Development of circular economy indicators starting from the in-use occupation of materials

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FACULTY OF BIOSCIENCE ENGINEERING

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For the master's tools will never dismantle the master's house.
Audre Lorde

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Development of circular economy indicators starting from the in-use occupation of materials

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Dissertation submitted to Ghent University in partial fulfilment of the requirements for the degree of Doctor of Bioscience Engineering: Environmental Sciences and Technology.

Academic year: 2020–2021

Dutch translation of the title:

Ontwikkeling van indicatoren voor de circulaire economie vanaf het gebruik van materialen

Cover image credit:

Flickr/CC BY 2.0/Carbon Visuals, One day's carbon dioxide emissions from above (quantity of carbon dioxide human activity added to the atmosphere every day in 2012 in scale over the island of Manhattan).

Please refer to this work as:

Moraga, G. L. (2021). Development of circular economy indicators starting from the in-use occupation of materials. PhD thesis, Ghent University, Belgium.

ISBN: 978-94-6357-429-7

Funding:

This work was financially supported by the Flemish administration of Belgium via the Steunpunt Circulaire Economie (Policy Research Centre Circular Economy).

Summary

Our society has an astounding and increasing consumption of materials. By 2050, three planets could be needed to provide resources for our current lifestyle. The world's climate is the direct subject of how the global economy manages natural resources, and resource efficiency will be vital to meet the Paris Agreement's temperature goals. In this sense, the circular economy concept can influence how we manage resources. The circular economy can promote the responsible and cyclical use of resources. In recent years, a circular economy has been endorsed as a policy to minimise burdens to the environment and stimulate the economy. The recent New Circular Economy Action Plan intends to achieve carbon neutrality and more efficiency in resources and materials management in the European Union. The supply chain of materials can much benefit from circular economy strategies to recover materials and products. However, despite the benefits of keeping materials in the loop, there will always be environmental burdens and cumulative use of resources associated with a chosen circular economy strategy. Thus, to target better policy towards a circular economy, indicators considering sustainability are needed. In the context of the Policy Research Centre for Circular Economy (Steunpunt Circulaire Economie) promoted by the Flemish Administration, this doctoral dissertation aimed to develop circularity indicators of materials in supply chains.

As the initial step, this dissertation focused on understanding the state-of-the-art of circular economy indicators in chapter 2. A classification framework is proposed to categorise indicators according to the reasoning of what indicators measure (circular economy strategies) and how they do so (measurement scopes). There are plenty of circular economy strategies, but they can be grouped according to their attempt to preserve functions, products, components, materials, or embodied energy. The measurement scope can show how indicators account for technological cycles (with or without a life cycle thinking approach) or the effects of such cycles on environmental, social, or economic aspects. We illustrated the framework with micro-scale indicators from literature and macro-scale indicators from the European Union's 'circular economy monitoring framework'. The framework illustration showed that most of the indicators focus on preserving materials, with strategies such as recycling. Although micro-scale indicators can assess strategies considering a life cycle thinking approach, the

European indicators often account for materials-based strategies without much life cycle thinking consideration. From the indicators considering life cycle thinking, few indicators assessed time, despite many circular economy definitions explicitly referring to an economy ‘where resources are kept for as long as possible.’ Furthermore, none of the analysed indicators could assess the preservation of functions (related to circular economy strategies such as sharing platforms). Finally, the framework illustration suggested that a set of indicators should be used to assess circular economy instead of a single indicator.

Circular economy strategies of slowing and closing loops of resources have the ultimate goal of keeping materials useful (i.e., in-use) while avoiding losses (dissipation). With this reasoning, this dissertation proposed a set of indicators in chapter 3. We proposed measuring the circularity of materials by quantifying their in-use occupation, that is, the maintenance of materials in a useful state in products for as long as possible while avoiding dissipation or hibernation. Specifically, two indicators were developed: in-use occupation ratio (UOR) and final retention in society (FRS). These indicators were applied in two case studies (materials in laptops and wood products) with three scenarios each: linear, product preservation (reuse), and material preservation (recycling). The reuse scenarios generally presented a higher UOR (41–48% for laptop materials and 53% for wood) compared to recycling scenarios (29–45% for laptop materials and 52% for wood). Only two scenarios of wood products resulted in retaining materials for the next generation (FRS > 0%). We argue that the differentiation between supply, in-use, and hibernation phases is essential for a circular economy.

UOR and FRS can measure the use of materials over time while considering life cycle thinking. In this sense, the in-use occupation-based indicators are a proxy for the benefit, or handprint, that materials provide in society. However, these indicators miss the connection with sustainability, particularly the environmental footprint caused by using such materials. Hence, in chapter 4, we further developed these indicators using the concept of resource efficiency to indicate the handprint and footprint of the used materials. We illustrated the developed indicators, resource efficiency of in-use occupation (Eff_{Occ}) and of final retention (Eff_{FRS}), with a case study of four materials (aluminium, copper, iron, and plastics) embedded in laptops. The study included scenarios with different circular economy strategies: energy recovery, recycling, refurbishing, and reuse. The scenarios showed the use of the materials in several cycles of laptops over a 25-year time horizon. Scenarios with cycles of refurbishment and reuse showed an improvement in Eff_{Occ} up to 189% and 157%, respectively, when compared to energy recovery. Nonetheless, it was remarkable that the average Eff_{Occ} and Eff_{FRS} showed a preference for refurbishing scenarios over reuse, considering the 25-year time horizon.

Finally, we concluded this dissertation in chapter 5 with further analysis, perspectives, and concluding remarks. Firstly, we critically assessed the proposed indicators (chapter 3 and 4) against the classification framework (chapter 2). The proposed indicators can measure a wide range of circular economy strategies. However, more work is still needed to assess function-related strategies. Still, we suggested possible pathways so that indicators could analyse such strategies. Secondly, we suggested future development of the in-use occupation concept with life cycle assessment, particularly the development of impact assessment methods for material inaccessibility and suggestions for using the indicators in policy-making. Finally, we presented this dissertation concluding remarks.

Samenvatting

De consumptie van materialen in onze samenleving neemt met een verbazingwekkende snelheid toe. Tegen 2050 zouden drie planeten nodig zijn om onze huidige levensstijl te kunnen behouden. Efficiënt gebruik van hulpbronnen zal van vitaal belang zijn om de temperatuurdoelstellingen van de Klimaatovereenkomst van Parijs te behalen. Circulaire economie kan een belangrijke invloed hebben op het beheer van hulpbronnen en het kan een verantwoord en cyclisch gebruik van hulpbronnen bevorderen. De afgelopen jaren is een circulaire economie onderschreven als beleid om de belasting van het milieu te minimaliseren en de economie te stimuleren. Het recente nieuwe actieplan voor de circulaire economie is bedoeld om koolstofneutraliteit en meer efficiëntie in het beheer van hulpbronnen en materialen in de Europese Unie te bereiken. De toeleveringsketen van materialen kan veel baat hebben bij deze strategieën om materialen en producten terug te winnen. Ondanks de voordelen van het in de kringloop houden van materialen, zullen er altijd milieubelastingen en cumulatief gebruik van hulpbronnen zijn die samenhangen met een gekozen strategie voor een circulaire economie. Om het beleid de juiste richting in te kunnen sturen, zijn dus indicatoren nodig die de duurzaamheid van systemen in kaart brengen. In het kader van het Steunpunt Circulaire Economie, gepromoot door de Vlaamse Overheid, had dit doctoraatsproefschrift tot doel de circulaire indicatoren van materialen in toeleveringsketens te ontwikkelen.

Als eerste stap richtte dit proefschrift zich op het begrijpen van de state-of-the-art indicatoren voor de circulaire economie in hoofdstuk 2. Er wordt een classificatiekader voorgesteld om indicatoren in te delen op basis van wat deze meten (circulaire economiestrategieën) en op basis van hun meetbereik. Er zijn tal van strategieën voor circulaire economie, maar ze kunnen worden gegroepeerd op basis van hun poging om functies, producten, componenten, materialen of belichaamde energie te behouden. Het meetbereik kan laten zien hoe indicatoren rekening houden met technologische cycli (met of zonder levenscyclusbenadering) of de effecten van dergelijke cycli op milieu-, sociale of economische aspecten. Het framework met microschaalindicatoren uit de literatuur en macroschaalindicatoren uit het ‘monitoringraamwerk circulaire economie’ van de Europese Unie werden gebruikt als leidraad. Deze liet zien dat de meeste

indicatoren gericht zijn op het conserveren van materialen, met strategieën zoals recycling. Hoewel microschaalindicatoren strategieën kunnen beoordelen waarbij een levenscyclusbenadering wordt overwogen, houden de Europese indicatoren vaak rekening met op materialen gebaseerde strategieën zonder veel aandacht voor levenscyclusdenken. Van de indicatoren die wel hiermee rekening houden, beoordeelden weinig indicatoren het tijdsaspect, ondanks dat veel definities van de circulaire economie expliciet verwijzen naar een economie 'waar hulpbronnen zo lang mogelijk worden bewaard'. Bovendien kon geen van de geanalyseerde indicatoren het behoud van functies beoordelen (gerelateerd aan circulaire economiestrategieën zoals deelplatforms). Ten slotte suggereerde het framework dat een set indicatoren zou moeten worden gebruikt om de circulaire economie te beoordelen in plaats van één enkele indicator.

Strategieën in de circulaire economie voor het vertragen en sluiten van kringlopen met allerlei hulpbronnen hebben het uiteindelijke doel om materialen bruikbaar (d.w.z. in gebruik) te houden en verliezen (dissipatie) te vermijden. Met deze redenering stelde dit proefschrift een reeks indicatoren voor in hoofdstuk 3. We stelden voor om de circulariteit van materialen te meten door hun gebruiksbezetting te kwantificeren. Hierbij worden materialen zo lang mogelijk behouden in hun bruikbare staat door deze te verwerken in producten in plaats van deze verloren te laten gaan in het milieu of door deze niet langer te gebruiken (bv. metalen in oude GSM die thuis bewaard wordt). Er werden twee specifieke indicatoren ontwikkeld: de bezettingsgraad in gebruik (UOR) en de uiteindelijke retentie in de samenleving (FRS). Deze indicatoren zijn toegepast in twee case studies (materialen in laptops en houtproducten) met elk drie scenario's: lineair, productbehoud (hergebruik) en materiaalbehoud (recycling). De scenario’s met hergebruik lieten over het algemeen een hogere UOR zien (41-48% voor laptopmaterialen en 53% voor hout) in vergelijking met recyclingscenario's (29-45% voor laptopmaterialen en 52% voor hout). Slechts twee scenario's met houtproducten resulteerden in behoudsmaterialen voor de volgende generatie (FRS > 0%). We stellen dat het onderscheid tussen fasen waarbij levering, in-gebruik en stockering van materialen optreedt, essentieel is voor een circulaire economie.

UOR en FRS kunnen het gebruik van materialen in de tijd meten, rekening houdend met levenscyclusdenken. In die zin geven de indicatoren voor in-use occupation een inschatting van het voordeel (ook wel handafdruk genoemd in LCA-kringen) dat materialen in de samenleving bieden. Deze indicatoren missen echter het verband met duurzaamheid, met name de ecologische voetafdruk die wordt veroorzaakt door het gebruik van dergelijke materialen. Daarom hebben we in hoofdstuk 4 deze indicatoren verder ontwikkeld met behulp van het concept van hulpbronnenefficiëntie om de handafdruk en voetafdruk van de gebruikte materialen aan te geven. De ontwikkelde indicatoren, hulpbronnenefficiëntie van in-use bezetting (EffOcc) en van uiteindelijke retentie (EffFRS) werden geïllustreerd via een casestudy met vier verschillende materialen (aluminium, koper, ijzer en plastic) die in een laptop zijn ingebed. De studie omvatte scenario's met verschillende strategieën voor de circulaire economie: energieterugwinning, recycling, opknappen en hergebruik. De scenario's toonden het gebruik van de materialen in verschillende gebruikscycli van de laptops over een tijdshorizon van 25 jaar. Scenario's met cycli waarbij opknappen en hergebruik optrad, toonden een verbetering in EffOcc tot respectievelijk 189% en 157% in vergelijking met energieterugwinning. Desalniettemin was het opmerkelijk dat de gemiddelde EffOcc en EffFRS een voorkeur lieten zien voor opknapscenario's boven hergebruik, gezien de tijdshorizon van 25 jaar.

