data questionnaires, interviews, and from LCA databases. This was analyzed through an LCA performed with the SimaPro v3.5 software. The methods to collect data are explained in more detail in the following sub-sections. LCA methodology is elaborated as part of the literature review in Section 3.1 including a focus on the assessment of climate impacts and resource efficiency. The LCA model developed in the course of the thesis is then explain in detail in Section 4.2.

2.2 Case study quality

Yin (2009) argues that the quality of case studies can be judged based on four criteria: construct validity, internal validity, external validity, and reliability.

Construct validity means that correct operational measures are established for the studied concepts by selecting "specific types of changes that are to be studied (in relation to the original objective of the study)" and demonstrating "that the selected measures of these changes do indeed reflect the specific types of changes that have been selected" (Yin, 2009, p. 32). Construct validity is strengthened by reviewing and carefully selecting measures of resource efficiency and the GHG mitigation potential in LCA methodology as part of the literature review and by validating these measures with experts. Further strategies to improve construct validity that were implemented in this study include a review of the case study description by key informants, using multiple sources of evidence, and a detailed documentation of the research approach and data sources. The case study description and developed LCA model were reviewed by Sysav and RISE.

Internal validity is relevant for explanatory studies only and means that causal relationships are established during data analysis (Yin, 2009). Two strategies are employed to support internal validity in this study: pattern-matching and explanation building. This includes an in-depth analysis of the quantitative data guided by LCA methodology (e.g. different sensitivity analyses) in combination with a detailed case study description to further facilitate the interpretation of the results (explanation building). In addition, the results of the study were interpreted in the context of previous studies identified during the literature review (pattern-matching).

External validity is concerned with the "domain to which a study's findings can be generalized" (Yin, 2009, p. 33). While a single case study does not allow for any generalization based on statistical inference, its strength is that it provides detailed and "close to reality" (Flyvbjerg, 2006, p. 223) knowledge on a phenomenon (Yin, 2009). Through this, it can contribute to knowledge accumulation and scientific development by "the force of example" (Flyvbjerg, 2006, p. 229). Therefore, a different understanding of generalizability based on judgement and not on statistical tests is required. Stake (2005) refers to this as "natural generalization" based on identifying similarities between issues and reoccurring patterns. External validity is then established by "making judgements on the basis of knowledge of the characteristics of the case and the target population" (Blaikie & Priest, 2019, p. 187). Furthermore, R.K. Yin (2009) argues that "analytical generalization" as opposed to statistical generalization can be achieved by generalizing a particular set of results to a broader theory. For example, to the proposition made in the introduction that secondary material use leads to resource efficiency and climate benefits.

Finally, reliability requires that the study can be repeated with the same results. This is achieved by a detailed documentation that makes transparent what data was collected, and how it was collected and analyzed. This was particularly facilitated by using the SimaPro LCA software.

2.3 Interviews

Interviews are regarded as an important source of information for case studies (Myers, 2013; Yin, 2009). They were used to collect different forms of information:

- Qualitative information on the case study and its narrow context. This included interviews with Sysav employees, the City of Malmö, and Swerock AB.
- Qualitative information on the wider context of the case study. For example, interviews with practitioners in the field of bottom ash recycling in Germany and Denmark.

- Qualitative descriptions of processes to develop the LCA model in SimaPro and quantitative data to build the life cycle inventory. These included employees from Sysav and AB Sydsten, a company that supplies construction aggregates from a local quarry.
- Information to verify assumptions made in the LCA. This included experts in road construction, waste incineration, bottom ash recycling, and environmental impact assessment.
- Interviews with LCA practitioners to discuss and review the developed LCA model.

Interview partners were identified based on recommendations by partner companies, during the literature review, and based on snowballing. Participants were considered to be experts when they held specific practice-oriented knowledge or experience relevant for the case study (Bogner et al., 2014). There was no uniform approach to the interviews. They ranged from unstructured, e.g. spontaneously talking to people during site visits to the Sysav bottom ash treatment plant, to semi-structured, e.g. interviewing experts on issues related to the LCA methodology. Written notes were taken during interviews which were then summarized and structured in an Excel sheet. Although data surveys would have been a more suitable method to collect quantitative data in some instances, delays in responses made it necessary to also collect quantitative information in the form of interviews. See Appendix II for a comprehensive list of people interviewed.

2.4 Data questionnaires, documents, and site visits

Further data was collected by using data questionnaires, reviewing company documents, and by conducting site visits. Several questionnaires were sent to Sysav and AB Sydsten to gather both quantitative and qualitative data on the product systems in view (see Table 12 in Appendix IV). Data questionnaires were usually followed up with a short interview. Further data was derived from documents such as company annual reports and previous studies conducted by or in collaboration with Sysav. Finally, several site visits to the bottom ash treatment facility operated by Sysav were conducted. They served to gain an adequate understanding of the studied product system, for example, with regard to the relevant environmental aspects and impacts and material flows.

3 Literature review

3.1 Environmental and Life Cycle Assessment

The assessment framework is developed based on LCA methodology. LCA is an environmental system analysis tool based on the concept of life cycle thinking. Environmental system analysis stems from the broader field of system analysis and is concerned with its application in evaluating the interplay between anthropogenic systems and their environments. Its goal is to provide knowledge to make more sustainable decisions (Moberg, 2006). There is a large number of different tools available to assess anthropogenic impacts on the environment (Finnveden & Moberg, 2005). LCA is special in that it focuses on product systems (Yang, 2019). A product system describes "any good, service, event, basket-of-products, average consumption of a citizen, or similar object that is analyzed in the context of the LCA study" (JRC-IES, 2010a, p. 23). LCA methodology has been continuously developed over the past decades and is now widely applied in practice (Finnveden et al., 2009).

The concept of life cycle thinking evolved during the 70s and 80s and helps to conceptualize environmental challenges from a systems perspective (Blomsma & Brennan, 2017). LCA can be seen as one practical realizations of life cycle thinking (Mont & Bleischwitz, 2007). It is an environmental system analysis tool that is used to quantify all relevant emissions and resource consumption and the resulting health and environmental impacts and depletion of resources from a product system. With regard to products and services, this means that all environmental aspects and potential impacts are considered throughout all life cycle stages from cradle to grave. Environmental impacts in LCA are understood as the "potential impact on the natural environment, human health or the depletion of natural resources, caused by the interventions between the technosphere and the ecosphere" (JRC-IES, 2010a, p. 23) such as emissions, resource extraction and land use.

The method is internationally standardized by the ISO 14040 and 14044 standards, which provide principles and frameworks, and requirements and guidelines to perform LCA studies, respectively. Specific guidance for this study is provided by two sources, Tillmann and Baumann (2004) and the International Reference Life Cycle Data (ILCD) Handbook (JRC-IES, 2010a).

According to ISO 14040, an LCA study includes four stages: goal and scope definition, inventory analysis, impact assessment, and interpretation (see Figure 3). In the following, a short introduction to the steps required in each stage is given. A particular focus lies on reviewing the relevance of the two general types of LCA studies, attributional and consequential LCA, in relation to the studied topic, and the choice of the right modelling approach and indicators to assess the GHG mitigation and resource efficiency potential of secondary resource use.

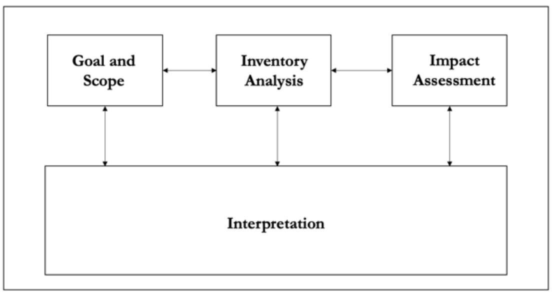


Figure 3: The life cycle assessment framework proposed by the International Organization for Standardization.

3.1.1 Goal and scope definition

In the goal and scope definition stage the product that is to be studied, the purpose of the study, and its context are defined. According to ISO 14040, information on the intended application, the audience, and the reason for doing the study are to be included. The goal and scope needs to be consistent with the intended application. Key decisions to be made in this phase involve the setting of system boundaries, the selection of a functional unit, the types of environmental impacts to be included, and the level of detail sought for. In essence, the goal and scope definition needs to answer the following question: "Who wants to know what about what for what reasons?" (Baumann & Tillman, 2004)

Environmental impacts are related to a specific function of a product system. It is necessary to express this function in quantitative terms, the functional unit. It serves as a basis for calculations. A reference flow is the quantified amount of a product that is needed for the product system to provide the performance that is described by the functional unit. All modelled flows are related to the reference flow. The functional unit is of particular importance if the comparison of two product systems is the aim (Baumann & Tillman, 2004).

Setting system boundaries determines which processes are included in the study. They are defined in several dimensions, including:

- in relation to the natural system,
- geographical boundaries,
- time boundaries, and
- boundaries with the technical systems including cut-off criteria (in relation to, e.g., production capital) and in relation to other product life cycles. The latter requires allocation procedures (Baumann & Tillman, 2004).

The goal and scope definition for the LCA carried out as part of this thesis is presented in Section 4.2.1. The goal of providing a comparison between two scenarios of the same road was set by the Commissioner, Sysav Utveckling AB. Scoping decisions, e.g. regarding the functional unit and system boundaries, were made on the basis of literature research and consultations with experts and in close dialogue with the Commissioner.

Allocation

Allocation is required when a process has more than one function and leads to more than one product or service that are part of multiple life cycles (Ekvall, 2020). These processes are called multifunctional. Two types of multifunctional processes exist: multi-output, i.e. a process that leads to multiple products, and multi-input, such as waste treatment processes with several different input streams (Ekvall & Tillman, 1997). Multifunctionality requires that the environmental impacts of a system are assigned to the functions of that system in an appropriate share. Allocation is also required when recycling leads to energy or material being used in more than one product. This is called open-loop recycling (Ekvall & Tillman, 1997).

Two general allocation procedures are used: partitioning and methods to avoid allocation by either system sub-division or system boundary expansion. The ILCD Handbook defines partitioning as dividing "the input or output flows of a process or a product system between the product system under study and one or more other product systems." (JRC-IES, 2010a). System expansion is understood as "adding specific processes or products and the related life cycle inventories to the analyzed system … [to] make several multifunctional systems with an only partly equivalent set of functions comparable within LCA" (JRC-IES, 2010a).

The ISO standard provides a general hierarchy of approaches to multifunctional processes:

1. Avoiding allocation as much as possible through system sub-division or system boundary expansion.
2. If allocation cannot be avoided, partitioning between a system's different products or functions should be based on underlying physical relationships.
3. As a solution of last resort, allocation can be based on other relationships, for example, economic ones (Ekvall & Tillman, 1997).

Two mathematically equivalent variants of system boundary expansion exist. On the one hand, system enlargement serves to make different multifunctional systems comparable in a comparison study. This is done by adding a function and the corresponding inventory to the system that lacks this function (JRC-IES, 2010a). On the other hand, the "avoided burden approach" (or "substitution") assumes that the additional or not required function "substitutes" the provision of that same function by other means, which thus is avoided. In this case, the environmental impacts of the "avoided burden" is subtracted from the product system in view. This is also called "crediting" (Finnveden et al., 2009).

While the ISO standard provides a ranking of allocation principles, the appropriate approach also depends on the aim of the study, the characteristics of the multifunctional process, and available data (Baumann & Tillman, 2004; Ekvall & Tillman, 1997; JRC-IES, 2010a). Economic allocation is considered a valid basis for partitioning because the economic value of a good or service is seen as a proxy for their expected contribution to the profit from the analyzed process indicating its relative importance (Ekvall, 2020). It also serves to distinguish waste from other outputs (Ardente & Cellura, 2012; Goedkoop et al., 2016). It is often used in practice due to its simplicity and due to other allocation approaches not being feasible, for example, because of a lack of data or the resources necessary to generate such data. On the contrary, the main drawback of using economic allocation is that prices fluctuate. This may influence results significantly (Goedkoop et al., 2016). Other allocation keys such as mass have the advantage that they remain relatively constant over time (Chen et al., 2010). The ISO standard requires a sensitivity analysis if several procedures seem applicable in order to show the impact of the selected procedure on the results (JRC-IES, 2010a).

Different approaches to system boundary expansion and partitioning were considered in this study. System enlargement is applied to consider the alternative disposal of MIBA in a landfill. Allocation based on economic values is used to partition the environmental burdens from the sorting of IBA between MIBA and scrap metals, which is an open loop recycling process. The system boundaries considered in this study are elaborated in detail in Section 4.2.1.

Consequential and attributional modelling

It is argued that there are two general types of LCA studies. While Tillmann and Baumann (2004) refer to these as either "change oriented" studies or "accounting style" studies, the ILCD Handbook and the general scientific discourse refer to these as consequential and attributional LCA (CLCA and ALCA) (Brander et al., 2009; Ekvall, 2020; Ekvall & Weidema, 2004; Finnveden et al., 2009; Goedkoop et al., 2016). The ILCD Handbook defines consequential modelling as a life cycle inventory "modelling principle that identifies, and models all processes in the background system of a system in consequence of decisions made in the foreground system" (p.21). ALCA is defined as a "modelling frame that inventories the inputs and output flows of all processes of a system as they occur" (JRC-IES, 2010a, p. 21). This categorization into two types of LCAs originates from the question regarding the type of input data that is to be used and how to deal with allocation problems (Ekvall, 2020).

In essence, the two approaches to modelling answer different questions (Ekvall, 2020): While ALCAs investigate questions of the type "What environmental impact can be associated with this product?" (p.78) CLCA deals with "What would happen if…" (Baumann & Tillman, 2004, p. 78) questions. Therefore, ALCA aims to "describe the environmentally relevant physical flows to and from a life cycle and its subsystems" (p. 3) and CLCA aims "to describe how environmentally relevant flows will change in response to possible decisions" (Finnveden et al., 2009, p. 3). It is argued that ALCA studies are comparative and retrospective whereas CLCA studies are comparative and prospective (Baumann & Tillman, 2004).

There are significant differences between CLCAs and ALCAs. Above all, the choice affects the selection of system boundaries and with this, the allocation approach. While both types consider the same life cycle stages, CLCA studies intend to show the total effect of a marginal change in the output of a product. This includes all processes and material flows which are directly or indirectly affected both within as well as outside the life cycle of the product that is under investigation. Indirect effects include, for example, the use of constrained products, negative and positive feedbacks (rebound effects), other market effects, or behavioral effects (Brander et al., 2009; Ekvall, 2020). On the contrary, ALCA considers only processes within the life cycle of the product that is studied. Environmental benefits and further indirect effects outside the life cycle of the product are not included (Ekvall, 2020).

As a result, a consequential modelling approach may cause overlapping product systems leading to "double-counting" of emissions (Brander et al., 2009). Moreover, because CLCA tries to quantify changes in emissions relative to the marginal production scenario and does not quantify the absolute existing emissions from a products life cycle, the results can include negative inventory flows or even yield negative overall environmental impacts (JRC-IES, 2010a). This may be the case if a change in the production level leads to a reduction in emissions from other product systems that is greater than the actual emissions from within the product's life cycle (Brander et al., 2009). In this case, there is a net benefit from the production of the analyzed system because the overall impact is overcompensated by the avoided burden from the product's co-functions (JRC-IES, 2010a).

While this feature makes CLCA unsuitable for certain purposes such as consumption-based carbon accounting (Brander et al., 2009), CLCA is considered to be the more accurate approach to modelling (Ekvall, 2020). It can provide precise information about the consequences of, for example, buying a product and, therefore, is suited to assess alternative options that may be produced or implemented in future (Ekvall & Weidema, 2004). As such, it is often argued that it is the right modelling approach to provide information for decision making (Finnveden et al., 2009).

Two further differences between ALCA and CLCA relate to the use of input data and the general allocation approach. While the use of average data is common for attributional studies, marginal data is used for CLCA studies. The former represents "the average environmental burdens for producing a unit of the good and/or service in the system" (p. 3). The latter shows "the effects of a small change in the output of goods and/or services from a system on the environmental burdens of the system" (Finnveden et al., 2009, p. 3). In ALCA, emissions are typically allocated by partitioning, while CLCA uses system boundary expansion.

Although CLCA is supposed to be more accurate and the right approach to model the consequences of decisions, it comes with significant limitations, including the availability of data and large uncertainties. It is also argued that results of CLCAs are less easily comprehensible for decision makers and sensitivity analyses often leave them without clear conclusions (Ekvall, 2020). System expansion is a difficult process and depends on the judgement of the practitioner.

Often it is difficult or even impossible to find a replacement process for substitution, making other allocation approaches necessary (Ekvall, 2020).

As a consequence, LCA studies are often hybrids, between a consequential and attributional modelling approach (Brander et al., 2009). For example, LCA studies based on attributional principles of modelling but with expanded system boundaries to account for potential benefits from recycling or the reuse of waste products (JRC-IES, 2010a). In that case, the major differences between a CLCA and a ALCA are reduced to the input data (Finnveden et al., 2009).

The thesis aims to support decision-making. As such, a consequential approach would be the more suitable approach. However, due to the limitations of CLCA described above, an attributional modelling approach with expanded system boundaries is chosen. How the principles of ALCA are operationalized in this study is elaborated in Section 4.2.

3.1.2 Inventory analysis

In the life cycle inventory phase, a system model based on the requirements set in the goal and scope definition is developed. This phase includes three major activities:

1. Constructing a flow model of the activities and flows between activities included in the product system based on the defined system boundaries.
2. Data collection on the inputs and the outputs of the product system. This covers raw materials, products, solid wastes, and emissions to air and water from the product over its entire life cycle.
3. Calculation of the emissions and resource use in relation to the functional unit (Baumann & Tillman, 2004).

This is the most laborious phase of the LCA. A system model was developed, discussed with experts, and refined in several iterations. Data was collected during different measuring campaigns from Sysav and other companies. Calculations where aided by the SimaPro LCA software. The system models and a description of the inventory is presented in Section 4.2.

3.1.3 Impact assessment

The goal of the life cycle impact assessment (LCIA) stage is to describe and evaluate the significance of the potential environmental impacts of the investigated product system (Finnveden et al., 2009). It serves two general purposes: to aggregate inventory results in fewer parameters and to translate environmental loads described in the life cycle inventory into information regarding impacts on entities that should be protected (Baumann & Tillman, 2004). These entities are called Areas of Protection (AoP). There is general acceptance that they include human health, the natural environment, and natural resources (Finnveden et al., 2009).

According to the ISO standard, the LCIA comprises of two mandatory (1 and 2) and two optional (3 and 4) steps:

1. Classification: assigning inventory data to impact categories they contribute to.
2. Characterization: multiplying inventory data with a substance specific characterization factor in order to get the relative contribution of the resource consumption or emission to each environmental impact category.
3. Normalization: multiplying characterization results with normalization factors that represent the total inventory of a reference to obtain dimensionless LCIA results.
4. Weighing (or valuation): multiplying results from normalization with weighing factors indicating the distinct relevance of different impact categories to further aggregate information into a single value (JRC-IES, 2010a). Weighing necessarily includes value

choices, which is why the ISO standard does not allow weighing in public studies that compare different products (Goedkoop et al., 2016).

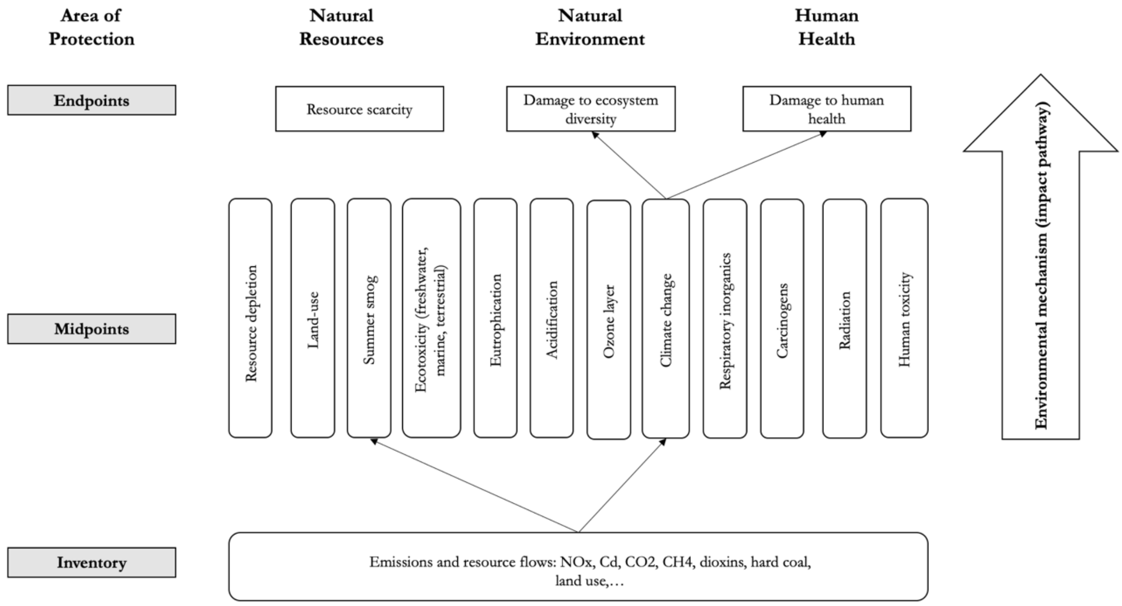


Figure 4: Schematic representation of the environmental mechanisms from inventory to endpoints. Normalization and weighing are not displayed.

Source: Prepared based on JRC-IES, 2010a, p. 108.

Impact categories are differentiated into two groups: midpoint and endpoint indicators. Midpoint methods are defined early in the cause-and-effect chain between the emission or resource extraction and the AoP (see Figure 4). Endpoint methods are defined at the level of the AoP (Finnveden et al., 2009). In general, characterization modelling at midpoint is further developed than at endpoint, which was "in clear need for further development" (Hauschild et al., 2013, p. 684) during the last comprehensive assessment of LCIA methods.

The SimaPro software offers a range of the most common and most advanced LCIA methods. In this thesis, two of these "ready-made" LCIA methods were used. The justification for the choice of the method is provided in Section 4.2.3. In the following two sub-sections, the assessment of the GHG mitigation potential and resource efficiency in LCA methodology is further elaborated.

Assessing the climate change mitigation potential

The term climate change describes the human induced warming of the Earth's climate system. The major cause for climate change is an increase in the concentration of GHGs in the atmosphere leading to an increase in radiative forcing (Levasseur, 2015). The major anthropogenic GHGs are carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4), and halocarbons. Their major sources include the combustion of fossil fuels, land use change, agriculture, and industrial processes (Levasseur, 2015). Climate mitigation, on the other hand, means a human intervention that reduces the source or enhances the sink of GHGs (IPCC, 2018). The assessment of the GHG mitigation potential in the case at hand has in view potential reductions of GHG emissions by using secondary material in lieu of primary material in road construction.

LCA is a common method to quantify the GHG emissions over the full life cycle of a product and to assess their potential impacts (Plevin et al., 2014; see e.g. Nußholz et al., 2019). The Global Warming Potential (GWP) characterization method developed by the Intergovernmental Panel on Climate Change (IPCC) is used as a proxy to quantify climate related impacts (Forster et al., 2007; Plevin et al., 2014). It is a midpoint method that is chosen early in the environmental mechanism, i.e. at the level of radiative forcing (Levasseur, 2015). The effects on the climate are defined as the weighted sum of the life cycle emissions of major GHGs per functional unit measured in kg-CO_2 eq. Three versions of the method with different time horizons exist: 20, 100, and 500 years. The most widely used is the 100-year timeframe (Hauschild et al., 2013).

The choice of the modelling approach influences how the mitigation potential is calculated and how results need to be interpreted. In the framework of ALCA, the difference between the GWP impact of the incumbent product system and an alternative product system is often interpreted as a proxy for the climate mitigation potential of the investigated product system (Plevin et al., 2014). ALCA is the appropriate choice to calculate consumption-based indicators such as the carbon footprint of a product because it only considers GHG emissions from within the investigated product system (Brander et al., 2009; Čuček et al., 2015). However, it is argued that ALCA should only be used to obtain a qualitative understanding of the product system, for normative analyses, or to conduct sensitivity analysis (Plevin et al., 2014).

On the other hand, decision support is best provided by CLCA that anticipates the effects of a decision (Plevin et al., 2014). In this case, the mitigation potential is approximated by the GWP impact caused by the potential change minus the GWP impact of a baseline scenario without the change (Plevin et al., 2014). CLCA is not suitable for carbon accounting because it does not quantify the absolute existing emissions due to the system expansion approach to allocation that leads to possible double counting of emissions (Brander et al., 2009).

In this thesis, an attributional modelling approach with extended system boundaries is applied. GWP is seen as a proxy for climate related impacts and, therefore, as an indicator for the GHG mitigation potential of the road scenario including MIBA, compared to a conventional road scenario. The limitations of using an ALCA to approximate the mitigation potential needs to be acknowledged in the interpretation stage and is further discussed in Section 6.2.

Assessing resource efficiency

The transition to a resource efficient economy requires indicators to measure resource consumption and associated impacts. A variety of indicators have been developed in science and policy contexts addressing different levels of economic activity, from single processes and products to the assessment of resource efficiency at the national and international level (Schneider et al., 2016).

Resource efficiency is generally understood as "achieving higher outputs with lower inputs" (IRP, 2020, p. 10) and, therefore, is the "relation of economic output (added value) and required resource input" (Schneider et al., 2016, p. 181). The European Commission uses a somewhat broader definition: "using the Earth's limited resources in a sustainable manner while minimizing impacts on the environment" (EC, n.d.). This definition highlights that resource efficiency includes 1) the use of natural resource itself as well as 2) the impact their extraction and use has on the natural environment. The focus of resource efficiency assessment is therefore on the two AoPs natural environment and natural resources.

The evaluation of resource efficiency further depends on the understanding of what natural resources entail (Schneider et al., 2016). In general, an anthropocentric perspective is taken in

LCA. AoPs are considered entities "of value to human society and what needs to be sustained for achieving human welfare" (Schneider et al., 2016, p. 183). Natural resources can then be defined as "objects of nature which are extracted by man from nature and taken as useful input to man-controlled processes, mostly economic processes" (Haes et al., 2003 in Huysman et al., 2015, p. 69). The AoP natural resources therefore includes mainly input-related environmental interventions (e.g. abiotic resource extraction, biotic resource extraction, and land use). How these interventions are categorized into impact categories varies between different LCIA methodologies. For example, into resource depletion water, land use, and resource depletion mineral, fossil, renewable in the ILCD 2011 Midpoint+ method (Goedkoop et al., 2016).

Based on this understanding, resource efficiency can be defined and measured at two levels: at the inventory level (i.e. the relation between benefits and the inventoried flows) and at impact level (i.e. the relation between benefits and environmental impacts) (Huysman et al., 2015). Inventory flows may be emissions (e.g. CO_2), natural resources (e.g. water), industrial resources (e.g. diesel), or waste-as-resource (e.g. secondary material). In order to get from inventoried flows to environmental impacts LCIA methodologies are used. Resource efficiency may then be expressed as the benefit over the impacts calculated though either midpoint or endpoint characterization methods, or further aggregated into a single score (see Table 1). Indicators can further be subdivided into resource efficiency indicators (also called resource efficiency indicators in "sensu stricto") and emission efficiency indicators (also called resource efficiency indicators in "sensu lato") (Huysman et al., 2015).

Table 1: Systematized framework of resource efficiency indicators at life cycle level presented by Huysman et al. (2015).

Level 1		Level 2		
Resource efficiency at inventory level	***Emission efficiency at inventory level***	***Resource efficiency at impact level***	***Emission efficiency at impact level***	***Overall efficiency at impact level***
Benefit over (kg) resources in life cycle	Benefit over (kg) emissions in life cycle	Benefit over (e.g. land use) impact in life cycle	Benefit over (e.g. GWP) impact in life cycle	Benefits over single score impact in life cycle

Source: Prepared based on Huysman et al., 2015, p. 71.

However, the integration of resource efficiency into LCIA methodology is not yet well developed (Schneider et al., 2016). In comparison to emission related impact categories, resource-related environmental impact methods are not mature and further research and consensus building is needed (Hauschild et al., 2013; Schneider et al., 2016). It is particularly highlighted that impact categories relating to resource depletion are not well defined and that there is a lack of understanding about what the AoP natural resources is and how it can be delimited from the AoP natural environment (Hauschild et al., 2013; JRC-IES, 2010b).

A comprehensive assessment of the impacts of natural resource use needs to be twofold: On the one hand, the consequence of resource extraction on the availability of the resource, and, on the other, impacts caused by the extraction process on the natural environment need to be assessed (Schneider et al., 2016). For an evaluation of the former, all relevant aspects of availability need to be considered. This includes the physical availability as well as access to the resource. While methods to assess the impacts of the extraction process are fairly well developed (i.e. emissions related impact categories), the evaluation of the removal of natural resources from nature are not well researched. Natural resources are only considered for three interventions: extraction of abiotic resources, extraction of biotic resources, and the allocation of land areas to human-controlled processes. Within current assessment methodologies, a distinction is usually made into the categories land use, water use, and use of biotic and abiotic resources. Land and water can be included in the category of abiotic resources, but a separate assessment is often done (Schneider et al., 2016).

Special attention is given to the depletion of abiotic and biotic resources. Depletion means the process of decreasing the abundance of resources. Abiotic resources are "chemical elements and minerals from the Earth's crust" (p. 186) and classified as non-renewable, meaning that they do not regenerate within a lifetime of a human. Biotic resources are living organism such as animals or trees and are classified as renewable. Biotic resources can deplete when their use is higher than their replenishment rate (Schneider et al., 2016).

A comprehensive assessment of biotic resource depletion needs to entail their availability as well as the impact that their removal from nature has on the natural environment. This includes an assessment of biodiversity loss as well as the loss of certain life-support functions that biotic resources fulfill. However, this is challenging due to missing inventory and applicable impact assessment data. Until now, the assessment of biotic depletion is not included in most LCIA methods and only rudimentarily in some (Schneider et al., 2016).

The assessment of abiotic depletion focuses on their availability only. The ILCD Handbook proposes a categorization of impact assessment methodologies into three groups of methods: those that are based on the inherent properties of materials, methods that are based on diminishing geological stocks, and methods that are based on the future consequences of resource extraction (Schneider et al., 2016). The first category includes thermodynamic resource indicators such as the Cumulative Energy Demand (CED) (Huysman et al., 2015; Schneider et al., 2016).

The CED uses energy as the basis for characterization. All energy embodied in the natural resources that are required along the entire life cycle of a product system is considered (Huysman et al., 2015). Thermodynamic resource indicators are located at the first step in the environmental mechanism between resource use/emissions and the AoP. It is argued that they are suited to assess the depletion of natural resources (Huysman et al., 2015; Risse et al., 2017) and that they are indicative for other environmental impacts because energy demand is an important driver for different environmental loads (Chen et al., 2010). A further advantage is that they provide a single score that facilitates comparisons (Risse et al., 2017).

The second category of methods to assess the depletion of abiotic resources is based on their reserves in nature and the annual extraction rates. Examples include midpoint methods such as the "abiotic depletion potential." These methods provide information regarding the geological availability of resources and the static lifetime of their stocks (Schneider et al., 2016). The final category addresses the damage of resource use. These methods are based on, for example, increasing future costs or energy requirements resulting from the extraction of natural resources. Exemplary LCIA methods include endpoint indicators such as "damage to resource availability" in the ReCiPe 2016 method (Schneider et al., 2016).

In order to assess resource efficiency in the Malmö case, a combination of different indicators is applied. Emission and resource related impact categories included in the ILCD 2011 Midpoint+ method are used to assess the depletion of abiotic resources as well as the impact that the extraction process has on the natural environment. This is complemented with a thermodynamic indicator, the CED, in order to give a more comprehensive picture. A more detailed description of the impact assessment conducted in this study is provided in Section 4.2.4.

3.1.4 Interpretation

The interpretation stage includes the assessment of results from previous stages in order to draw conclusions and give recommendations (Baumann & Tillman, 2004). Results are evaluated in relation to the goal and scope of the study. According to ISO 14044, the interpretation typically involves the following analytical aspects:

- Identification of significant issues, including significant methodological choices as well as the main contributors to inventory results.
- Completeness, sensitivity, and consistency tests. For comparative assertions, such tests should include different assumption scenarios on data and methods such as reasonably best and reasonably worst cases for inventory data values, parameters, relevant system properties, allocation approaches, and the mix of superseded processes used in substitution.
- Drawing conclusions and giving recommendations considering limitations (JRC-IES, 2010a).

3.2 Bottom ash utilization in road construction

In order to conduct an LCA, knowledge on the investigated product systems is necessary. This section provides context with regard to the system's components and their assessment using LCA methodology. This includes information on residues from waste incineration and their treatment, road construction, landfilling, and primary construction aggregates. Landfilling is introduced because it is assumed to be the alternative treatment for MIBA in the specific case. Primary construction aggregates such as crushed rock is the material substituted by MIBA in the road. Section 3.3 then presents a synthesis of previous research regarding the environmental assessment of secondary material use in road construction with a focus on MIBA. The literature is complemented with findings from multiple expert interviews in the field of waste incineration, bottom ash treatment and utilization, environmental assessment, and road construction.

3.2.1 Residues from waste incineration

Waste incineration with energy recovery is the most common thermal waste treatment technology applied. Globally, more than 1000 facilities exist in over 40 countries treating approximately 10% of global MSW (Kahle et al., 2015). From the incineration process, different solid residues remain including bottom ash, fly ash, and flue gas residues (see Figure 5).