Ten slotte hebben we dit proefschrift in hoofdstuk 5 afgesloten met verdere analyse, perspectieven en slotopmerkingen. Ten eerste hebben we de voorgestelde indicatoren (hoofdstuk 3 en 4) kritisch getoetst aan het classificatiekader (hoofdstuk 2). De voorgestelde indicatoren kunnen een breed scala aan strategieën voor de circulaire economie meten. Er is echter nog meer werk nodig om functiegerelateerde strategieën te beoordelen. Toch stelden we mogelijke trajecten voor zodat indicatoren dergelijke strategieën zouden kunnen ondersteunen. Ten tweede suggereerden we toekomstige ontwikkeling van het concept in-use occupation binnen de levenscyclusanalyse, met name de ontwikkeling van effectbeoordelingsmethoden voor materiële ontoegankelijkheid en suggesties voor het gebruik van de indicatoren in beleidsvorming. Tot slot presenteerden we dit proefschrift met de slotopmerkingen.

Acknowledgments

Not often I think about this: Life is a precious thing. Getting a PhD amidst a pandemic certainly makes one appreciate and be grateful for the beautiful gift that life is. Life is essential to be present, but it is (arguably) a unique chance to make a difference—to do things right.

To try to do things a bit more right (or less wrong), I took this PhD challenge to develop circular economy indicators. In the first place, I shall thank the Flemish administration for providing financial support. Likewise, I thank Ghent University and the Circular Economy Policy Research Centre for their technical and administrative support. More than technically, I should thank my supervisors Jo, Steven, and Sophie: this PhD would have been impossible without your smart and focused help. I especially thank my main supervisor, Jo, for believing in me when I couldn't. It might be common place to say that I'm a different person than four years ago when I started this challenge; one would have been changed anyway. Still, I'm thankful for all that I have learned and—dare I say—evolved during this period. Also, I thank my fellow colleagues in STEN and EnVOC for all the interesting discussions and fun moments during lunch, coffee breaks, and Summer and Christmas activities. I also thank my research colleagues at the PRC—especially Luc, for the many words of support. Over the last year, I also thank Fábio for keeping me healthy.

However, not only of PhD a PhD student lives. It is impossible to forget the friendships that lighten up my life in Belgium. Some of them from work, some of them from encounters and reencounters. You are very much in my heart forever: Laia, Fernanda, Moritz, Vanessa, Roger, Isa, Victor, Laís, Lígia, Karl, Gaby, Pawel, Kamila, Thibaud, Roberto, Joana, Mel, Lena, Steven, Angelina, and David.

Presently, I'm somewhere over the Atlantic Ocean. I'm returning from a long-awaited trip to visit some of my most beloved people. First, I would like to thank my mother (in memoriam) for the unconditional love that taught me kindness and hope: I miss you so much, but I know you would be so much happy right now. To my father, I'm thankful for believing in me and cheering up for me. To my sister, Mari, the most beautiful mind that I know, thank you for being a continuous source of inspiration. Moreover, I thank my cousins for being always there when I needed them, especially: Be, Bel, Bru, and Jo. And, to my extended family, my grandparents, uncles, and aunts: thank you. I'm also grateful to my friends: Matheus, Riggs, Jaque, and Vivi.

Lastly, Rodrigo, my partner in crime, my heart and my soul, my first and last thought every day. You have waited a long time for this; so, here it is.

International waters—Atlantic Ocean,

August 2021

Table of contents

List of abbreviations

ADP	Abiotic Depletion Potential (ADP)
AF	Allocation factor
Al	Aluminium
Ave	Avoided environmental impacts of energy production due to energy recovery
Avm	Avoided environmental impacts of the production of the virgin raw material after the TH
BSI	British Standard Institute
C&D	Construction and demolition (waste)
CC	Climate change
CE	Circular economy
CEC	The Council of the European Communities
CEENE	Cumulative exergy extraction from the natural environment
CEI	Circular economy index
CIRAIG	Centre for the Life Cycle of Products Processes and Services
CIRC	Material circularity indicator
CPI	Circular economy performance indicator
CR	Old scrap collection rate
Dis	Environmental impacts of final disposal
DMC	Domestic material consumption
dMFA	Dynamic material flow analysis
e&m-recovery	Energy recovery and material recovery from bottom ash
EASAC	European Academies' Science Advisory Council
EC	European Commission
eDiM	Ease of disassembly metric
EEA	European Environmental Agency
EEE	Electrical and electronic equipment
Eff_{FRS}	Resource efficiency of the final retention in society
Eff_{Occ}	Resource efficiency of the in-use occupation
EMF	Ellen MacArthur Foundation
EoL	End-of-life
EoL-RIR	End-of-life recycling input rates

EOL-RR	End-of-life recycling rate
e-recovery	Energy recovery
EU	European Union
EVR	Eco-cost value ratio
Fe	Iron
FRS	Final retention in society
Ftp	Footprint associated with the employment of 1 kg of material (resource consumption or emissions)
FU	Functional unit
GDP	Gross domestic product
GPP	Green public procurement
GRI	Global resource indicator
GWP100	Global warming potentials in 100a time horizon
IO-LCA	input-output life cycle assessment
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
JRC	European Commission’s Joint Research Centre
LCA	Life cycle assessment
LCD	Liquid crystal display
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LCT	Life cycle thinking
l_H	Loss of material from the hibernation phase
LiVES	Lifespan database for Vehicles, Equipment, and Structures
LMA	Lifetime of materials on anthroposphere
l_S	Loss of material from the supply phase
l_U	Loss of material from the in-use phase
M	Environmental impacts of manufacturing a product
MCI	Material circularity indicator
MFA	Material flow analysis
m_H	Mass of material going into the hibernation phase
m_S	Mass of material going into the supply phase
m_{TH}	Mass of material at the time horizon
m_U	Mass of material going into the in-use phase
NACE	Nomenclature statistique des activités économiques dans la Communauté Européenne (Statistical Classification of Economic Activities in the European Community)
NTUM	Number of times of use of a material

Occ_U	In-use occupation
Occ_{Umax}	Theoretical maximum in-use occupation
OSR	Old scrap ratio
P	Environmental impacts of the primary or secondary processing of the material
PCB	Printed circuit board
PET	Polyethylene terephthalate
PL	Plastics
PLCM	Product-level circularity metric
PM	Precious metals
PSS	Product-service systems
Rc	Environmental impacts of recycling the materials embedded in a product
Re	Environmental impacts of reusing a product
Rf	Environmental impacts of refurbishing a product
RIR	Recycling input rate
RR	Recycling process efficiency rate
SCI	Sustainable circular index
SuMMa	Steunpunt Duurzaam Materiaalbeheer (Support Centre for Sustainable Materials Management)
SUPRIM	Sustainable Management of Primary Raw Materials
TH	Time horizon
t_H	Time occurrence of the hibernation phase start
TRP	Total restored products
t_S	Time occurrence of the supply phase start
t_U	Time occurrence of the in-use phase start
UNEP/IRP	United Nations Environment Programme’s International Resource Panel
UNEP/LCI	United Nations Environment Programme’s Life Cycle Initiative
UOR	In-use occupation ratio
V	Environmental impacts of the production of 1 kg of the virgin raw material
VRE	Value-based resource efficiency
WEEE	Waste electrical and electronic equipment
Δt	Time of a product cycle
Δt_H	Duration of the hibernation phase
Δt_S	Duration of the supply phase
Δt_U	Duration of the in-use phase

1 Introduction

In the 1960s, we had the opportunity to see ourselves from space for the first time – the images from Earth largely influenced the environmental movement starting in that decade.[1] It gave us empiric evidence of Earth's limited boundaries and resources. Rising concern about our limited access to resources prompted the circular economy's early ideas. Metaphorically, Kennet Boulding (1966) wrote that Earth is

> *'(…) a single spaceship, without unlimited reservoirs of anything, either for extraction or for pollution, and in which, therefore, man must find his place in a cyclical ecological system which is capable of continuous reproduction of material form even though it cannot escape having inputs of energy.'*

Nonetheless, we are still a resource-intensive society. From 1970 to 2017, we increased the use of materials from 26.7 to 88.6 billion tonnes (UNEP/IRP, 2017). In the future, these trends in material use tend to be intensified with our growing population (Figure 1.1). It is projected that by 2100, the world's population will be close to 11 million people (UN, 2019). Likewise, the extraction of materials may pass 180 billion tonnes if we

[1] Earthrise, the photograph of Earth taken by Apollo-8-astronaut Bill Anders in 1968, has been credit for sparking the environmental movement (https://www.nasa.gov/centers/johnson/home/earthrise.html).

continue the historical trend of extraction (UNEP/IRP, 2017). By 2100, the global annual demand may have increased two-fold for iron and steel and fourfold for aluminium and copper (Figure 1.1) (Watari et al., 2021). Putting into perspective, Elhacham et al. (2020) estimated that the accumulated human-made material output (i.e. metals, minerals, and wood – excluding waste) surpassed all living biomass on the planet in 2020. The mass of plastics only is already bigger than the mass of all animals on the planet (wild and livestock) (Elhacham et al., 2020).

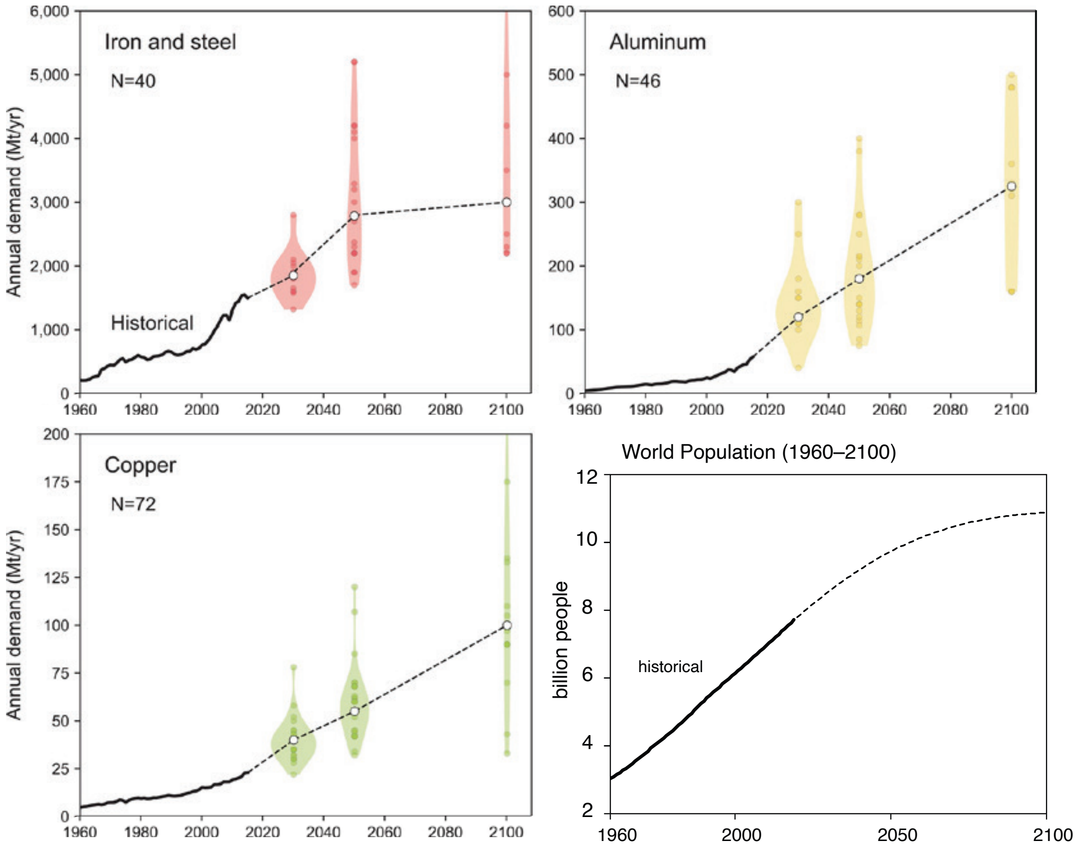

Figure 1.1: Historical and projected annual demand for three metal groups: iron and steel, aluminium, and copper (Watari et al., 2021); World population (UN, 2019). Global-scale from 1960 to 2100

The circular economy (CE) aims that resources stay in the loop so that the extraction of new materials could be displaced. Efficiency in keep materials in the loop is a key aspect for a CE. Much of the worldwide environmental pollution is related to the use of materials, not to mention that material depletion is a problem in itself. Abiotic material resources (i.e. minerals and metals) are finite. Nevertheless, many materials are lost or dissipated in production and consumption processes. Hence, a CE is a promise to halt or decrease the consumption of new raw materials and promote well-being while reducing environmental burdens. In the following subsections, we give an overview of the CE's origins and implementation, its indicators, and possible ways in which a CE could contribute to the sustainable use of materials. The chapter concludes with the objectives of this doctoral research.

1.1 The circular economy origins and implementation

The promotion of a CE is needed for more effective use of resources, in which losses are avoided. Many countries and regions are promoting a CE. The EU developed the CE Action Plan in 2015 (recently reviewed in 2020), but the concept of a more circular society is on the EU's political agenda since the 1970s. In 1975, the EU encouraged waste prevention and recovery 'to conserve natural resources' with its first directive on waste (CEC, 1975). Based on this direction, Germany was at the forefront of CE legislation with the law on kreislaufwirtschaft (circular economy) in 1996 (Ogunmakinde, 2019). In other countries, China has a specific CE law since 2009, but concepts of cleaner production and industrial ecology were already applied before the law came into force (Ogunmakinde, 2019). Japan promoted the reuse and recycling of materials since the early 1990s with the Law for Effective Utilisation of Recyclables. More recently, the British Standard Institute (BSI, 2017) published a first-of-a-kind standard for implementing a CE in organisations.

Despite its use in policy, the CE has heterogeneous principles and definitions. Kirchherr et al. (2017) found no less than 114 different CE definitions. This diversity can be attributed to the different schools of thought of the CE. These schools include, for example, industrial ecology, industrial metabolism, cradle-to-cradle, biomimicry, performance economy, blue economy, and regenerative design (BSI, 2017; CIRAIG, 2015; EMF, 2013a; Geissdoerfer et al., 2017; Homrich et al., 2018). Amusingly, Reike et al. (2017) named CE a refurbished concept. Amidst this plethora of definitions, the CE is often labelled as an umbrella concept (CIRAIG, 2015). Umbrella concepts are broad ideas that encompass diverse phenomena, which arise when a discipline lacks one specific guideline or paradigm (Blomsma and Brennan, 2017). Typically, umbrella concepts direct attention to the shared characteristics of its forming theories (Blomsma and Brennan, 2017). Therefore, the CE centralises some common elements that are present in all its different schools of thought.

Common elements in the core of a CE are to close and slow down the loops of materials. Authors often agree that the CE is about a cyclical closed loop system regarding the use of materials (Murray et al., 2017). Bocken et al. (2016) reasoned about the distinction between closing and slowing loops when consumers are involved. On the one hand, the closing loops are related to actions between a product's post-use and manufacturing, such as recycling or recovery. On the other hand, the extension of loops is related to actions intended to prolong the use of products, such as repair or remanufacture. In this dissertation, these actions of closing and slowing loops are called CE strategies.

As CE itself, CE strategies are not brand new. For example, the well-known 3Rs – reduce, reuse, and recycle – are used in a CE but are on the waste management agenda for many decades. In a CE, the strategies are often in a long list of 're-words'. A review found 38 're-words' in combinations ranging from 3Rs to 10Rs (Reike et al., 2017). The combinations of strategies can be organised in so-called R-ladders in order of preference for their circularity. Such a comprehensive ladder is proposed by Potting et al. (2017b) with ten CE strategies followed by their definitions (Figure 1.2(a)). According to these authors, a rule of thumb is that the higher the strategy, the fewer the needed natural resources. By means of comparison, Figure 1.2(b) shows the EU's waste hierarchy based on their most recent Waste Framework Directive (EC, 2008). Hence, because of the diverse implementation and conceptualisation, the CE can be taught as a policy on the go.

From the related CE concepts, the methodology of industrial ecology is particularly influential. Industrial ecology offers an inspirational model to plan industrial systems in consonance with natural ecosystems. In ecosystems, energy and limited resources produce a cyclical or quasi-cyclical use of materials, in which waste is hardly produced (Graedel, 1996). The connection of industrial ecology with a cyclical use of materials is evident since the early stages of this concept. In their seminal paper, Frosch and Gallopoulos (1989)[2] described that in industrial ecology, the 'waste from one process can serve as the raw material for another.' Moreover, these authors argued that the cyclical use of materials has the potential to reduce environmental impacts of production and manufacturing. Interestingly, before the industrial ecology's seminal paper, a study named L'Écosystème Belgique. Essai d'Écologie Industrielle was publish in Brussels in

[2] Ideas of industrial ecology existed since 1940s, but the Frosch and Gallopoulos's paper is regarded as the 'official birth of the industrial ecology concept' (CIRAIG, 2015).

1983. This independent publication gave an overview of the Belgian economy based on national statistics. However, instead of monetary values, they expressed the economy 'in terms of circulation of materials' disaggregated into several streams (e.g. iron, glass, plastic, lead, wood, and paper) (Erkman, 1997).

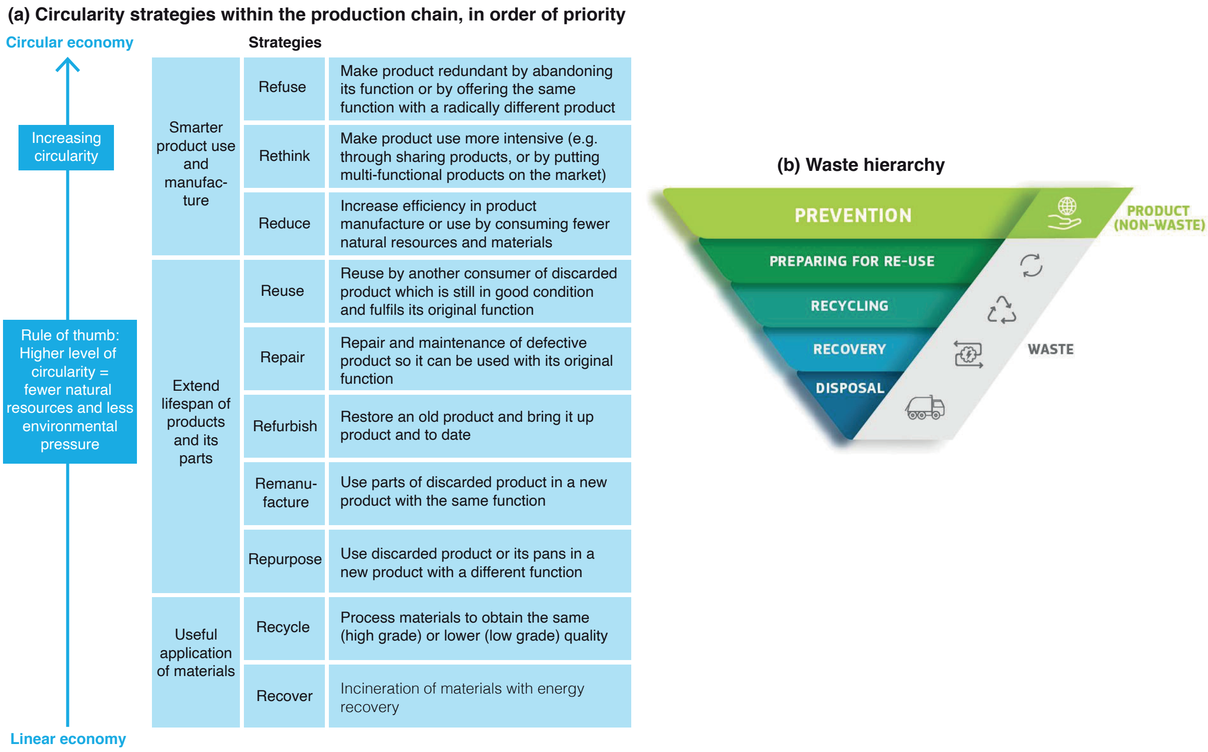

Figure 1.2: (a) Circularity strategies within the product chain, in order of priority (adapted from Potting et al. (2017b)); (b) EU's waste hierarchy (EC, 2021)

The influence of industrial ecology can be drawn upon the example of the European Commission's (EC) definition of a CE. An economy is circular when the value of resources, materials, and products is kept for as long as possible and the generation of waste minimised (EC, 2015a). This value is maintained as long as the flow of resources is kept in the cycle, avoiding a take-make-dispose economy, which generates waste. Indeed, some definitions of a CE include the term resources, which can present multiple interpretations. Natural resources can be classified into many groups, such as biotic (e.g. wood) and abiotic (e.g. metals); renewable (e.g. wind) and non-renewable (e.g. natural gas); and fund (i.e. similar to biotic), flows (i.e. similar to renewable), and stocks (i.e. similar to non-renewable) (Swart et al., 2015). We initially distinguish natural resources based on their general application: materials or energy. For the purpose of this dissertation, we analyse and discuss resources in terms of their use in material applications.

On a regional scale, the Flemish SuMMa (Steunpunt Duurzaam Materiaalbeheer)[3] investigated actions to promote a more sustainable and circular use of materials. SuMMa was in operation from 2012 to 2015 and was the predecessor of the current Support Centre for Circular Economy (CE Centre). The CE Centre combines researchers from Ghent University, KU Leuven, University of Antwerp, and VITO since 2017.[4] It aims to give scientific-based support to policy-making towards a CE in Flanders. The centre has seven subject areas: measure the transition, model systems, learning effects, market acceptance, financing and revenue models, employment and actor analysis, and indicators for circularity. The latter situates this doctoral research. Evidently, from the variety of subjects above, the CE Centre vision of a CE is broad. Indeed, the CE Centre may be an example of an organisation influenced by industrial ecology but also other schools of thought.

Moreover, the influence of industrial ecology is also relevant with private organisations. One such organisation is Ellen MacArthur Foundation (EMF), which stimulates an economic model 'that is restorative or regenerative by intention and design' (EMF, 2013a). The EMF is a think-tank founded in 2010 whose mission 'is to accelerate the transition to a circular economy' (EMF, 2015a). This foundation is among the most important actors in popularising the term 'circular economy' in European policy (Kovacic et al., 2019). One example of EMF influence is what has become a mainstream illustration of a CE – the so-called butterfly diagram (Figure 1.3). In this diagram, they show the flow of materials in two groups – biological and technical nutrients. These flows are presented as inner (faster) and outer (slower) cycles pointing out the preference strategies (in a similar fashion as Figure 1.2(a)). One of the main messages from EMF is to promote an economic model that imitates, or take inspiration from, natural processes where nutrients are assimilated in nature (Kovacic et al., 2019). Hence, the human-made biological and technical nutrients should be designed to be assimilated by the economy or nature. Nonetheless, critics point out that this differentiation only

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[3] Support Centre for Sustainable Materials Management (https://ce-center.vlaanderen-circulair.be/nl/over-ons/voorloper-2012-2015)
[4] https://ce-center.vlaanderen-circulair.be/

complicates the recovery of materials (Kovacic et al., 2019) since natural and processed materials can have organic and inorganic elements (Velenturf et al., 2019).