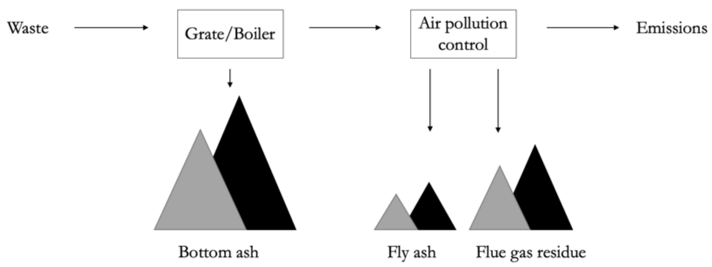


Figure 5: Different solid residues remaining from the incineration process.

Source: Prepared based on Zero Waste Europe, 2019, p. 3.

The mass of IBA produced corresponds to approximately 20% of the mass of the incinerated waste and to 5-10% of its volume. As a result, the concentration of metals is 4-6 times higher in IBA than in the source material. Their concentrations vary between 4-10% for Fe metals and 2-5% for NF metals (Hykš & Hjelmar, 2018; Respondent 1, personal communication, February

4, 2020). For metals such as copper, the concentration is comparable to naturally occurring ores (Arm et al., 2017). The remaining mass of IBA is composed of a mineral fraction including non-combusted materials like soil minerals and melt, sintered products of different mineral composition, and waste glass (80-85%) as well as a small fraction of unburned organic material (<1%) (Hykš & Hjelmar, 2018).

Its high metal content and good mechanical properties make IBA a resource from which valuable metals can be extracted and which can replace increasingly scarce resources in the construction industry (Hykš & Hjelmar, 2018). The economic potential of recycling metals from IBA has made this process common practice in Europe (Blasenbauer et al., 2020). Modern sorting facilities are capable of recovering 85–95 wt% of the Fe metals and 40–75 wt% of the NFe metals depending on the technology and if secondary use of the mineral fraction is sought for (Blasenbauer et al., 2020; Respondent 1, personal communication, February 4, 2020).

3.2.2 Treatment of incinerator bottom ash

The secondary use of IBA in the civil engineering sector requires its pre-treatment in order for the bottom ash to fulfill environmental and mechanical properties placed on construction materials. For its use as unbound aggregate in road constructions, this includes the drying of IBA before sorting in case the output of the incineration plant is wet, a sorting process, and aging (also called "weathering").

Drying is achieved by stockpiling IBA in open air conditions. This process does not require any further inputs and takes up to three months. It facilitates a more efficient sorting of materials. The sorting process includes the extraction of organic and oversize materials, Fe and NF metals, as well as crushing in order to get a more homogeneous aggregate. Subsequent to the sorting process, the MIBA is stockpiled in open-air conditions again in order to age for 4-6 months. Multiple chemical reactions take place during aging, including the oxidation of elemental metals, precipitation of salts, corrosion of glass, slaking of lime, and carbonation, among others. Most notably, carbonation leads to a reduction of the pH from about 12 to 9-10.5, to an immobilization of heavy metals and reduced leaching if used in constructions, and further improved geotechnical properties (Hykš & Hjelmar, 2018).

Carbonation also leads to the uptake of CO_2. The amount of CO_2 sequestrated in MIBA was measured to be 37 kg of CO_2 per ton of bottom ash at the case study company (Johansson & Lönnebo Stagnell, 2016). Potential benefits from carbonation have been analyzed as part of environmental assessments of secondary use of construction and demolition waste in road construction (e.g. Butera et al., 2015), however, so far, it was disregarded in studies on MIBA. Carbonation will therefore be included in this study.

While the treatment of IBA contributes to the immobilization of heavy metals contained in MIBA, the high content of salt and trace elements is still considered the main potential risk to human health and the environment stemming from the use of MIBA in civil engineering (Astrup et al., 2016). In particular, when used in unbound form, this raises concerns about the potential leaching of heavy metals into percolating rainwater and subsequently into the soil, and ground and surface water (Hykš & Hjelmar, 2018; Lynn et al., 2018). Therefore, the treatment of IBA needs to be geared towards minimizing the leaching of contaminants, and the handling of IBA and MIBA needs to be done in a way so as to minimize the exposure by direct contact and ingestion – particularly regarding untreated IBA (Hykš & Hjelmar, 2018). This includes the handling of MIBA during road construction and possibly implies different resource use and emissions from the road construction phase.

3.2.3 MIBA in road construction

Roads are one of the most significant types of infrastructures we use. They consume extensive amounts of raw materials and activities throughout their entire life cycle cause significant emissions. However, there is no agreement on the categorization of activities into life cycle phases and studies identify a different number of phases ranging between three to eleven (Santero et al., 2010). For example, while Olsson et al. (2006) consider the resource consumption and emissions from five stages: production, transportation, location in road structure, use, and disposal, Birgisdóttir et al. (2006) only analyze the design, construction, and operation and maintenance.

A categorization that was found to be useful for this study is provided by Santero et al. (2010), who identifies five general phases based on a review of road LCA studies: materials production, construction, use, maintenance, and EoL.

- **Materials production**: Includes the production and transportation of materials used in the road structure; from the extraction of raw materials (e.g. IBA or granite) to the finished construction material (e.g. MIBA or crushed rock). This phase involves energy intensive activities like crushing and sorting.
- **Construction**: This phase consists of heavy-duty earthworks like levelling and the placement of the construction materials in the road.
- **Use:** Includes all activities that take place during the lifetime of the road, with the exception of maintenance.
- **Maintenance:** Activities like winter road maintenance, reconstruction, and rehabilitation during the life of the road.
- **End-of-life:** Activities involved in taking the road out of service. Different scenarios include that the road remains in place permanently or its demolition and either disposal in a landfill and/or recycling of the materials used in the road.

Roads have several characteristics that require special attention in LCA methodology. This includes, for example, the large volumes of materials used, the longevity of the product, and the impact of longevity on the need for repair (Mroueh et al., 2001). Every road is unique with significant variations due to factors such as its geographic location, meteorological and geo-technical conditions, and traffic intensity, among others. For example, the type of terrain may require more or less excavation and the routing has influence on emissions and resource consumptions in the use phase (Stripple, 2001).

Furthermore, the technical composition of a road can vary greatly. The type of road needs to be suitable for the projected traffic flow, be adapted to the land characteristics, and different construction techniques are available (Stripple, 2001). The latter is also influenced by the availability of materials (e.g. natural gravel or bedrock to produce crushed rock) and techniques in a specific geography. As a result, two roads with the same length can have fundamentally different characteristics (Santero et al., 2010). This needs to be taken into account when developing a life cycle model.

As has been mentioned, road structures are heterogenous. On the horizontal axis, they include elements such as different means of traffic like vehicles, bikes and pedestrians, and parking or reserves to separate traffic. Vertically, roads consist of different material layers that vary considerably depending on the traffic load of the road as well as on the road elements included. Typically, the vertical road structure includes a wearing course, a base course, a subbase, and subgrade (see Figure 6) (Birgisdóttir, 2005).

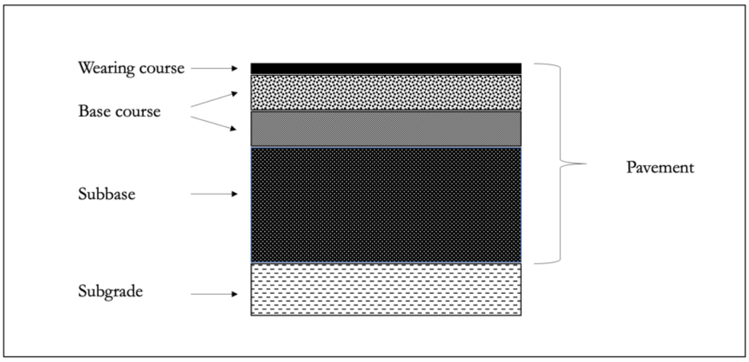


Figure 6: General structure of a road.

Source: Prepared based on Birgisdóttir, 2005, p. 6.

The layers of roads consist of different materials, including bound and unbound materials. While bound material are used in the upper layers, unbound materials are typically used in the lower layers. Bound materials can be concrete or bituminous (Birgisdóttir, 2005). Unbound materials include different primary and/or secondary aggregates that vary in their grain sizes (Arm, 2003).

The use of primary materials in roads such as sand, gravel, and crushed stone predominates. Particularly, in regions and countries like Sweden where there is abundant primary material available (Arm, 2003; Respondent 4, personal communication, December 12, 2019). However, numerous residual materials and industrial by-products are suitable to be used in road constructions. This includes the use of reclaimed asphalt material or fly ash from thermal power plants in the wearing course (Birgisdóttir, 2005) and construction and demolished waste, steel slag, and MIBA in the lower (unbound) layers (Arm, 2003). Interviewees estimated that asphalt layers in Sweden contain between 10-50% reclaimed asphalt, while the base and subbase of roads are almost exclusively built with primary aggregate (Respondent 10, personal communication, March 10, 2020).

After treatment, MIBA has been shown to fulfill the geotechnical requirements to be used as unbound aggregate in road subbases (Arm, 2003; Hykš & Hjelmar, 2018). This also includes the material considered in the case study (Arm et al., 2017). Ongoing studies in Denmark suggest that MIBA could be used in the base course layer of roads as well as in roads with higher traffic load (Hykš & Hjelmar, 2018; Overgaard et al., 2020; Respondent 8, personal communication, March 16, 2020). However, its utilization in the subbase constitutes the common practice so far.

Similarly, after pre-treatment, MIBA fulfills the environmental requirements posed for its use in the subbase of roads in countries such as Denmark, the Netherlands, and Germany (Hykš & Hjelmar, 2018). In focus is the total content and leaching of salts and heavy metals, which, in turn, depends on the material composition, the amount of water passing through the material, and the way the material is laid in the road (Mroueh et al., 2001).

There are strict regulative measures in place to control the use of MIBA as secondary material in civil engineering, particularly for its use as unbound aggregate in road constructions. In most cases, regulations are in the form of quality criteria including limits on the total content of substances and on leaching accompanied by conditions on its use. These include, for example, requirements as to where roads including MIBA can be constructed as well as requirements with regard to construction components and in which structural parts MIBA can be utilized (Saveyn et al., 2014).

In the case of Sweden, limit values for total contents and leaching are defined based on the concept of a “negligible remaining risk” (“ringa risk”). Three scenarios are possible. The risk associated with the use of MIBA can either be below, equal to, or greater than the negligible remaining risk. In the first case, the “free” use without regulatory restrictions outside ecologically sensitive areas is possible. This means that no monitoring or environmental control is necessary and that the waste can be moved after the life of the construction without further restrictions. Currently, due to the exceptionally strict limit values, this regulation effectively prevents using MIBA freely (Van Praagh et al., 2018). Specifically, the total contents of various heavy metals (except mercury) in MIBA exceed legal limits, and the same goes for the leaching levels of copper (although leaching levels for all other substances are below requirements) (Arm et al., 2017).

In case two and three, i.e. the risk is equal to or greater than the negligible remaining risk, a “non-free” use is still possible under strict conditions. In the former case, the approval by the local government is required while in the latter the approval must be granted by regional authorities. Both cases demand resource intensive impact assessments, as well as control and monitoring measures (Respondent 1, personal communication, February 4, 2020). As a result of the strict requirements, the use of MIBA outside landfills in Sweden is non-existent with the exception of a few test sites and pilot projects (Blasenbauer et al., 2020).

However, because the regulation considers the “free” use of MIBA, it has been pointed out that it neglects the fact that MIBA would only be used within road subbases covered by an impermeable layer that greatly reduces the potential risk posed for human health and the environment (Respondent 1, personal communication, February 4, 2020; Van Praagh et al., 2018). If such factors are incorporated in risk assessments, MIBA would fulfill environmental requirements (Arm et al., 2017; Van Praagh et al., 2018).

MIBA remains a relatively new material used in civil constructions. It is highlighted that its use requires a broader involvement with multiple actors including the public, environmental planners, and individual users (Arm et al., 2017). Furthermore, in order for its reuse in road subbases to be a feasible option in terms of its technical properties and environmental requirements, it is important that the material is treated by sorting out metals and aging, and that it is used only in specific controlled applications under conditions that limit negative impacts from, for example, direct exposure or leaching, and that the material complies with regulations on leaching that are developed based on the real risk that is associated with the specific application scenario (Hykš & Hjelmar, 2018).

Together with general technical requirements for construction materials that MIBA needs to fulfill when used in road subbases, environmental limit values and conditions for structural components need to be taken into account when modelling a street in LCA. The case study company and material complies with these requirements as expressed in their strategy on IBA (see Avfall Sverige, 2015). Therefore, in the course of this study, these prerequisites serve as guidance to develop a realistic LCA model. The assessment of such a scenario on a regional level in Malmö strengthens the basis for the afore mentioned actors to take informed decisions.

3.2.4 Alternative treatment of MIBA: landfilling

In the case that MIBA cannot be used in the civil engineering sector, it can either be used as construction material in a landfill or can be directly disposed of in a landfill. In this thesis, due to the limited demand for landfill construction material in the region of Skåne, Sweden, its direct disposal in a landfill is assumed to be the more realistic alternative.

MIBA is usually disposed of in a special compartment for slag within a sanitary landfill. This compartment can also be considered a type of inorganic landfill, i.e. one that receives waste with a low carbon content. Necessary features of a sanitary landfill include a boundary and base sealing, a leachate collection and treatment system, and a gas collection and utilization system (Bilitewski & Härdtle, 2013).

Emissions from landfills include direct and indirect emissions. Direct emissions result from processes such the transportation and handling of waste as well as from the waste itself. Over time, waste is transferred either into leachate or landfill gas. While gas production occurs for several decades, leachate is produced continuously over an unlimited time period (Doka, 2003). Indirect emissions originate from the production of fuels and materials used on the landfill (Doka, 2003).

Several processes throughout the life cycle of a landfill contribute to the depletion of resources. These include, for example, the fuel consumption for the energy used during the construction, operation, and aftercare, and the demand for infrastructure materials as well as other capital goods such as construction machinery. A further significant resource-related impact is the use of land areas (Doka, 2003).

A typical lifetime scenario for a landfill considered in LCA includes a five-year construction phase, a 30-year operation and filling phase, and 75 years of aftercare including reclamation, leachate treatment, and monitoring (Doka, 2003). However, leachate emissions may occur in small amounts over very long time periods with a large part of the pollutants remaining in the landfill after the first controlled hundred years (Hauschild et al., 2008). If considered that engineered barriers such as sealing sheets have a limited functional lifetime, it can be assumed that the pollutants remaining in the landfill after the first 100 years will be completely released if very long-time horizons are considered (Hauschild et al., 2008). Hence, from an LCA perspective, landfills only postpone potential emissions into the future (Doka, 2003).

The aforementioned makes accounting for landfilling and particularly leaching in LCA methodology a challenging task. Emissions are typically inventoried in a way that assumes that they are released at one reference point in time and geographic region. Temporal, spatial or dose-response information is not included (Baumann & Tillman, 2004). Emissions occurring in the future are therefore also not weighted, they are treated in the same way as short-term emissions (Doka, 2003). To address this problem, the inclusion of different time scenarios is suggested (Hauschild et al., 2008). For example, one surveyable scenario of 100 years and one infinite time period (Finnveden et al., 1995).

This perspective is also important for studies on MIBA utilization that do include leaching, as leaching of heavy metals from MIBA in road structures is also a slow process. However, while it is assumed that leaching from MIBA used in road structures happens all the time, leaching from MIBA in an engineered landfill only takes place once the landfill is closed and leachate treatment stops (Toller, 2008). Taking only a short or surveyable time perspective could lead to underestimating the potential environmental impacts of MIBA leachate, as only a fraction of the total amount of pollutants would be emitted during this period (Finnveden et al., 1995).

Developing a leaching model for a landfill and the necessary consideration of different time scenarios adds to the difficulties of including leaching in an LCA model for MIBA utilization. Combined with the challenges posed by considering leaching from MIBA in the subbase of roads that are further elaborated in Section 3.3.2, this has further contributed to the decision to exclude impacts from leaching from the scope of the analysis.

3.2.5 Alternative material: crushed rock

MIBA can substitute primary raw material in road subbases. Natural aggregates such as crushed rock and natural gravel are exclusively extracted from open pit mines, either by dredging in lakes, rivers, or gravel pits or by blasting bed rock in quarries. They can include different mineral components and are only processed mechanically.

While the material itself does not pose any risks to human health or the environment, the extraction operations are associated with environmental impacts. Primary environmental aspects of the quarry process include the consumption of energy in the form of fuels and electricity for machinery, changes in the landscape, dust emissions, as well as the contamination of water with fine particles that usually require a mechanical wastewater treatment (Bundesministerium des Innern, für Bau und Heimat, n.d.).

The material considered in this study is crushed rock of the size 0-90 mm. The major difference between construction aggregates from crushed rock from bedrock and natural gravel is that natural gravel is extracted by dredging and not blasting. The general process steps to obtain crushed rock include:

- the removal of overburden such as soil to uncover the rock formation;
- drilling and blasting of rock;
- transportation of rock to crushing;
- several rounds of crushing depending on the material output; and
- transportation to construction site (Respondent 14, personal communication, March 18, 2020).

3.3 Reuse of residues in roads in environmental and Life Cycle Assessment

This section aims to elaborate the current stance of the assessment of environmental impacts from MIBA utilization in road construction with a focus on LCA studies. First, different approaches to the environmental assessment are reviewed. This is followed by an analysis of previous studies for important methodological choices, including system boundaries and impact assessment, which are also relevant to this study.

3.3.1 Different assessment approaches

The potential environmental impacts from reusing MIBA in road constructions and more generally secondary materials have been analyzed using different environmental assessment methods (Roth & Eklund, 2003). The variety of tools used ranges from specific leaching tests to broad LCA studies for which different categorizations have been proposed (e.g. Finnveden & Moberg, 2005). Roth and Eklund (2003) provide a categorization specifically of studies regarding the reuse of by-products in road construction in Sweden based on the system boundaries that the studies consider. Studies are classified into four types of different organizational levels. Ranked by system boundary complexity, these include: the material only; road environment; a narrow life cycle; and a study on an industrial level (see Figure 7).

- The **material level** studies only consider the impacts of a road construction material (e.g. MIBA), its contents, and properties. Examples include different types of chemical analyses on the total content and leaching behavior of pollutants.
- The **road-environment level** studies mainly include a substance-flow analysis, considering the entire road environment and studying the construction material in its spatial context.

- A **narrow life cycle approach** typically includes an assessment of further aspects, such as the pre-treatment of materials and their transportation.
- An **industrial system level LCA** includes an assessment of other industrial sectors, such as the generation of the by-product (Roth & Eklund, 2003).

The choice of system boundaries yields different results and focusses different issues. LCAs address a much broader variety of environmental aspects than assessments with narrow system boundaries and, in addition, include the use of natural resources. This is specifically recommended for assessing the use of wastes and by-products in road construction in the Swedish context, as the use of primary aggregates such as natural gravel leads to significant environmental impacts such as spoilage of drinking water or the destruction of landscapes (Roth & Eklund, 2003).

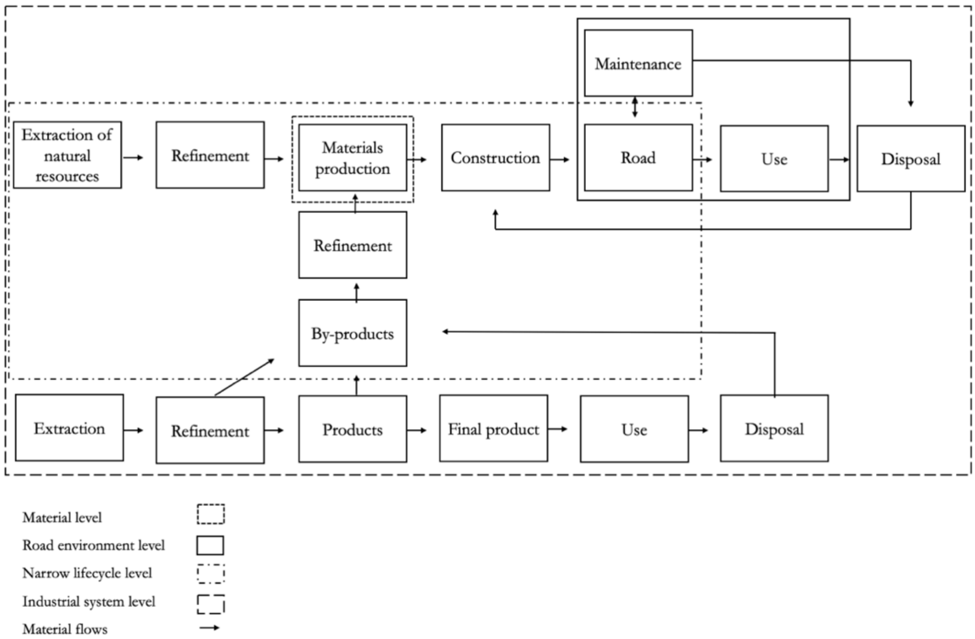


Figure 7: Categorization of environmental assessment methods regarding the reuse of by-products in road construction in Sweden proposed by Roth & Eklund (2003).

Source: Prepared based on Roth & Eklund, 2003, p. 113.

While the leaching of heavy metals has been the focus of evaluations with narrow system boundaries (e.g. Bruder-Hubscher et al., 2001; Hjelmar et al., 2007), studies of MIBA utilization with broader system boundaries have been conducted in the form of LCA studies mostly falling into the third category described by Roth and Eklund (2003). Such studies are either waste LCAs, assessing different management strategies for secondary materials (Allegrini et al., 2015; Birgisdóttir et al., 2007; Butera et al., 2015; Deviatkin et al., 2017; Toller et al., 2009), or studies with a product focus that examine different road construction scenarios with and without MIBA (Birgisdóttir et al., 2006; Geng et al., 2010; Olsson et al., 2006).

A further characteristic that distinguishes studies is if they look at the road as a whole or if the focus lies on comparing two scenarios. Whereas the former aims at considering the full environmental impacts and resource use from all life cycle phases (e.g. Birgisdóttir et al., 2006; Mroueh et al., 2000) the latter only looks at the parts of the life cycle that are different (e.g. Olsson et al., 2006). Therefore, it only considers the relative environmental impact of one scenario (e.g. conventional road) to another (e.g. road with MIBA).

The thesis takes a narrow life cycle perspective. It focuses on the comparison of two variations of the same road. As such, it is a product LCA as opposed to a waste LCA. Because the focus lies on the comparison of the two product systems, only the parts that are different between the two scenarios are considered. Results therefore need to be interpreted as the relative environmental impacts.

3.3.2 Methodological choices

In the following subsections, earlier LCA studies are reviewed with respect to key methodological choices. First, two instructive sources for this thesis are presented. Subsequently, further sources are analyzed with regard to the selection of system boundaries and their approach to the impact assessment.

Reference studies

An important guidance for the thesis is provided by two doctoral dissertations and their corresponding publications (Birgisdóttir, 2005; Toller, 2008). Both include a comparative LCA with a product focus on the use of MIBA as unbound aggregate in the subbase layer of a road in Denmark and Sweden respectively.

Toller (2008) conducted a broad environmental assessment of incineration residue utilization using a wider system perspective (Olsson et al., 2006; Toller et al., 2009) as well as a more specific view regarding the chemical composition of MIBA from MSW incinerators and their leaching behavior (Olsson et al., 2007, 2009). The work uses a case study applying the model in a consequential LCA comparing the same road built with and without MIBA in the Swedish context (Olsson et al., 2006). One kilometer of road and the disposal of 5,200 tons MIBA is used as the functional unit. The study includes two time scenarios, 100 years and an infinite time horizon. All significant life cycle stages are regarded, including production, transportation, location in road structure, and use-phase, and disposal. Life cycle stages that are similar for both scenarios are excluded as the study focusses on the comparison and the relative environmental impacts. The study is limited to an analysis of the normalized inventory results, no classification and characterization into impact categories is done.

Birgisdóttir (2005) developed a general LCA model in order to assess road constructions and the disposal of incineration residues ("ROAD-RES model"). It serves to assess the resource consumption and environmental impacts of roads constructed with virgin material and MIBA, as well as to evaluate different disposal options for incineration residues. The model has been applied in different case studies, including an LCA comparing two secondary roads with and without MIBA in Denmark (Birgisdóttir et al., 2006) and a study evaluating its disposal on a landfill compared to its use in road construction in Denmark (Birgisdóttir et al., 2007). One kilometer of secondary road is taken as a functional unit and the analysis is conducted for a time horizon of 100 years. The impact categories Stored Ecotoxicity (water) and Stored Ecotoxicity (soil) were introduced to account for pollutants that remain in the road after the first 100 years. The study considers the full environmental impact throughout the design, construction, and operation and maintenance life cycle stages.

System boundaries

Multiple studies focus on the comparison of different scenarios rather than showing the full environmental impact (Butera et al., 2015; Deviatkin et al., 2017; Geng et al., 2010; Mroueh et al., 2000; Toller et al., 2009). These studies generally omit those parts that are similar for the compared systems. For example, the clearance and road construction, structural components that are the same for both roads, and functionalities connected to the use of the road, such as road markings, traffic signs, lights, seasonal and regular maintenance work, and emissions from traffic (Mroueh et al., 2000). While it is acknowledged that the use of MIBA could potentially lead to a different performance in the use phase as well as to differences in the maintenance and EoL scenario, these stages are also excluded due to a lack of data and the need for introducing further assumptions (Geng et al., 2010; Santero et al., 2010).

The exclusion of similar parts also extends to the EoL phase in most examined studies because road constructions often remain in place after their useful service life (e.g. Birgisdóttir, 2005; Butera et al., 2015; Mroueh et al., 2001; Olsson et al., 2006; Stripple, 2001). Stripple (2001) argues that the majority of roads have no EoL because they are continuously reconstructed or replaced by a new road while the old road remains in place. Santero et al. (2010) call this a "perpetual pavement".

This serves to justify the exclusion of certain structural parts of the road as well as life cycle stages that can be considered similar for both product systems. To verify such assumptions, experts in the field of waste incineration, road construction, and bottom ash treatment and utilization were consulted. The results are summarized in Appendix III. The material extraction phase is not included in this summary because it is fundamentally different for both systems.

On the contrary, multiple studies highlight the importance of accounting for the potential benefits from avoiding the use of primary raw material as well as from the reduced demand for landfilling MIBA. In waste LCA, benefits from avoided primary material are usually credited to the product system including MIBA (Allegrini et al., 2015; Birgisdóttir et al., 2007; Butera et al., 2015). Allegrini et al. (2015) further extend system boundaries to include the avoided burdens from scrap metal recycling. Product LCAs do not credit benefits from avoided primary material but consider the alternative treatment of MIBA, which is assumed to be landfilling in all reviewed studies. This is done in one of two possible ways: the avoided burden is included as negative impacts (i.e. benefits) in the road scenario including secondary material (e.g. Birgisdóttir et al., 2006; Mroueh et al., 2001) or processes related to the landfilling of MIBA are included as environmental burdens in the conventional road scenario, thus increasing the overall environmental impacts of this scenario (e.g. Geng et al., 2010; Olsson et al., 2006). Both approaches yield equivalent results in a comparison. However, it can be argued that the second approach is more accurate as it more closely resembles the reality.

Both waste and product LCAs are similar in that multiple inputs and outputs to the product system are excluded from the studies due to a lack of data. In general, the production of fixed capital goods such as machines and vehicles as well as the manufacture of production plants used in the production of primary raw material and sorting of MIBA are excluded from studies (e.g. Allegrini et al., 2015; Deviatkin et al., 2017; Mroueh et al., 2001). Only Butera et al. (2015) consider capital goods in their assessment but come to the conclusion that their contribution to the overall environmental impacts are negligible for road construction scenarios and low for landfilling. Further excluded processes with respect to landfilling include the production and transport of materials used in the landfill, the energy consumption for the construction of the landfill (Birgisdóttir et al., 2007), materials used for the final covering of the landfill, and the production and use of chemicals for leachate treatment (Olsson et al., 2006). Finally, the use of some substances in the extraction and processing of primary aggregates such as blasting material (Stripple, 2001) as well as emissions in the form of noise and dust from the production and occupation of land are generally not considered (Stripple, 2001; Toller et al., 2009).

With regard to capital goods and other parts excluded from the system boundaries mentioned above, in general, this study follows previous approaches and bases their inclusion on the availability of data and the expected significance to the results of the study. However, compared to older studies (e.g. Birgisdóttir, 2005; Mroueh et al., 2000; Stripple, 2001; Toller, 2008), this study benefits from the further development of LCA software and linked databases such as Ecoinvent v3.5 that provide datasets that cover capital goods and infrastructure processes.

Various different time horizons are used in LCAs on roads. Decisive factors for the chosen time boundary are the assumed lifetime of the road as well as whether impacts from leaching are considered. The assumed useful lifetime of roads varies greatly between 25 and 200 years (Carlson, 2011; Carpenter et al., 2007). Studies that include leaching usually include an infinite time horizon, or "maximum impact scenario", in order to account for long term environmental impacts (e.g. Toller, 2008). However, if in comparative studies the EoL scenario of the road is assumed to be a continuous reconstruction of the road, the assumed lifetime becomes irrelevant. Furthermore, if impacts from leaching are not considered, an infinite time horizon is not necessary. In such cases, the time boundary is mainly defined by the considered LCIA methodology, e.g. GWP100 (i.e. 100 years).

A further important point with regard to the considered system boundaries is the exclusion of the incineration process from the product system including MIBA. All reviewed studies, waste LCAs and product LCAs, have taken a "zero-burden" assumption. This means that the emissions and resource use from the incineration process are entirely disregarded. In LCAs with a product focus, it is argued that industrial by-products are considered wastes according to waste legislation for which no environmental burdens are allocated (Mroueh et al., 2001). Furthermore, in ALCA, partitioning according to recommended allocation principles (i.e. economic value) would lead to an allocation of the environmental burdens to the co-products of the incineration process with a positive economic value: heat used in the district heating system and electricity. Similarly, it is argued that in a CLCA MIBA should be provided burden free to the system under investigation because its production is entirely dependent on the demand for incineration (Ekvall, 2020; Ekvall & Weidema, 2004).

The thesis builds on earlier LCA studies and excludes the incineration process from the system boundaries. The potential significance of this decision for the results of the LCA is further discussed in section 6.2.1.

Impact assessment

Different approaches are taken to assess the environmental impacts and resource use in the reviewed studies. While evaluations in some studies are based on normalized inventories (e.g. Stripple, 2001; Toller, 2008), other studies include classification and characterization of environmental loads into impact categories (e.g. Birgisdóttir et al., 2007; Deviatkin et al., 2017; Geng et al., 2010). The main reasons for not classifying and characterizing inventory data include the limited interest of decision makers in Sweden, limited data availability hindering the coverage of some impact categories, and uncertainties related to the consistent and accurate calculation of inventory data (Toller, 2008).

According to Toller (2008), possible environmental impacts from the use of MIBA in road construction can be categorized into "i) the use of natural resources, ii) the use of energy and the associated emissions to air and water, and iii) the direct emissions from the material" (p. 27). While procedures and impact categories regarding the second category are abundant (e.g. GWP), the first and third category receive special attention in studies.

Regarding the assessment of leaching of toxic substances from MIBA, there is a certain consensus among practitioners regarding two points. First, a reasonably accurate estimate of the environmental and human health risks from leaching is a challenging task. Second, it is rather difficult to include the results of risk assessments in an LCA methodology (Lynn et al., 2018; Silva et al., 2019). Leaching and the potential impacts it causes is highly dependent on time and space that determine precipitation, soil constitution at the base of the road, and percolation rate. The quality of the MIBA and the construction characteristics also play an important role (Schwab et al., 2014). Despite there being a large amount of data available on leaching, its analysis is difficult due to a wealth of different leaching criteria and test methods (Lynn et al., 2018). As a result, the inclusion of leaching effects in an LCA is a typical area of uncertainties (Butera et al., 2015).

A further challenge is the impact assessment of leaching. In general, the toxicology part of LCA is underdeveloped and impact assessment methodologies included in LCA software, such as USEtox, are not capable of calculating environmental impacts from leaching as only the total content of toxic substances is considered (Respondent 9, personal communication, September 3, 2020; Respondent 12, personal communication, April 3, 2020). Furthermore, the USEtox model does not account for the potential immobilization of pollutants in the soil layer before reaching ground water or different types of soils and their properties to retain constituents (Butera et al., 2015; Schwab et al., 2014). The characterization of toxic impacts is therefore prone to many uncertainties (Allegrini et al., 2015) and for many substances, there are no characterization factors (Butera et al., 2015). To address these challenges, several studies develop their own models or adjust the USEtox model in order to approximate leaching (e.g. Elisa Allegrini et al., 2015; Butera et al., 2015; Schwab et al., 2014).

As a result of the uncertainties and difficulties related to assessing the impacts of leaching, some scholars recommend to include leaching through qualitative assessment in LCAs as long as no better quantitative models are available (Santero et al., 2010). Furthermore, while it can be argued that including leaching data is important to accurately reflect potential environmental impacts, it is also stated that "the accuracy of its modelling should be consistent with the accuracy of modelling other relevant emissions, in order to avoid unfair and misleading assessments" (Allegrini et al., 2015, p. 483). Leaching is therefore excluded from the scope, because a sufficiently accurate modelling cannot be done within the thesis project.