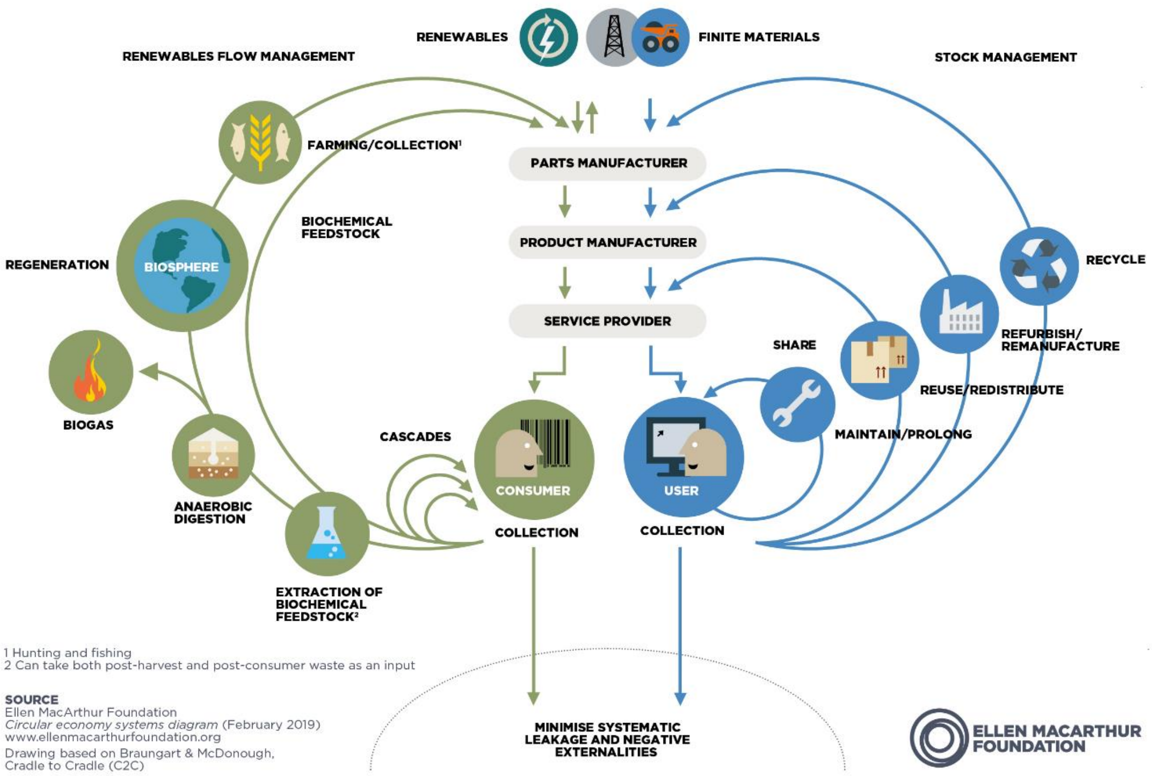

Figure 1.3*: Ellen MacArthur Foundation's butterfly diagram* (source: EMF, 2013b)

Although some cohesion can be found in the development of a CE policy, critics argue that the CE is much focused on waste and recycling and that there is no fundamental shift from other policies, such as resource efficiency or waste management (Allwood, 2014; Kovacic et al., 2019; Murray et al., 2017). Neither clear is the promise of reduced environmental impacts through CE strategies (Kirchherr et al., 2017). Often, CE strategies are analysed as single actions, which can create a burden shift. To further develop the CE concept, Blomsma and Brennan (2017) argue that strategies should be analysed in sequence or in parallel configurations.

Finally, although the CE lacks a widely agreed definition, its evolution and policy usage can help us grasp the directions that a CE can lead. However, some measurement is needed to assess the correctness of these directions. In the following subsection, we introduce indicators as a way to assess CE, which is the main focus of this dissertation.

1.2 How to measure the progress towards a circular economy?

A possible way to measure the progress towards a CE is using indicators. However, before we introduce CE indicators, the term indicator needs clarification. An indicator is ‘a thing that indicates the state or level of something’ (Oxford Dictionary, 2020a). In other words, the level of fuel in a car or the charge level in a mobile phone are everyday examples of indicators. Indicators can be qualitative or quantitative. Qualitative indicators can measure intangible concepts, such as happiness or empowerment. On the other hand, quantitative indicators concern tangible measures, such as the amount of materials flowing in the economy. The focus of this dissertation is on quantitative indicators.

Indicators as tools can be used for a diversity of phenomena. But it is important to distinguish between the phenomena and the indicator. Indicators give selected information about a phenomenon that cannot be directly measured as a whole (Kovacic et al., 2019). Thus, an indicator is a proxy for something. For example, a pluviometer indicates the amount of rainfall over a period of time in a specific area – it does not measure the exact amount of the whole rainfall. The amount of precipitation is a selected piece of information about the whole phenomenon of rainfall but says nothing about its other characteristics, such as droplets size and shape, distribution, velocity, and so on. This differentiation between indicator and phenomenon may be straightforward for a natural phenomenon but not when the phenomenon is a concept, such as the CE (Kovacic et al., 2019). What follows is that a CE (or circularity) as a whole cannot be directly measured with indicators. What we can measure through indicators is a selection of characteristics about a CE with a specific definition.

CE is such a debated framework that the development and use of its indicators are abstruse. In other words, how would one measure the rainfall amount if rainfall was not defined? Despite the lack of agreement about CE, many indicators were proposed in a variety of scopes. This variety can cause governments or companies to cherry-pick results that are suitable with a specific circularity message. For example, although the standard BS 8001:2017 proposes that the CE should be measured and monitored, it leaves the organisations to choose appropriate indicators. Aware of this problem, Pauliuk (2018) proposed a dashboard with indicators to be used with the BS 8001:2017. However, other existing indicators were left out because they might measure scopes outside this standard's boundary. Evidently, CE indicators are measuring something, but what they show is not always clear because the concept itself is under construction.

Nonetheless, for the development of CE indicators in policy, Potting et al. (2018a) showed the overarching aspects of the CE policy process (Figure 1.4). The policy towards a CE transition is composed of means (input), activities (throughput), achievements (output), and effects (outcome). The first step in a policy process is the goal(s) definition. This definition can include desired output (e.g. 50% more plastics recycling in 5 years) and the possible outcome (e.g. reduced environmental impacts from plastics waste management). The policy can also include the needed inputs for these goals (e.g. promoting plastics recycling) and their implementation process (e.g. tax-reduction for recycling companies). All in all, indicators in all these four steps are needed. However, in the scope of this dissertation, we focus on indicators for output and outcome.

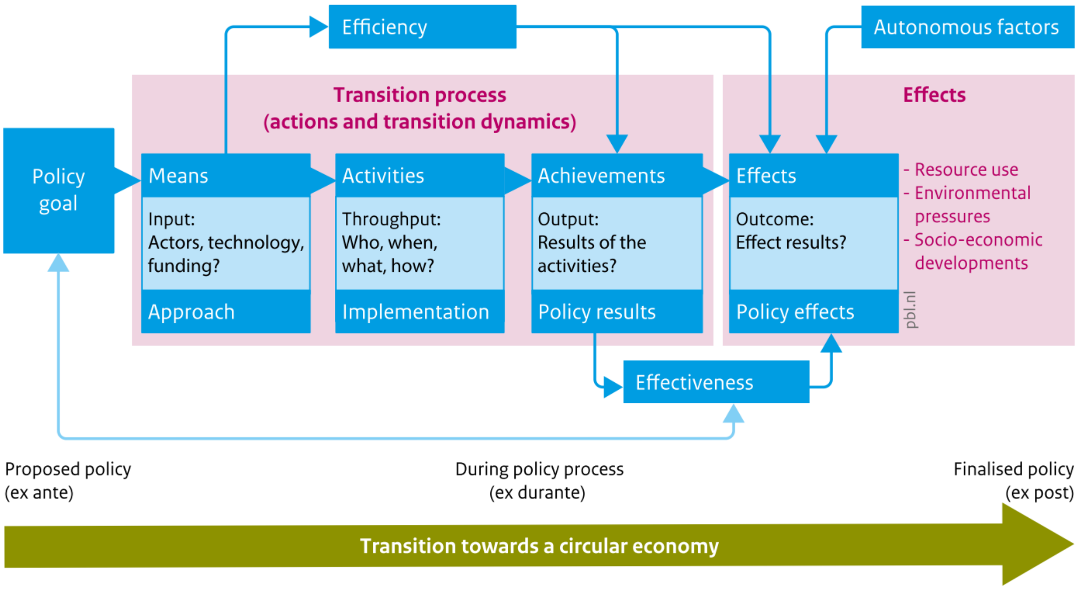

Figure 1.4: Assessment framework for measuring the progress towards a CE (Potting et al., 2018a)

To understand what output and outcome indicators measure, a first differentiation can be made upon their scale. This scale can range between product, business, city, region, country, and planet (Kirchherr et al., 2017). This differentiation in scale is also mentioned as micro, meso, and macro-scale. Another differentiation is about their methodology. CE indicators can measure the use of materials, products, or other resources concerning, for example, economic value (Linder et al., 2017), environmental burdens (Scheepens et al., 2016), energy (Cullen, 2017), exergy (Huysman et al., 2017), and time (Vanegas et al., 2018). Moreover, indicators can measure indexed combinations or ratios of the mentioned examples. Lastly, indicators can have different scopes about the life cycle of products and materials. An indicator with a narrower scope can measure, for example, one specific company's output (e.g. waste generation). In comparison, an indicator with a broader scope can include the whole upstream and downstream supply chain. These scopes can be linked to the circularity itself (output) or the effects of such circularity (outcome).

Some frameworks were proposed to assess the correctness of CE indicators (e.g. Elia et al., 2017; Iacovidou et al., 2017; Pauliuk, 2018). They all, however, abide by strict definitions of a CE. Considering that the CE is an evolving concept (Blomsma and Brennan, 2017; Reike et al., 2017), we should not disregard indicators that do not fit a strict-sense CE. Nonetheless, without a clear CE definition is challenging to define what CE indicators evaluate. On the other hand, instead of delimitating CE based on a definition, we could delimitate the CE based on its existing indicators. Or, to use the previously mentioned example, it may be possible to define rainfall by understanding its measuring instruments. We argue that understanding indicators for a CE, a concept on the go, can frame the concept as it currently is and point out possible directions. A classification as such can identify opportunities for the development of new indicators, which is discussed in chapter 2.

Furthermore, the intersection of sustainability and CE is important. In this dissertation, we take the term sustainability as sustainable development – a 'development that meets the needs of the present without compromising the ability of future generations to meet their own needs' (Brundtland et al., 1987). An issue pointed out by several authors is that the CE does not show clear connections with sustainability (Geissdoerfer et al., 2017; Kirchherr et al., 2017; Kovacic et al., 2019); hence, this is a key issue for the development of CE indicators. An essential topic to sustainable development is related to environmental impacts, which also situates a focus of this dissertation. Indeed, the circular use of materials is related to important effects on the environment. Hertwich et al. (2019) estimated that the absolute emissions related to the global material production were about 11 Gt CO_2-eq in 2015 (Figure 1.5). If we are to meet the Paris Agreement's goal of 1.5° C temperature increase, resource efficiency of materials will be critical considering the world's growing population (UNEP/IRP, 2020). Indeed, a type of CE indicator can be related to resource efficiency.

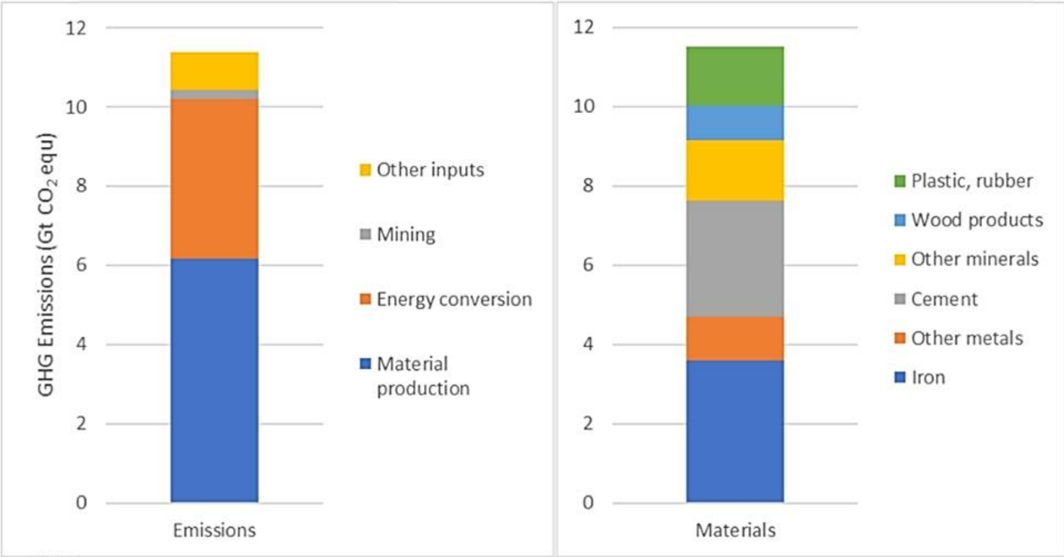

Figure 1.5: Cradle-to-gate global warming emissions from world's key materials in 2015, identified per emitting process (left – material production itself, energy inputs, mining, or other inputs) and identified per material (right) (source: Hertwich et al., 2019)

Resource efficiency is a term that means achieving more benefits with fewer negative consequences. The International Resource Panel (UNEP/IRP, 2017) defines resource efficiency as 'achieving higher outputs with lower inputs and can be reflected by indicators such as resource productivity (including GDP/resource consumption).' In the sense of resource efficiency in a CE, the definition of the proxies used to measure the CE benefit is essential. For example, GDP may be satisfactory to evaluate economic production, but a growing GDP is also a characteristic of a linear economy. However, indicators build to measure a CE must somehow aim at an economy in consonance with sustainability. Therefore, in the following subsection, we analyse possible ways to characterise and quantify a sustainable use of materials to derive CE indicators.

1.3 Characterising and quantifying the sustainable use of materials

Often, the characterisation and quantification of CE indicators rely on the use of industrial ecology tools, such as material flow analysis (MFA) and life cycle assessment (LCA); we introduce them in turn. MFA is a methodology for the assessment of the flows and stocks of materials in a defined space and time (Brunner and Rechberger, 2004). MFA is conducted with mass balances between inputs, stocks, and outputs of a process or system and is widely used to visualise materials in a CE context (Iacovidou et al., 2017). MFA results can be illustrated through Sankey diagrams, in which one can easily identify relevant CE information, such as the size and direction of linear and return flows. Because of these qualities, MFA and Sankey diagrams are essential tools in developing indicators (see, e.g. EC, 2018a; Mayer et al., 2018; Pauliuk, 2018). MFA results show output information, as described in Figure 1.4. Hence, for the development of indicators, this methodology is limited about effect results and can be complemented with, for example, LCA.

While MFA focuses on the whole picture of the flow of materials, LCA is more concerned with the effects of a particular product (good or service). LCA is a standardised methodology (ISO 14040/14044:2006) for the ‘compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle’ (ISO, 2006a). It is a well-known and commonly used tool for assessing a product’s potential environmental impacts from raw materials sourcing to use, and end-of-life management (Iacovidou et al., 2017). The standard (ISO, 2006a) lists four steps for an LCA completion: definition of goal and scope, life cycle inventory analysis, life cycle impact assessment, and interpretation. For its versatility, LCA is another commonly used tool for the development of indicators from a CE perspective.

Considering either MFA or LCA many indicators for the sustainable management of materials are developed using resource efficiency concepts. Huysman et al. (2015b) ponder that the several different types of resource efficiency indicators can be expressed by Eq. (1.1) or Eq. (1.2).

$$Efficiency\ 1 = \frac{benefits}{inventoried\ flows} \tag{1.1}$$

$$Efficiency\ 2 = \frac{benefits}{environmental\ impacts} \tag{1.2}$$

The nominator benefits is a useful output from the production system (e.g. GDP). The denominator inventoried flows is, for example, natural resources, industrial resources, wastes, or emissions. The other denominator, environmental impacts, measures the environmental effects caused by the inventoried flows. Eq. (1.1) originated from thermodynamics in engineering. The original thermal efficiency equation is the dimensionless ratio of the net work delivered over the net heat absorbed by a Carnot engine. As the nominator is always lower than the denominator, the thermal efficiency always predicates a theoretical maximum bound (Heijungs, 2007). However, such a bound is not always present in resource efficiency indicators, such as the resource productivity indicator mentioned from UNEP/IRP. Eq. (1.2) can also be defined as eco-efficiency – or a ratio between intended benefits and generated environmental impacts. An example of an eco-efficiency indicator is GDP/climate change potential (Huysman et al., 2015b). The equations' denominators can be assessed using the already mentioned industrial ecology's tools, MFA and LCA.

Environmental impacts are often referred to as environmental footprint. Footprint is a wide-ranging term for measuring space, area, size, or effect of something (Oxford Dictionary, 2020b). Herein, we use the word footprint as the potential environmental impact or cumulative use of resources related to anthropogenic activity and calculated with LCA. Footprints are, therefore, the sum of direct environmental pressures (e.g. CO_2 released at a cement production plant) and indirect environmental pressures (e.g. upstream CO_2 released in the production of fuels used in the cement kiln). Moreover, in the original definition of eco-efficiency, benefit is the value of production, which often refers to economic value (Huppes and Ishikawa, 2007). In the resource efficiency realm, benefits were used, for example, as monetary values, created environmental benefit, the output of energy or exergy, and economic and social welfare (Huysman et al., 2015b). Therefore, the term benefit embraces a multitude of uses.

On the quantification of (environmental) benefits, the handprint concept can be promising. Alvarenga et al. (2020) distinguished three types of handprint: direct, indirect, and relative. Direct handprint occurs when a product brings positive impacts to its intended user. In a practical sense, direct handprints can be, for example, the human health effect from ingesting food or medicine, or the human well-being and health effects from using a bicycle as transportation. An indirect handprint occurs when an unintended user receives this positive impact. Examples of indirect handprint are increased pollination because of honey production or carbon sequestration with biomass production. Lastly, relative handprint is the positive impact in relation to a benchmark (intended or unintended users may be affected). For example, the environmental improvement of repairing and reusing a product compared to landfilling the old product and buying a new product. In this dissertation, we use the terms handprint and benefits interchangeably. Additionally, when we use the term handprint, we mean direct handprint unless stated otherwise.

Furthermore, much of the discussion about the sustainable use of materials is related to abiotic materials extraction. The over-extraction of those materials can lead to the depletion, or quality decrease, of their natural reserves. It is essential to distinguish, however, the depletion of materials from the depletion of natural reserves. It is common sense that the depletion of natural reserves is unsustainable and should be avoided. In LCA, to measure reserves depletion, a reference of the total reserve must be established. However, the estimation of reserves at a global scale is imprecise and scientifically insubstantial (Dewulf et al., 2021) because the technologically dependent 'ultimately extractable reserves' will never be known (Sonderegger et al., 2020). Far from this discussion is the depletion of materials. In a physical sense, the elements in materials are not destroyable according to the law of mass conservation. Unless elements in such materials are used in nuclear fission or sent to space, transferring materials from the ecosphere to the technosphere cannot make them vanish (Frischknecht, 2016). Notwithstanding, this transfer can decrease the technical-economic availability of materials through, for example, dissipation. Recently, van Oers et al. (2020, 2019), Schulze et al. (2020a, 2020b), and Dewulf et al. (2021) discussed the use of materials not in terms of their depletion but in terms of the human actions leading to their inaccessibility. The Life Cycle Initiative (UNEP/LCI, 2019) recommended the causes of inaccessibility as the way forward to assess the use of minerals (and metals) in LCA. In this dissertation, we take a similar position about materials inaccessibility.

The human actions causing material inaccessibility can be grouped in dissipation into the environment, hibernation in the technosphere, and in-use occupation (van Oers et al., 2020). Dissipation to the environment is the emissions of materials in such low concentration that they become inaccessible for humankind. Dewulf et al. (2021) explain that dissipation is a wide-range term that identifies processes' irreversibility (with, e.g. energy), whereas the term dispersion identifies the scattering of a substance within another. Herein, we refer to dissipation as the scattering of materials. Moreover, hibernation in the technosphere refers to a series of actions that keep resources in the technosphere but without functional use. Hibernation actions can include, for example, landfills, tailings, abandoned infrastructure, hoarded products, and dissipation in the technosphere. Lastly, in-use occupation can be understood as the functional stock of materials. This occupation is classified as a cause for inaccessibility because it generates competition. In other words, the material in product A (e.g. gold in a mobile phone) cannot be in product B simultaneously (e.g. gold in jewellery). However, in-use occupation is the goal of the use of materials and, as such, can be questioned as a cause for inaccessibility (Dewulf et al., 2021).

Therefore, although the first two human actions are undesirable (in the sense that they can be prevented or eliminated), the third action is desirable. In this way, in-use occupation is also the benefit of the use of materials for a specific consumer. Indeed, CE strategies intend to keep the value of materials for as long as possible. To put it another way, CE strategies can increase the in-use occupation of materials. Hence, for the development of CE indicators, the concept of in-use occupation could be used to assess the benefits of circularity, which is discussed in chapter 3. However, the in-use occupation will have an environmental footprint over time and could be used within resource efficiency indicators to assess materials in a CE, which is discussed in chapter 4.

1.4 Objective of this dissertation

The CE has a definition under construction. Yet, there is a need for measurement. A multitude of indicators measuring circularity was developed, but many are not circular-economy specific. This ongoing discussion is critical to understand the benefits and adverse impacts caused by the use of materials considering different CE strategies. The resource efficiency of strategies capable of retaining materials in a longer time horizon is yet to be determined. Indeed, a research question can be formulated as 'how to measure the beneficial use of materials in current and future applications in a circular economy considering environmental concerns?' Hence, to answer this question, this doctoral dissertation's general objective is to develop circular economy indicators that evaluate the beneficial and adverse-environmental effects of the circular use of materials.

To achieve this general objective, we formulated four specific objectives addressed in chapters 2 to 4. The specific objectives are:

1. to understand the gaps with existing CE indicators. This is done with a framework that organises and classifies existing indicators claiming to measure a CE or circularity (chapter 2);

2. to develop indicators that measure the benefit, or handprint, of the use of materials in functional products following different CE strategies (chapter 3);
3. to measure the global warming impact and cumulative resource consumption, herein footprint, associated with the use of materials in functional products (chapter 4);
4. to illustrate the indicators use with case studies (chapter 3 and chapter 4).

As an outlook, chapters 2–4 present independent sections for introduction, methodology, results, and discussion. Chapter 5 discusses the previous chapters altogether and draws conclusions and perspectives regarding both methodology and case studies. Figure 1.6 schematically illustrates the structure of this doctoral dissertation.

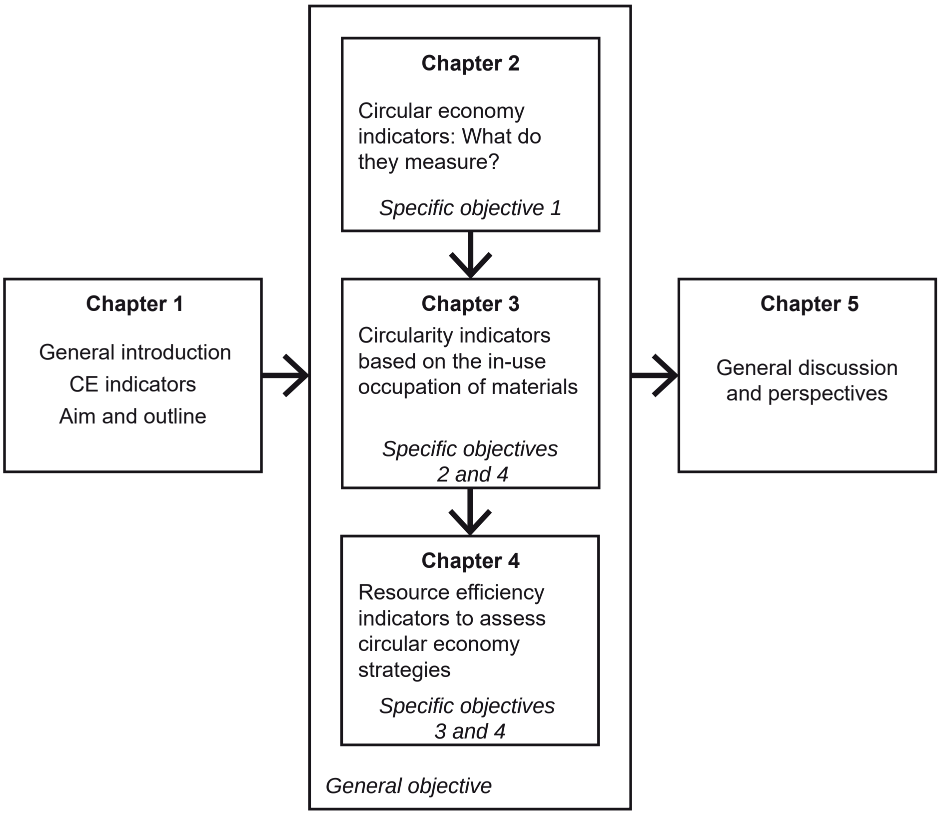

Figure 1.6: Schematic presentation of the contents in this dissertation

2 Circular economy indicators: What do they measure?

Redrafted from:

Title: Circular economy indicators: What do they measure?