With regard to the first category of impacts identified by Toller (2008), the use of natural resources, the general difficulties and uncertainties were already elaborated in detail in Section 3.1.3. None of the reviewed studies classifies and characterizes the use of natural resources into impact categories. Particularly the use of land areas for landfilling and the production of primary raw material is excluded from the scope of studies. This is despite the fact that it is cited as a potentially important environmental aspect (Birgisdóttir et al., 2006; Mroueh et al., 2000; Santero et al., 2010). The thesis uses the resource-related impact categories included in the ILCD 2011 Midpoint+ LCIA method.

3.3.3 Impact assessment results

Some key findings from previous LCAs are analyzed below. This includes the two dissertations presented above and the results of another similar but less detailed study by Geng et al. (2010). This serves as a guidance for the interpretation phase for this LCA as well as a basis for the discussion of the LCA results presented in Section 6.

The study by **Toller (2008)** indicates that emissions are strongly related to either the use of energy or leaching of pollutants in different life cycle stages. Different impacts prevail in different life cycle phases; whereas natural resource use and energy consumption were largest during the raw material extraction and refinement, the leaching of trace elements occurred during the use phase. All emissions to air, and to some extent the emissions to water, depended on the amount of energy used in the different unit processes of the system. The production of crushed rock and landfilling are the two most energy intensive processes. As a result, there is a large difference in terms of energy use between the two scenarios as both processes are part of the conventional road scenario. It was further found that the transportation distances have a high influence on the results.

The general conclusion of the study is that the scenario with virgin material utilizes more energy and natural material while substituting primary material with MIBA would lead to more leaching of trace elements. A road with MIBA would be the more energy-saving alternative even when considering a 140 km longer transportation distance for the MIBA than for primary aggregates. The use of primary material and emissions of chromium, copper, and cadmium were identified to be the most important elementary flows. Although it is stated that the results regarding the leaching of trace elements are hard to interpret, the results confirm that it is an important impact to be considered in the analysis of MIBA management. However, it is also stated that the environmental impacts resulting from trace element leaching are likely to be more dependent on when, where and at what concentrations elements leach than on the total amount that leaches over 100 years.

Birgisdóttir (2007) concludes that the comparison of a road built with and without MIBA shows only marginal differences regarding resource consumption and environmental impacts. Furthermore, the results indicate that the overall impacts from using MIBA in roads are insignificant. However, the study takes into consideration the full environmental impact of the two road scenarios as well as the use of the road. Therefore, impacts caused by the use of MIBA in the subbase layer are relatively small compared to the impacts contributed by other layers of the road, such as the asphalt layer, which contains a considerable amount of heavy metals and requires more energy than the production of granular material.

According to Birgisdóttir (2007), the majority of the environmental impacts are related to emissions from the combustion of fossil fuels in all life cycle stages. Furthermore, spoiled groundwater resources are identified to be a significant potential resource consumption, however, this is not due to the leaching of heavy metals or salts from MIBA but from road salting during the use phase. With regard to the conventional road scenario, CO_2 and NO_x emissions from diesel combustion in transport vehicles and machinery as well as the heavy metal content of construction material are identified to be the major drivers of environmental impacts. On the other hand, the road including MIBA would save approximately 20% of the used gravel material in the design stage and lead to an overall reduction in the consumption of natural aggregate of 10%. The results further indicate an increase of 10% in potentially spoiled groundwater resources caused by the leaching of salts from MIBA as well as marginally lower long-term impacts caused by leaching (i.e. stored ecotoxicity) due to the avoided landfilling.

The introduced stored ecotoxicity impact categories account for the stored heavy metals in the road structure that lead to potential impacts after the assumed service life of the road. They constitute the greatest potential environmental impact; however, they are assumed to take place over many centuries, and it is argued that they should carry a lower weight than other potential impacts.

Geng et al. (2010) compared the energy consumption and environmental impacts of a road structure with and without MIBA for a highway in Shanghai, China. The results indicate that the use of MIBA leads to lower impacts in all impact categories by around 40% except for ecotoxicity water and human toxicity caused by heavy metal leaching, which are 84% and 147% higher compared to the conventional road scenario. The road with MIBA leads to lower resource consumption including 51% less gravel, 51% less electivity, and 41% less diesel. This is mainly due to the quarrying and processing stage of the primary material, avoided landfilling of MIBA, and shorter assumed transportation distances for incineration residues.

While the reviewed studies are not directly comparable, results point in similar directions. The comparisons show a clear tradeoff between impact categories: the use of MIBA leads to a reduction in resource use and non-toxic impacts and an increase in toxic impacts due to potential leaching. This tradeoff is also highlighted by Carpenter et al. (2007) who conducted a semi-industrial level LCA based risk assessment combined with site specific data that compared the use of different secondary materials in road constructions to the use of virgin material. Similarly, if MIBA is used, burdens shift from the materials production stage to the use phase: resources and energy are saved by avoiding landfilling and the production of primary aggregates and leaching of contaminants increases during the use of the road.

Energy use in the form of electricity as well as diesel combusted in transportation vehicles and construction machinery is a major driver of environmental impacts. As a result, the assumed transportation distances to construction sites and final disposal are important assumptions. The importance of energy consumption and transportation on the overall environmental impacts and resource consumption is corroborated by studies on the recycling of other secondary material such as construction and demolition wastes in road structures in Denmark (Butera et al., 2015). Results indicate that transportation represents the greatest overall positive and negative contributions, including transportation to facilities to process secondary material as well as saved transportation to a landfill. Finally, Birgisdóttir et al. (2007) show that if the full environmental impacts of a road are considered, the share of impacts resulting from the use of MIBA is rather small.

3.4 Summary

Section 3.1 presented the concept of LCA, the chosen approach to environmental assessment. Different modelling approaches were explained and the various methodological considerations that need to be made in an LCA were also described. A particular focus was on the concepts of system boundaries and allocation, and how these concepts are operationalized in different modelling approaches. Two sub-sections then detailed how climate benefits and resource efficiency can be evaluated using LCA. The first part of the literature review provides the foundation to address the first research question regarding how climate benefits and resource efficiency can be assessed using LCA methodology in the specific case.

Section 3.2 introduced context with regard to the components of the product system in scope as well as the important points with regard to their assessment using LCA methodology. Literature on different technologies and practices, as well as legal standards in terms of the environment and geotechnical requirements, were reviewed in order to be able to develop an LCA model that most accurately resembles the current situation in the Malmö Harbor case.

Section 3.3 then presented different environmental assessment approaches as applied to secondary material use in road construction. Methodological choices of previous studies were reviewed, and impact assessment results of these studies presented. Combined with information from Section 3.1 and 3.2, this serves to address RQ1 which asks for a suitable LCA model to assess the environmental performance in terms of resource efficiency and GHG mitigation potential of a road using MIBA, compared to a conventional road, in the specific case. The LCA model that was developed for the case is detailed in the following chapter. The LCA model is considered to be part of the key "findings" of this study, as a large share of this research went into developing it, verifying its assumptions, and implementing it in the SimaPro software. The synthesis of previous research in Section 3.3 guides the analysis and interpretation of the LCA "results" of this study in Section 5 and 6 (answering RQ2 and RQ3).

4 The case of using incineration residues in road construction in Malmö

4.1 Case specific context: bottom ash utilization and Sysav AB

Sysav seeks to promote the utilization of MIBA in the North Harbor industrial area in Malmö that is to be developed over the next ten years. The company operates an incineration plant in the harbor area and is permitted to treat 630,000 tons of waste annually by means of incineration (Sysav, n.d.). In 2018, 576,500 tons of MSW and industrial waste were treated (Sysav, 2018). The recovery of energy from the facility includes heat and electricity. It contributes about 60% of the district heating in Malmö and the neighboring city of Bürlöv, as much as 70,000 small houses. The electricity production is approximately 250,000 MWh per year (Sysav, n.d.). The incinerator is located 4.6 km from the new North Harbor industrial area (see Figure 8).

Figure 8: Map of the Malmö harbor area.

Source: Prepared using Google Earth.

The incineration process produces around 110,000 tons of IBA annually. 90,000 tons of MIBA remain after the treatment process. Currently, the entire amount is either used as construction material on regional landfills operated by Sysav and other companies or disposed of without further use on regional landfills. Transportation distances to landfills range between 36 and 81 km, with an average distance of 64.52 km/ton over the last two years.

Managing bottom ash induces significant economic cost for logistics, landfill fees (if not disposed of on a landfill operated by Sysav), and landfill tax (if not used as construction material on a landfill). Meanwhile the demand for MIBA as construction material on landfills is limited. The capacity for the direct disposal of MIBA on landfills is also limited in the region. In the short-term future (<10 years), Sysav expects to have to landfill all MIBA on its own landfill site, Spillepeng, in the vicinity of the incineration plant. This would lead to a reduction of the landfill's expected lifetime from 30 to 5 years, necessitating the development of a new site. This would imply further significant economic cost for Sysav as well as environmental and social costs for the society as a whole because a new landfill site would require scarce bio-productive land, which in the region of southern Sweden is among the most fertile in the EU.

Sysav expects significant economic and environmental benefits from the utilization of MIBA in the North Harbor area. Currently, its management incurs a significant cost. In a road construction scenario, it is expected that a local company, Swerock AB, which sources construction materials, will take MIBA for free and both delay the need for a new landfill site and reduce transportation distances for MIBA from an average of 65.52 km to 4.6 km. In addition, primary raw material and the corresponding transportation otherwise necessary for the construction of the road provided by a quarry located 30 km from the North Harbor could be avoided.

Sysav both supported and conducted several research projects exploring the technical and environmental properties of MIBA as road construction material. One important outcome is that Sysav's MIBA fulfills the geotechnical requirements for the use as subbase layer for roads (Arm et al., 2017). Regarding environmental impacts from leaching, monitoring and analysis of several pilot projects in and around Malmö over a period of 8-14 years, concluded that no negative environmental impacts to surroundings or groundwater resources were caused by the use of MIBA in different applications (Avfall Sverige, 2015). Nevertheless, the material does not fulfill the legal requirements for the "free" use, as the total contents of various substances and the leaching levels of copper exceed the negligible remaining risk threshold. The use of MIBA is therefore contingent upon local government approval and requires an environmental impact assessment, as well as control and monitoring measures (see Section 3.2.3).

This thesis seeks to describe and better understand the potential of using MIBA as secondary material in road construction in terms of resource efficiency and GHG mitigation, compared to a conventional road construction scenario using primary materials. The results will improve the basis for decision making in the local context with regard to the use of secondary materials in roads. Actors that particularly benefit from the generated knowledge include Sysav, companies organized in the Copenhagen Malmö Port, and the City of Malmö. For Sysav, the results could help to demonstrate potential benefits from using MIBA, compared to primary raw material, and thus provide an argument in a future approval procedure for its use in the Malmö harbor area. The LCA results could further feed into a required environmental impact assessment.

The City of Malmö is particularly interested in encouraging industrial symbiosis in the Malmö harbor area, and has expressed interest in using MIBA for its potential low-carbon benefits (Respondent 10, personal communication, March 10, 2020). The city has adopted ambitious environmental targets with regard to resource efficiency and climate (Malmö Stad, 2014). The authorities' goal is to reach climate neutrality in 2020 (Malmö Stad, 2009). These targets also extend to the demands placed on public procurement and infrastructure projects including road construction (Respondent 10, personal communication, March 10, 2020).

4.2 The Life Cycle Assessment model

4.2.1 Goal and scope

The goal of applying an LCA to the Malmö Harbor case is to describe and compare the environmental impacts and resource use of two variants of a hypothetical road built in the Malmö harbor area. Two scenarios are considered in this study. Scenario (A), refers to a road built with primary aggregates, crushed rock, in the subbase layer. Scenario (B) refers to a road in which the primary raw material is substituted by MIBA in the form of unbound aggregates from the Sysav waste incineration facility in the Malmö harbor area.

The geographical scope of the study is Malmö. The time boundary considered for the life cycle impact assessment is 100 years based on the selected LCIA methodology (e.g. GWP100). The study was performed with the SimaPro 9 software package including the Ecoinvent v3.5 life cycle inventory database.

The LCA was particularly guided by the ILCD Handbook (JRC-IES, 2010a) following the suggestions for goal situation A "Micro-level decision support." As such, the assessment followed an attributional modelling approach but with extended system boundaries in order to include emissions and resource use from the alternative disposal of MIBA. The LCA therefore serves as decision support for different actors and the product is assumed to be produced only as a consequence of decisions that are supported by the LCA study (JRC-IES, 2010a).

Situation A decision support is typically aimed at short to medium term decision support (present to 10 years from present). It particularly serves to inform the purchase of products that are already developed and that are foreseen to enter the market. A further important criteria for the use of the provisions of goal situation A is that the product has a limited market share of the total production of its sector so that the production decision does not have large-scale structural consequences in terms of changes in the installed capacity in the background system or other systems (JRC-IES, 2010a).

Functional unit and reference flows

The product system under study provides two functions: the production of 1 km of road in the Malmö harbor area and the disposal of 6,960 tons of MIBA. By including the disposal of MIBA the effects of avoiding landfilling on the overall environmental performance of the system can be considered. 6,960 tons is the amount of material that would replace crushed rock to produce 1 km of road and that otherwise would be landfilled.

According to Stripple (2001), using a section of a road is the most representative and simplest functional unit. It is common practice in earlier LCA studies (Birgisdóttir et al., 2006; Carlson, 2011; Mroueh et al., 2001; Olsson et al., 2006) and was confirmed with experts (Respondent 11, personal communication, January 3, 2020). The type of road was decided in consultations with the City of Malmö in order to ensure the suitability for the case study area. It is based on a layout of an existing road in the North Harbor (see Appendix I). A simplified version of the road is displayed in Figure 9. Calculations are based on the simplified version.

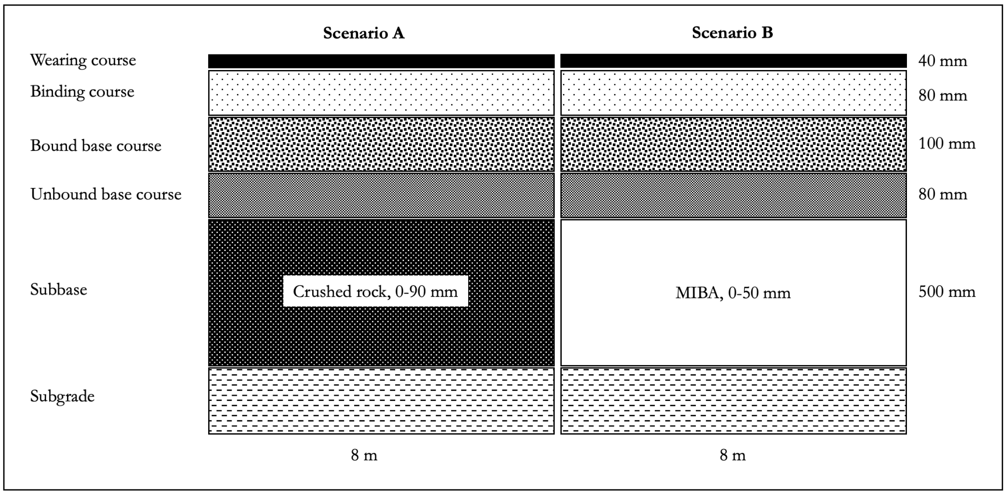


Figure 9: Simplified layout of the two considered road scenarios.

The reference flows were calculated based on the maximum dry bulk density provided by AB Sydsten, the company operating the quarry that according to the Real Estate and Street Office of the City of Malmö would provide primary aggregates for road constructions in this specific context. The density for MIBA from the waste incineration facility in Malmö was provided by Sysav and is based on measurements conducted by Arm et al. (2017). The maximum dry bulk density indicates the density after compaction by means of a proctor test. It is therefore a theoretical value for maximum compaction. Although the level of compaction is likely to deviate from this value when the material is worked during road construction, it more accurately resembles the amount of material used in the road than the bulk density (Respondent 7, personal communication, March 13, 2020).

Table 2: Material intensity of 1 km of road for scenario A and scenario B.

	Material	**Volume (m^3)**	**Bulk density (t/m^3)**	**Maximum dry bulk density (t/m^3)**	**Reference flow (t)**
Scenario A	Crushed rock, 0-90 mm	4000	1.7	2.3	9200
Scenario B	MIBA, 0-50 mm	4000	1.25	1.74	6960

System boundaries

Due to a focus on the comparison, only life cycle stages which are different between the two scenarios are considered in the system boundary. Identical life cycle stages such as the use, maintenance, and EoL are excluded from the scope of the analysis. A further delimitation is the exclusion of potential leaching of toxic substances from MIBA in all life cycle stages.

It is assumed that MIBA with the texture of 0-50 mm has the same functionality as virgin rock materials of the size 0-90 mm when used in the subbase layer of the investigated road and that it leads to a product of equivalent durability. The subbase layer is the only layer that is considered for substitution of virgin material. As a result, no other relevant structural components of the street change due to the use of MIBA and all other road layers (unbound base course, bound base course, binding course, and wearing course) are excluded from the study (see Figure 9).

Interviews with experts in the field of road construction, bottom ash utilization (Respondent 1, personal communication, February 4, 2020; Respondent 7, personal communication, March 13, 2020; Respondent 8, personal communication, March 16, 2020) as well as with stakeholders potentially affected by the LCA study (Respondent 10, personal communication, March 10, 2020) were conducted to confirm the equal durability of the two products. The results were that within the scope of this study the use of MIBA leads to only minor changes in the excluded parts that are assumed to have insignificant influence on the overall results of the study. Interview results with regard to potential differences in the life cycles of a road with and without MIBA are summarized in Appendix III. This assumption has also been made by previous LCA studies on MIBA utilization in roads (e.g. Birgisdóttir et al., 2006; Geng et al., 2010).

Furthermore, the use of MIBA in the road subbase is assumed to have no influence on the maintenance and use of the road. Maintenance work is usually limited to the wearing and binding course that only constitutes the first few centimeters of the road structure. This includes the reconstruction of the road after its first useful lifetime of approximately 50 years. The subbase layer remains unaffected. The maintenance and use stage are therefore not considered. With regard to the construction stage, only the different transportation distances for MIBA and crushed rock to the construction site are included. In line with previous LCA studies on roads (e.g. Birgisdóttir, 2005; Butera et al., 2015; Mroueh et al., 2001; Olsson et al., 2006; Stripple, 2001) and recommendations from experts (Respondent 7, personal communication, March 13, 2020; Respondent 10, personal communication, March 10, 2020), it is further assumed that the road is continuously reconstructed and remains in place after its useful lifetime. The EoL stage is therefore also excluded.

Two cut-off criteria were used to determine the initial inclusion of inputs and outputs: the availability of data and the potential significance to the results of the study. In general, fixed capital goods (i.e. products not intended for consumption and with a lifetime of more than one year (Weidema et al., 2013)) were included if they were accounted for in datasets derived from Ecoinvent. However, the infrastructure of the sorting facility as well as any other environmental aspects other than its energy consumption and from the vehicles used at the plant were excluded. This was done in the majority of reviewed LCA studies due to a lack of available data (e.g. Allegrini et al., 2015; Mroueh et al., 2001; Stripple, 2001). It is assumed that omitting infrastructure in this case has no significant influence on the results because the major environmental impacts are caused by the use of construction machinery and the related energy consumption (Butera et al., 2015).

Data requirements

Two general types of data were used to conduct the LCA. Data on the foreground system (i.e. the newly developed product system) including qualitative descriptions of processes, material and energy flows as well as quantitative information on material intensities. Foreground data was obtained by means of either data questionnaires or interviews mainly from Sysav, AB Sydsten, and NCC Industry Nordic AB. On the other hand, background data was obtained from the Ecoinvent v3.5 database accessible through the SimaPro software. Data was exclusively obtained from this database in the same system model version for all processes in order to ensure methodological consistency.

The Allocation at Point of Substitution Ecoinvent (APOS) System Model was chosen because it resembles the overall modelling approach of the study. Based on the attributional approach, it uses the average or unconstrained supply of products and partitioning to solve multifunctionality. Furthermore, credits are given to recyclers for keeping materials out of the waste stream. Credits are based on the avoided amount of primary material consumption minus the resource use and emissions necessary to refurbish the secondary product (Weidema et al., 2013).

The research took into account the most recently available and accessible data on the two product systems (time-related coverage). Due to the local focus of the study, site-specific data was preferred over regional, national or international averages (geographical and technological coverage). This includes, for example, data on resource consumption specific to the Sysav treatment facility for bottom ash. Where this was not possible, average data was used that most accurately resembled the process in view.

4.2.2 Inventory analysis

Scenario A: primary raw material

Scenario A includes primary material of natural origin in the subbase layer. To fulfill the functional unit, system boundaries are expanded to include all processes related to the materials production and construction of the road as well as the landfilling of 6,960 tons of MIBA (i.e. the amount of MIBA that could have replaced crushed rock in 1 km of road) (see Figure 10). Processes related to the landfilling of MIBA are included as environmental burdens in scenario

A as opposed to benefits in scenario B because it more closely resembles reality and because it makes the interpretation and communication of results easier.

It is assumed to be closer to reality because emissions and resource use from landfilling MIBA can actually be expected to occur if crushed rock is used, i.e. the MIBA that could have been used needs alternative disposal. On the other hand, if MIBA is used in the road, the emissions and resource use from landfilling are zero, but not negative. If scenario B is credited with benefits from avoided landfilling, the impact scores for various impact categories are negative, which is difficult to interpret against the positive impacts of scenario A. This would also make the communication of results problematic, because results showing a net benefit for scenario B could be falsely interpreted to suggest that the production of a further unit of product B leads to a reduction of the overall environmental burden. Including landfilling of MIBA as environmental burdens in scenario A hence simplifies the interpretation of results as both systems have positive impacts, in the sense that they both result in environmental burdens. Finally, the chosen approach facilitates the comparability with studies conducted within similar geographical contexts (e.g. Olsson et al., 2006).

The material used in the subbase of the road is crushed rock of the size 0-90 mm. An interview with the Real Estate and Street Office of the City of Malmö revealed that the material would most likely come from AB Sydsten's largest quarry located close to the town of Dalby, 30 km northeast of Malmö. The material extracted consists mostly of two granite-like rock types, Gneiss and Diabase, at a ratio of 1:1. The interview further confirmed that the product goes through the typical process steps for the production of construction aggregates from bedrock described in Section 3.2.5. This includes one to two crushing steps to obtain a grain size of 0-90 mm. The number of crushing steps is important because it is the most energy-intensive process at the quarry. Apart from on-site transportation, all machinery is run by electricity from the local grid (Respondent 14, personal communication, March 18, 2020).

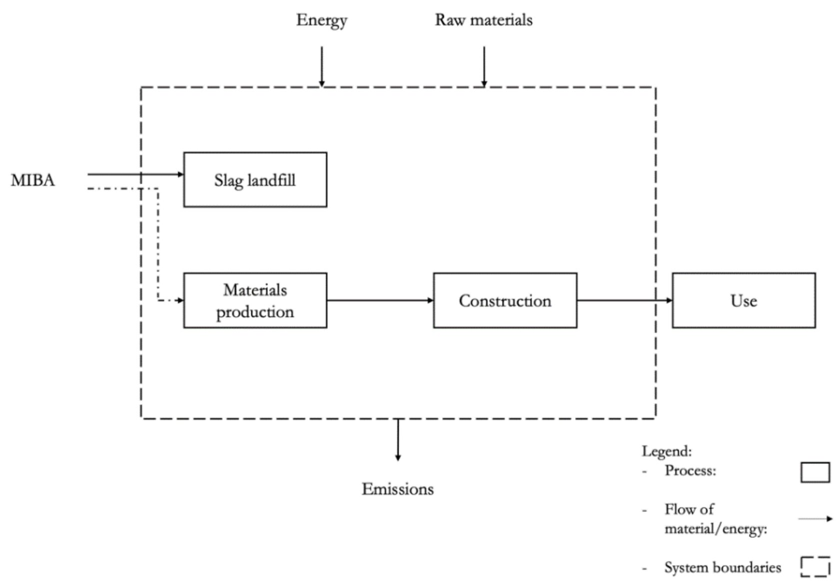


Figure 10: System boundaries considered in scenario A.

Quarry process

Different datasets were considered to resemble the production conditions at the quarry in Dalby. Ecoinvent processes for the production of crushed gravel based on data from Swiss quarries and for the production of crushed rock based on Brazilian quarries, the ELCD dataset "Crushed stone 16/32 mm, wet and dry quarry, production mix, at plant, undried, EU", and data from an Environmental Product Declaration (EPD) for different construction aggregates from a quarry in Glimmlingen, Sweden (NCC, 2019).

It was decided to use the dataset "gravel production, crushed, Brazil" (Ecoinvent v 3.6). The process was adjusted to Sweden by changing, where possible, background data on elementary, product, and waste flows to Swedish conditions. Furthermore, foreground data was adjusted based on the EPD data after an interview with AB Sydsten had confirmed that the processes for the product group 0-90 mm including the number of crushing steps are identical in both quarries. Adjusted foreground data includes the water demand, electricity consumption, amount of steel to refurbish machinery, lubricating oil, hazardous waste, and MSW. Background processes that were changed to more closely resemble Swedish conditions are the electricity mix, waste management scenarios, and the tap water.

The final process included in the model covers the mining of granite by drilling and blasting, transportation, crushing, storage, land use, equipment maintenance, and infrastructure. Machinery uses electricity and on-site transportation is done by diesel-powered trucks. A lifetime of 80 years for the quarry is considered. The removal of overburden, administrative activities, the transportation of the material to the construction site (included in the construction stage), and the recultivation of the mine are not included.

The main reasons for choosing the dataset based on Brazilian quarries were that, first, the dataset based on Swiss quarries covers the production of crushed gravel based on natural gravel and not bedrock. As opposed to crushed rock from bedrock, natural gravel is extracted based on dredging and not blasting. Second, the ELCD dataset was not available as unit process and therefore could not be adjusted to Swedish conditions. Finally, the EPD data lacked information regarding infrastructure and land use. A detailed elaboration on the choice of data regarding the material production stage of scenario A can be found in Appendix V.

Transportation

The transportation distance to the construction site in Malmö of 30 km was measured using Google Earth. The transportation mode and vehicle type were identified through a data questionnaire sent to AB Sydsten and Akka Frakt, a local logistics company. The transport is assumed to be carried out by a diesel-powered truck with a total weight of >32 metric tons and emission class Euro 6. The truck returns empty. Detailed information on the chosen Ecoinvent datasets for transportation can be found in Appendix V.

With regard to the landfilling of MIBA, it is assumed that MIBA is deposited in a slag compartment of a sanitary landfill similar to the description in Section 3.2.4. The assumed transportation distance to the landfill is based on the average transportation distance of MIBA from the Sysav incinerator to final disposal in 2018 and 2019, 64.52 km. It is further assumed that MIBA is transported by a Euro 6 lorry with a total weight of 28 metric tons and using biodiesel as required by Sysav tendering documents. The lorry returns empty. See Appendix V for a description of the different disposal sites and the respective amount of material and transportation distances.

Landfill process

Two Ecoinvent datasets were used to model the landfilling of MIBA. The process "slag landfill construction, CH" considers the infrastructure materials, operation, and aftercare of a sealed off slag compartment in a sanitary landfill for bottom ash from a MSW incinerator. The dataset is based on Swiss technology in the year 2000, however, it is still applicable to current landfilling practices in Europe (Ecoinvent, 2019). The time coverage considers 5 years construction, 30 years use, and 75 years aftercare. The avoided landfilling was calculated based on the landfill capacity (375,000 m^3) and the maximum dry bulk density of MIBA originating from the Sysav incinerator. The maximum dry bulk density was used because waste is compacted by heavy machinery in landfills. In addition, the dataset for "process-specific burdens, slag landfill, CH" was used. This dataset is based on the same technology and considers the land-use and energy demand for the treatment of a specific amount of waste independent of its composition. Both processes were adjusted to Sweden by changing the electricity consumption to the Swedish market mix.

Scenario B: secondary material

Road scenario B includes MIBA from the municipal waste incinerator in Malmö in the subbase layer. Its material production stage is composed of several consecutive steps: first, IBA is transported by lorry from the waste incineration facility to a stockpile on the Spillepeng recycling center for drying (1 km). After drying for about 2 months, dry IBA is transported by a wheeled loader to a sorting facility (0.1 km). After the sorting process, MIBA is transported by a wheeled loader to a second stockpile for aging for another 4-6 months (0.3 km). Drying and aging require no further inputs. After the aging process, MIBA can be used as secondary material in road construction and is transported by lorry to the construction site (4.6 km).

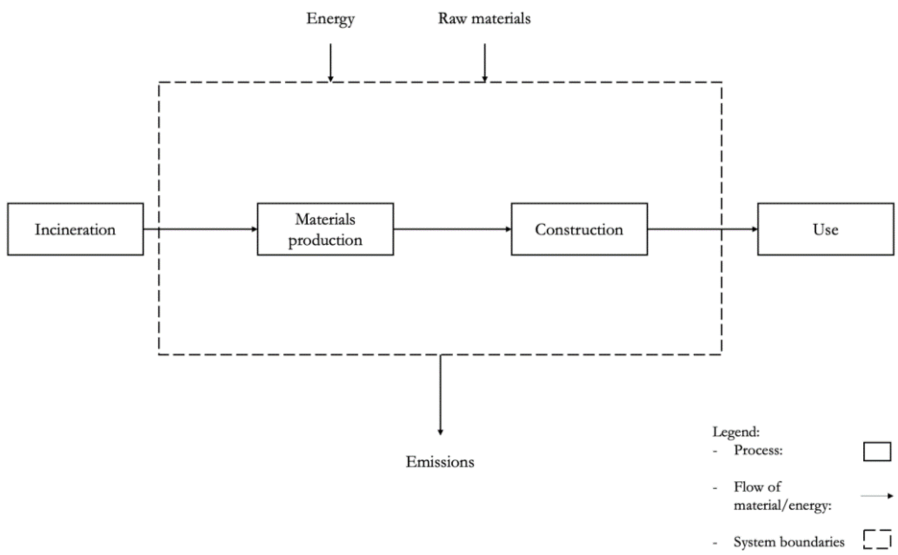


Figure 11: System boundaries considered in scenario B.

The product system under consideration starts with the reception of IBA from the incinerator and its transport to the drying stockpile (see Figure 11). The incineration process is not included in the product system. This has been the approach in all reviewed studies done on the use of secondary material in road constructions that have a product focus (e.g. Birgisdóttir et al., 2006; Geng et al., 2010; Mroueh et al., 2001; Olsson et al., 2006). Here, it is assumed that IBA enters the system burden free because economic allocation would lead to the environmental burdens being attributed to the co-products of the incineration process with a positive economic value.

Sorting process

The sorting and aging process are necessary treatment steps for the bottom ash to fulfill geotechnical and environmental requirements for its use in road construction. A new sorting facility has been installed by Sysav at the Spillepeng recycling center in Malmö, 1 km from the considered incineration plant. It started operating at the start of the research (January 2020). As a result, data received from Sysav regarding the energy demand, sorting efficiency, and quality of sorted metals are mostly estimates based on tender documents, comparable sorting facilities in Denmark and Germany, as well as machine documents.

With the new sorting plant, Sysav aims to recover Fe, NF, and stainless-steel metals and at the same time produce a construction aggregate that can be used in civil engineering by crushing the IBA and extracting not-burned (organic) objects. The plant includes a combination of separation technologies for different material types including magnets (Fe metals), wind shifters (organic material), eddy-current separators (NF metals), and screens (grain size separation). The capacity of the facility is 100 tons of dried IBA per hour.

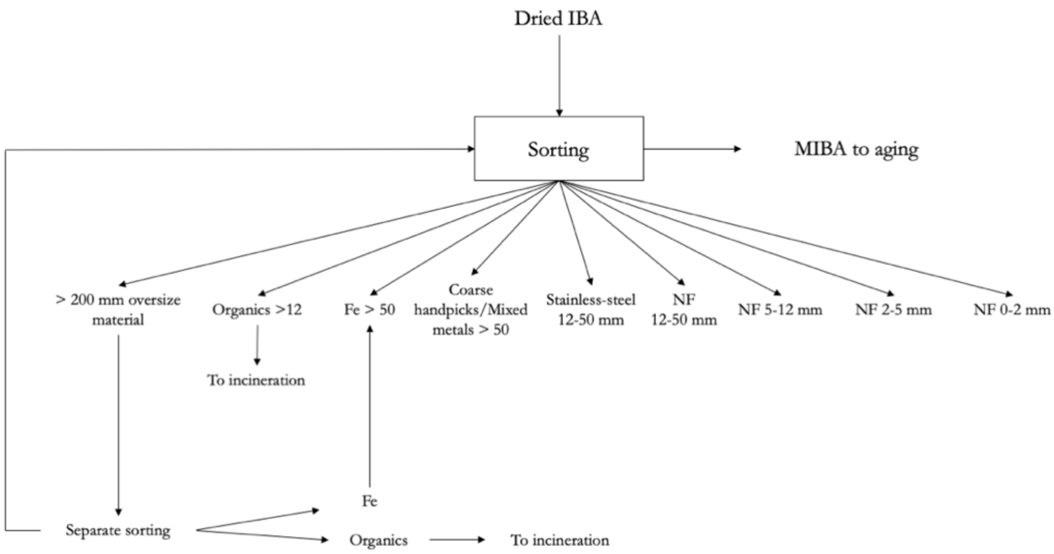


Figure 12: Simplified illustration of the sorting process and its outputs.