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Journal: Resources, Conservation and Recycling

Date: July 2019

Abstract

Circular Economy (CE) is a growing topic, especially in the European Union, that promotes the responsible and cyclical use of resources possibly contributing to sustainable development. CE is an umbrella concept incorporating different meanings. Despite the unclear concept, CE is turned into defined action plans supported by specific indicators. To understand what indicators used in CE measure specifically, we propose a classification framework to categorise indicators according to reasoning on what (CE strategies) and how (measurement scope). Despite different types, CE strategies can be grouped according to their attempt to preserve functions, products, components, materials, or embodied energy; additionally, indicators can measure the linear economy as a reference scenario. The measurement scope shows how indicators account for technological cycles with or without a Life Cycle Thinking (LCT) approach, or their effects on environmental, social, or economic dimensions.

To illustrate the classification framework, we selected quantitative micro-scale indicators from literature and macro-scale indicators from the European Union 'CE monitoring framework'. The framework illustration shows that most of the indicators focus on the preservation of materials, with strategies such as recycling. However, micro-scale indicators can also focus on other CE strategies considering LCT approach, while the European indicators mostly account for materials often without taking LCT into account. Furthermore, none of the available indicators can assess the preservation of functions instead of products, with strategies such as sharing platforms, schemes for product redundancy, or multifunctionality. Finally, the framework illustration suggests that a set of indicators should be used to assess CE instead of a single indicator.

2.1 Introduction

Circular economy (CE) is an approach to promote the responsible and cyclical use of resources. In recent years, CE has been endorsed as a policy to minimise burdens to the environment and stimulate the economy. CE is an umbrella concept (Blomsma and Brennan, 2017; CIRAIG, 2015; Homrich et al., 2018) that includes lowering material input and minimising waste generation (EASAC, 2016; EEA, 2016) to decouple economic growth from natural resource use (Cullen, 2017; EASAC, 2016; Pauliuk, 2018). Despite these actions, so far there is no commonly agreed concept of CE. Different actors have distinct interpretations of what CE could or should depict (Blomsma and Brennan, 2017), where the connection with sustainability is not always clear (Kirchherr et al., 2017). Despite the blurred boundaries of CE definition, there is a need for specific methods to measure the CE progress. In this context, indicators can be useful in various implementation scales and as a tool to assess CE (EASAC, 2016; Geng et al., 2012). However, what to be measured in the sense of CE is subject to debate as the definition is ambiguous, and indicators might lead to different or even incoherent conclusions. Some authors reviewed tools and methodologies already in use.

Elia et al. (2017) assessed a set of selected methodologies and indicators according to five CE characteristics[6] deducted from the European Environmental Agency (EEA, 2016). The authors showed that none of the indicators and methodologies alone was capable of monitoring all the characteristics. Iacovidou et al. (2017) reviewed the methods to assess resource recovery from waste to promote CE. Their results showed that none of the methods alone could account for the retention of value in waste resources, and a holistic evaluation was necessary to encompass the environmental, economic, social, and technical dimensions of CE. Pauliuk (2018) proposed a dashboard of indicators to be used with the standard BS 8001:2017 from the British Standard Institute (BSI, 2017). This standard aims to help the CE implementation in businesses, organisations, and production systems; however, this standard does not contain compliance requirements (Pauliuk, 2018). The proposed dashboard used existing indicators to assess five characteristics promoted by the BSI standard (restore, regenerate, maintain utility, maintain financial value, and maintain nonfinancial value)

[6] According to Elia et al. (2017): Reducing input and use of natural resources; reducing emission levels; reducing valuable materials losses; increase share of renewable and recyclable resources; and increasing the value durability of products.

and existing indicators for complementary characteristics (resource efficiency, climate, energy, and sufficiency).

Notwithstanding, the mentioned studies assessed CE tools considering restrictive CE characteristics (i.e. EEA and BSI) or restrictive scopes (i.e. resources recovered from waste). To bear CE as an umbrella concept, the classification of indicators has to consider CE encompassing different understandings. To our knowledge, such classification is yet to be made. The classification of existing CE indicators according to their capability can map the state of play for the development of new CE indicators. Hence, the objective of this chapter is to understand what quantitative indicators used to assess CE measure specifically, and how they do so. This chapter is limited to analyse output and outcome indicators according to the terminology given by Potting et al. (2017a); thereon, we do not focus on input and throughput indicators. The aims are: (1) to propose a framework to classify indicators according to CE strategies (what) and measurement scopes (how) (section 2.2); (2) to apply and discuss the framework with existing micro-scale indicators from literature (section 2.3); (3) to apply and discuss the framework with macro-scale indicators using the European 'CE monitoring framework' as an example (section 2.4). Finally, we present a closing discussion and conclusions (section 2.5).

2.2 Establishing the classification framework

To establish the classification framework, we propose a rationale to clarify concepts in the CE context. In subsection 2.2.1 and 2.2.2, we present a rationale about critical topics for CE indicators. In subsection 2.2.3, we present the framework overview.

2.2.1 Finding *what* indicators measure in CE

2.2.1.1 CE definitions: sensu stricto and sensu latu

As an umbrella concept, CE is challenging to grasp. While some authors have proposed a consensual and broader definition (Kirchherr et al., 2017; Prieto-Sandoval et al., 2018), others have argued that the attempt of a single definition is merely unachievable (Korhonen et al., 2018b). We understand that by including only one CE definition, we potentially exclude possible meanings. However, to classify the indicators, we need to establish the boundaries encompassing the different CE approaches. As guidance, we use two definitions representing CE in sensu stricto and sensu latu.

The sensu stricto definition has a narrow focus. It is adapted from the rationale of Bocken et al. (2016), where CE is distinguished from the linear economy by two characteristics: slowing and closing resource loops. Slowing happens 'through the design of long-life goods and product-life extension (i.e. service loops to extend a product's life, for instance through repair, remanufacturing),' therefore 'the utilisation period of products is extended and/or intensified, resulting in a slowdown of the flow of resources.' Closing happens when 'the loop between post-use and production is closed, resulting in a circular flow of resources,' meaning the linear flows of waste are turned into secondary resources.[2] Thus, the sensu stricto focuses on the technological cycle of resources.

On the other hand, the sensu latu definition has a broader focus. It is given by Murray et al. (2017) where CE 'is an economic model wherein planning, resourcing, procurement, production and reprocessing are designed and managed, as both process and output, to maximise ecosystem functioning and human well-being.' Thus, the sensu latu definition pushes the focus to sustainability and the effects CE strategies have on the economy, environment, and society. We do not discuss which definition (sensu stricto or latu) is more or less appropriate for CE, but we use the definitions as a basis to establish the framework to understand and map CE indicators.

2.2.1.2 What to measure: CE strategies grouped by common aspects

CE strategies are largely defined in the scientific and grey literature (Blomsma and Brennan, 2017). However, there is no consensual definition of each strategy promoting CE (Reike et al., 2017). For example, reduce can refer either to waste generation, raw materials input, eco-design (e.g. lightweight of products), or consumption. In this context, several ladders, or R-frameworks, position three or more strategies (Kirchherr et al., 2017). One R-framework uses ten strategies to increase circularity: refuse, rethink, reduce, reuse, repair, refurbish, remanufacture, repurpose, recycle, and recover (Potting et al., 2017a). Despite the definitions, CE strategies can preserve products, their parts (modules and components), or the materials (and substances) present in each product's part (Ghisellini et al., 2016; Iacovidou et al., 2017; Potting et al., 2017a). Additionally, CE

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[2] In this paper, closing (resource loops) is used as a reference to prevent waste generation, as far as possible, in the post-use phase. It does not preserve the product or components and includes open/closed loop recycling, downcycling, and energy recovery.

strategies can preserve the energy embodied in resources that cannot be preserved by other strategies (Kirchherr et al., 2017; Potting et al., 2017a); landfilling and incineration for energy recovery should be used in the lack of other CE strategies.

CE strategies may also promote innovative business models that go beyond product preservation. Strategies for redundancy, multifunctionality, or use intensification of products promote CE by preventing the consumption of new products or creating new consumption patterns. For example, consumers may refuse to buy new products if services or multifunctional products create redundancy in the expected function (Potting et al., 2017a). Renting, sharing, and pooling through product-service systems (PSS) can be important instruments to promote CE because goods will be used in a more intensive way (Tukker, 2015). PSS can be oriented towards the product, use, and result (Kjaer et al., 2018; Tukker, 2015). Product-oriented PSS are related to additional services after the product sale (e.g. maintenance); thus, they focus on products. However, use- and result-oriented PSS focus on the preservation of the function provided by a product (Kjaer et al., 2018). For example, EMF (2015a) mentions sharing (such as car-sharing) and virtualisation (such as telemeetings instead of physical meetings) as CE actions; the first example is use-oriented, and the second example is result-oriented. In the case of product-oriented PSS, the strategies preserve the product, but in use- and result-oriented PSS, the strategies preserve the function.

All in all, the specific strategies in ladders can vary depending on the CE definition. Our aim is, rather than define specific strategies, to acknowledge the strategies' capacity to promote CE considering common aspects. Hence, we propose a classification to group the existing CE strategies recommended by diverse authors. Inspired by the hierarchical ladder from Potting et al. (2017b), we identified six common groups. The first five groups acknowledge strategies for preservation, and the last group considers the reference scenario measurement. For the sake of simplicity, we call these six groups CE strategies. Although the classification is numbered from 1 to 6, we do not claim an order of preference.

Strategy 1. Preserve the **function** of products or services provided by circular business models such as sharing platforms, PPS (use and result-oriented), and schemes promoting product redundancy and multifunctionality.

Strategy 2. Preserve the **product** itself through lifetime increase with strategies such as durability, reuse, restore, refurbish, remanufacture.

Strategy 3. Preserve the product's **components**, through the reuse, recover and repurpose of parts.

Strategy 4. Preserve the **materials** through recycling and downcycling.

Strategy 5. Preserve the **embodied energy** through energy recovery at incineration facilities and landfills.

Strategy 6. Measure the linear economy as the **reference scenario** or the absence of a preservation strategy to show the status, progress or regress towards CE. For example, the indicator for waste generation per person in a year (EC, 2018a) might show whether the promotion of CE is generating less waste.

2.2.1.3 Measurement type according to CE definition and CE strategies

CE does not work under a closed system. Circularity has direct and indirect effects on the economy (Potting et al., 2017a). Its assessment can rely on direct and indirect indicators when data is unavailable, e.g. the proportion of PSS related to CE cannot be assessed yet, but indirect data from companies' report and surveys could provide a preliminary analysis (EEA, 2017). However, it is difficult to define what direct or indirect mean since the CE definition itself is debatable. To further address the problem, we propose that indicators may be direct or indirect concerning the definition in sensu stricto or latu. In this way, both measurement types can be held in the classification framework without excluding views of CE in sensu stricto or latu. Moreover, Direct CE indicators may assess specific or non-specific strategies considering the rationale from subsection 2.2.1.1. To summarising, CE indicators can be classified into three measurement types:

a. **Direct CE with Specific Strategies**: indicators can focus on one or more identifiable CE strategies, e.g. Recycling Rate (Graedel et al., 2011) is specific to materials and the strategy, recycling, is clear.
b. **Direct CE with Non-specific Strategies**: indicators always focus on more than one strategy, and it is not possible to recognise the explicit strategies, e.g. water withdrawal (Geng et al., 2012) – Water withdrawal indicates how much water has been taken from the environment, but it does not say much about the 'circularity' of water.
c. **Indirect CE**: indicators may evaluate aspects of CE strategies but with the use of ancillary approaches to assess CE, e.g. the indicator 'Eco-innovation index' form the Resource Efficiency Scoreboard (EC, 2016) rank European countries in relation to eco-innovation factors; the indicator may provide information on CE, but it is not direct to a CE definition.

2.2.2 Finding *how* indicators measure CE

2.2.2.1 Measurement scopes according to Life Cycle Thinking (LCT) and modelling levels

CE acts on several steps of the production and consumption chain so that indicators may use a Life Cycle Thinking (LCT) approach. LCT is the capacity to look at products or services over the cycles of design, production, consumption, use, and disposal, including interactions with sustainability (UNEP/LCI, 2005). LCT is considered the state-of-the-art for analysing potential impacts (EC, 2003), and several reviews on CE show the necessity of a systemic view of the life cycle of resources (Ghisellini et al., 2016; Iacovidou et al., 2017; Reike et al., 2017). Moreover, LCT is at the heart of the Circular Economy Action Plan in the European Union that is actually split into sections concerning production, consumption, waste management, and production of secondary raw materials (EC, 2015a). Sustainability is divided in environmental, economic, social, and technical (technological) areas of concern (Dewulf et al., 2015). For the sake of simplicity in the cause-and-effect modelling, we consider that the technological cycles of materials, products, and services cause the effects on environmental, economic, and social domains (Figure 2.1).