The sorting process has seven groups of outputs: NF metals in different grain sizes, Fe metals in different grain sizes, handpicks, unburned organic material, stainless steel, oversize material, and MIBA (see Figure 12). Oversize material consists of Fe metals, organics, and a mineral part. The mineral part is fed back into the sorting process. Handpicks consist of other magnetic material such as copper. The organic fraction is transported to the incineration facility and incinerated again. Table 20 in the Appendix VI shows the composition of IBA delivered to the sorting facility, the expected future sorting efficiency, and the future expected amount of material sorted out per fraction.

Only the electricity demand of the sorting facility and the use of different construction machinery operated at the plant were accounted for in the model. Land use was not considered because the plant is located in an area that is part of the incineration and landfill facilities at Spillepeng. The water consumption of the sorting plant was not taken into account as it is estimated to be negligible. The total energy demand of the facility amounts to 345 kWh per hour and includes the sorting machinery, air treatment, cleaning, high-pressure equipment as well as lighting and heating for office space in the sorting facility. About 80 kWh is consumed by the office space.

Two wheeled loaders of different sizes (Volvo L350 and L150G) and an excavator (Doosan DX255LC) are used at the sorting facility. Loaders transport IBA from the drying stockpile to the sorting facility, load the plant, and transport MIBA from the sorting facility to the aging stockpile. The excavator sorts coarse material and mixes IBA before it is loaded into the plant. All three machines are assumed to operate 7 hours daily. Based on the measured fuel consumption per hour of each vehicle, the Ecoinvent process "diesel, burned in building machine, GLO" was used to inventory resource consumption and emissions from their operation.

The recycling of scrap metals constitutes an open-loop recycling process. To solve multifunctionality, partitioning based on the economic value was applied. Recommendations of the ILCD guide for attributional modelling for EoL and waste products of negative market value that produce secondary goods were considered (JRC-IES, 2010a, p. 269). The absolute value of the cost to manage MIBA after sorting in scenario A (i.e. MIBA is landfilled) was used on the one hand, and the expected future market price for different scrap metals including Fe metals and handpicks, stainless steel, and NF metals on the other. The underlying assumption for taking the absolute value of the disposal cost of MIBA is that the sorting process is seen as the upgrading process that makes possible the use of MIBA in road construction in scenario B. Therefore, it increases the value of MIBA from negative to zero, as Sysav expects local construction companies to receive MIBA at a price of zero if its use in the case study area would be permitted by the local authorities.

The negative market price for MIBA was calculated based on the average cost for the management of MIBA during the years 2018 and 2019. The calculation includes the landfill handling fee and the average cost for logistics provided by the sales department of Sysav (see Table 21 in Appendix VI). Scrap metals are sold to recycling companies in Sweden, the Netherlands, Germany, and Italy. However, due to the short amount of time that the plant has been running so far, prices have only been received for some of the materials. The quality of sorted metals is also expected to further increase. Because the achieved prices are dependent on the quality of the material, prices are expected to increase in future as well. Therefore, and because market prices are volatile, the expected future average prices based on estimations by the Sysav sales department were used to calculate allocation factors (see Table 3).

The following formula was used to calculate allocation factors:

$$P_i = \frac{n_i \times x_i}{\sum n_i \times x_i}$$

P_i = allocation factor of co-product i

n_i = the quantity of product i

x_i = the price of product i

Table 3: Allocation factors used for the economic allocation applied to the multifunctional sorting process based on 100 t of sorted IBA. Mass allocation factors are indicated for a later sensitivity analysis.

Material	Prices (SEK/t)	Quantities (t)	Allocation Factor based on economic value (%)	Allocation factor based on mass (%)
Fe metals and hand picks	1,950.00	5.29	15.39	5.59
NF metals	6,750.00	4.03	40.59	4.26
MIBA	320.78	85.00	40.68	89.81
Stainless steel	7,000.00	0.32	3.34	0.34
Rest fraction*	0	5.11	0	0

**the rest fraction consists of unburned organic material and mineral material from the oversize fraction that is fed back into the sorting process. Table 20 in Appendix VI includes more detailed information about the material received by the sorting plant, the sorting efficiency, and the expected amount of material sorted out.*

The treatment of the sorted organic fraction is disregarded as it only constitutes 0.25% of IBA sorted and is assumed to have minor influence on the results of the study. Similarly, oversize materials are not included in the allocation because they are either included in the organic fraction or Fe metal fraction or are fed back into the sorting process.

Transportation

Sysav requires all logistic companies to use biodiesel and fulfill the Euro 6 emission standard. Biodiesel is also used in vehicles at the Spillepeng recycling center. Therefore, all transportation by lorry on site as well as transportation to the final disposal of MIBA and to the construction site are assumed to be based on biodiesel. Only the construction machinery used at the sorting facility was modelled using data on conventional diesel due to a lack of data on the use of biodiesel in construction machinery. Appendix VI includes a detailed description of transported material, mode of transportation, distances, and data sources. All distances were measured with Google Earth.

Aging process

The final process included in the materials production stage is the aging of MIBA. During the carbonation process, CO_2 is bound in the material. The measured amount of 37 kg of CO_2 per ton of MIBA was derived from a previous study conducted by Sysav (Johansson & Lönnebo Stagnell, 2016). CO_2 sequestration was inventoried as negative emissions to air.

4.2.3 Impact assessment

The LCIA focuses on the assessment of the GHG mitigation potential and resource efficiency from using secondary material. The environmental performance of the two scenarios is compared in terms of different environmental impact categories included in the updated version of the ILCD 2011 Midpoint+ method and the single-issue indicator CED.

The ILCD 2011 Midpoint+ method is an updated version of a LCIA methodology developed based on a comprehensive review of best existing practices for characterization modelling conducted by the Joint Research Centre of the European Commission. For a set of sixteen impact categories, the assessment identified the best among existing characterization models at mid-point and end-point level (Hauschild et al., 2013).

The method only uses indicators at the midpoint level and is developed for the European context. It provides normalization factors for the average European citizen (EU 27) based on domestic inventories in the year 2010 (Benini et al., 2014). It is the preferred method for LCAs in the construction sector. A list of the 15 included impact categories and their indicators is shown in Table 4. The category ionizing radiation environment was excluded from the analysis after a first screening had shown that the impacts are negligible as well as because it is classified as "interim" by the Joint Research Center and therefore not recommended for use.

Table 4: List of impact categories of the ILCD 2011 Midpoint+ method used for the impact assessment.

Impact category	Abbreviation	Unit
Climate change	GWP	kg CO2 eq
Ozone depletion	ODP	kg CFC-11 eq
Human toxicity, non-cancer effects	HTnc	CTUh
Human toxicity, cancer effects	HTc	CTUh
Particulate matter	PM	kg PM2.5 eq
Ionizing radiation human health	IR	kBq U235 eq
Photochemical ozone formation	POF	kg NMVOC eq
Acidification	AP	molc H+ eq
Terrestrial eutrophication	TEP	molc N eq
Freshwater eutrophication	FEP	kg P eq
Marine eutrophication	MEP	kg N eq
Freshwater ecotoxicity	ETfw	CTUe
Land use	LU	kg C deficit
Water resource depletion	WRD	m3 water eq
Mineral, fossil & renewable resource depletion	RD	kg Sb eq

The focus lies on the evaluation of resource efficiency and climate impacts. Resource efficiency is assessed by analyzing the impacts related to the provision of the benefits provided by the system, i.e. the functional unit (1km of road in the Malmö harbor area and the disposal of 6,960 t MIBA). The impact on the availability of natural resources is assessed based on three impact categories (i.e. "resource efficiency indicators" or resource efficiency indicators in "sensu stricto", see Section 3.1.3): resource depletion water, land use, and resource depletion, mineral, fossil and renewable.

The remaining impact categories included in the ILCD method that are used here are indicative for the impacts on the natural environment and human health resulting from the extraction process and use of natural resources (i.e. "emission efficiency indicators" or resource efficiency indicators in "sensu lato", see Section 3.1.3*)*. Among others, this includes the GWP100 impact category that is used as a proxy for the assessment of the GHG mitigation potential of scenario B in comparison to scenario A.

The CED indicator provides a single score to assess impacts on the availability of natural resources as well as the impacts caused by the extraction and use of resources on the natural environment. It provides characterization factors for energy resources split into five impact categories that are summed up without weighing: non-renewable, fossil; non-renewable, nuclear; renewable, biomass; renewable, wind, solar, geothermal; and renewable, water (Huysman et al., 2015).

The CED falls into the first category of indicators for the assessment of abiotic resources in LCA that are based on the inherent properties of materials (see Section 3.1.3) (Schneider et al., 2016). Here, it used in addition to the ILCD method to provide a single score indicator for resource efficiency and to give a more complete picture of the impacts and potential benefits of using secondary materials compared to virgin material. The CED indicator is redundant to the ILCD method. Therefore, it should be seen as a separate assessment (Chen et al., 2010).

4.2.4 Interpretation

The interpretation stage typically includes the identification of significant issues, tests to assess completeness, sensitivity and consistency, as well as drawing conclusions and recommendations. Significant issues in terms of inventory items and impact categories were identified by conducting different contribution analysis (e.g. for single processes and process groups) in the SimaPro software. Different sensitivity analyses were conducted to evaluate the significance of certain assumptions and methodological choices on the results of the study. The influence of different transportation distances was tested because it was identified to be an influential variable with regard to the environmental load in the literature review as well as during pre-screening results. A second allocation procedure was tested to investigate the influence of the methodological choice of the partitioning approach for the sorting process on the results of the study as recommended by the ISO standard.

5 Life Cycle Assessment results

This chapter presents the results of the LCA. Results are presented separately for each scenario (5.1 and 5.2) and then compared (5.3). The content of this section addresses RQ2 that asks for the environmental impacts in terms of selected impact categories from using post-incineration residues for road construction in the specific case, compared to virgin raw materials, based on the developed LCA model presented in Section 4.

5.1 Life cycle impact assessment of scenario A

5.1.1 ILCD 2011 Midpoint+

Figure 13 shows the normalized potential environmental impacts of scenario A. The results are expressed in "Person Equivalents" (PE) and show the relative importance of the impacts by relating the characterization results to the flow of emissions or resources of an average European person in the year 2010. It needs to be emphasized that the results for both scenarios are not the full environmental impacts of the road construction. Only those parts that are different for both scenarios were considered and, therefore, results need to be interpreted as relative to the other. The values for the total environmental impacts of scenario A without normalization are displayed in Table 5, Section 5.3.1. The exact numeric values of the normalized characterization results are included in Table 22 in Appendix IX. Figure 24 in Appendix VII further shows a network diagram of scenario A including all processes that contribute more than 0.1% to the overall environmental burdens (single score) of the product system.

The different colors in Figure 13 show the contribution of different process groups to the impact categories. Orange shows the contribution of all processes related to the alternative disposal of MIBA in a slag compartment of a sanitary landfill including its transportation. Grey shows the contribution of all processes related to the production and transportation of crushed rock.

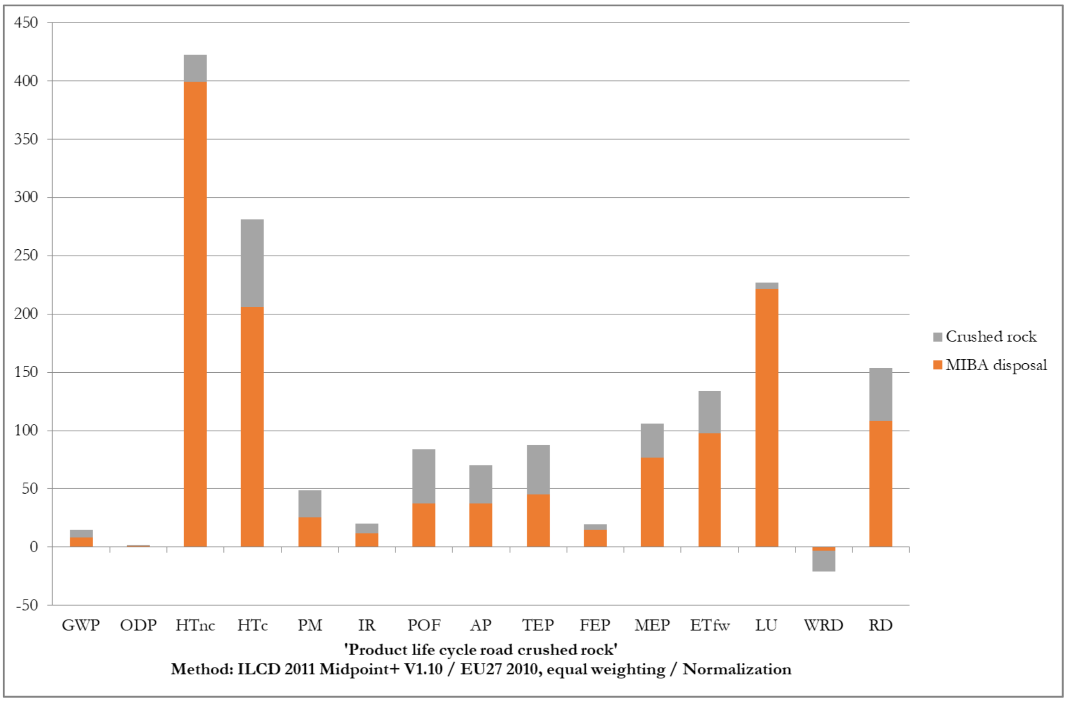


Figure 13: Normalized LCIA results for scenario A grouped into processes related to the alternative disposal of MIBA and processes related to the production of crushed rock.

The product system shows positive results for all impact categories except WRD. The negative results for WRD are mainly contributed by the production of crushed rock (86.4%) and are predominantly caused by the blasting and production of explosives and to a lesser extent by processes related to the transportation with biodiesel. In both cases, negative scores result from the production of aluminum and related energy processes in Saudi Arabia. The highest impact scores after normalization are HTnc, HTc, LU, RD, and ETfw. The major contributor to all environmental impacts except WRD are processes related to the alternative disposal of MIBA.

With regard to natural resource related indicators other than WRD, the alternative disposal of MIBA contributes 94.7% of LU and 70.5% of RD. For both, transportation is the major contributor. Land use is mainly driven by the production of rape seeds for biodiesel. LU impacts from the quarry and landfill processes appear negligible with 1.6% and 3.8% of the normalized impacts, respectively. RD results mainly from transportation and to a lesser extend from processes related to blasting.

Figure 14 shows the results of a group analysis where all major transportation processes for both the production of crushed rock and MIBA disposal were included in one group. The analysis indicates that transportation also contributes a large share of the impacts for emission related indicators including HTnc (95.5%), ETfw (76.6%), FEP (73.4%), MEP (67.1%), and HTc (66.4%). Due to the comparatively long transportation distance of MIBA to its final disposal, it holds the largest share of impacts resulting from transportation.

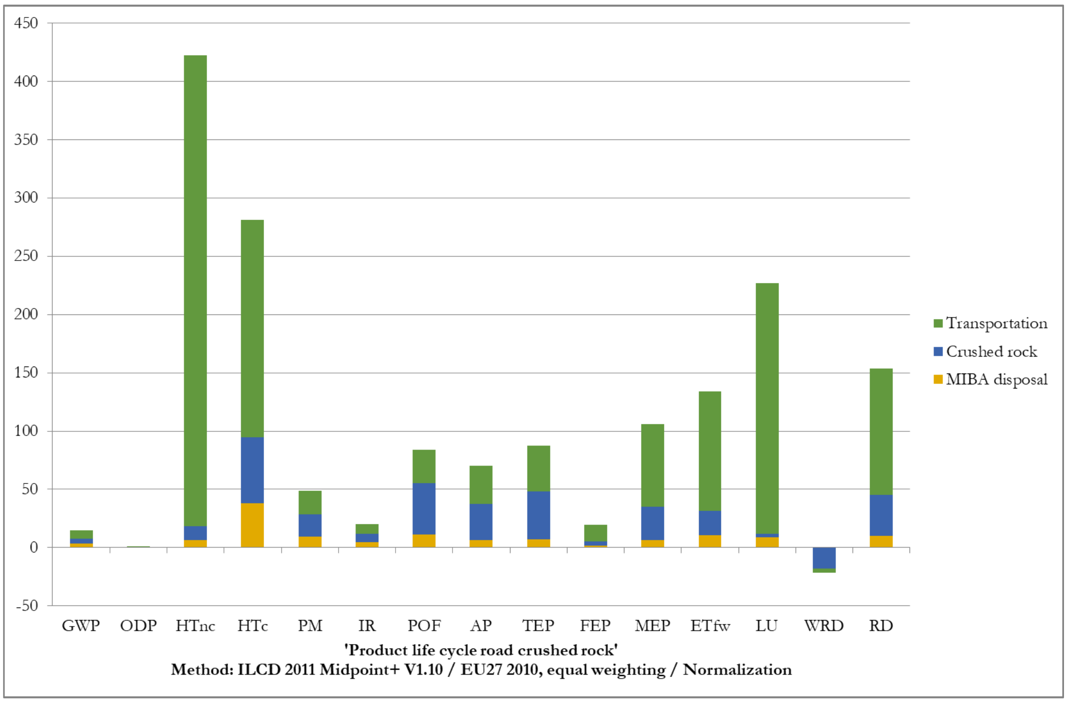


Figure 14: Normalized LCIA results of scenario A grouped into contributions of transportation processes, the crushed rock production process, and processes related to the alternative disposal of MIBA.

Figure 14 further shows the share of impacts that result only from the production process of crushed rock and the disposal of MIBA in a landfill. With regard to natural resource related categories, their share in RD is noticeable. The extraction process of rock material by means of blasting is the major contributor. Emission related impact categories for which the production of crushed rock and landfilling of MIBA contribute significant shares include MEP, TEP, AP, POF and HTc. Important contributing processes include blasting and to a lesser extend the use of diesel in construction machinery during the production of crushed rock as well as on the landfill.

Climate impacts amount to 137,560 kg-CO_2 eq. The normalized result is low compared to other emission related impacts such as HTnc or resource related impacts such as LU. Transportation processes contribute 48.0%, the production of crushed rock 27.3%, and processes related to the disposal of MIBA 24.7% to the GWP. In the production of crushed rock, blasting and diesel burned in building machines contribute relatively more than other processes. Similarly, diesel consumption contributes relatively more to the GWP than other landfill processes.

5.1.2 Sensitivity analysis

Transportation of MIBA over a relatively large distance to its final disposal is a major contributor to all environmental impacts. Therefore, and because it is a major varying assumption in previous papers, a second (future) transportation scenario was considered. The second scenario assumes the transportation over 1 km and should resemble the disposal of MIBA in the Spillepeng landfill in the immediate vicinity of the incineration plant.

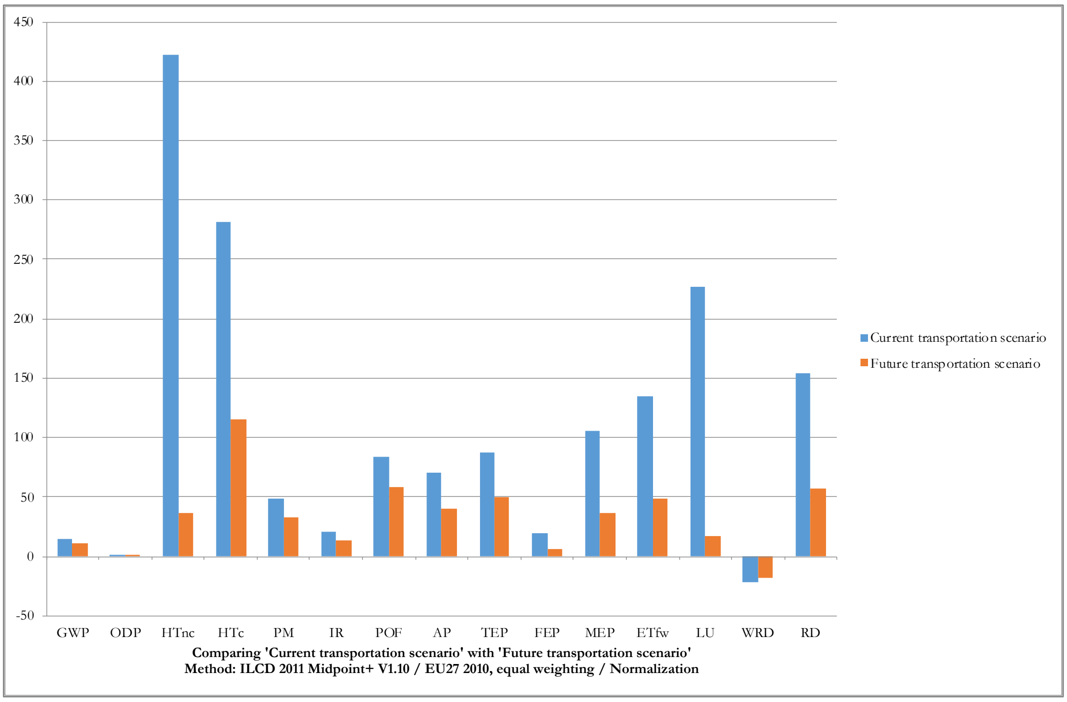


Figure 15: Comparison of the normalized LCIA results of scenario A considering two transportation scenarios for MIBA to its final disposal.

Figure 15 shows a drastic decrease in almost all impact categories if a shorter transportation distance is assumed. Particularly, HTnc and LU and to a slightly lesser extend MEP, ETfw, RD and HTc. This is, for example, due to the reduced LU for biodiesel production. Reduced transportation distance has a slight effect on the GWP and WRD. The small impact on the GWP can be explained by the relative importance of other processes. For example, blasting, diesel burned in building machines in the quarry and landfill, and the transportation of primary material to the construction site using fossil diesel.

5.1.3 Cumulative Energy Demand

The impact assessment using the CED as an LCIA methodology for scenario A results in a primary energy demand of 4.1 TJ (see Figure 16). The majority of primary energy demand is nonrenewable (71.65%); 65% fossil, 5.73% nuclear, and 0.02% nonrenewable biomass. The share of renewable primary energy demand (28.35%) is almost entirely renewable biomass (26.10%). Only a minor share comes from renewable, wind, solar, and geothermal (0.24%) and water (2.01%).

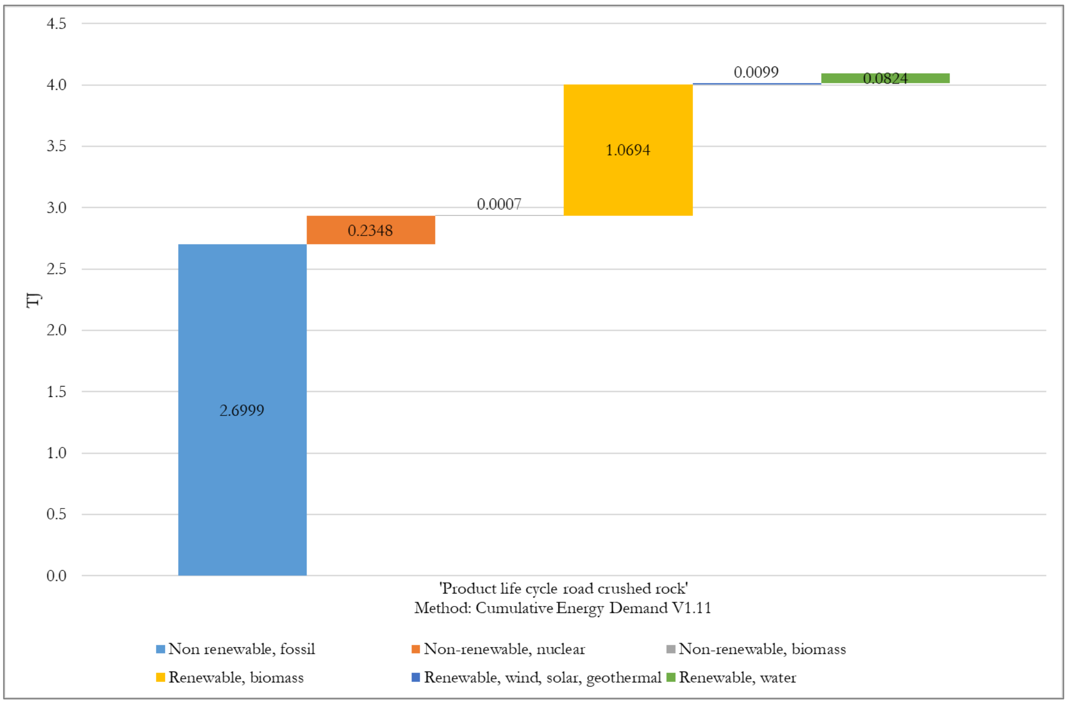


Figure 16: The CED for scenario A. Different colors indicate different sources of energy.

Processes that cause the highest impact on fossil energy demand are transportation by biodiesel and diesel, the production of pitch for landfill construction, and diesel burned in building machines. Nuclear energy stems from the use of electricity. Nuclear currently has a share of around 34% in the Swedish electricity production (IAEA, 2020). Demand for nonrenewable biomass comes from transportation with biodiesel and to a lesser extend from processes related to blasting. Impacts on the renewable energy demand are primarily contributed by using biodiesel for transportation (renewable biomass) and electricity consumption in different processes (water). See Figures 29 and 30 in Appendix VII for a contribution analysis of transportation, the production of crushed rock, and processes related to the landfilling of MIBA.

5.2 Life cycle impact assessment of scenario B

5.2.1 ILCD 2011 Midpoint+

The normalized potential environmental impacts of scenario B are displayed in Figure 17. For the analysis, groups were made for processes related to the treatment of MIBA including all on-site transportation on the one hand, and the transportation to the construction site in the Malmö harbor area on the other. The values for the total environmental impacts of scenario B without normalization are displayed in Table 5, Section 5.3.1. The exact numeric values of the normalized characterization results are included in Table 22 in Appendix IX. Figure 31 in Appendix VIII further shows a network diagram including all processes of the product system.

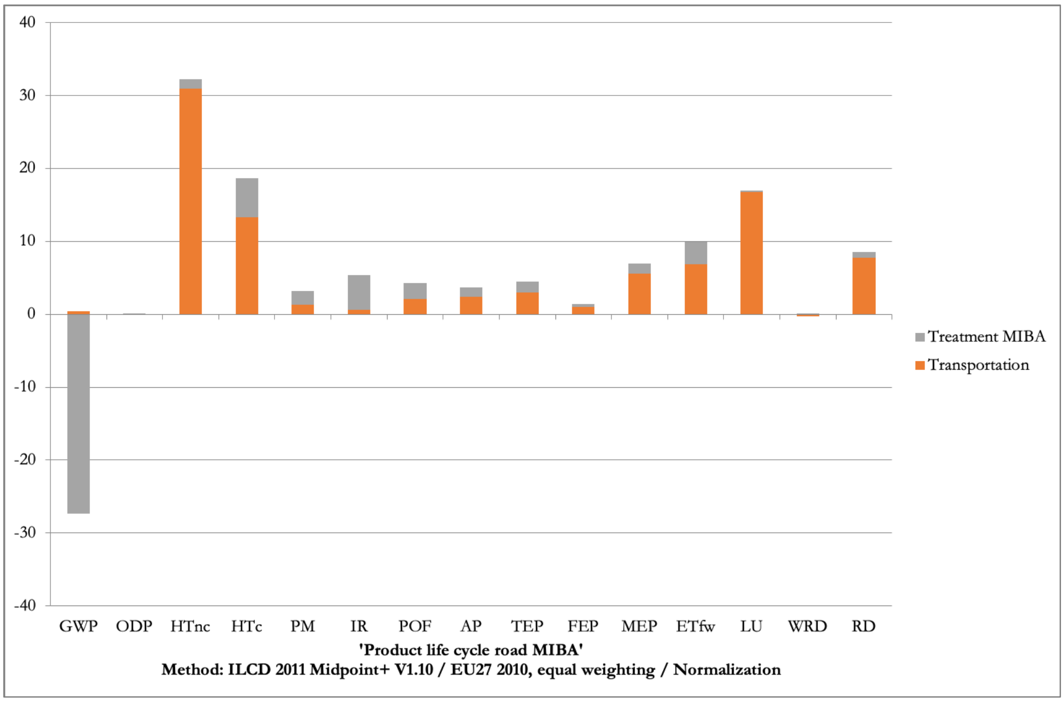


Figure 17: Normalized LCIA results for scenario B grouped into processes related to the treatment of MIBA and transportation processes.

The product system shows positive results for all impact categories except GWP and WRD. The negative impact for GWP results from the aging process. Due to carbonation, MIBA takes up carbon from the ambient air. In total, approximately 257,520 kg per functional unit calculated based on results from tests conducted on the MIBA from the Sysav incineration plant. The slight negative results for WRD result from processes related to the transportation with biodiesel.

The highest positive impact scores after normalization are HTnc, HTc, LU, and ETfw. All impact category scores except GWP, IR, and to a lesser extend PM, are driven by the transportation to the construction site. All impacts related to the depletion of natural resources are almost exclusively contributed by the transportation process. This is also because only emissions and resource use from the on-site transportation, handling of bottom ash with construction machinery, and the electricity consumption of the sorting facility were considered. Land use or water consumption related to the sorting facility were not included in the inventory. On the contrary, transportation contributes to a greater extend due to the production of biomass for biodiesel (LU) and the resources that go into the manufacture and maintenance of the vehicles (RD). Contributions from the MIBA treatment to emission related impact categories mostly stem from the diesel consumption of the wheeled loaders and the excavator.

The sequestration of carbon during the aging process overcompensate emissions of GHGs from other processes including transportation. Therefore, the product system leads to net environmental (climate) benefits. The benefit amounts to 248,584 kg-CO_2 eq. The normalized climate benefits are in the same order of magnitude than the highest positive impact categories, HTnc and HTc.

5.2.2 Sensitivity analysis

A sensitivity analysis was performed on the methodological choice for partitioning environmental burdens from the sorting process. The allocation factors are shown in Table 3 in Section 4.2.2. As for the economic allocation, the rest fraction was not considered in calculating allocation factors.

The normalized values of the environmental impacts resulting from the different allocation approaches are shown in Figure 18. Mass allocation leads to higher impacts across all impact categories. This is because MIBA constitutes the major material outflow in terms of mass from the sorting process. If the rest fraction is disregarded, 89.81%. Although different metal fractions constitute a relatively small output in terms of mass, they have a much higher economic value which is why a larger share of the environmental burden is allocated to them if economic allocation is applied.

Using mass allocation instead of economic allocation influences emission related impacts more than resource related categories. ODP, PM, IR and POF show the highest relative increases. However, the normalized values for these impact categories remain relatively low, particularly for ODP. The least influence is on GWP and WRD because the processes with the most impact on these categories, aging and transportation, happen after the sorting process. Figure 34 in Appendix VIII shows the characterization results for both scenarios using a percentage scale.

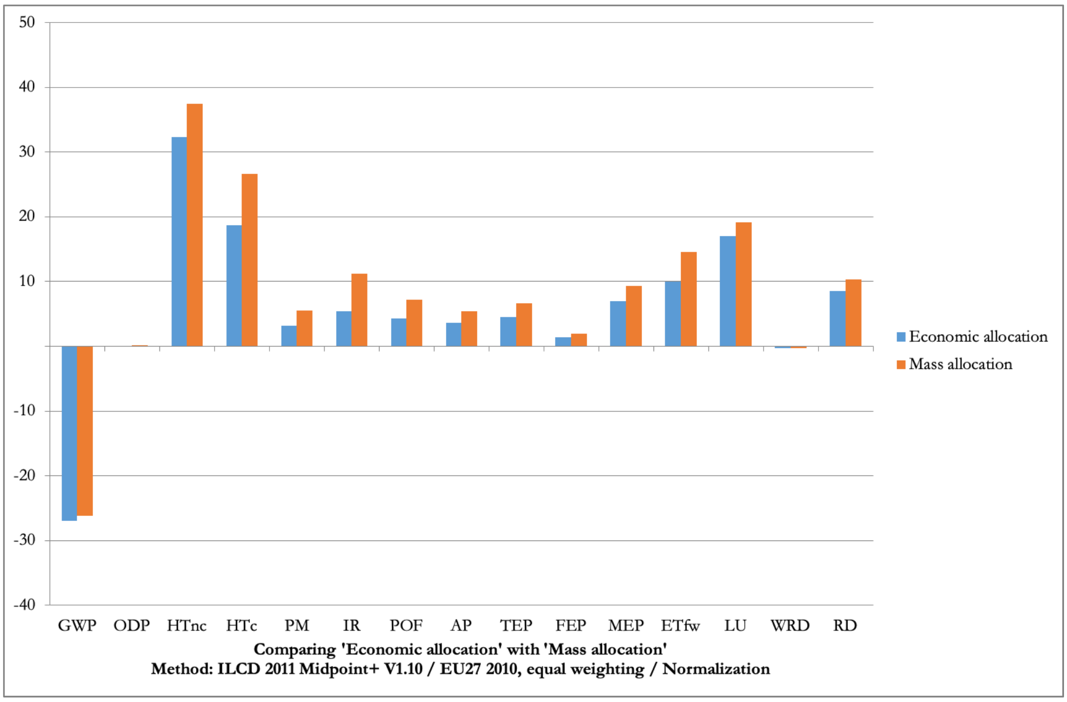


Figure 18: Comparison of the normalized LCIA results for scenario B using economic allocation and mass allocation for partitioning the environmental burdens of the sorting process.

5.2.3 Cumulative Energy Demand

The CED of scenario B amounts to 0.333 TJ (see Figure 19). The majority of primary energy demand is nonrenewable (64.25%); 41.50% fossil, 22.74% nuclear, and 0.01% nonrenewable biomass. The share of renewable primary energy demand (35.75%) consists mostly of renewable biomass (26.94%) and water (7.56%). Only a minor share of 1.24% comes from renewable, wind, solar, and geothermal.

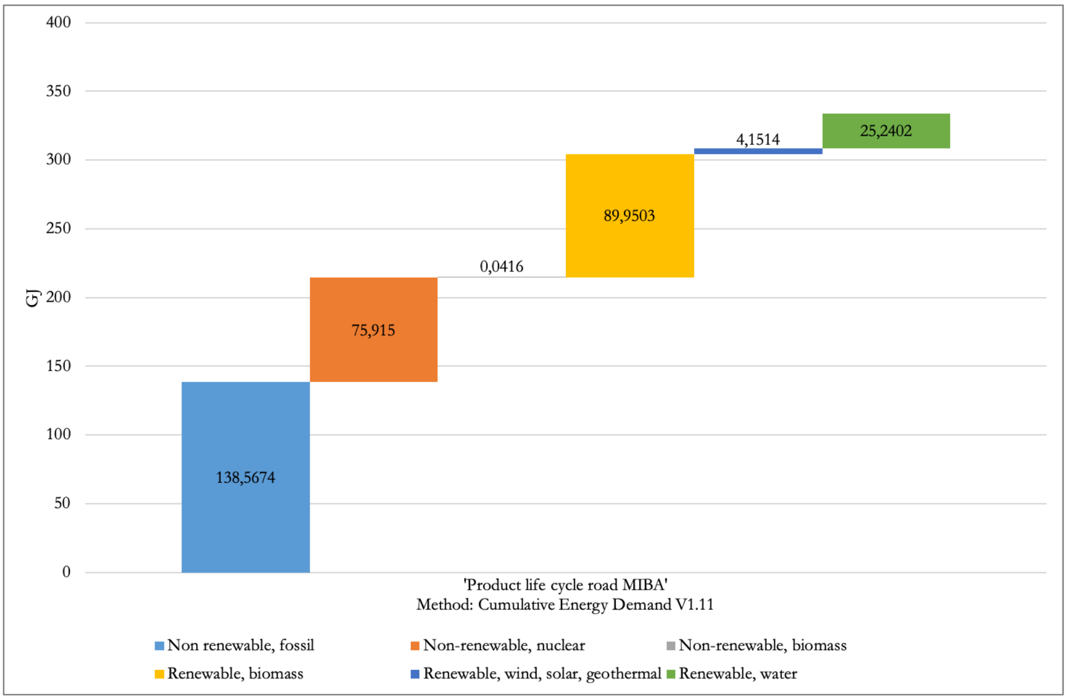


Figure 19: The CED for scenario B. Different colors indicate different sources of energy.

The overall primary energy demand is contributed in relatively equal shares by the electricity consumption of the sorting facility, diesel burned in the construction machinery used at the sorting facility, and the transportation to the construction site. Nonrenewable fossil energy is consumed by construction machinery while nuclear energy demand comes from the electricity consumption of the sorting facility. Renewable biomass is contributed by the transportation with biodiesel and renewable water by electricity consumption. See Figures 35 and 36 in Appendix VIII for a contribution analysis of transportation and processes related to the treatment of bottom ash to the CED.

5.3 Comparison between scenario A and B

5.3.1 ILCD 2011 Midpoint+

Table 5 shows the characterization results for scenario A, scenario B, and the difference between them. The normalized values for both scenarios are shown in Table 22 in Appendix IX. Figure 20 shows the normalized results in the form of a bar chart. The difference can be interpreted as the environmental impact reductions that could be achieved by producing product system B instead of product system A respecting limitations in terms of the methodology, assumptions, and the limited impact coverage. These limitations are further discussed in the Section 6.

Table 5: Characterization results for scenario A, scenario B, and the difference between the two scenarios.

Impact category	Unit	Scenario A	Scenario B	Difference (A – B)
Climate change (GWP)	kg CO_2 eq	137,560.6628	-248,584.1237	386,144.7865
Ozone depletion (ODP)	kg CFC-11 eq	0.0306	0.0022	0.0284
Human toxicity, non-cancer effects (HTnc)	CTUh	0.2251	0.0172	0.2079
Human toxicity, cancer effects (HTc)	CTUh	0.0104	0.0007	0.0097
Particulate matter (PM)	kg PM2.5 eq	184.8633	11.9573	172.9060
Ionizing radiation HH (IR)	kBq U235 eq	23,150.1908	6,058.5398	17,091.6510
Photochemical ozone formation (POF)	kg NMVOC eq	2,654.4113	136.7153	2,517.6959
Acidification (AP)	molc H+ eq	3,311.0685	171.9971	3,139.0714
Terrestrial eutrophication (TEP)	molc N eq	15,392.6888	791.2961	14,601.3928
Freshwater eutrophication (FEP)	kg P eq	28.8236	2.0231	26.8005
Marine eutrophication (MEP)	kg N eq	1,794.3931	117.5583	1,676.8347
Freshwater ecotoxicity (ETfw)	CTUe	1,173,214.8213	87,122.3809	1,086,092.4404
Land use (LU)	kg C deficit	16,970,471.3542	1,270,466.1919	15,700,005.1623
Water resource depletion (WRD)	m^3 water eq	-1,724.1727	-19.6250	-1,704.54780
Mineral, fossil & renewable resource depletion (RD)	kg Sb eq	15.5169	0.8606	14.6563

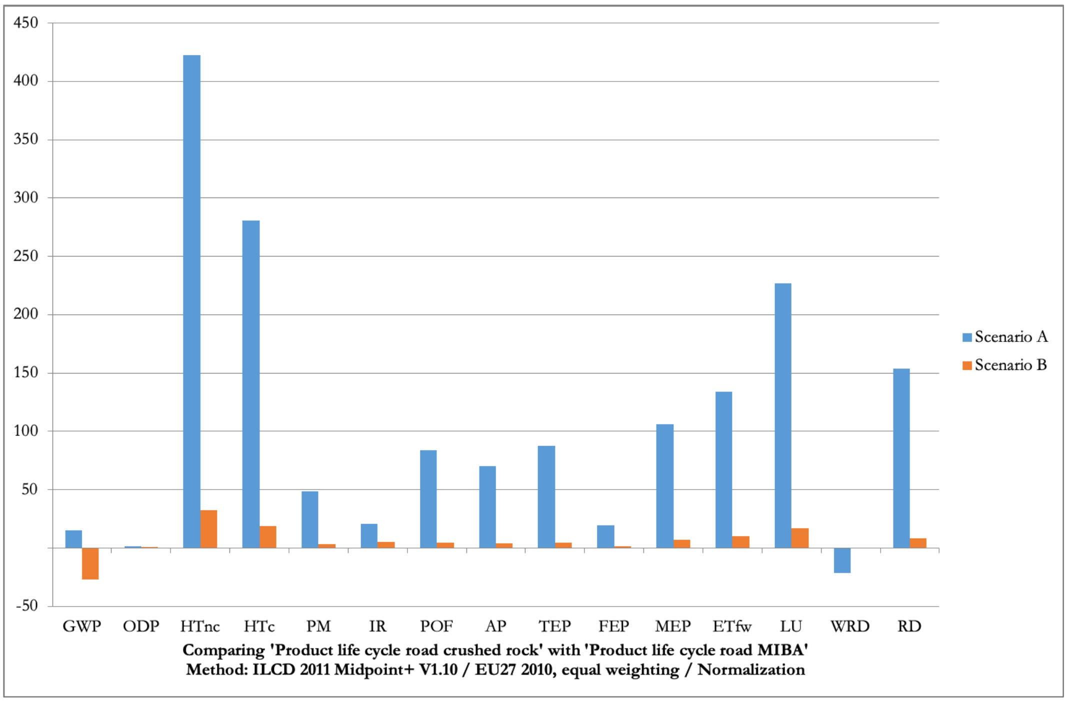


Figure 20: Comparison of the normalized LCIA results of scenario A and scenario B.

Scenario B results in significantly lower potential environmental impacts than scenario A. The three impact categories with the highest absolute difference are LU, HTnc, and HTc. In both product systems, the impacts contributed by transportation processes are significant. With regard to resource depletion related impact categories, LU and WRD are in both systems predominantly caused by the production of biomass and aluminum for transportation processes as well as for explosives for blasting in scenario A. Similarly, in both scenarios resource use for the manufacture and maintenance of the transportation vehicles is the major contributor to RD.

The highest relative difference between the scenarios is achieved for some emission related impact categories, including POF, AP, TEP, and to a lesser extend MEP, ETfw, PM, and HTc. These are the impact categories where the quarry process and the alternative disposal of MIBA contribute significant shares to the environmental load.

With regard to GWP, the results are different. While the impact for scenario A is positive but rather small compared to other impact categories, scenario B shows a net benefit that is large compared to other impact categories. Table 6 shows the weighted sum of the life cycle emissions of major GHGs per functional unit for both product systems. The difference between the two product systems amounts to **386,145 kg-CO_2 eq**. Therefore, scenario B constitutes a **281%** improvement in terms of GWP compared to scenario A. This could be interpreted as the GHG mitigation potential of product system B, road with MIBA, compared to product system A, road with crushed rock. Limitations to this interpretation are discussed in the following section.

Table 6: Characterization results (in kg-CO2 eq) for the impact category GWP for scenario A and scenario B.

Scenario A	Scenario B	Difference	Improvement (%)
137,561	-248,584	386,145	281

5.3.2 Sensitivity analysis

Different tests were conducted in order to analyze the significance of important assumptions on the results of the study. These were introduced in the previous sections and include the choice of the allocation method for partitioning the environmental burdens from the sorting process between MIBA and scrap metals in scenario B (B2) as well as the transportation distance of MIBA to its final disposal in scenario A (A2) (see Table 7).

Table 7: Different scenarios tested in the sensitivity analysis.

Scenario	MIBA disposal (km)	Allocation	Description
Baseline scenario A (A1)	64.5	-	Scenario described in section 4.2.2
Short transportation A (A2)	1	-	Assumes that all MIBA is disposed of in the Spillepeng landfill
Baseline scenario B (B1)	-	Economic	Scenario described in section 4.2.2
Mass allocation B (B2)	-	Mass	Uses mass allocation instead of economic allocation to partition the environmental burdens of the sorting process

Figure 21 shows the normalized values of the environmental impacts for each scenario. Reducing the transportation distance for MIBA in A2 drastically reduces its environmental impacts in almost all impact categories. However, if compared to B1, even with a transportation distance of only 1 km the road using primary material has higher impacts in all impact categories except LU and WRD. The latter are mainly caused by transportation with biodiesel of MIBA to the construction site in B1.

Using mass allocation instead of economic allocation increases the overall potential impacts of scenario B. If A2 is compared to B2, the road with MIBA also shows higher impacts for HTnc.

In the context of this study, the comparison of scenario A2 and B2 can be seen as a comparison of the assumed reasonably best case for scenario A (i.e. short transportation distance for MIBA to final disposal) and a reasonably worst case for scenario B (i.e. mass allocation).

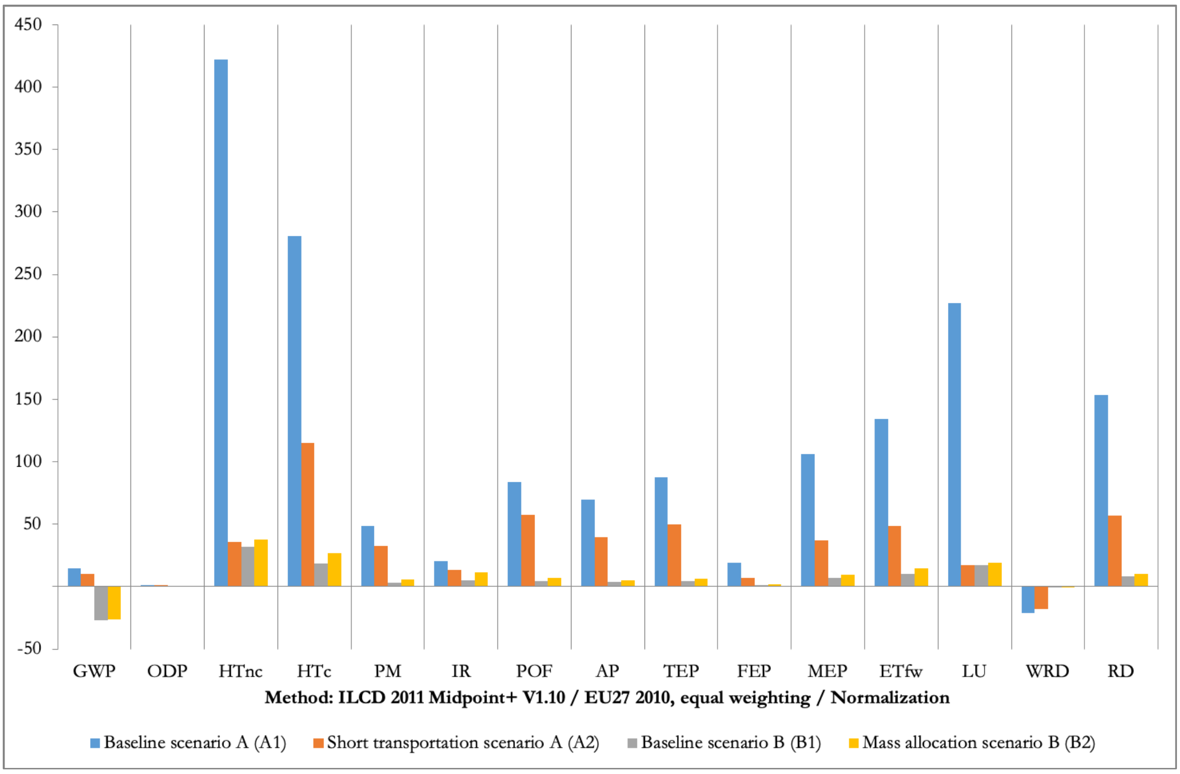


Figure 21: Comparison of the normalized LCIA results for scenarios A1, A2, B1, and B2.

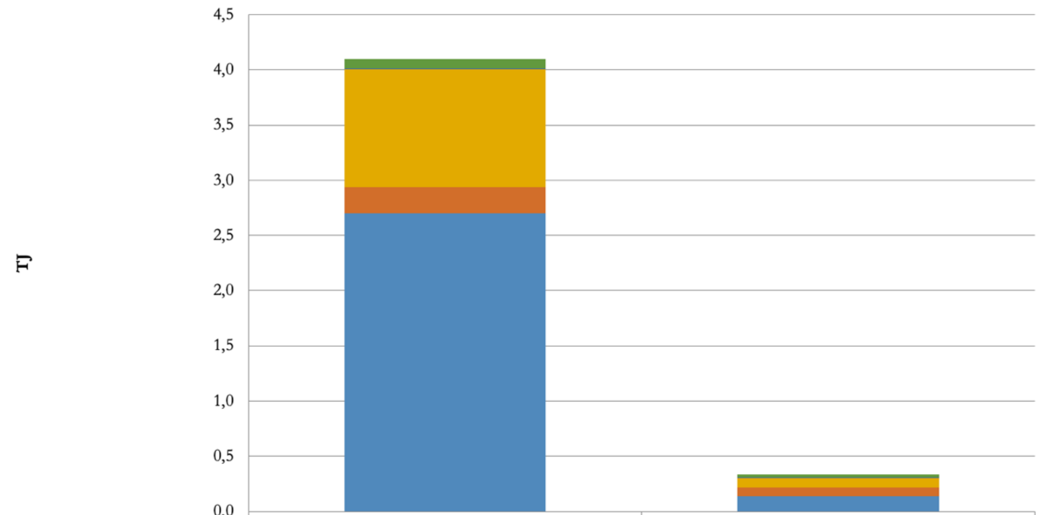
The chosen scenarios have less influence on climate related impacts (see Table 8). The short transportation scenario leads to a reduction of the climate impacts of scenario A by 41,460 kg-CO_2 eq. Considering A2 and B1 results in a CO_2 mitigation potential of **344,685 kg-CO_2 eq**. The relative improvement of B1 compared to A2 is **359%**. Mass allocation for scenario B leads to a slight reduction in the climate benefits of scenario B by 7,165 kg-CO_2 eq. Considering A2 and B2, results in a difference of **337,520 kg-CO_2 eq**. The improvement then amounts to **351%**.

Table 8: Characterization results for the GWP impact category (in kg-CO2 eq) for different scenarios.

A1	A2	B1	B2	Difference 1 (A2-B1)	Improvement (%)	Difference 2 (A2-B2)	Improvement (%)
137,561	96,101	-248,584	-241,419	344,685	359	337,520	351

5.3.3 Cumulative Energy Demand

Figure 22 shows the CED for both product systems (A1 and B1). The primary energy demand for the road with MIBA is significantly lower compared to the road with crushed rock. The largest share of primary energy demand comes from non-renewable sources in both product systems. While the majority of nonrenewable energy in A1 is based on fossil, in B1 a considerable part is based on nuclear. This is because a greater share of energy in B1 stems from electricity. The larger share of renewable biomass in A1 compared to B1 is caused by the transportation with biodiesel, whereas the larger share in energy from hydropower in B1 compared to A1 results from the use of electricity for sorting.



	Scenario A	Scenario B
Renewable, water	0,0824	0,0252
Renewable, wind, solar, geothe	0,0099	0,0042
Renewable, biomass	1,0694	0,09
Non-renewable, biomass	0,0007	4,16E-05
Non-renewable, nuclear	0,2348	0,0759
Non renewable, fossil	2,6999	0,1386

Figure 22: Comparison of the CED for scenario A and scenario B.

6 Discussion

The aim of this thesis is to improve the current understanding of, as well as quantify the resource efficiency and greenhouse gas (GHG) mitigation potential, from using MIBA as construction material, compared to virgin material. While the last section has contributed the quantification, in the following, the results are discussed with the intention to contribute to a better understanding of the potential environmental impacts and benefits from using MIBA as a secondary construction material and thereby addressing RQ3.

First, the results are discussed against previous findings of LCA studies on MIBA utilization in road construction. The discussion particularly examines the specific context of the case study and identifies parameters that are influential in determining the environmental performance of using secondary material in lieu of virgin raw material. The discussion is expanded with findings from conversations with practitioners in the field of bottom ash recycling from Germany and Denmark as well as toxicologists from Sweden and Denmark. The second part of the discussion focusses on methodology. A particular focus lies on the limitations of LCA, the assessment of resource efficiency and climate benefits, and data quality. The section closes with some more general reflections on the use of MIBA in road constructions and the type of environmental assessment needed to provide practitioners and particularly policy makers with the right information to make informed decisions on the preferable alternative.

6.1 Results in the context of previous knowledge

The LCA results indicate significantly less environmental impacts from the product system including MIBA, compared to the same system using virgin raw material. Previous product oriented LCA studies on MIBA utilization in roads in Denmark and China, came to results that point in a similar direction. However, different scopes and assumptions render studies not directly comparable. For example, Geng et al. (2010) have chosen a different functional unit ($1m^2$ of road) and assume the substitution of bottom ash in more than one structural layer of the road. Olsson et al. (2006) do an impact assessment based on normalized inventory flows and assume that the use of bottom ash makes necessary changes in the strength of other road layers. Furthermore, potential environmental impacts from the content and potential leaching of toxic substances from MIBA were considered in both studies, while these were excluded from the scope of this study.

6.1.1 Transportation

One general conclusion from this study that coincides with earlier studies is that the distance materials are transported is an important variable. The transportation of materials is a major contributor to the overall environmental impacts of both scenarios. The sensitivity analysis for scenario A shows that environmental impacts are highly sensitive to the transportation distance of MIBA to the landfill. A shorter transportation distance leads to reductions in both emission and resource related impact categories particularly HTnc, HTc, LU, RD, and ETfw. The environmental impacts of scenario B in most categories are significantly lower compared to those of scenario A because the transportation distance of MIBA towards the construction site is only about one sixth of that of crushed rock. In addition, scenario A includes the transportation of MIBA to its final disposal, which is more than twice as long as the transportation of crushed rock to the construction site and about 14 times longer than the transportation distance of MIBA in scenario B.

The transportation distance is also an important assumption in previous studies. Olsson et al. (2006) assume equal transportation of MIBA and natural aggregates, which is supposed to resemble conditions in the Stockholm area. Geng et al. (2010) assume a shorter transportation distance for MIBA (10 km), than for crushed rock (60km) based on a specific case in Shanghai.

Birgisdóttir et al. (2006), on the other hand, assume a 2.5 times longer transportation distance for MIBA than for natural gravel. It is argued that this resembles conditions in Denmark because there are significantly more quarries than waste incinerators. All three studies consider the alternative disposal of MIBA in a landfill. However, only Geng et al. (2010) state the distance considered for the transportation to a landfill of 20 km.

The transportation distance is very much dependent on the local context. This is not only shown by the different assumptions taken in studies but was also corroborated in discussions with practitioners in the field of bottom ash recycling from Germany and Denmark. According to representatives of Strabag AG (Germany), it is not uncommon that MIBA is transported for several hundred kilometers for its final disposal, such as in the case of Munich (Respondent 7, personal communication, March 13, 2020). Similarly, representatives of Afatek (Denmark) mentioned that often, long transportation distances towards construction sites are necessary (Respondent 8, personal communication, March 16, 2020). In the case of Copenhagen, although bottom ash is usually produced where construction aggregates are needed (i.e. in cities), it happens frequently that the supply does not correspond to the demand in terms of time. MIBA is continuously produced but storage capacities are limited. As a result, the company is often forced to deliver MIBA to construction sites that are further away (Respondent 8, personal communication, March 16, 2020).

In the case of Sysav, the current conditions lead to a result that favors the product system with secondary material because a potential construction site is close by. At the same time, the relatively long transportation distance towards the alternative disposal, contributes to the positive picture with regard to the reuse of MIBA. These favoring factors should be taken into account when considering the use of MIBA for its potential environmental benefits in the specific context, as well as elsewhere. It could also be argued that in the future the increasing scarcity of natural aggregates and landfill capacities might lead to increasingly longer transportation distances. For example, one expert mentioned that it is estimated that in Copenhagen within the next five years, the transportation distance of natural gravel will increase to an average of 100 km due to the depletion of gravel pits that are closer (Respondent 8, personal communication, March 16, 2020). In this case, the depletion of resources increases MIBAs environmental performance relative to primary material in the long run. Thus, the transportation distances prove to be a crucial parameter for the environmental performance.

6.1.2 Fuel choice and land use

A potential tradeoff is caused by the fuel used in transportation vehicles. Renewable fuels (biodiesel) are chosen by Sysav for their supposedly better environmental performance, particularly with respect to climate impacts. However, while renewable fuels do contribute less to the impact category GWP than transportation with fossil diesel, they contribute to eutrophication, cause toxic impacts on the human health, and particularly lead to resource depletion in the form of land use. Land use impacts in scenario A and B are almost exclusively due to the choice of fuel type. Climate benefits are here gained at the expense of resource depletion, i.e. by a decrease in resource efficiency in the strict sense. However, it needs to be noted that a transportation process for vegetable methyl ester with an inventory modelled for Switzerland was chosen to resemble the use of biodiesel in Sweden. To a certain extent, the production of biodiesel in Sweden could cause different impacts.

The large share of transport in the environmental impact LU compared to the small share of the landfilling process and the production of crushed stone, is a rather surprising result. The use of land for landfilling as well as the destruction of scenic landscapes through the extraction of primary aggregates in quarries, is frequently mentioned as important environmental impacts that could be avoided by using secondary material. However, the importance of these potential benefits is not reflected in the results. Previous studies have not classified and characterized resource use into impact categories at all (Birgisdóttir et al., 2006; Geng et al., 2010; Olsson et al., 2006). This shows a potentially important gap in the current literature and highlights a key limitation of the LCA method, namely the lack of temporal and geographical differentiation, which is further discussed in Section 6.2.2.

6.1.3 Type of substituted material

While transportation is identified to be a key parameter influencing the environmental performance of both systems, the sensitivity analysis done for scenario A also shows that even under the most favorable conditions in terms of transporting MIBA to its final disposal in Spillepeng, scenario B still shows less environmental impacts in almost all impact categories. Potential impacts that remain significantly higher in scenario A than scenario B include HTc, PM, POF, AP, TEP, MEP and ETfw among emission related impacts and RD for resource related impacts.

One process that contributes relatively more to this difference compared to other processes is the blasting of rock and the production of explosives. This shows that the potential benefits are also dependent on the type of material MIBA replaces in the road. For example, natural gravel which is dredged as opposed to crushed rock which is normally blasted. While almost no natural gravel is used anymore in Sweden, it is the main source of construction aggregates in Denmark. Sensitivity analyses regarding the type of primary material that is substituted were conducted by Butera et al. (2015) and Allegrini et al. (2015) in the Danish context. The use of crushed rock instead of natural gravel led to an increase of potential environmental impacts in most categories, but not to an extent that the overall conclusions of the studies changed. However, if natural gravel is used, other potential environmental aspects such as ground water quality might become more prominent particularly in Sweden (Roth & Eklund, 2003).

6.1.4 Energy use and the importance of considering the alternative disposal

Previous studies found that emissions in both product systems are mostly dependent on energy use. A road with primary raw materials includes two processes that require larger quantities of energy: the production of crushed rock and landfilling. The quarrying of rock materials includes several energy-intensive processes such as the operation of construction machinery, the crushing of rock, and the manufacture of auxiliary products such as explosives. Operating a landfill also requires the use of heavy machinery. As a result of the higher energy demand, the road including crushed rock causes more emissions and potential impacts. In Olsson et al. (2006), the production of crushed rock was identified to be twice as energy intensive as the landfill process.

In this thesis, the CED was used to indicate the primary energy demand of both scenarios. In general, the results agree with Olsson et al. (2006) in that emissions and potential impacts are closely related to the use of energy, as indicated by the large difference between the CED of scenario A and B. Furthermore, if the transportation that differs greatly between studies is disregarded, the production of crushed rock and landfilling are also the most energy intensive processes. However, here, the landfill process is identified to be twice as energy intensive as the production of crushed stone. A possible explanation for the difference compared to Olsson et al. (2006) could be the more detailed datasets used for the landfill process and the methodology for measuring the primary energy demand.

The CED considers the total direct and indirect energy input of a product system. This means that not only the direct energy input for the different life cycle stages of the investigated product system is considered but also the energy demand for the production processes of auxiliary materials and consumables used in the investigated product system. Furthermore, the provision of the final energy used is related to processes like mining, transformation, and transportation that in turn require energy. Either way, the results further corroborate the previously stated importance of considering the alternative disposal of MIBA in comparative studies. In this particular case, the alternative disposal is even more important due to the long transportation distance to the landfill.

Processes related to the treatment of MIBA, i.e. the materials production of the secondary material, only require one third as much energy as the production of crushed stone. This shows that the production of secondary materials is not burden free, but in the particular case of Malmö, and considering the limited scope of the LCA, the environmental consequences from producing the secondary aggregates, compared to the primary aggregates that fulfill the same technical requirements, are significantly less.

6.1.5 Climate benefits and carbonation

Results with regard to the GWP are different from previous studies. While earlier research agrees that a road built with MIBA has less potential climate impacts (either characterized into GWP (Geng et al., 2010) or by showing the amount of emissions of major GHGs (Olsson et al., 2006)) this study finds a much higher difference between scenario A and B. This is mainly caused by considering the sequestration of carbon during the aging of MIBA that was not included in previous studies regarding MIBA. However, it received some attention in LCAs on construction and demolition waste management.

Butera et al. (2015) compared the use of construction and demolition waste in road subbases against its disposal in landfills in Denmark. It was concluded that the inclusion of carbonation results in a significant decrease of GWP, leading to considerable net environmental benefits. The study assumes the full carbonation of the waste resulting in 57 kg CO_2 being bound per ton of waste. While it is acknowledged that complete carbonation is unrealistic, carbonation of only 15% in this particular case would already outbalance GWP impacts from transportation. Transportation is assumed to be 60 km, 30 km from the demolition site to a treatment facility and 30km from the treatment facility to the construction site.

However, the values used for carbonation must be considered with care. The sequestration of 37 kg per ton of MIBA was measured at the Spillepeng recycling center under normal conditions on MIBA that was aged for six month (Johansson & Lönnebo Stagnell, 2016). Literature shows a wide spread of values between 12-251 kg/t (e.g. Cho et al., 2011; Rendek et al., 2006). The amount of carbon sequestrated depends on different factors such as the exposure to air, the chemical composition, the grain size of the material, the moisture content, and most notably the storage time. The inclusion of carbonation in scenario B, is also based on the assumption that MIBA that is disposed of in a landfill in scenario A, is not stored in open air conditions for a considerable amount of time – otherwise carbonation would take place in scenario A as well. This assumption is supported by the fact that storage capacity is limited and therefore MIBA, which is not reused, is transported directly to its final disposal.

Nevertheless, the results show that although the inclusion of carbonation in this study is subject to many uncertainties and assumptions, it appears to be an important issue that has not yet been considered in LCAs of MIBA reuse. This finding contradicts statements of practitioners that were of the opinion that carbonation would be insignificant compared to the benefits that could be achieved by avoiding the production of primary material (Respondent 8, personal communication, March 16, 2020). The case of carbonation also shows that additional properties of materials, such as the ability to sequestrate carbon, can have a significant influence on the environmental performance of secondary materials.

6.2 Reflections and limitations

In the following, some major limitations are discussed. Limitations correspond to the assumptions made for the LCA model, the modelling approach chosen, the limited impact coverage of this study, and data quality. Subsequently, the generalizability of the case to other contexts is discussed. The section concludes with some reflections regarding the question of whether MIBA should be used as road construction material and what kind of assessment would be needed to identify an environmentally preferable option in the specific case.

6.2.1 Critical assumptions

An LCA study models a product systems life cycle. Modelling necessarily requires assumptions and a simplification of a complex reality (Goedkoop et al., 2016). Assumptions regarding the characteristics of the product system and different considered scenarios can limit the usability of the results (JRC-IES, 2010a). Some exemplary assumptions made in this study are summarized in Table 9.

Table 9: Examples of assumptions made to develop the LCA model.

Scenario A	Scenario B	General
- Alternative disposal is in a slag compartment in a sanitary landfill - MIBA sent to alternative disposal in a landfill is not aged for a significant amount of time	- "zero-burden" assumption - No land use of the sorting facility - Negligible water use during MIBA treatment - MIBA is aged for six month	- MIBA has the same functionality as crushed rock and makes no changes in the structure of the road necessary - Using MIBA leads to a product of the same durability and only insignificant changes in the construction, maintenance, and use life cycle stages - Roads are continuously reconstructed and have therefore no EoL

The effect of assumptions regarding the transportation distance to the alternative disposal of MIBA, and the allocation procedure for partitioning the potential environmental burdens from the sorting process, have been tested in sensitivity analyses. With regard to the allocation approach, partitioning based on mass values leads to higher impact scores in all categories. Nevertheless, economic allocation is chosen as a default for the model. It seems to be more adequate, because, although the different metal fractions only have a small mass, they have a considerable economic value that constitutes a major motivation to treat bottom ash in the current form.

In fact, although it is the least preferable option as indicated by the ISO standard, the price is often times the only allocation approach that can be used consistently throughout a products life cycle (Ekvall, 2020). It is one of the most widely applied partitioning approach in LCA including studies on recycling and the use of secondary materials in the construction sector (Ardente & Cellura, 2012; Chen et al., 2010). The influence of the assumed transportation distance was already discussed in detail in Section 6.1.1.

Further potentially influential assumptions were made as part of this study. While due care was taken during the research to justify these assumptions, for example, by interviewing experts and reviewing scientific literature, it needs to be acknowledged that the assessment was done based on many subjective decisions by the LCA practitioner, while not all assumptions could be tested in sensitivity analyses.

One example that could have a significant influence on the results and that demonstrate how influential modelling choices can be is the zero-burden assumption. All reviewed LCA studies took the decision to not assign any environmental burdens from incineration to the investigated product system. If assumed that the incineration process is the production process of the product (and not waste) "MIBA", in an ALCA, one could argue that a share of the environmental burden would need to be allocated to the investigated product system.

In CLCA, the product system could be credited with the avoided burden from the production of heat and/or electricity. Allegrini et al. (2015) used MIBA utilization in road construction as a case study to assess the influence of altering the system boundaries on the potential environmental impacts caused by leaching of toxic substances in a consequential approach. The study concludes that while leaching impacts are dominant in scenarios with narrow system boundaries that only consider the road and the avoided production of primary aggregates, impacts are negligible if wider system boundaries are used that include the waste management system (i.e. metal recycling from MIBA, incineration, and collection of MSW) due to the dominance of the incineration process.

Although the study by Allegrini et al. (2015) is based on a consequential approach, is conducted from a waste management perspective and therefore, includes further processes that are not part of a product LCA, it still does demonstrate the importance of the incineration process. If incineration would be included as the production process of MIBA in this LCA, its environmental performance in comparison to a road with crushed rock could be expected to decrease significantly.

6.2.2 Method-related limitations

LCA has been continuously developed throughout the last decades and by now is widely applied in many sectors as well as in support of legislative processes (Finnveden et al., 2009; Richter, 2019). However, results of LCAs should not be regarded as "the truth" or the "real environmental impact" of a product system. There are numerous limitations resulting from the methodology and research in multiple areas of its application is needed to further develop it (Finnveden et al., 2009; Schneider et al., 2016).