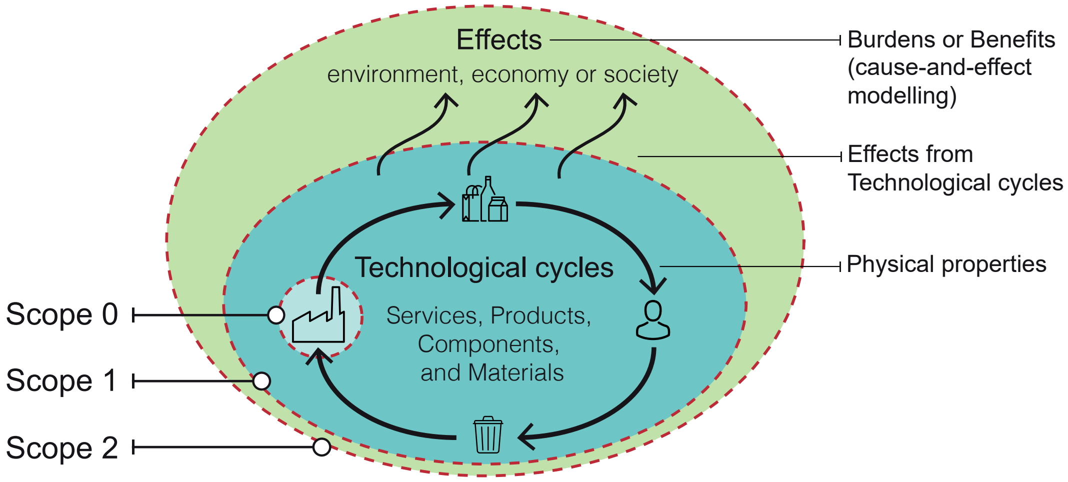


Figure 2.1: Proposed classification for the three measurement scopes from CE indicators

We argue that indicators measuring CE can be classified into three measurement scopes considering their LCT approach and modelling level (technological cycles and their cause-and-effect chain):

a. **Scope 0**: the indicators measure physical properties from the technological cycles without LCT approach, e.g. Recycling Rate (Graedel et al., 2011);
b. **Scope 1**: the indicators measure physical properties from the technological cycles with full or partial LCT approach, e.g. the indicator Reusability/Recyclability/Recoverability (RRR) in terms of mass includes the potential rate to reuse (products, components), recycle (materials), and recover (energy) (Ardente and Mathieux, 2014);
c. **Scope 2**: the indicators measure the effects (burdens/benefits) from technological cycles regarding environmental, economic, and/or social concerns in a cause-and-effect chain modelling, e.g. RRR benefit rate (RRR in terms of environmental effects) (Huysman et al., 2015a).

2.2.2.2 Implementation scale

CE has different implementation scales. The taxonomy from two reviews outlines three main scales: micro as a single product, company, or consumer; meso as eco-industrial parks and industrial symbiosis; and macro as a city, province, region, or nation (Ghisellini et al., 2016; Kirchherr et al., 2017). However, we have noticed that the micro, meso, and macro definition is neither consistently used nor clearly defined among different authors. The micro-scale usually focuses on a single product, service, or organisation. The meso-scale usually incorporates eco-industrial parks (Ghisellini et al., 2016). China is promoting industrial parks intensively (Geng et al., 2012), but they have specific characteristics: they integrate industrial, residential, business, research, and service areas (Geng and Doberstein, 2008). In this sense, the Chinese industrial parks are closely related to cities, indicating a macro-scale. Additionally, Geng et al. (2012) also refer to meso as the development of networks beneficial to regions and the natural environment. The macro-scale is usually limited to include the national level; where global can be an additional scale (EASAC, 2016). However, some authors suggest macro goes beyond countries, including the globe (CIRAIG, 2015; Kirchherr et al., 2017). Regions, with the scope between cities and countries, are considered macro-scale for the Chinese CE law, but Smol et al. (2017) propose regions are the connection between macro and micro-scales when measuring CE eco-innovation; indicating a meso-scale. For the sake of understanding, we argue that the micro, meso, macro terminology should be followed by the specific range of the analysis (e.g. consumer, product, service, business, technology, city, park, region, nation, continent, or globe).

2.2.2.3 Equation types of indicators

Generally, indicators are variables providing relevant information for decision-making (Gallopín, 1996). Variables are the representation of quantitative and qualitative attributes (Waas et al., 2014). Indicators can be either individual variables or a function of variables, e.g. ratio (number relative to a reference value), index (single number resulting from the aggregation of two or more variables), or the result of a complex simulation model (Gallopín, 1996). To indicate, indicators refer to a comparison value or reference (Waas et al., 2014). The reference value can be a baseline with undefined targets or baseline with specific (quantitative) or non-specific (qualitative) targets (Moldan et al., 2012). The reference can be either built-in or external to the indicator. To illustrate, the Sustainable Development Goal 12.3 aims to reduce by half the waste food per capita by 2030 (UN, 2015); an indicator for waste food could use the target as an external reference value (e.g. as parameters in a temporal evaluation) or as a built-in reference (e.g. as a ratio with the reference value as denominator).

The terminology used in this chapter is following Sala et al. (2013) who made a clear distinction amongst methodology, method, model, and indicator. The CE evaluation has methodologies (e.g. LCA), which are a set of methods (e.g. LCA impact categories). A method groups models, tools, and indicators relevant for showing information on circularity (technological cycles or its cause-and-effect modelling). A model is a mathematical description of calculating an indicator, which can be obtained through a tool. An indicator is a variable (parameter) or a function of variables to provide information about circularity (technological cycles) or the effects (cause-and-effect modelling). Additionally, an indicator may be the result of the composite information on quantitative and qualitative data.

2.2.3 Classification framework

The framework joins the rationale presented before for quantitative indicators (Figure 2.2). CE strategies are grouped for the preservation of functions, products, components, materials, and embodied energy. Additionally, a reference scenario may be used for the assessment. The framework considers three scopes for the LCT approach: two measuring physical properties of the technological cycles (scopes 0 and 1), and one measuring the effects of the technological cycles (scope 2). The framework incorporates bio and non-bio materials; however, their cycles are treated equally. Once bio-based materials are inside economic cycles, they can be recovered by strategies with similar preservation focus as the non-bio materials, e.g. food composting is a downcycling process to recover nutrients, hence with focus on the materials; particle boards can be incinerated to recover the energy, hence with focus on the embodied energy.

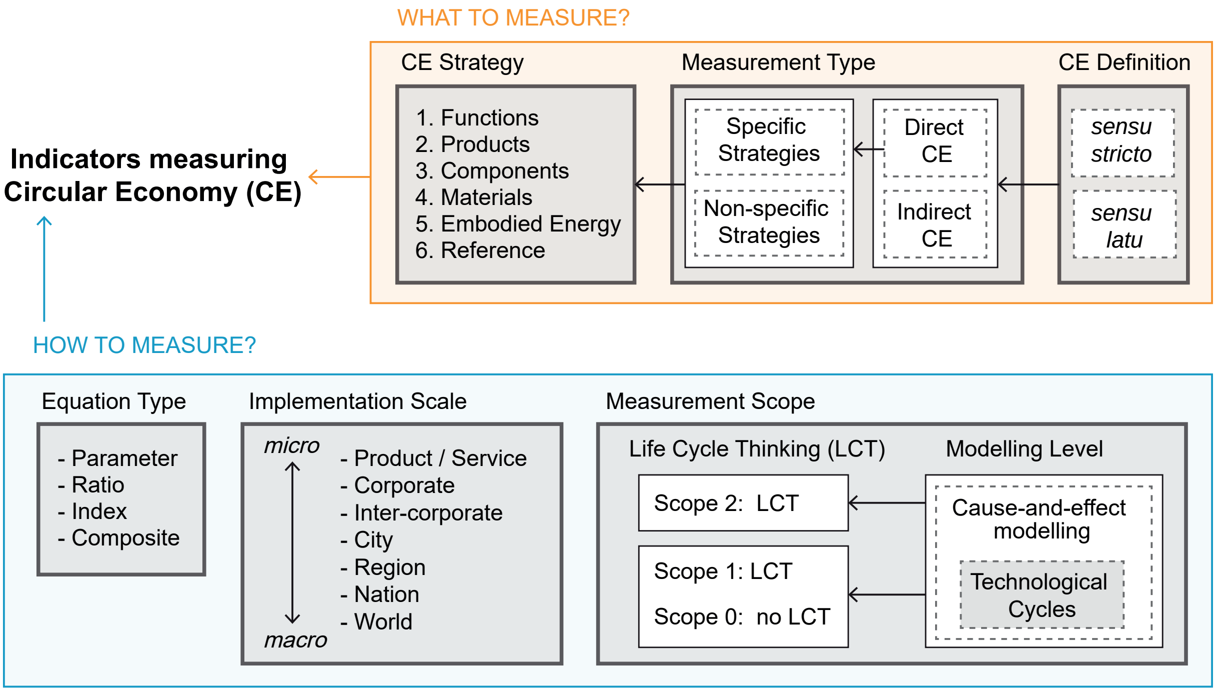


Figure 2.2: Classification framework for CE indicators

2.3 Illustrating the classification framework: micro-scale indicators

To illustrate the framework, we performed a literature review focusing on micro-scale CE indicators. The review included three steps. (1) We searched on Scopus and Web of Science databases the string indicator OR score OR metric OR measur* AND ‘circular economy’ in the title, abstract or keywords. We restricted the results by the English language and peer-reviewed documents. The Scopus database returned 251 results; the Web of Science returned 222 results. From the total, 154 were duplicates; hence 319 papers were analysed. (2) From a screening on title and abstract, we selected 11 documents proposing or discussing indicators for micro-scale: products, services, and companies. (3) To the mentioned documents, we added three documents (grey and scientific literature) discussed by other authors. In total, we analysed 20 indicators from 14 documents. For all the selected indicators and equations, see tables A1–A3 in appendix A.

In section 2.3.1, we present a short overview of the framework illustrated with a set of indicators. In section 2.3.2, we provide a critical analysis of the indicators and their classification.

2.3.1 Classification of the CE indicators: overview

Some patterns can be deducted from the framework illustration (Figure 2.3). The measurement type of all analysed indicators is Direct CE with Specific Strategies because the indicators can discriminate the measured strategies. This is evident as our literature review was focused on indicators designed to measure circular economy. Considering the CE strategy, most of the indicators measure the preservation of materials - strategy 4. Considering the measurement scope, indicators are distributed mainly in scope 1 and scope 2 - they examine a partial or full LCT approach. Some indicators are in scope 1 by measuring more than one strategy of technological cycles, e.g. the Material Circularity Indicator (MCI) (EMF, 2015b) gauges properties in a product, components, materials, and potential waste generation. However, the measurement of more than one strategy is not a requirement for the classification in scope 1 or 2. The Lifetime of Materials in the Anthroposphere (LMA) (Pauliuk, 2018) and the Number of Times of Use of a Material (NTUM) (Matsuno et al., 2007) measure the cascading of materials over different product groups. The two indicators focus on recycling and downcycling to account for the residence time of materials; therefore, only strategy 4 is measured but LCT approach is achieved.

Finally, none of the analysed indicators measures strategy 1, which focuses on the preservation of functions. This CE strategy is achieved through dematerialisation of products with PPS, sharing platforms, products refusing through multifunctionality. Interestingly, few indicators measure more than one CE strategy group, with a maximum of four strategies with MCI and SCI, and three strategies with Longevity.

2.3.2 Classification of the CE indicators: analysis

2.3.2.1 Indicators focusing on functions

Although none of the reviewed indicators assesses functions, some of them attempt to measure functions using a composition of quantitative and qualitative indicators. For example, Scheepens et al. (2016) used the Eco-costs Value Ratio (EVR) (a quantitative LCA-based indicator) and the Circular Transition Framework (a qualitative structure) to assess a PSS for water tourism. While the EVR provided an analysis of the products used in the PPS, the qualitative framework focused on the required steps for the PSS implementation. It is unclear what was the function-related strategy; the assessment analysed the substitution of a PPS with a diesel engine by a PPS with an electrical engine. In this sense, the EVR provided the eco-design improvement in the product used by the service, but the preservation of functions is not clearly depicted.