Results of LCA studies are static, since models are largely linear. Local or regional specificities such as different sensitivity to pollutants or their ecological value (e.g. high biodiversity or bio-productivity of land) are not taken into account (Baumann & Tillman, 2004). There is also a lack of temporal information on the simultaneous emissions of processes outside the product system and on background concentrations of other substances. While this is less important for some impacts that are global like GWP, it can be critical for local impacts including human and environmental toxicity, or the abundance of certain resources (Finnveden et al., 2009). Considering the special circumstances of the analyzed case, i.e. an area with highly bio-productive land, it could be argued, for example, that the potential LU impacts resulting from landfilling and the production of primary road construction material are higher in reality.

Further methodological limitations relate to the choice of the modelling approach. While a strictly attributional approach would not consider any indirect effects outside the investigated product system, here, system boundaries are expanded in order to include the benefits/burdens from the alternative disposal of MIBA. This suggests that the results show the environmental consequence of producing one scenario (A or B) instead of the other, however, it does not. What the results show is an estimation for how much lower or higher the potential environmental impact of one scenario is, compared to the other. This does not mean that the production of scenario B (road with MIBA) actually leads to a reduction of the environmental impact as indicated by the results of the study (e.g. by 386,145 kg CO_2 eq per km of road). The difficulties to interpret LCA results that are neither strictly attributional nor consequential have been highlighted by several scholars (e.g. Brander et al., 2009; Plevin et al., 2014).

In order to investigate the consequence of the decision to build a road with MIBA instead of crushed rock, a strictly CLCA would be needed. And even then, because of the large uncertainties inherent to prospective LCA studies, one could never know if the results show the actual consequences (Ekvall et al., 2005).

A CLCA would need to investigate long and short-term indirect effects of the decision to produce scenario B. Particularly long term effects can have significant environmental consequences (Ekvall, 2020). Such a study would, for example, need to consider if the demand for the next unit of landfill capacity makes necessary the development of a new landfill in an area with bio-productive land. Ideally, it would also investigate potential positive and negative feedbacks and behavioral effects. One long-term feedback could be that the use of MIBA leads to its establishment on the market for construction aggregates and to a normalization of the practice to use MIBA as secondary material. This, in turn, could contribute to solving a major challenge for waste incineration in Sweden: What to do with bottom ash? This reduces the need to search for alternative, potentially more sustainable waste management practices, and to a strengthening of the framing of incineration as a circular practice because its residues are used in "circular" applications.

While there are many would's and could's to this stream of thought, it stands to reason that this could further contribute to a lock in of the current waste management system to less sustainable practices (Corvellec et al., 2013). While such an investigation would be the right one to answer the question "what would happen if we decide to use MIBA from the Sysav waste incinerator in the Malmö harbor area instead of virgin raw material?", CLCA are complex and require a considerable amount of resources, competences and speculative inductions, which may render the insights unreliable.

6.2.3 Limited impact coverage

LCA has as its aim to quantify all relevant emissions and resource consumption as well as the resulting health and environmental impacts and depletion of resources from a product system. However, the complexity of product systems, methodological limitations, and a general lack of data and resources necessary to generate data on inventory flows make the complete coverage of all relevant impacts impossible. Most notably, in this study this includes the deliberate exclusion of the potential leaching of heavy metals. This was done despite previous studies indicating that this is a major contributor to potential toxic impacts, as well as to the overall environmental burden of road constructions that include MIBA.

The decision to exclude leaching from the study was taken based on discussions with experts from the field of toxicology. The toxicology part of LCA was said to be underdeveloped and local impacts such as from leaching are particularly prone to the methodological limitations of LCA regarding the lack of spatial and temporal differentiation. Toller (2008) highlights this when concluding that the potential environmental impacts from trace element leaching are "probably more dependent on when, where, and at what concentrations (…) than on the total leaching" (p. 29). At the same time, the integration and interpretation of both global and local impacts and their comparison in the same model presents itself difficult. One respondent referred to it as "comparing apples to oranges" (Respondent 9, personal communication, September 3, 2020). A further reason was the lack of resources and background data for the development of a sufficiently accurate leaching model and its integration in the LCA model.

Other impacts are not covered because they are not part of the latest LCIA methodologies. This particularly relates to the assessment of natural resource depletion elaborated in Section 3.1.3. As a result, the depletion of biological resources and its effect on the natural environment, i.e. impacts on biodiversity and important life support functions, are not included. Similarly, although the GWP characterization method is regarded as one of the most developed models, it does not give a complete picture of the climate impacts. It misses characterization factors for GHGs and important impact pathways. The consideration of the latter is of particular importance if the focus of the study is to assess resource efficiency and potential climate benefits.

Furthermore, missing inventory data for multiple processes limits the extent to which this study can be regarded as "complete." Among others, these include the infrastructure and capital goods for the sorting facility; processes that are different between the two investigated scenarios but that are considered insignificant (see Appendix III); and dust emissions during MIBA management.

Finally, while LCA tries to include "everything," what is not yet known cannot be included in the inventory or characterization. This was particularly highlighted in a discussion with an environmental NGO regarding the emissions of persistent organic pollutants from the management of MIBA in the Netherlands (Respondent 13, personal communication, March 3, 2020; Zero Waste Europe, 2019). New chemicals are developed on a daily basis and so changes the composition of waste and the substances that end up in MIBA.

6.2.4 Inventory data quality

Data quality criteria help to evaluate to what extent the quality of data supports conclusions and recommendations. An important concept in LCA is the representativeness in terms of technology, geography, and time. It "characterizes in how far the inventory as a whole is depicting the functional unit(s) and/or reference flow(s) of the process or system" (JRC-IES, 2010a, p. 325). Key to representativeness is the appropriateness of the chosen individual dataset to represent the actual process in the context of the product system. For example, if an appropriate dataset for the production of crushed rock in terms of time, geography, and technology was chosen, it contributes positively to the representativeness of the inventory.

The appropriateness of chosen background datasets is of particular importance. In several instances, datasets from the Ecoinvent database were not available for the geographical scope of the study, for the specific technology used, or were based on extrapolations from studies that date back more than 20 years. In terms of technology, for example, a dataset for the burning of diesel in construction machinery was used to resemble two loaders and one excavator operating at the sorting facility. While this can be considered "common practice" in LCA, the appropriateness of this dataset is lower than, for example, the datasets used for transportation processes, which were rather precisely chosen based on the Euro emission class, gross-weight tonnage, the fuel used, and the specific use (e.g. freight) of the transport vehicles. The technological data representativeness is therefore likely to cause some error.

Multiple Ecoinvent datasets were not available for the geographic context of Malmö/Sweden. This includes the processes used to resemble landfilling (Switzerland), the production of crushed rock (Brazil), and transportation with biodiesel (Switzerland). Where possible, such datasets were adapted to Swedish conditions particularly by adjusting energy related processes, which are usually the most important ones with regard to their contributions to environmental impacts.

However, this was not possible for all processes such as for the transportation with biodiesel. The geographic representativeness is therefore a particular issue with regard to this study.

Foreground data was collected for the most recent time period. Data received from Sysav was based on measurements from within the last two years or estimates made for now or the short-term future. However, most Ecoinvent datasets are based on studies from the early 2000s (e.g. Doka, 2003; Kellenberger et al., 2007). While, again, this is common practice in LCA, it introduces further uncertainties.

Test runs with datasets from different databases, showed that results differ depending on what background data is selected, for example, for the production of crushed rock from the ELCD and Ecoinvent databases. The discrepancies between results in terms of potential environmental impacts that using different datasets lead to have also been highlighted by previous studies on the energy consumption and GHG emissions from pavement rehabilitation. Four datasets were compared, including Ecoinvent datasets and inventory data from a study by Stripple (2001) which is one of the most widely used inventory studies in road construction (e.g. Birgisdóttir et al., 2006; Olsson et al., 2006). According to this, the environmental burdens vary by 25% depending on what dataset is chosen (Azari Jafari et al., 2016).

Why one should not rely excessively on databases is also shown by the results obtained for the impact category WRD. Both road scenarios show a negative impact for WRD. In both cases, the negative impact results from energy processes related to the production of aluminum in Saudi Arabia that is used in the supply chain for biodiesel as well as explosives. In these processes, water is included from the global market and then released in Saudi Arabia. Hence, water is withdrawn from a region where it is abundant and released in a region where it is scarce, resulting in a significant negative impact. However, water from the global market was probably only selected in this case because no other data was available (Delem et al., 2019). The results for WRD are therefore most like not accurate.

Considering these limitations regarding data quality, it becomes apparent that the job of an LCA practitioner with regard to data selection is to deal with tradeoffs and to balance different data quality criteria. Should a dataset that fits the geographical scope be selected, or the one that is more appropriate with regard to technological representativeness but inventoried for a different country or region? Should a recently developed dataset for a slightly different technology be preferred over an older dataset that is more appropriate for the investigated process in terms of technology? Therefore, data quality is an important consideration and inherent limitation in conducting an LCA.

6.2.5 Generalizability

The single-case study approach chosen in this study naturally does not allow for generalizations based on statistical inference. Similarly, results of LCAs are usually not generalizable beyond the specific case without certain model and data adjustments. Among other factors, production practices, electricity mixes, and available materials are all processes that are often times specific to the location. Results are therefore unlikely to be transferable over national or regional boundaries. This applies all the more to studies of roads, since roads are very diverse (Santero et al., 2010). However, generalizing based on statistical inference was not the aim when selecting a single case study research design. A case study was chosen to identify relevant data and as "a mode of organizing data in terms of some chosen unit" (Goode and Hatt, 1952, p. 339 in Blaikie & Priest, 2019). Therefore, it served as "a choice of what is to be studied" (Stake, 2005, p. 445).

Nevertheless, the results of this study provide rich and close to reality learnings. It provides an example for the environmental performance of the secondary use of incineration residues compared to primary material in a specific case. It identifies parameters that influence the environmental performance as well as potential tradeoffs. Generalizations for such elements can be established based on judgement and identifying similarities and reoccurring patterns (Stake, 2005).

6.2.6 Should bottom ash be used in road construction in Malmö? – Some further reflections

Due to the limitations elaborated above, this study alone is not suited to identify a preferable alternative, because it only covers a limited number of potential impacts. The aim of this study was also not to provide an answer to the question of whether MIBA should be used in road constructions or not. However, the exploration of the case study generated a lot of knowledge regarding the use of MIBA and the assessment of its environmental performance from literature, interviews, and by conducting an LCA study. Some more general reflections regarding this knowledge are provided in the following.

Sustainability and environmental assessment are value-based. Depending on what you consider important, the method is adjusted, and different aspects are prioritized (Roth & Eklund, 2003). This includes the process of setting system boundaries. For example, while the toxicologist considers the leaching of pollutants as the key impact and prefers to use narrow system boundaries, a sustainability scientist may prefer to look at the "bigger picture," taking into account regional and global impacts and benefits. The case of bottom ash shows how difficult a "fair" assessment becomes when clear tradeoffs are involved. In this case, between leaching/local/toxic impacts and the climate/global/non-toxic benefits. The integration of local risk assessment into LCA methodology is difficult and more research is needed so that an assessment does not end up comparing "apples to oranges." LCA does not substitute for local risk assessments (Finnveden et al., 2009) and decision makers are better advised to use multiple environmental assessment tools to decide between such alternatives, particularly when tradeoffs are involved (e.g. Berlin & Iribarren, 2018; Carpenter et al., 2007; Olsson et al., 2006).

The results of the LCA show that there could be some application contexts where using MIBA in lieu of primary material, creates larger benefits than in others. However, in the case of Malmö, the legal basis for the use of incineration residues in secondary applications, is solely based on the total contents and leaching values of pollutants. Whereas the potential benefits that were identified in this study are not considered in local regulations.

Previous risk assessments also conducted in the context of Malmö, suggest a rather low risk to the human health and the environment due to leaching (Avfall Sverige, 2015; Van Praagh et al., 2018). Furthermore, its application in an industrial area limits the potential of exposure to humans or damage to water protection areas. A change of policy priorities towards the protection of natural resources and climate change provides a further argument to consider assessments using broader system boundaries, in addition to local risk assessments. Ideally, such an assessment would then assess the actual consequence of building a road with secondary aggregates. This would ensure that effects on other systems beyond the investigated product system, are also taken into account. For example, if the decision contributes to a lock in of the waste management system to a lower level of the waste management hierarchy, and therefore causing potentially significant negative long-term environmental effects.

Conclusion

This thesis sought to improve the current understanding of, as well as quantify the resource efficiency and greenhouse gas (GHG) mitigation potential, from using MIBA as construction material, compared to virgin material. This was done by first developing an appropriate LCA model to assess the environmental performance of a road built with MIBA, compared to a road built with primary raw material (RQ1). The model was then applied to a case study of reusing MIBA from the Sysav MSW incinerator in Malmö, in road construction (RQ2). Finally, the analysis of the results identified important parameters that influence the environmental performance of MIBA as a secondary material, compared to primary materials, in the specific case (RQ3). The thesis thereby contributes to a better understanding of the environmental consequences of utilizing incineration residues in road construction, as well as helping to address the current lack of understanding regarding the potential low carbon benefits, and emission savings from material efficiency, of CE strategies more broadly. By applying the model to a real case, it further provides evidence for local practitioners and political decision makers to make more informed decisions regarding the use of MIBA.

The assessment shows significantly less potential environmental impacts for the road scenario with MIBA, across all categories except WRD. Even for the considered best case in terms of transportation to a landfill in scenario A, and a mass allocation approach in scenario B, a road with secondary material performs better across almost all impact categories. The GHG mitigation potential of scenario B compared to scenario A amounts to a difference of 386,145 kg-CO2 eq, a 281% improvement. In other words, the GWP would decreases by 281% if crushed rock is replaced with MIBA. This significant improvement is mainly due to the carbonation of MIBA during aging. The total primary energy demand as indicated by the CED for scenario B is 3.763 TJ less than for scenario A. With regard to the materials production stage for both scenarios, the assessment shows that, in the specific case, the primary energy demand for producing secondary material is only one third that of the primary material.

Several important parameters were identified that are critical for the environmental performance of the road with secondary material, compared to the same road with primary material, in the specific case. The transportation distance of different materials was shown to be the largest contributor to environmental loads in both scenarios. Choosing biodiesel for transportation led to a tradeoff between GWP and resource related impact categories. The results further corroborate the importance of considering the alternative disposal scenario in assessments of secondary materials. The high energy demand of landfill processes and potentially long transportation distances to alternative disposal sites, can be important contributors to the environmental burden of the product system including primary material.

A further important consideration impacting the environmental performance of a road with MIBA relative to a conventional road scenario, is the type of primary material that is substituted. While crushed rock is used in Sweden, the improvements achieved by using MIBA in other countries like Denmark, where the use of natural gravel as construction aggregate is common, can be less significant. Finally, properties specific to the investigated material, i.e. carbonation and the ability to sequestrate carbon, were most influential in determining the climate benefits that could potentially be achieved by using secondary material instead of crushed rock.

The empirical findings are mostly in line with previous LCA studies on MIBA utilization from a product perspective. This thesis goes beyond earlier studies by also including resource related impact categories and by applying the CED to investigate the primary energy demand of the investigated product systems. It also highlights the importance of the carbonation process for the GHGs mitigation potential that was not part of earlier studies.

By including novel aspects, new areas that need consideration were identified. The characterization of resource use into impact categories has received less attention in reviewed studies. In a local context, however, the depletion of certain resources including bio-productive land, can be of particular importance. Methodological development is needed to more adequately account for such impacts. Further exploration on the impacts of carbonation on the GHG mitigation potential could be of interest as well.

However, the contribution of this study lies not only in its' novelty outlined above, but also in that it adds to the literature and research testing CE strategies in a specific case and context. By expanding the body of knowledge on life cycle-based assessments for different CE strategies, this study also contributes to the theoretical development of the CE concept, as highlighted by Blomsma & Brennan (2017). More work is needed to quantify and assess the diversity of material uses and circumstances from a life cycle perspective, in order to provide decision makers with the right information to advance a CE that actually decouples environmental consequences from economic development. While the specificity of this case naturally does not allow to copy paste the generated learnings to other cases, it does provide decision makers with important insights on what to look out for in similar contexts.

For actors that seek to make environmentally sound decisions on whether to use primary or secondary material, be it planning departments of public authorities or private companies, this study shows that such a decision is an act of balancing trade-offs and carefully considering the specific circumstances. Using secondary material is not environmentally preferable per se. However, the results indicate that there could be some application contexts where using MIBA in lieu of primary material, creates larger benefits than in others. For example, the Malmö Harbor case appears to be one such application context.

LCA can provide useful insights for decision makers. Considering LCA results in addition to local risk assessments seems particularly important when considering secondary materials, as they might provide benefits that are not captured by local studies with narrow system boundaries. However, this study also highlights potential limitations of LCAs due to limited impact coverage, subjective assumptions, methodological shortcomings, and general data quality. What questions an LCA really answers and which conclusions its limitations actually allow, needs careful consideration.

Finally, the LCA model developed in this thesis should be regarded as a starting point for further developing it into a more robust tool. Numerous things could be improved and refined. Adding currently missing aspects such as potential toxic impacts from leaching of heavy metals, or dust emissions during different life cycle stages, would improve the extent to which the model can be regarded as "complete." Further considerations include the infrastructure used for bottom ash treatment, other capital goods, or the inclusion of aspects that were identified during the thesis but deemed insignificant (e.g. see Appendix III). A different alternative disposal scenario such as the use as construction material on a landfill could be considered, and comparisons to other secondary materials, like construction and demolition wastes, be made. This thesis nevertheless gives a strong basis for further work and should be used for developing future research and informing decision making.

Bibliography

Allegrini, E., Butera, S., Kosson, D. S., Van Zomeren, A., Van der Sloot, H. A., & Astrup, T. F. (2015). Life cycle assessment and residue leaching: The importance of parameter, scenario and leaching data selection. *Waste Management*, *38*, 474–485. https://doi.org/10.1016/j.wasman.2014.12.018

Allegrini, Elisa, Vadenbo, C., Boldrin, A., & Astrup, T. F. (2015). Life cycle assessment of resource recovery from municipal solid waste incineration bottom ash. *Journal of Environmental Management*, *151*, 132–143. https://doi.org/10.1016/j.jenvman.2014.11.032

Ardente, F., & Cellura, M. (2012). Economic allocation in Life Cycle Assessment. *Journal of Industrial Ecology*, *16*(3), 387–398. https://doi.org/10.1111/j.1530-9290.2011.00434.x

Arm, M. (2003). *Mechanical properties of residues as unbound road materials: Experimental tests on MSWI bottom ash, crushed concrete and blast furnace slag* [PhD, KTH Royal Institute of Technology]. https://pdfs.semanticscholar.org/e737/4f5a9d7f9eefa910482494bbce5fbee2898c.pdf

Arm, M., Flyhammar, P., Grönholm, R., Kristensson, M., Lind, B., & Wik, O. (2017). *Användning och modifiering av metallseparerat slaggrus – potential och matchning, AMOD* (No. 10; Issue 10, p. 84). Avfall Sverige.

Astrup, T., Muntoni, A., Polettini, A., Pomi, R., Van Gerven, T., & Van Zomeren, A. (2016). Treatment and Reuse of Incineration Bottom Ash. In *Environmental Materials and Waste* (pp. 607–645). Elsevier. https://linkinghub.elsevier.com/retrieve/pii/B978012803837600024X

Avfall Sverige. (2015). *Utvärdering av miljöpåverkan vid användning av slaggrus baserat på utförda projekt* (No. 2; Utvecklingssatsningen för energiåtervinning, Issue 2, p. 96). Avfall Sverige.

Avfall Sverige. (2019a). *Swedish Waste Management 2018* (p. 47) [Annual Report]. Avfall Sverige. https://www.avfallsverige.se/fileadmin/user_upload/Publikationer/SAH_2019_EN.pdf

Avfall Sverige. (2019b). *Uppdaterat beslutsstöd för återvinning av slaggrus i specifika asfalttäckta anläggningskonstruktioner* (2019:14; Issue 2019:14, p. 111). Avfall Sveriges Utvecklingssatsning. https://www.avfallsverige.se/kunskapsbanken/rapporter/rapportera/article/uppdaterat-beslutsstod-for-atervinning-av-slaggrus-i-specifika-asfalttackta-anlaggningskonstruktione/

Azari Jafari, H., Yahia, A., & Ben Amor, M. (2016). Life cycle assessment of pavements: Reviewing research challenges and opportunities. *Journal of Cleaner Production*, *112*, 2187–2197. https://doi.org/10.1016/j.jclepro.2015.09.080

Balaguera, A., Carvajal, G. I., Albertí, J., & Fullana-i-Palmer, P. (2018). Life cycle assessment of road construction alternative materials: A literature review. *Resources, Conservation and Recycling*, *132*, 37–48. https://doi.org/10.1016/j.resconrec.2018.01.003

Baumann, H., & Tillman, A.-M. (2004). *The hitch hikers's guide to LCA: An orientation in life cycle assessment methodology and application*. Studentlitteratur AB.

BDI. (2020). Wirtschaft stemmt sich gegen Vorschläge der Länder zur Änderung der Mantelverordnung. *EUWID Recycling und Entsorgung*, *30*(7), 1–2.

Benini, L., Mancini, L., Sala, S., Manfredi, S., Schau, E. M., & Pant, R. (2014). *Normalisation method and data for environmental footprints*. Publications Office of the European Union. http://dx.publications.europa.eu/10.2788/16415

Berlin, J., & Iribarren, D. (2018). Potentials and limitations of combined life cycle approaches and multidimensional assessment. In E. Benetto, K. Gericke, & M. Guiton (Eds.), *Designing Sustainable Technologies, Products and Policies: From Science to Innovation* (pp. 313–316). Springer International Publishing. https://doi.org/10.1007/978-3-319-66981-6_34

Bilitewski, B., & Härdtle, G. (2013). *Abfallwirtschaft*. Springer Berlin Heidelberg. http://link.springer.com/10.1007/978-3-540-79531-5

Birgisdóttir, H. (2005). *Life cycle assessment model for road construction and use of residues from waste incineration.* [PhD, Technical University of Denmark (DTU)]. https://orbit.dtu.dk/en/publications/life-cycle-assessment-model-for-road-construction-and-use-of-resi

Birgisdóttir, H., Bhander, G., Hauschild, M. Z., & Christensen, T. H. (2007). Life cycle assessment of disposal of residues from municipal solid waste incineration: Recycling of bottom ash in road construction or landfilling in Denmark evaluated in the ROAD-RES model. *Waste Management*, *27*(8), 75–84. https://doi.org/10.1016/j.wasman.2007.02.016

Birgisdóttir, H., Pihl, K. A., Bhander, G., Hauschild, M. Z., & Christensen, T. H. (2006). Environmental assessment of roads constructed with and without bottom ash from municipal solid waste incineration. *Transportation Research Part D: Transport and Environment*, *11*(5), 358–368. https://doi.org/10.1016/j.trd.2006.07.001

Blasenbauer, D., Huber, F., Lederer, J., Quina, M. J., Blanc-Biscarat, D., Bogush, A., Bontempi, E., Blondeau, J., Chimenos, J. M., Dahlbo, H., Fagerqvist, J., Giro-Paloma, J., Hjelmar, O., Hyks, J., Keaney, J., Lupsea-Toader, M., O'Caollai, C. J., Orupõld, K., Pająk, T., … Fellner, J. (2020). Legal situation and current practice of waste incineration bottom ash utilisation in Europe. *Waste Management*, *102*, 868–883. https://doi.org/10.1016/j.wasman.2019.11.031

Blomsma, F., & Brennan, G. (2017). The emergence of Circular Economy: A new framing around prolonging resource productivity. *Journal of Industrial Ecology*, *21*(3), 603–614. https://doi.org/10.1111/jiec.12603

Bocken, N. M. P., de Pauw, I., Bakker, C., & van der Grinten, B. (2016). Product design and business model strategies for a circular economy. *Journal of Industrial and Production Engineering*, *33*(5), 308–320. https://doi.org/10.1080/21681015.2016.1172124

Brander, M., Tipper, R., Hutchison, C., & Davis, G. (2009). *Consequential and attributional approaches to LCA: A guide for policy makers with specific reference to greenhouse gas LCA of biofuels* (Technical Paper TP-090403-A; Issue TP-090403-A, p. 14). ecometrica press.

Bruder-Hubscher, V., Lagarde, F., Leroy, M. J. F., Coughanowr, C., & Enguehard, F. (2001). Utilisation of bottom ash in road construction: Evaluation of the environmental impact. *Waste Management & Research*, *19*(6), 545–556. https://doi.org/10.1177/0734242X0101900611

Brunner, P. H., & Rechberger, H. (2015). Waste to energy – key element for sustainable waste management. *Waste Management*, *37*, 3–12. https://doi.org/10.1016/j.wasman.2014.02.003

Butera, S., Christensen, T. H., & Astrup, T. F. (2015). Life cycle assessment of construction and demolition waste management. *Waste Management*, *44*, 196–205. https://doi.org/10.1016/j.wasman.2015.07.011

Carlson, A. (2011). *Life cycle assessment of roads and pavements: Studies made in Europe* (Technical Report No. 736A; VTI Report, Issue 736A, p. 22). Environment. http://www.diva-portal.org/smash/get/diva2:670607/FULLTEXT01.pdf

Carpenter, A. C., Gardner, K. H., Fopiano, J., Benson, C. H., & Edil, T. B. (2007). Life cycle based risk assessment of recycled materials in roadway construction. *Waste Management*, *27*(10), 1458–1464. https://doi.org/10.1016/j.wasman.2007.03.007

Chen, C., Habert, G., Bouzidi, Y., Jullien, A., & Ventura, A. (2010). LCA allocation procedure used as an incitative method for waste recycling: An application to mineral additions in concrete. *Resources, Conservation and Recycling*, *54*(12), 1231–1240. https://doi.org/10.1016/j.resconrec.2010.04.001

Council Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. OJ L 312, 22.11.2008, p. 3–30.

Čuček, L., Klemeš, J. J., & Kravanja, Z. (2015). Chapter 5 – Overview of environmental footprints. In J. J. Klemeš (Ed.), *Assessing and Measuring Environmental Impact and Sustainability* (pp. 131–193). Butterworth-Heinemann. http://www.sciencedirect.com/science/article/pii/B9780127999685000051

de Wit, M., Verstraeten-Jochemsen, J., Hoogzaad, J., & Kubbinga, B. (2019). *The Circularity Gap Report – 2019* (p. 56). Circle Economy. https://bfc732f7-80e9-4ba1-b429-7f76cf51627b.filesusr.com/ugd/ad6e59_ba1e4d16c64f44fa94fbd8708eae8e34.pdf

Delem, L., Allacker, K., & Wastiels, L. (2019). Effects of LCA impact categories and methodology on the interpretation of a building's environmental performance. In R. Caspeele, L. Taerwe, & D. Fangopol (Eds.), *Life Cycle Analysis and assessment in civil engineering: Towards an integrated vision* (pp. 863–869). Taylor & Francis Group.

Deviatkin, I., Havukainen, J., & Horttanainen, M. (2017). Comparative life cycle assessment of thermal residue recycling on a regional scale: A case study of South-East Finland. *Journal of Cleaner Production*, *149*, 275–289. https://doi.org/10.1016/j.jclepro.2017.02.087

Doka, G. (2003). *Life cycle Inventories of waste treatment services, part III: Landfills - Underground Deposits - Landfarming* (Ecoinvent Report No. 13; Issue 13, p. 137). Swiss Centre for Life Cycle Inventories. https://www.doka.ch/13_III_Landfills.pdf

EC. (2020a). *Circular Economy Action Plan*. European Commission.

EC. (2020b). *Resource Efficiency*. European Commission Environment. https://ec.europa.eu/environment/resource_efficiency/

Ecoinvent. (2019). *Dataset information (UPR): Slag landfill construction*.

Ekvall, T. (2020). Attributional and Consequential Life Cycle Assessment. In *Sustainability Assessment in the 21st century* (p. Ch.-4). IntechOpen. https://doi.org/10.5772/intechopen.89202

Ekvall, T., & Tillman, A.-M. (1997). Open-loop recycling: Criteria for allocation procedures. *The International Journal of Life Cycle Assessment*, *2*(3), 155–162. https://doi.org/10.1007/BF02978810

Ekvall, T., Tillman, A.-M., & Molander, S. (2005). Normative ethics and methodology for life cycle assessment. *Journal of Cleaner Production*, *13*(13), 1225–1234. https://doi.org/10.1016/j.jclepro.2005.05.010

Ekvall, T., & Weidema, B. P. (2004). System boundaries and input data in consequential life cycle inventory analysis. *The International Journal of Life Cycle Assessment*, *9*(3), 161–171. https://doi.org/10.1007/BF02994190

Enkvist, P., & Klevnäs, P. (2018). *The Circular Economy - A powerful force for climate mitigation: Transformative innovation for prosperous and low-carbon industry* (p. 176). Material Economics. https://materialeconomics.com/publications/the-circular-economy-a-powerful-force-for-climate-mitigation-1

Finnveden, G., Albertsson, A.-C., Berendson, J., Eriksson, E., Höglund, L. O., Karlsson, S., & Sundqvist, J.-O. (1995). Solid waste treatment within the framework of life cycle assessment. *Journal of Cleaner Production*, *3*(4), 189–199. https://doi.org/10.1016/0959-6526(95)00081-X

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. *Journal of Environmental Management*, *91*(1), 1–21. https://doi.org/10.1016/j.jenvman.2009.06.018

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D. W., Haywood, J., Lean, J., Lowe, D. C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., & Dorland, R. van. (2007). Changes in atmospheric constituents and in radiative forcing. In *Climate Change 2007. The physical science basis* (pp. 129–234). Cambridge University Press. https://www.research-collection.ethz.ch/handle/20.500.11850/8576

Geng, C., Chen, D., Sun, W., & Liu, P. (2010). Life cycle Assessment for Roadbase Construction Using Bottom Ash from Municipal Solid Waste Incineration in Shanghai. *2010 International Conference on E-Product E-Service and E-Entertainment*, 1–6. https://doi.org/10.1109/ICEEE.2010.5660547

Ghisellini, P., Cialani, C., & Ulgiati, S. (2016). A review on circular economy: The expected transition to a balanced interplay of environmental and economic systems. *Journal of Cleaner Production*, *114*, 11–32. https://doi.org/10.1016/j.jclepro.2015.09.007

Goedkoop, M., Oele, M., Leijting, J., Ponsioen, T., & Meijer, E. (2016). *Introduction to LCA with SimaPro* (p. 80). PRé-sustainability.

Haes, H. A. U. de, Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E. G., Hofstetter, P., Jolliet, O., Klopffer, W., Krewitt, W., Lindeijer, E., Muller-Wenk, R., Olsen, S. I., Pennington, D. W., Potting, J., & Steen, B. (2003). *Life cycle impact assessment. Striving towards best practice.* SETAC-Press. https://www.rug.nl/research/portal/publications/life cycle-impact-assessment-striving-towards-best-practice(07e2ee6f-8a29-4fd4-92a8-b927526837f2)/export.html

Hammervold, J. (2014). *Towards greener road infrastructure: Life cycle assessment of case studies and recommendations for impact reductions and planning of road* [PhD, Norwegian University of Science and Technology (NTNU)]. https://ntnuopen.ntnu.no/ntnu-xmlui/handle/11250/274029

Hauschild, M., Olsen, S. I., Hansen, E., & Schmidt, A. (2008). Gone…but not away – Addressing the problem of long-term impacts from landfills in LCA. *The International Journal of Life Cycle Assessment*, *13*(7), 547–554. https://doi.org/10.1007/s11367-008-0039-3

Hauschild, M. Z., Goedkoop, M., Guinée, J., Heijungs, R., Huijbregts, M., Jolliet, O., Margni, M., De Schryver, A., Humbert, S., Laurent, A., Sala, S., & Pant, R. (2013). Identifying best existing practice for characterization modelling in life cycle impact assessment. *The International Journal of Life Cycle Assessment*, *18*(3), 683–697. https://doi.org/10.1007/s11367-012-0489-5

Hertwich, E. G., Ali, S., Ciacci, L., Fishman, T., Heeren, N., Masanet, E., Asghari, F. N., Olivetti, E., Pauliuk, S., Tu, Q., & Wolfram, P. (2019). Material efficiency strategies to reducing greenhouse gas emissions associated with buildings, vehicles, and electronics—A review. *Environmental Research Letters*, *14*(4), 043004. https://doi.org/10.1088/1748-9326/ab0fe3

Hjelmar, O., Holm, J., & Crillesen, K. (2007). Utilisation of MSWI bottom ash as subbase in road construction: First results from a large-scale test site. *Journal of Hazardous Materials*, *139*(3), 471–480. https://doi.org/10.1016/j.jhazmat.2006.02.059

Huysman, S., Sala, S., Mancini, L., Ardente, F., Alvarenga, R. A. F., De Meester, S., Mathieux, F., & Dewulf, J. (2015). Toward a systematized framework for resource efficiency indicators. *Resources, Conservation and Recycling*, *95*, 68–76. https://doi.org/10.1016/j.resconrec.2014.10.014

Hykš, J., & Hjelmar, O. (2018). Utilisation of Incineration Bottom Ash (IBA) from waste incineration – Prospects and limits. In *Removal, Treatment and Utilisation of Waste Incineration Bottom Ash* (pp. 11–24). Thomé-Kozmiensky Verlag GmbH.

IAEA. (2020). *PRIS - Country Details*. International Atomic Energy Agency (IAEA). https://pris.iaea.org/pris/CountryStatistics/CountryDetails.aspx?current=SE

IPCC. (2015). *Climate change 2014: Synthesis report* (p. 151) [Full Report]. Intergovernmental Panel on Climate Change (IPCC). https://www.ipcc.ch/site/assets/uploads/2018/02/SYR_AR5_FINAL_full.pdf

IPCC. (2018). *Global warming of 1.5°C* (p. 616) [Full Report]. Intergovernmental Panel on Climate Change (IPCC). http://www.ipcc.ch/report/sr15/

IRP. (2020). *Resource Efficiency and Climate Change: Material efficiency strategies for a low-carbon future* (p. 44) [Summary for Policy Makers]. International Resource Panel (IRP), United Nations Environment Programme (UNEP). https://www.resourcepanel.org/reports/resource-efficiency-and-climate-change

Johansson, J., & Lönnebo Stagnell, K. (2016). *Slagg – En koldioxidsänka? En studie av karbonatisering i slagg från förbränning av hushålls- och industriavfall.* Institutionen för Teknik och samhälle, Lunds Tekniska Högskola.

JRC-IES. (2010a). *ILCD handbook: General guide for life cycle assessment*. Publications Office of the European Union. http://lct.jrc.ec.europa.eu/pdf-directory/ILCD-Handbook-General-guide-for-LCA-DETAIL-online-12March2010.pdf

JRC-IES. (2010b). *International Reference Life Cycle Data System (ILCD) handbook: Framework and requirements for life cycle impact assessment models and indicators.* (First Edition). Publications Office of the European Union.

Kahle, K., Kamuk, B., Kallesøe, J., Fleck, E., Lamers, F., Jacobsson, L., & Sahlén, J. (2015). *Bottom ash from WtE plants – Metal recovery and utilization* (p. 30). International Solid Waste Association (ISWA).

Kellenberger, D., Althaus, H.-J., Künninger, T., Lehmann, M., Jungbluth, N., & Thalmann, P. (2007). *Life cycle inventories of building products* (Final Report Ecoinvent Data v2.0 No. 7; Issue 7, p. 914). EMPA Dübendorf, Swiss Centre for Life Cycle Inventories.

Kirchherr, J., Piscicelli, L., Bour, R., Kostense-Smit, E., Muller, J., Huibrechtse-Truijens, A., & Hekkert, M. (2018). Barriers to the Circular Economy: Evidence from the European Union (EU). *Ecological Economics*, *150*, 264–272. https://doi.org/10.1016/j.ecolecon.2018.04.028

Kirchherr, J., Reike, D., & Hekkert, M. (2017). Conceptualizing the circular economy: An analysis of 114 definitions. *Resources, Conservation and Recycling*, *127*, 221–232. https://doi.org/10.1016/j.resconrec.2017.09.005

Lynn, C. J., Ghataora, G. S., & Dhir, R. K. (2018). Environmental impacts of MIBA in geotechnics and road applications. *Environmental Geotechnics*, *5*(1), 31–55. https://doi.org/10.1680/jenge.15.00029

Malmö Stad. (2009). *Environmental Programme for the City of Malmö 2009-2020* (p. 16). Malmö Stad. https://malmo.se/download/18.6301369612700a2db9180006215/1491304408540/Environmental+Programme+for+the+City+of+Malmö+2009-2020.pdf

Malmö Stad. (2014). *Comprehensive Plan for Malmö* (p. 16). Malmö Stad. https://malmo.se/download/18.1256e63814a61a1b34c1ade/1491298771893/OP_english_summary_hemsida.pdf

Milios, L. (2016). *Policies for resource efficient and effective solutions – A review of concepts, current policy landscape and future policy considerations for the transition to a Circular Economy* (p. 81). International Institute for Industrial Environmental Economics (IIIEE). https://portal.research.lu.se/portal/files/18427043/Mistra_REES_Report.pdf

Moberg, Å. (2006). *Environmental systems analysis tools for decision-making: LCA and Swedish waste management as an example* [PhD, KTH Royal Institute of Technology]. http://www.diva-portal.org/smash/record.jsf?pid=diva2%3A9966&dswid=3873

Mont, O., & Bleischwitz, R. (2007). Sustainable consumption and resource management in the light of life cycle thinking. *European Environment*, *17*(1), 59–76. https://doi.org/10.1002/eet.434

Moretti, L., Mandrone, V., D'Andrea, A., & Caro, S. (2018). Evaluation of the environmental and human health impact of road construction activities. *Journal of Cleaner Production*, *172*, 1004–1013. https://doi.org/10.1016/j.jclepro.2017.10.250

Mroueh, U.-M., Eskola, P., & Laine-Ylijoki, J. (2001). Life cycle impacts of the use of industrial by-products in road and earth construction. *Waste Management*, *21*(3), 271–277. https://doi.org/10.1016/S0956-053X(00)00100-8

Mroueh, U.-M., Eskola, P., Laine-Ylijoki, J., Wellman, K., Juvankoski, E. M. M., Ruotoistenmäki, A., & Finnish National Road Administration. (2000). *Life cycle assessment of road construction*. Finnish National Road Administration. https://www.doria.fi/bitstream/handle/10024/139521/4734tie.pdf?sequence=1

NCC. (2019). *Environmental Product Declaration for aggregates from Uddevalla quarry – Glimmingen* (p. 11) [Environmental Product Declaration]. NCC Industry Nordic AB. https://www.environdec.com/Detail/?Epd=16080

Nußholz, J. L. K., Rasmussen, F. N., Whalen, K., & Plepys, A. (2020). Material reuse in buildings: Implications of a circular business model for sustainable value creation. *Journal of Cleaner Production*, *245*, 118546. https://doi.org/10.1016/j.jclepro.2019.118546

Nußholz, J., Nygaard Rasmussen, F., & Milios, L. (2019). Circular building materials: Carbon saving potential and the role of business model innovation and public policy. *Resources, Conservation and Recycling*, *141*, 308–316. https://doi.org/10.1016/j.resconrec.2018.10.036

OECD. (2019). *Global material resources outlook to 2060: Economic drivers and environmental consequences*. OECD Publishing.

Officina dell'Ambiente S.p.A. (2019). *Environmental Product Declaration of secondary raw materials or aggregates of industrial origin – AG Matrix* (p. 12) [Environmental Product Declaration]. Officina dell'Ambiente S.p.A.

Olsson, S., Kärrman, E., & Gustafsson, J. P. (2006). Environmental systems analysis of the use of bottom ash from incineration of municipal waste for road construction. *Resources, Conservation and Recycling*, *48*(1), 26–40. https://doi.org/10.1016/j.resconrec.2005.11.004

Olsson, S., van Schaik, J. W. J., Gustafsson, J. P., Kleja, D. B., & van Hees, P. A. W. (2007). Copper (II) Binding to dissolved organic matter fractions in municipal solid waste incinerator bottom ash leachate. *Environmental Science & Technology*, *41*(12), 4286–4291. https://doi.org/10.1021/es062954g

Onkelbach, A. (2016). *Handbuch Ersatzbaustoffe Grundlagen für den Einsatz im Straßen- und Erdbau* (4th ed.). REMEX Mineralstoff GmbH.

Overgaard, T. B., Winkler, M., Thøgersen, F., & Dyhr-Jensen, S. (2020, January). Slaggegrus 0/31,5 mm som bærelagsmateriale i vejbygning. *Trafik & Veje*, *1*, 42–45.

Plevin, R. J., Delucchi, M. A., & Creutzig, F. (2014). Using attributional Life Cycle Assessment to estimate climate-change mitigation benefits misleads policy makers. *Journal of Industrial Ecology*, *18*(1), 73–83. https://doi.org/10.1111/jiec.12074

Rendek, E., Ducom, G., & Germain, P. (2006). Carbon dioxide sequestration in municipal solid waste incinerator (MSWI) bottom ash. *Journal of Hazardous Materials*, *128*(1), 73–79. https://doi.org/10.1016/j.jhazmat.2005.07.033

Richter, J. L. (2019). *Towards a Circular Economy with Environmental Product Policy: Considering dynamics in closing and slowing material loops for lighting products* [PhD, International Institute for Industrial Environmental Economics (IIIEE), Lund University]. https://portal.research.lu.se/ws/files/71968456/JLRichter_2019_dissertation_LundUniversity.pdf

Risse, M., Weber-Blaschke, G., & Richter, K. (2017). Resource efficiency of multifunctional wood cascade chains using LCA and exergy analysis, exemplified by a case study for Germany. *Resources, Conservation and Recycling*, *126*, 141–152. https://doi.org/10.1016/j.resconrec.2017.07.045

Roth, L., & Eklund, M. (2003). Environmental evaluation of reuse of by-products as road construction materials in Sweden. *Waste Management*, *23*(2), 107–116. https://doi.org/10.1016/S0956-053X(02)00052-1

Santero, N., Masanet, E., & Horvath, A. (2010). *Life cycle Assessment of pavements: A critical review of existing literature and research* (No. SN3119a; Issue SN3119a, p. 81). Portland Cement Association. https://escholarship.org/uc/item/8632v9cc#main

Saveyn, H., Eder, P., Garbarino, E., Muchova, L., Hjelmar, O., Sloot, H., van der, Comans, R., Zomeren, A., van, Hyks, J., & Oberender, A.. (2014). *Study on methodological aspects regarding limit values for pollutants in aggregates in the context of the possible development of end-of-waste criteria under the EU Waste Framework Directive: Final report.* Publications Office. http://dx.publications.europa.eu/10.2791/1125

Schneider, L., Bach, V., & Finkbeiner, M. (2016). LCA perspectives for resource efficiency assessment. In M. Finkbeiner (Ed.), *Special Types of Life Cycle Assessment* (pp. 179–218). Springer Netherlands. https://doi.org/10.1007/978-94-017-7610-3_5

Schroeder, P., Anggraeni, K., & Weber, U. (2019). The relevance of circular economy practices to the Sustainable Development Goals: Circular Economy and SDGs. *Journal of Industrial Ecology*, *23*(1), 77–95. https://doi.org/10.1111/jiec.12732

Schwab, O., Bayer, P., Juraske, R., Verones, F., & Hellweg, S. (2014). Beyond the material grave: Life Cycle Impact Assessment of leaching from secondary materials in road and earth constructions. *Waste Management*, *34*(10), 1884–1896. https://doi.org/10.1016/j.wasman.2014.04.022

SGU. (2017). *Grus, sand och krossberg 2016: Aggregates* (Grus, Sand Och Krossberg) [Annual Report]. Geological Survey of Sweden (SGU).

Silva, R. V., de Brito, J., Lynn, C. J., & Dhir, R. K. (2019). Environmental impacts of the use of bottom ashes from municipal solid waste incineration: A review. *Resources, Conservation and Recycling*, *140*, 23–35. https://doi.org/10.1016/j.resconrec.2018.09.011

Simion, I. M., Fortuna, M. E., Bonoli, A., & Gavrilescu, M. (2013). Comparing environmental impacts of natural inert and recycled construction and demolition waste processing using LCA. *Journal of Environmental Engineering and Landscape Management*, *21*(4), 273–287. https://doi.org/10.3846/16486897.2013.852558

Stake, R. E. (2005). Qualitative case studies. In *The Sage handbook of qualitative research, 3rd ed* (pp. 443–466). Sage Publications Ltd.

Stripple, H. (2001). *Life Cycle Assessment of roads: A pilot study for inventory analysis* (p. 182). Swedish Environmental Research Institute (IVL). http://seeds4green.net/sites/default/files/Life%20Cycle%20Assessment%20of%20Road_0.pdf

Sysav. (2018). *Annual Report 2018* (p. 16) [Annual Report].

Toller, S. (2008). *Environmental assessment of incinerator residue utilisation* [PhD, KTH Royal Institute of Technology]. http://kth.diva-portal.org/smash/record.jsf?pid=diva2%3A336509&dswid=-9417

Toller, S., Kärrman, E., Gustafsson, J. P., & Magnusson, Y. (2009). Environmental assessment of incinerator residue utilisation. *Waste Management*, *29*(7), 2071–2077. https://doi.org/10.1016/j.wasman.2009.03.006

Trunzo, G., Moretti, L., & D'Andrea, A. (2019). Life Cycle Analysis of road construction and use. *Sustainability*, *11*(2), 377. https://doi.org/10.3390/su11020377

Van Praagh, M., Johansson, M., Fagerqvist, J., Grönholm, R., Hansson, N., & Svensson, H. (2018). Recycling of MSWI-bottom ash in paved constructions in Sweden – A risk assessment. *Waste Management*, *79*, 428–434. https://doi.org/10.1016/j.wasman.2018.07.025

Weidema, B. P., Hischier, R., Mutel, C., Nemecek, T., Vandenbo, C. O., & Wernet, G. (2013). *Overview and methodology. Data quality guideline for the ecoinvent database version 3* (1(v3); Ecoinvent Report, Issue 1(v3), p. 169). The ecoinvent Centre.

Yang, Y. (2019). A unified framework of life cycle assessment. *The International Journal of Life Cycle Assessment*, *24*(4), 620–626. https://doi.org/10.1007/s11367-019-01595-w

Zero Waste Europe. (2019). *The hidden impacts of incineration residues* (p. 11) [Case Study]. Zero Waste Europe. https://zerowasteeurope.eu/wp-content/uploads/2019/11/zero_waste_europe_cs_the-hidden-impacts-of-incineration-residues_en.pdf

Appendix

APPENDIX I: Road layout provided by the City of Malmö

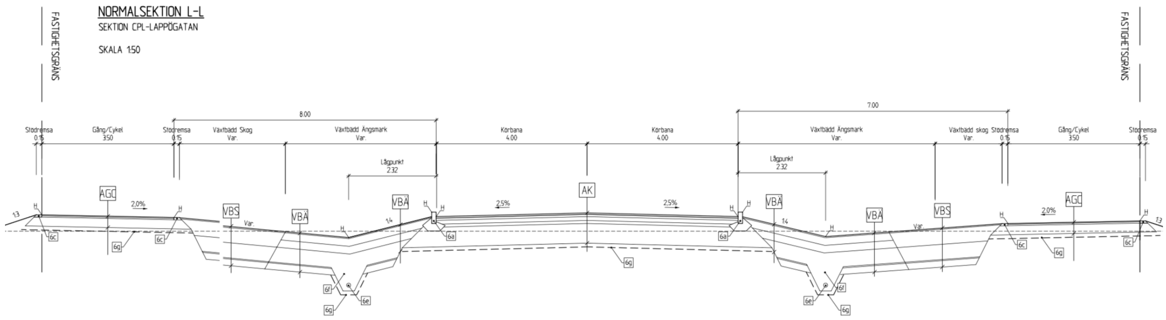


Figure 23: Road layout provided by the City of Malmö.

APPENDIX II: List of interviewees

Table 10: List of people that were consulted in the course of the study.

#	Organization	Position	Type of communication	Date	Relevance for the thesis
1.	Sysav Utveckling AB	Project Manager	Personal interview and review of data and case study report	multiple	Informant for case study and expert on waste incineration and bottom ash (utilization)
2.	RISE Research Institutes of Sweden AB	Researcher	Review of LCA model and report	multiple	LCA expert
3.	RISE Research Institutes of Sweden AB	Senior Researcher, Material Design	Written communication/data questionnaire	multiple	Expert on construction aggregates
4.	Swerock AB	Business Development Manager, Recycling	Written communication/data questionnaire and personal interview	multiple	Informant for case study and expert on construction material and recycling of construction material
5.	Amager Bakke Ressourcecenter	Environmental Manager	Personal interview	17.2.2020	Expert on waste incineration
6.	Amager Bakke Ressourcecenter	Chief Consultant	Personal interview	17.2.2020	Expert on waste incineration and bottom ash
7.	Strabag AG	Technical Director	Skype interview	multiple	Expert on bottom ash recycling (in Germany)
8.	Afatek	Development Manager	Skype interview	12.3.2020	Expert on bottom ash recycling (in Denmark)
9.	Danish Waste Solutions	Partner	Skype interview	9.3.2020	Expert on environmental impact assessment (leaching) and

					bottom ash utilization in road construction
10.	Real Estate and Street Office, City of Malmö	Project Manager	Personal interview	10.3.2020	Case study informant and expert on road construction in Malmö
11.	Department of Environmental Engineering, Technical University of Denmark (DTU)	Professor	Written communication	2.3.2020	Expert on bottom ash and LCA
12.	Centre for Environmental and Climate Research (CEC), Lund University	Researcher	Skype interview	4.3.2020	Expert on environmental impact assessment (leaching) and bottom ash utilization in road construction
13.	Toxicowatch	Founder and CEO	Skype interview	3.2.2020	Expert on environmental impacts from bottom ash
14.	AB Sydsten	Technical Operating Officer	Skype interview	18.3.2020	Case study informant and expert on primary construction aggregates
15.	Sysav AB	Head of Business Development	Skype interview	18.3.2020	Case study informant

APPENDIX III: Selected interview results

Table 11: Summary of literature and interview results with regard to potential differences in the life cycles of a road with and without MIBA.

Construction	Maintenance	Use	End-of-life
Key to an accurate comparison of the two material systems is to know whether the use of MIBA requires any other structural changes in the road layout. Arm et al. (2017) state that MIBA can be expected to degrade faster than crushed rock due to the amorphous phase, the organic content, and the porous structure. This should be taken into account when constructing roads with a very long lifetime. A test road in the Malmö area that includes MIBA and that was considered in Arm (2003) and Toller (2008) included a bigger base coarse layer compared to a road with natural aggregate. A study by Deviatkin et al. (2017) was based on a similar assumption. On the contrary, Birgisdottir et al. (2006) and Geng et al. (2010) assume two otherwise identical roads based on recommendations from the Danish Road Administration and a performance test against the relevant Chinese technical requirements for road structures respectively. Interviews with companies using MIBA in road construction confirmed that no other layers need to change due to the use of MIBA (2 and 3). This was also confirmed by the Real Estate and Road Department of the City of Malmö (4). It was mentioned that due to potential dust emissions construction companies would need to consider changing from a layer-by-layer procedure to a section-by-section procedure (1 and 4), however, other interviewees stated that this is not practiced in Germany (2). The effectiveness of compacting MIBA depends on its water content. Both companies using MIBA in road constructions in Germany and Denmark that were interviewed stated that the water consumption for optimizing compaction is a major difference in terms of	Currently, annual leaching tests and analysis are required for roads that include MIBA in Sweden (1). Because the material is used in the subbase with at least 0.5 m of other road layers above it, no uncontrolled excavation of the material needs to be expected (Arm et al., 2017). Interviewees were not aware of incidences where the use of MIBA had led to a change in the conventional maintenance routine, for example, in the form of early repairs (2 and 3).	Interviewees agreed that there are no differences to be expected in this phase other than potential leaching of salts and heavy metals from MIBA. Only the wearing course has influence on, for example, albedo and rolling resistance. (2 and 4).	Interviewees agreed that the common scenario would be that the road remains in place (2, 3, and 4). It is very rare that roads get deconstructed. Usually only a replacement of the wearing course is done. The city of Malmö assumes a 40-year lifespan for the subbase and 20 years for the wearing course, however, most streets are "hundreds of years old" (4). In reality, a reconstruction of the asphalt layer takes place every 40-50 years, depending on the traffic. What would happen to the material if a road is deconstructed entirely remains a theoretical question. Although MIBA is used since many decades in both Germany and Denmark (approx. 40-50 years), both companies interviewed had no experience with deconstructions of roads. While in the case of Germany, the company would take back and reuse the material in case of a complete deconstruction of a road, in Denmark, the material would need to be managed by a company that has a license to treat waste (2 and 3). In Sweden, the supplier of MIBA would also be required to take back the material in case a road is deconstructed (Arm et al., 2017).

resource consumption (2 and 3). The use of water can reach 20 w% of the MIBA (3). Potential dust emissions if MIBA is not handled appropriately were highlighted by one interviewee (5). Measures to manage dust emissions (e.g. covers and sprinkling) present a difference to primary aggregate use, however, it needs to be taken into account that MIBA is kept wet on the construction site for compaction anyways, which reduces the potential for dust emissions (2 and 3). In some countries like Denmark a temporary storage of MIBA on the construction site is not allowed (3). Resulting differences in work processes could lead to differences in resource consumption and emissions.			

Sources: 1 = Respondent 1, personal communication, February 4, 2020; 2 = Respondent 7, personal communication, March 13, 2020; 3 = S. Respondent 8, personal communication, March 16, 2020; 4 = Respondent 10, personal communication, October 3, 2020; 5 = Respondent 13, personal communication, March 3, 2020.

APPENDIX IV: Data questionnaire

Table 12: Questionnaire used to collect data from Sysav and AB Sydsten.

Description of data needed	Value/description	Unit	Date	Type of source			Comment
				Direct data (derived directly from administrative systems)	Indirect data (based on some sort of calculation)	Estimated data	
…	…	…	…	…	…	…	…

APPENDIX V: Data description and tables scenario A

Different datasets available in the ELCD and Ecoinvent database were considered for the production of crushed stone. The process originally found to be most suitable to represent the quarry in Dalby was available in the ELCD database, "Crushed stone 16/32 mm, wet and dry quarry, production mix, at plant, undried, EU." While the grain size differs from the grainsize used in the subbase of scenario A, it was confirmed with AB Sydsten that the material would go through the same number of crushing steps – the most energy intensive process in the production of crushed stone. A further difference between the quarry in Dalby and the ELCD dataset is that it considers the production of crushed rock from limestone. Whereas the quarry in Dalby produces crushed rock from a mixture of diabase and gneiss. However, it was confirmed with AB Sydsten that the process steps for the production of crushed stone from limestone that is to be used in road constructions (i.e. hard limestone) is similar to the processes at Dalby. This dataset was also used in an EPD for different secondary aggregates made from MIBA ("AGMatrix") for a comparison with primary aggregates (Officina dell'Ambiente S.p.A., 2019).

A major drawback to this dataset is that it is not available as a unit process in SimaPro. As a result, it was not possible to adjust the process to Swedish conditions. For example, environmental impacts resulting from electricity consumption, which is predominantly renewable in Sweden, are not accurate. Due to the focus on climate impacts, a dataset that could be adjusted to Swedish conditions was therefore preferred. The dataset is further not available in different system models like the Ecoinvent processes. In order to be consistent in the use of data, an Ecoinvent process was therefore considered. Furthermore, the ELCD dataset only includes "dummy processes." It is not possible to generate a detailed process network which limits the possibility to conduct a contribution analysis. Finally, the ELCD dataset does not include important aspects of resource use such as land use.

A second dataset considered is the Ecoinvent process "gravel production, crushed, CH." It covers the production of aggregates for the use in construction from natural gravel. As opposed to the production of crushed rock from bedrock, gravel is extracted by dredging and not blasting. The final product of this process consists of a mixture of mined round gravel, sand, and 15% crushed gravel. While it is a different product and extracted in a different way than crushed rock, process steps are similar to a large extend. The process could also be adjusted to more closely resemble the production of crushed rock in Sweden.

The Ecoinvent dataset "Gravel, crushed, Brazil" was used for the baseline scenario. It includes data on the production of crushed rock from bedrock based on Brazilian quarries. It is available as unit process and was adjusted to resemble Swedish conditions. Where possible, background data on elementary, product, and waste flows was adjusted. Foreground data was changed based on data from an Environmental Product Declaration for construction aggregates from a quarry in Glimmingen, Sweden provided by NCC Industry Nordic AB. An Interview with AB Sydsten had confirmed that processes for the product group 0-90 mm are identical in both quarries.

Table 13: Considered database processes and inventory data for the production of crushed rock in scenario A.

Description	Database/Source	Comment
Crushed stone 16/32 mm, open pit mining, production mix, at plant, undried, EU	ELCD	This dataset was not used due to the reasons elaborated above.
Gravel, crushed, Switzerland	Ecoinvent v 3.5	This dataset was not used due to the reasons elaborated above.
Gravel, crushed, Brazil	Ecoinvent v 3.6	This dataset was used combined with additional, product specific information from NCC (2019).
Environmental Product Declaration for aggregates from Uddevalla quarry, Glimmingen, Sweden	NCC (2019)	Data provided by NCC Industry Nordic AB.

Table 14: Average amount of MIBA transported from the incineration plant in Malmö to different landfill sites for final disposal during 2018 and 2019.

Destination	Amount (t)	Distance (km)	Comment
Trelleborg	15731	36	Landfill owned by Sysav
Måsalycke	10,322	81	Landfill owned by Sysav
Hedeskoga	277	62	Landfill owned by Sysav
Landskrona	13,685	38	Landfill not owned by Sysav
Nårab	51,436	77	Landfill not owned by Sysav

Table 15: Transportation considered in scenario A.

Transported material	Type of transportation	Ecoinvent process	Transportation distance (km)	Destination (from-to)
Crushed rock	Lorry >32 metric ton (Euro 6)	Transport, freight, lorry >32 metric ton, diesel, RER	30	Quarry – construction site

APPENDIX VI: Data description and tables scenario B

Table 16: Calculation of the energy demand of the Sysav sorting plant per kilogram of IBA treated.

	Value	Source	Comment
Power rating per machine documents (kW)	873.28	Provided by Sysav	
Energy consumption (kWh) at full capacity	345.00	Provided by Sysav	This is the maximum value of energy consumption of machinery. It also includes air treatment, cleaning, and high-pressure equipment.
Sorting capacity (t/h)	100.00	Provided by Sysav	
Energy consumption per ton of sorted IBA without considering start up and cool down of machines (kWh/t)	3.45	calculated	
Number of hours per daily shift	7.00	Provided by Sysav	
Energy consumption per day (kWh)	2,415.00	calculated	
Amount of IBA sorted per daily shift	700.00	calculated	
Number of hours for startup and cool down of machines	0.67	Provided by Sysav	
Energy consumption during start up and cool down (kWh)	150.00	Provided by Sysav	Starting plant: 0-300kW Mean value: 150kW Time: 40 minutes per day (20 min for start-up and 20 min for shut down) Particularly crushing uses less energy during start up and cool down
Total energy consumption per daily shift including start up and cool down of machines (kWh)	2,515.00	calculated	
Energy consumption per t of sorted IBA including start up and cool down (kWh/t)	3.592857143	calculated	
Energy consumption per kg of sorted IBA including start up and cool down (kWh/kg)	0.003592857143	calculated	

Table 17: Ecoinvent datasets included in the sorting process.

Description	Ecoinvent process	Amount (per kg IBA sorted)	Comment
Excavator	Diesel, burned in building machine, GLO	0.00575232 MJ	Excavator to mix IBA and sort out oversize material. Operated 7 h a day. The actual machine used is a Doosan DX255LC.
Electricity consumption of the plant	Electricity, medium voltage, SE	0.003592857143 kWh	For calculation of electricity consumption see Table 16.
Wheeled loader	Diesel, burned in building machine, GLO	0.006758976 MJ	Wheeled loader used for moving IBA and MIBA at the sorting plant and loading the plant. Operated 7 h daily. The actual machine used is a Volvo L350.
Wheeled loader	Diesel, burned in building machine, GLO	0.004026624 MJ	Wheeled loader used for moving IBA and MIBA at the sorting plant and loading the plant. Operated 7 h daily. The actual machine used is a Volvo L150G.

Table 18: Fuel consumption of construction vehicles at the bottom ash sorting facility and the resulting energy demand from diesel fuel.

Machine	Type	Diesel consumption (l/h)	Diesel consumption (kg/h)	Energy demand from diesel (MJ)	Energy demand from diesel per kg MIBA (MJ)
Volvo L350	Loader	18.8	15.792	675.8976	0.006758976
Volvo L150G	Loader	11.2	9.408	402.6624	0.004026624
Doosan DX 255	Excavator	16	13.44	575.232	0.00575232
Sum		46		1653.792	0.01653792

Source: Fuel consumption of vehicles was derived from the Sysav administrative system. Energy demand from diesel fuel was calculated based on the net heat value of diesel (42.8 MJ/kg) and fuel density (0.84 kg/l) derived from Kellenberg et al. (2007). The energy demand from diesel per kg of MIBA was calculated based on the sorting capacity of 100,000 kg/h.

Table 19: Transportation considered in scenario B.

Transported material	Type of transportation	Ecoinvent process	Transportation distance (km)	Destination (from-to)
IBA	Lorry run on biofuel (Euro 6)	Transport, freight, lorry 28 metric ton, vegetable oil methyl ester, CH	1	Incinerator - stockpile for drying
IBA	Wheeled loader: Volvo L350 or L150G	Diesel, burned in building machine, GLO	0.1	Stockpile for drying – sorting and loading of IBA into sorting plant
MIBA	Wheeled loader: Volvo L350 or L150G	Diesel, burned in building machine, GLO	0.3	Sorting – stockpile for aging
MIBA aged	Lorry run on biofuel (Euro 6)	Transport, freight, lorry 28 metric ton, vegetable oil methyl ester, CH	4.6	stockpile for aging – construction site
MIBA	Lorry run on biofuel (Euro 6)	Transport, freight, lorry 28 metric ton, vegetable oil methyl ester, CH	64.52*	Sorting facility – final disposal site

**Average transportation distance calculated based on MIBA quantities transported to different landfill sites in southern Sweden in 2018 and 2019. See Table 14.*

Table 20: Expected amount of material received by the Sysav sorting facility, the sorting efficiency, and the expected amount of material sorted out.

Fraction	Metal content of IBA (%)	Sorting efficiency (%)	Minerals (%)	Expected amount of metals sorted out (%)	Expected amount of material (%)
NF <2 mm	0.25	40	50	0.10	0.20
NF 2-5 mm	0.55	70	50	0.39	0.77
NF 5-50 mm	1.70	90	50	1.53	3.06
Sum	2.50	-	-	2.02	4.03
Ferrous 5-50 mm	1.40	90	10	1.26	1.40
Ferrous >50 mm	3.40	90	5	3.06	3.22
Sum	4.80	-	-	4.32	4.62
Stainless steel	0.20	95	40	0.19	0.32
Organics	-	-	-	-	0.25
Oversize	-	-	-	-	5.00
Handpick	0.50	100	25	0.50	0.67
MIBA	-	-	-	-	85

Table 21: Calculation of MIBA management cost used for the economic allocation.

	External disposal	**Internal disposal**
Landfill cost (SEK/t)	260	0
Logistics (SEK/t)	150	100
Sum (SEK/t)	410	100
Amount (t/year)	65161	26330
Average price paid per ton of MIBA managed in 2018 and 2019 (SEK/t)	320.78	

APPENDIX VII: LCA results, graphs, and figures scenario A

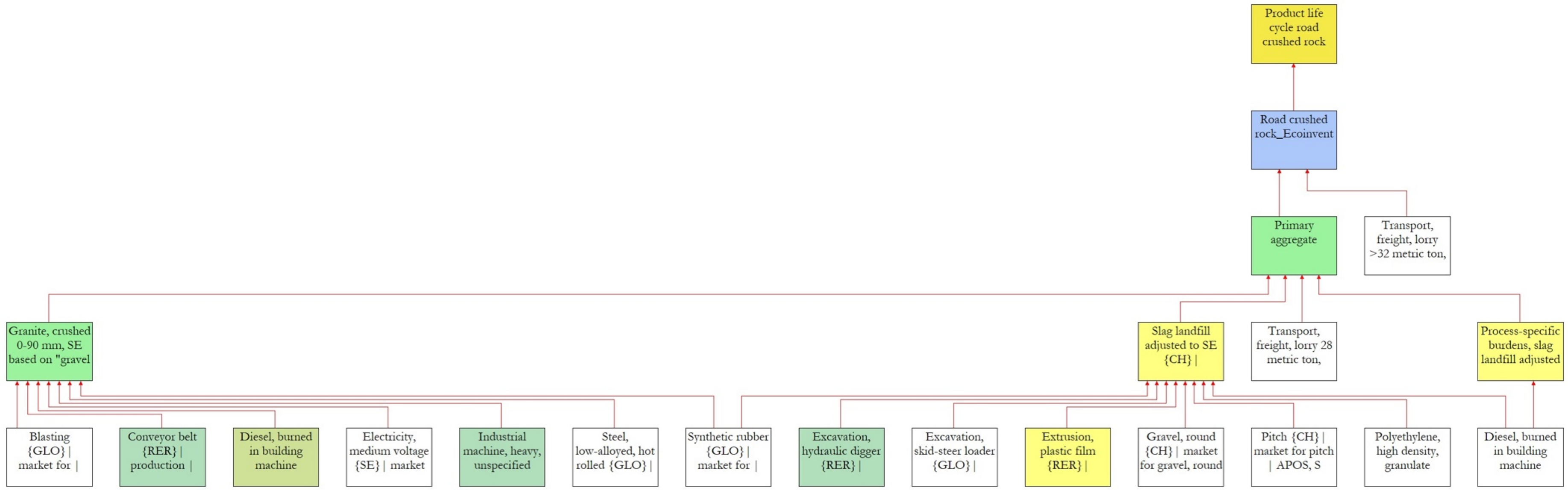


Figure 24: Network diagram of scenario A. The diagram does not show all processes. Processes that contribute less than 0.1% to the overall environmental burdens (single score) are cut off in order to be able to display it on one page.

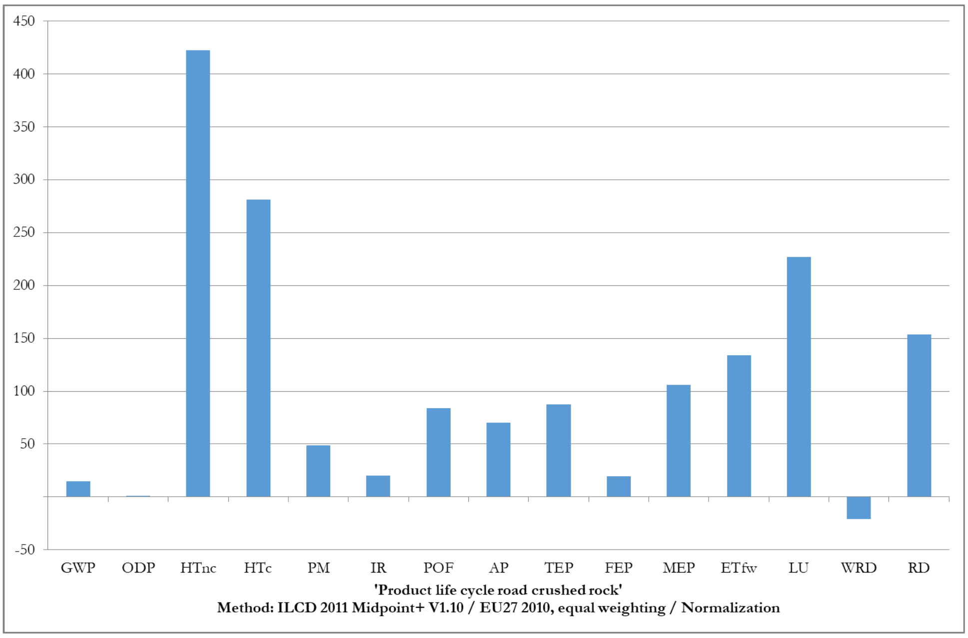


Figure 25: Normalized impact assessment results for scenario A without grouping.

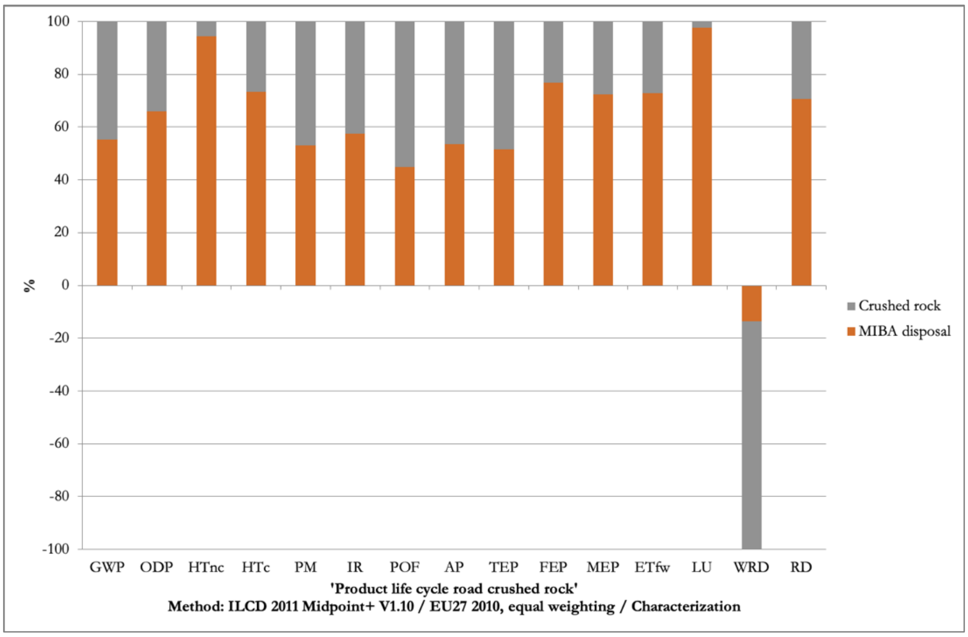


Figure 26: Characterization results for scenario A including two groups of processes. The y-axis is a percentage scale.

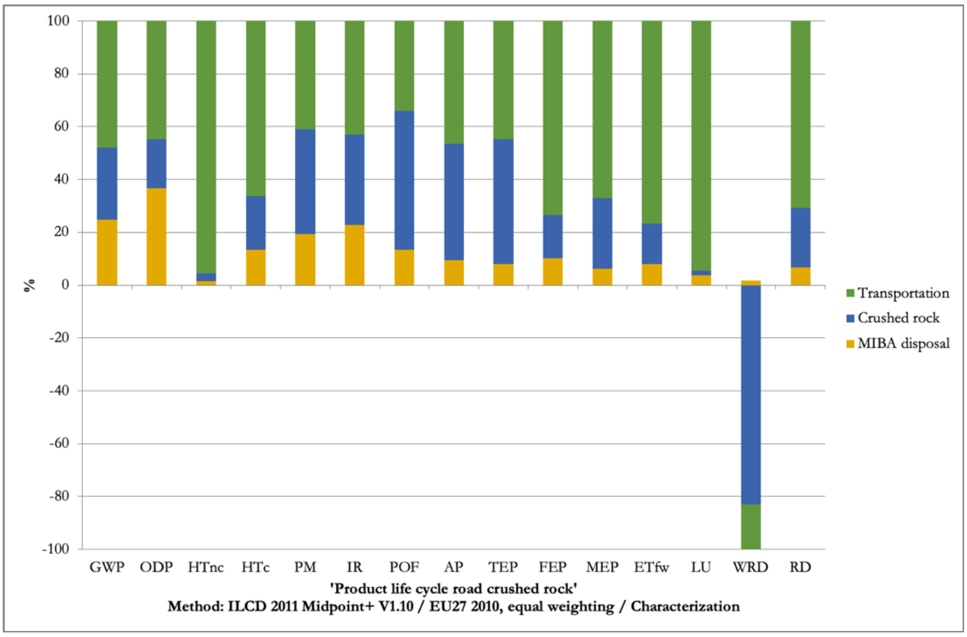


Figure 27: Characterization results for scenario A including three groups of processes. The y-axis is a percentage scale.

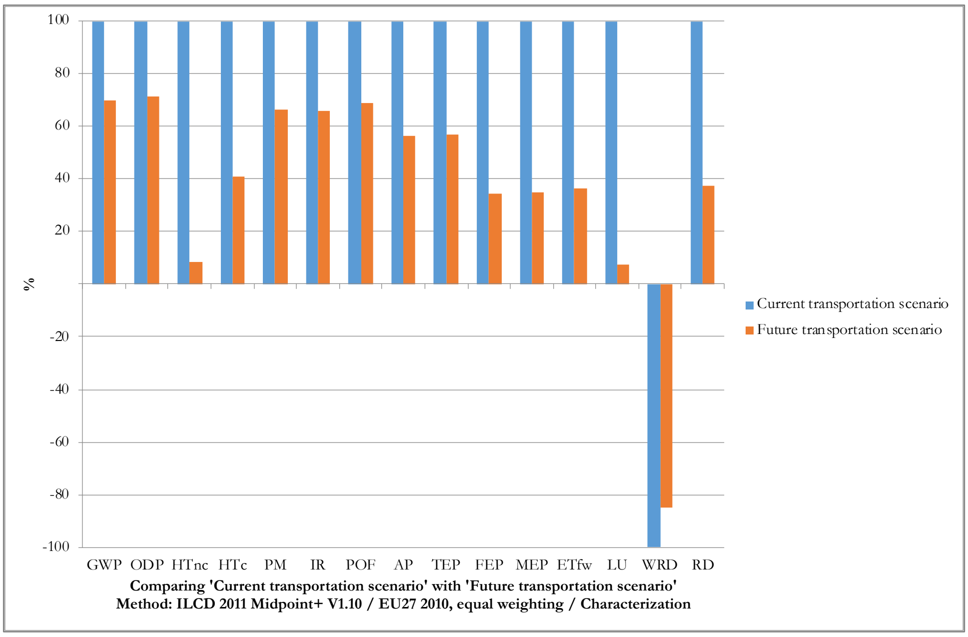


Figure 28: Characterization results for Scenario A for a comparison between the current (baseline) transportation scenario and a hypothetical future transportation scenario. The y-axis is a percentage scale.

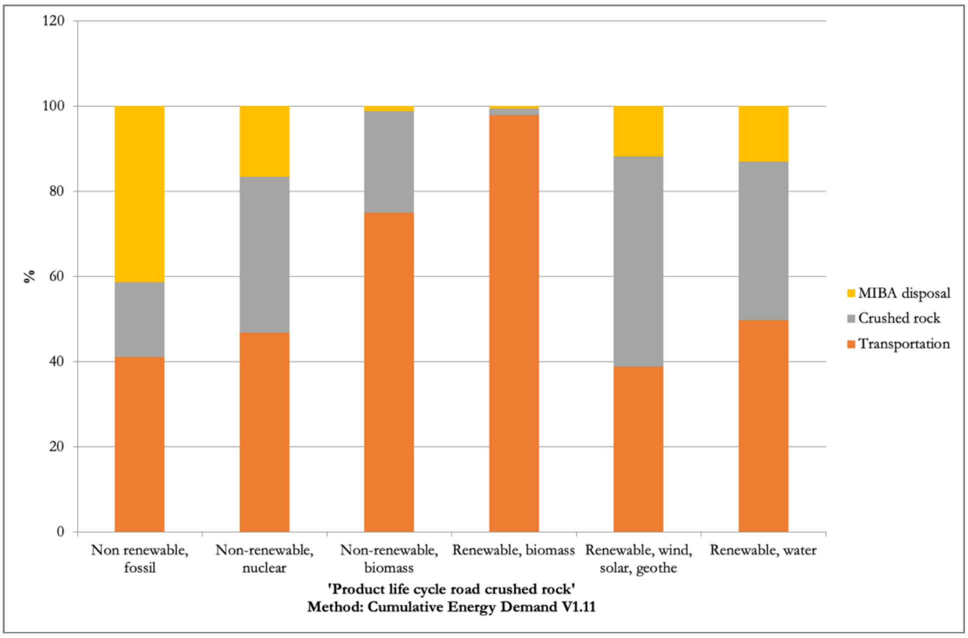


Figure 29: Characterization results for the CED of scenario A including three groups of processes. The y-axis is a percentage scale.

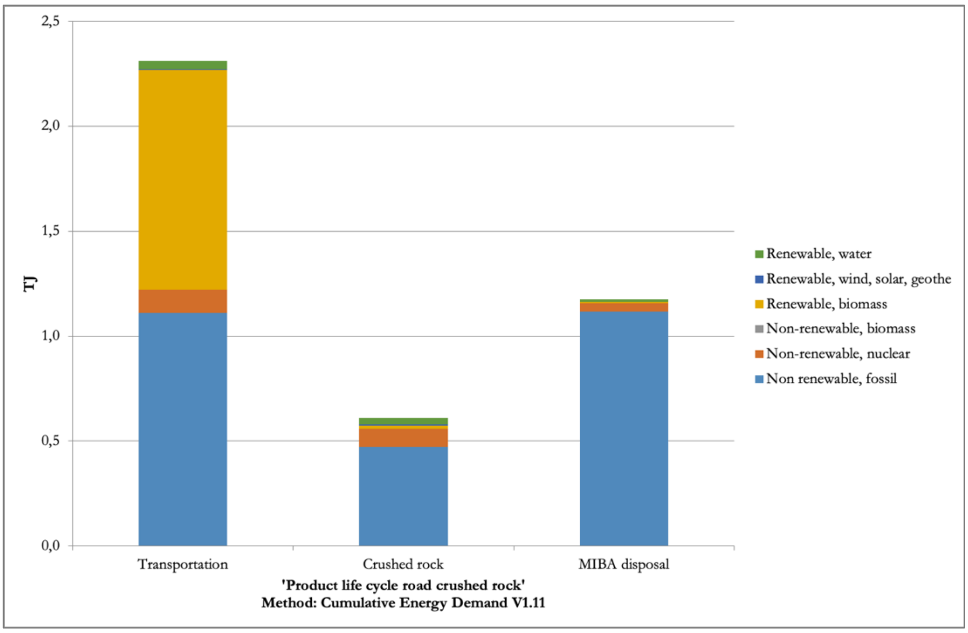


Figure 30: The CED of scenario A divided into three process groups.

APPENDIX VIII: LCA results, graphs, and figures scenario B

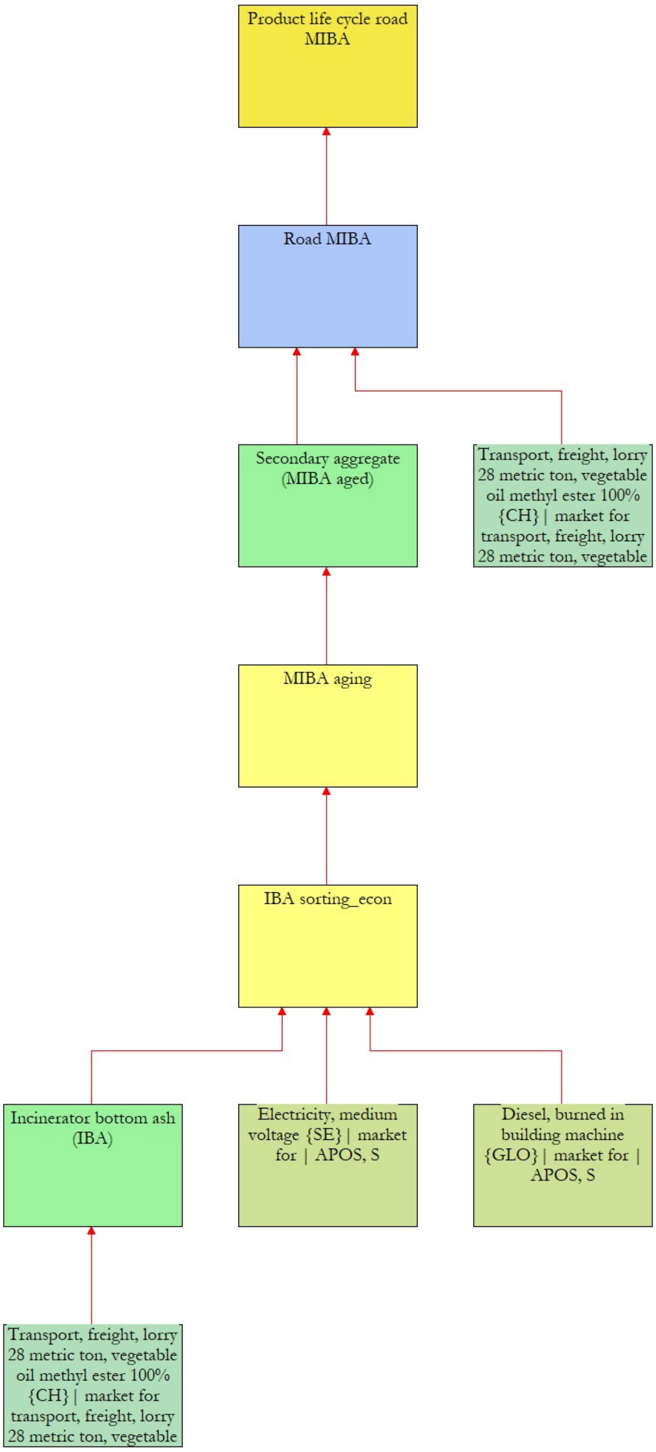


Figure 31: Network diagram of scenario B.

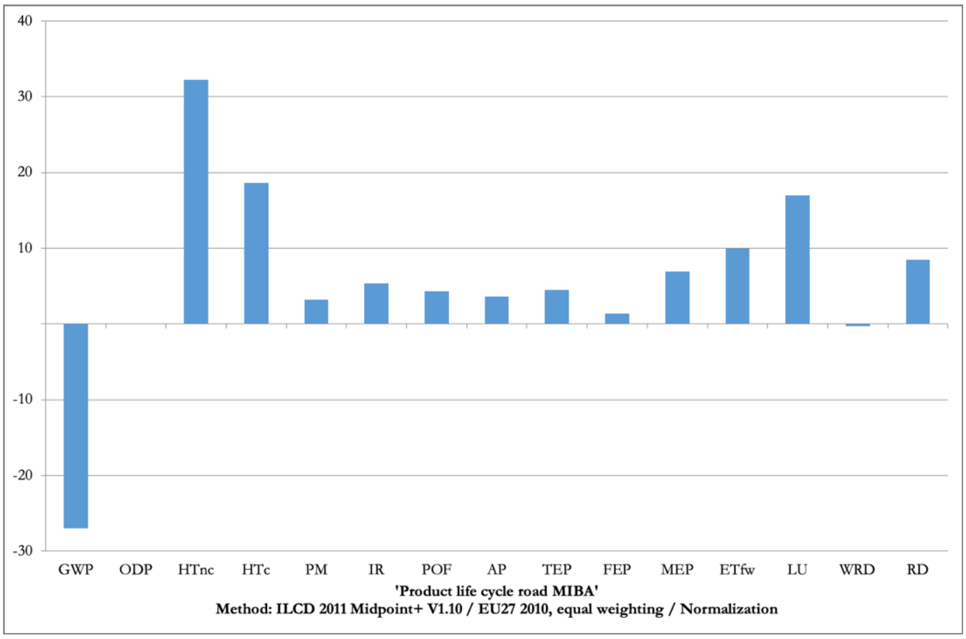


Figure 32: Normalized LCIA results for scenario B without grouping.

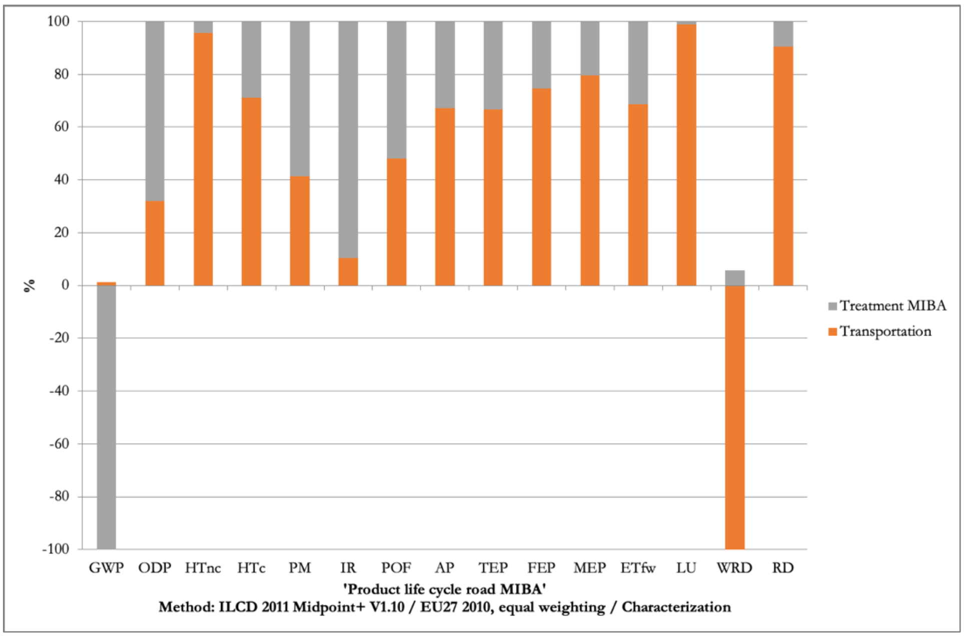


Figure 33: Characterization results for scenario B including two groups of processes. The y-axis is a percentage scale.

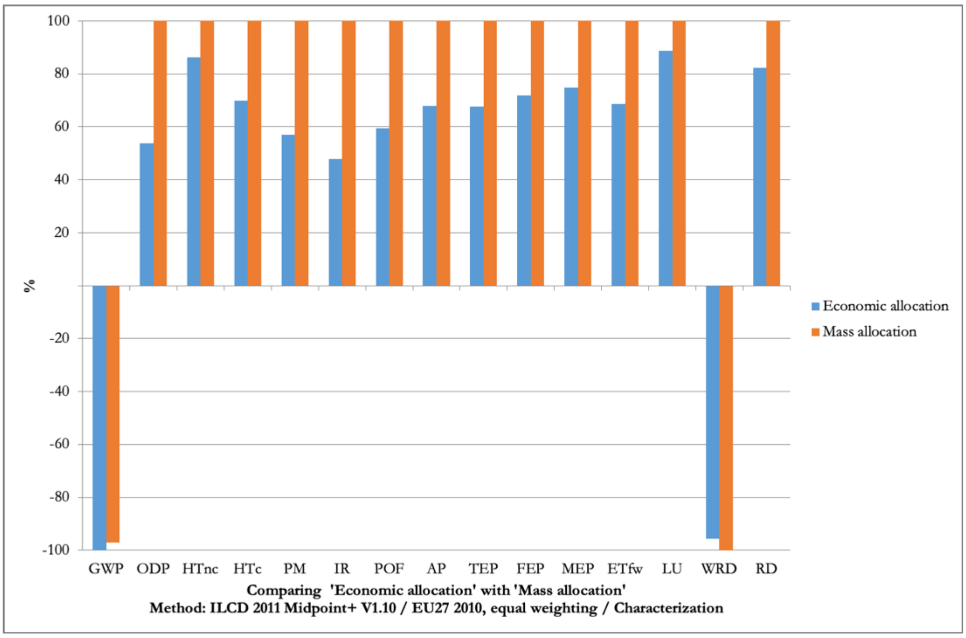


Figure 34: Characterization results of scenario B for a comparison between economic allocation and mass allocation for partitioning the environmental burdens of the sorting process. The y-axis is a percentage scale.

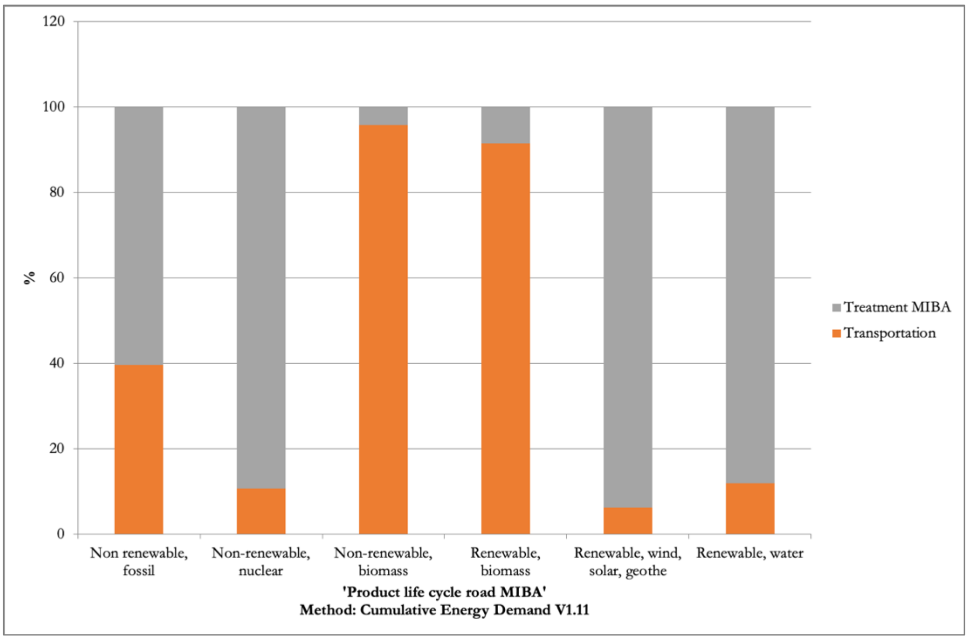


Figure 35: Characterization results for the CED of scenario B including two groups of processes. The y-axis is a percentage scale.

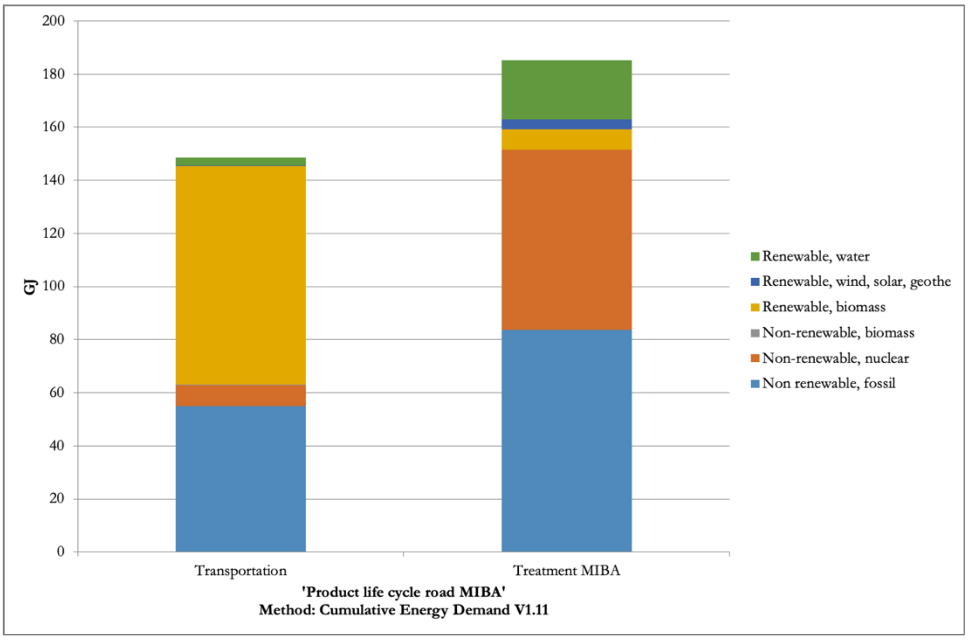


Figure 36: The CED of scenario B divided into three process groups.

APPENDIX IX: LCA results, graphs, and figures comparison scenario A and B

Table 22: Characterization and normalization results for scenario A and B.

Impact category	Unit	Scenario A	Scenario A normalized	Scenario B	Scenario B normalized	Difference	Difference normalized
Climate change (GWP)	kg CO2 eq	137,560.6628	14.9198	-248,584.1237	-26.9614	386,144.7865	41.8813
Ozone depletion (ODP)	kg CFC-11 eq	0.0306	1.4146	0.0022	0.1014	0.0284	1.3131
Human toxicity, non-cancer effects (HTnc)	CTUh	0.2251	422.3298	0.0172	32.2570	0.2079	390.0727
Human toxicity, cancer effects (HTc)	CTUh	0.0104	280.8942	0.0007	18.6410	0.0097	262.2532
Particulate matter (PM)	kg PM2.5 eq	184.8633	48.6483	11.9573	3.1467	172.9060	45.5016
Ionizing radiation HH (IR)	kBq U235 eq	23,150.1908	20.4869	6,058.5398	5.3615	17,091.6510	15.1254
Photochemical ozone formation (POF)	kg NMVOC eq	2,654.4113	83.7353	136.7153	4.3128	2,517.6959	79.4225
Acidification (AP)	molc H+ eq	3,311.0685	70.0013	171.9971	3.6363	3,139.0714	66.3650
Terrestrial eutrophication (TEP)	molc N eq	15,392.6888	87.4582	791.2961	4.4960	14,601.3928	82.9622
Freshwater eutrophication (FEP)	kg P eq	28.8236	19.4754	2.0231	1.3670	26.8005	18.1085
Marine eutrophication (MEP)	kg N eq	1,794.3931	106.1771	117.5583	6.9561	1,676.8347	99.2210
Freshwater ecotoxicity (ETfw)	CTUe	1,173,214.8213	134.2345	87,122.3809	9.9682	1,086,092.4404	124.2664
Land use (LU)	kg C deficit	16,970,471.3542	226.8782	1,270,466.1919	16.9849	15,700,005.1623	209.8934
Water resource depletion (WRD)	m3 water eq	-1,724.1727	-21.1815	-19.6250	-0.2411	-1,704.5478	-20.9404
Mineral, fossil & renewable resource depletion (RD)	kg Sb eq	15.5169	153.6333	0.8606	8.5207	14.6563	145.1125

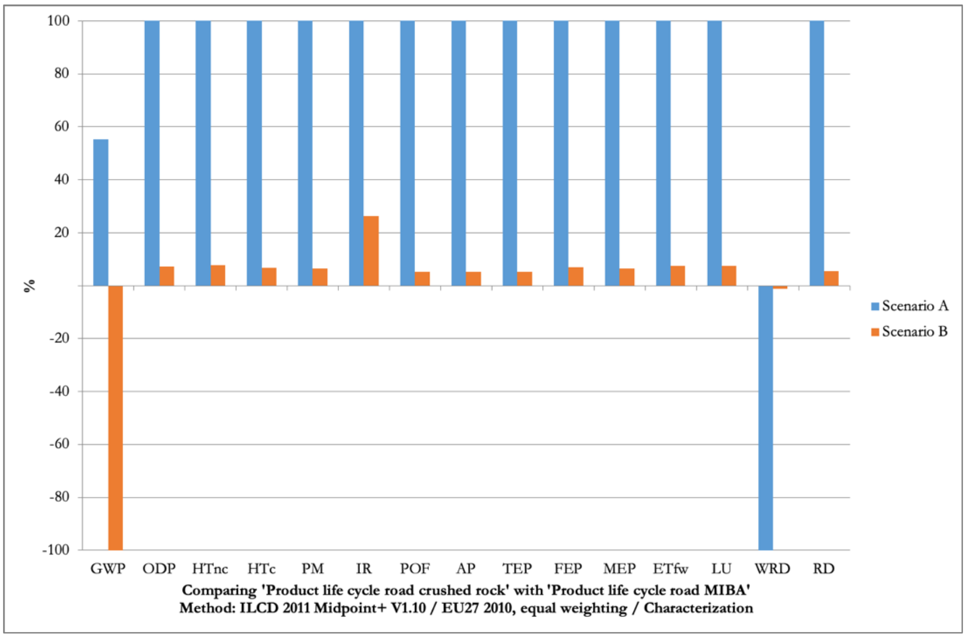


Figure 37: Comparison of the characterization results for scenario A and B. The Y-axis is a percentage scale.

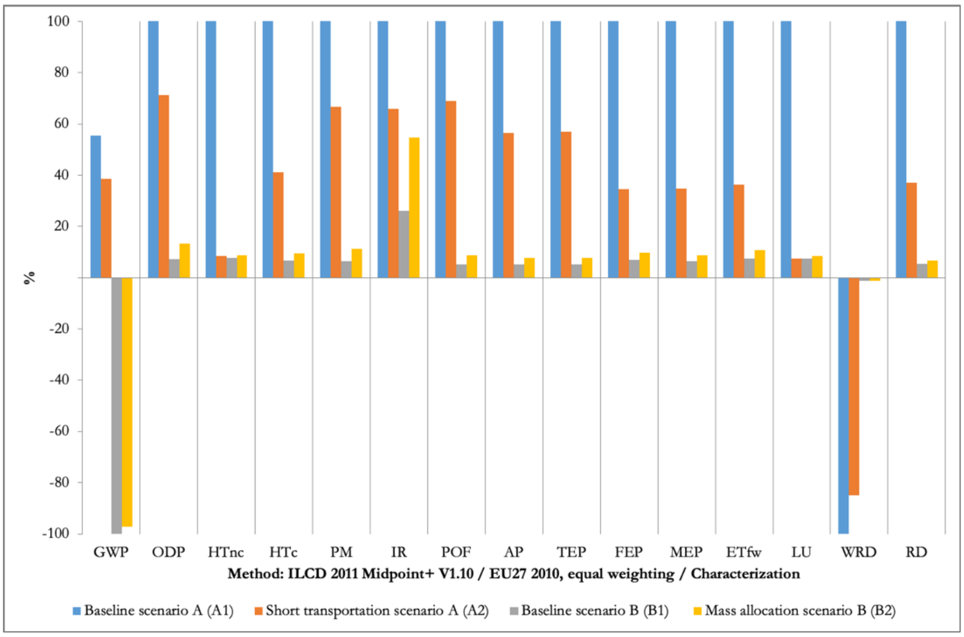


Figure 38: Comparison of the characterization results for scenarios A1, A2, B1, and B2. The y-axis is a percentage scale.

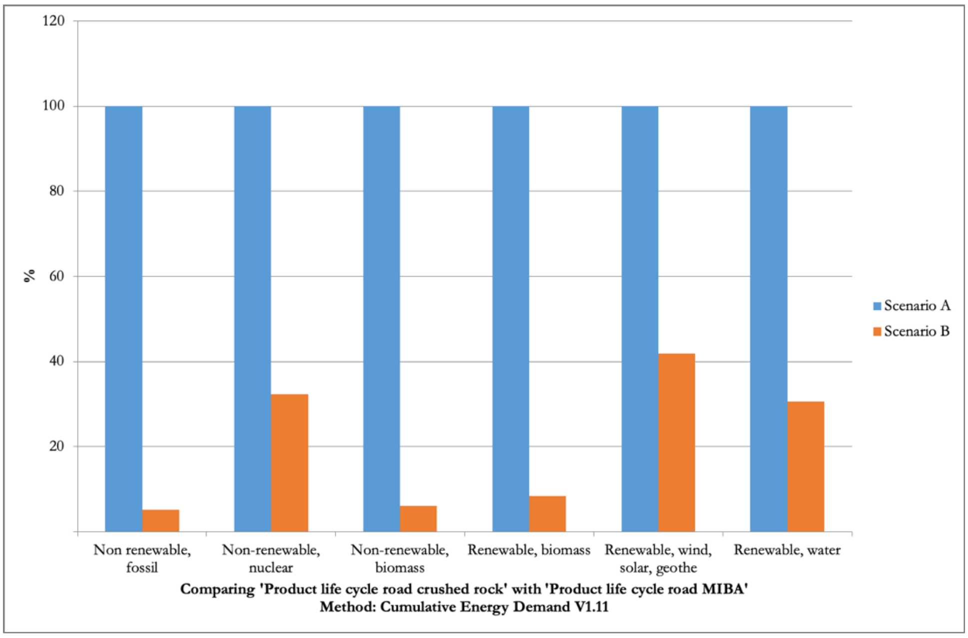


Figure 39: Comparison of the characterization results of the CED for scenario A and B. Results for scenario A are set to 100%. The Y-axis is a percentage scale.