HOW DO INDICATORS MEASURE?
Measurement scopes

WHAT DO INDICATORS MEASURE? **CE Strategies**	**Scope 0** Technological cycles without aspects of Life Cycle Thinking	**Scope 1** Technological cycles with aspects of Life Cycle Thinking (Technological cycles with physical properties)	**Scope 2** Cause-and-effect modelling with/without aspects of Life Cycle Thinking (Cause-and-effect modelling from Technological cycles)
Preservation strategies: **1 Function** e.g. refuse, rethink, reduce			
2 Product e.g. reuse, refurbish, remanufacture	eDiM	TRP Longevity MCI	EVR PLCM SCI
3 Component e.g. reuse, repurpose	eDiM	TRP	PLCM
4 Material e.g. recycle, downcycle	CR RR EOL-RR RIR OSR	NTUM Longevity MCI CIRC LMA	PLCM SCI GRI Displacement CEI CPI VRE
5 Embodied Energy e.g. energy recovery, landfilling with energy recovery		MCI	CPI SCI
Linear: **6 Reference** e.g. waste generation, landfilling without energy recovery		MCI Longevity	SCI

Figure 2.3: Indicators measuring CE at the micro-scale. The found indicators encompass the measurement type Direct CE with Specific Strategies. Indicators: eDiM (ease of Disassembly metric) from (Vanegas et al., 2018); CR (old scrap Collection Rate), RR (Recycling process efficiency Rate); EOL-RR (End-of-Life Recycling Rate); RIR (Recycling Input Rate); OSR (Old Scrap Ratio) from (Graedel et al., 2011); Longevity from (Franklin-Johnson et al., 2016); MCI (Material Circularity Indicator) from (EMF, 2015b); PLCM (Product-Level Circularity Metric) from (Linder et al., 2017); CPI (Circular economy Performance Indicator) from (Huysman et al., 2017); CEI (Circular Economy Index) from (Di Maio and Rem, 2015); VRE (Value-based Resource Efficiency) from (Di Maio et al., 2017); EVR (Eco-cost value ratio) from (Scheepens et al., 2016); NTUM (Number of Times of Use of a Material) from (Matsuno et al., 2007); CIRC (Material Circularity Indicator CIRC), TRP (Total Restored Products), LMA (Lifetime of Materials on Anthroposphere) from (Pauliuk, 2018); Displacement from (Zink et al., 2016); SCI (Sustainable Circular Index) from (Azevedo et al., 2017); GRI (Global Resource Indicator) from (Adibi et al., 2017)

The preservation of functions is not as straightforward as the other strategies. The comparison between services and products demands attention on specific aspects of CE, such as the consequences caused by consumers' behaviour change (Zink and Geyer, 2017). The Circularity Gap report (Wit et al., 2018) provides insights into what could be the evaluation of functions on a global scale. The authors used material flow analysis (MFA) and a Sankey diagram to show the transformation from natural and secondary resources into 'societal needs' (i.e. housing, communication, mobility, healthcare, services, consumables, and nutrition) in one year. These societal needs could also be expressed as the functions that materials provide to society. Methodologies such as LCA and MFA may evaluate the preservation of functions, but indicators are still necessary. In this sense, indicators to assess functions could be derived from MFA providing enough disaggregated information about each of the societal needs (e.g. private, leased, and shared housing). However, disaggregated macro-economic data is not easily available or existent for the assessment of functions.

2.3.2.2 Indicators focusing on products and components

The indicators measuring strategies on products or components consider at least the sensu stricto aspect of slowing resources loops. Indicators measure this aspect in several ways, but two ways deserve attention: the assessment of quantity and quality. Indicators measuring quantity can account for tangible properties that are not user- or market-related. For example, the Total Restored Products (TRP) (Pauliuk, 2018) is MFA-based; it accounts for the products reused, refilled, refurbished, redistributed, and remanufactured at the end-of-life (EoL). On the other hand, indicators measuring quality account for properties influenced by the user or markets, such as time or economic value. For example, the Product-Level Circularity Metric (PLCM) (Linder et al., 2017) is a ratio from the economic value from recirculated flows over the economic value of all flows. Another one measuring quality is the Longevity indicator (Franklin-Johnson et al., 2016); it uses lifespan estimations from statistical records and experts approximation to account for the duration of materials in products. In contrast with the Longevity indicator, the results from PLCM can be equal for similar products with different lifespans (products with identical function and recirculated flows). However, the Longevity indicator, by only including the average lifespan, has to deal with the data variability caused by different consumer behaviour. Additionally, the Material Circularity Indicator (MCI) (EMF, 2015b) uses information on mass (virgin and recycled materials and waste) and product lifespan in one index system. Interestingly, from the indicators considering life cycle thinking, few indicators assessed time, despite some CE definitions explicitly referring to an economy 'where resources are kept for as long as possible.' For the moment, quantity information may be more reliable, but quality is a measure that deserves the attention of indicators in CE and may show the influence of consumer behaviour.

2.3.2.3 Indicators focusing on materials, embodied energy, and the reference scenario

From our analysis, it is clear that most of the indicators focus on strategies to preserve materials. This result was expected because CE has a high emphasis on recycling (Ghisellini et al., 2016). Recycling is the most frequent strategy across different CE concepts (Kirchherr et al., 2017). The indicators measuring materials consider at least the sensu stricto aspect to close resource loops. However, it is not possible to identify a pattern showing how materials' preservation is measured. The indicators can gauge information based on different characteristics of materials, e.g. supply and demand interactions (Displacement indicator from Zink et al., 2016), or the creation of economic value (Circular Economy Index from Di Maio and Rem, 2015).

Additionally, the illustration seems to point out that authors developing CE indicators at the micro-scale are less concerned with the preservation of embodied energy and the assessment of waste generation. Energy recovery is usually understood as the least preferred option (EMF, 2013a; Potting et al., 2017b). However, the Circular economy Performance Indicator (CPI) shows that for plastics, the options should depend on the material quality; if the waste quality is low, recycling may result in higher environmental impacts than incineration (Huysman et al., 2017). In any case, both recycling and energy recovery are neither green nor burden-free (Allwood, 2014) – as they generate environmental impacts and often consume non-renewable energy (recycling). Furthermore, CPI and MCI can account for energy recovery; and MCI can account for unrecoverable waste.

2.3.2.4 Composite Indicators

Finally, some indicators use a composition of qualitative and quantitative information to assess CE. For example, the Sustainable Circular Index (SCI) for manufacturing companies from Azevedo et al. (2017) considers sustainability reports (Triple Bottom Line, Global Reporting Initiative, and others) and the MCI. SCI includes the weighting of the information with factors determined by a panel of experts. Another composed indicator is the Global Resource Indicator (GRI) from Adibi et al. (2017). GRI combines scarcity, geopolitical availability, and recyclability. The scarcity and recyclability are quantitative measures of resource availability, recycling rate, and dissipative losses. Geopolitical availability is a qualitative parameter for the geopolitical stability of the countries where the resource is available and a parameter for the homogeneity of distribution.

2.4 Illustrating the classification framework: 'CE monitoring framework' in the European Union

To illustrate the framework with macro-scale indicators, we selected the indicators recently proposed by the European Commission (EC, 2018a). The EC proposal is one possible example of CE indicators at a macro-scale; other examples could include the proposals from the Netherlands (Potting et al., 2018b), France (Magnier et al., 2017), or China (Geng et al., 2012).

In section 2.4.1, we present an overview of the framework illustration. In section 2.4.2, we provide critical analysis of the indicators classification.

2.4.1 Classification of the 'CE monitoring framework': overview

The CE monitoring framework is the EC proposal for measuring CE progress in the EU and Member States (EC, 2018a). The 'CE monitoring framework' divides indicators into four topics: production and consumption, waste management, secondary raw materials, and competitiveness and innovation. Those are closely related to the priority areas from the CE Action Plan in Europe: plastics, food waste, critical raw materials, construction and demolition, and biomass and bio-based products (EC, 2015a). The EC proposal presents ten indicators, but six of them also have so-called 'sub-indicators.' In total, the proposal uses twenty-four measurement guides. The indicators are based on existing information from Eurostat, the Raw Materials scoreboard, and the Resource Efficiency scoreboard (EC, 2018a).

Eight indicators from the 'CE monitoring framework' are present in other European frameworks[3] and are not unique to CE (Figure 2.4). The other indicators are under

[3] Waste-related indicators from Eurostat are considered proxies for the Waste Framework Directive.

development: ‘Food Waste’ and ‘Green Public Procurement’ (GPP). In any case, the measurement of the first was foreseen in the revision of the EU Waste Directive (EC, 2015b). For the GPP, data are still unavailable. GPP significance for CE may depend on the inclusion of relevant requirements (e.g. reparability, durability, and recyclability) in public contracts and procurements (EC, 2018a). Both indicators are also in the scope of the Sustainable Development Goals for responsible consumption and production (EC, 2018a).

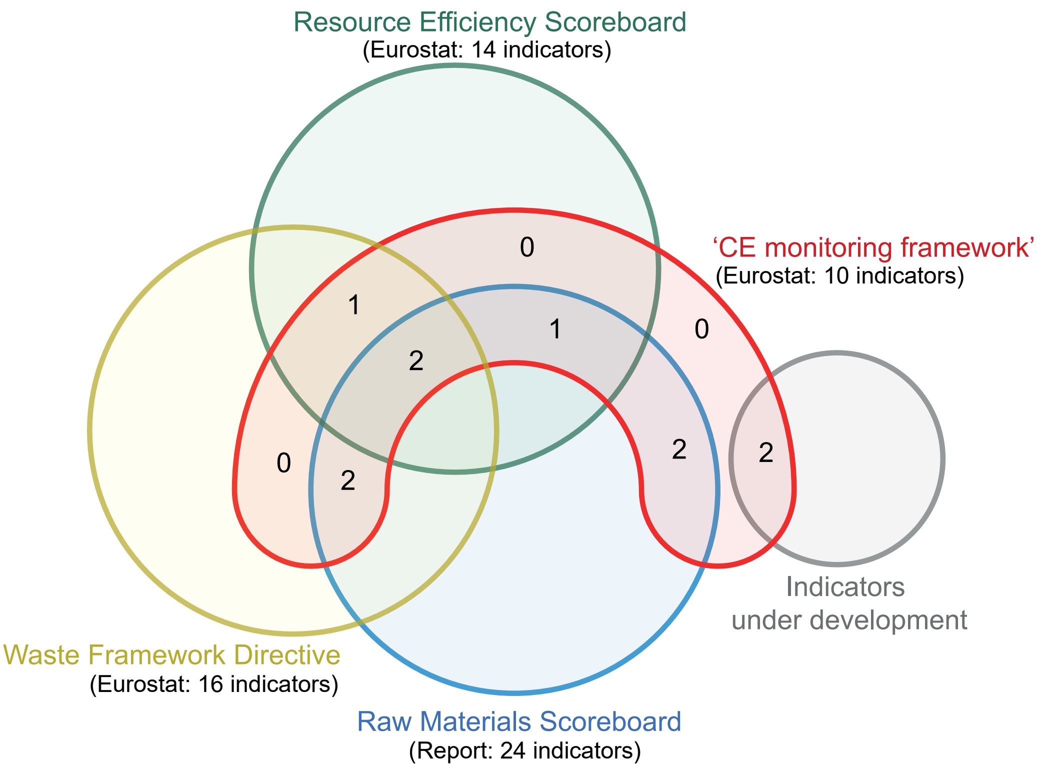


Figure 2.4: The interaction of the indicators from the ‘CE monitoring framework’ and other European directives shows that the indicators are not unique to the ‘CE monitoring framework’.

The ‘CE monitoring framework’ also uses material flow analysis (MFA) with Sankey diagrams to give an overview of materials flows in the EU. The diagrams show aggregated information of metallic and non-metallic materials, fossil energy, and biomass, providing an initial guide for a more detailed MFA. The diagrams may be used to extract indicators for CE, but at this point, the ‘CE monitoring framework’ does not describe those specific indicators. For this reason, the Sankey diagrams are not analysed in this chapter.

*Table 2.1: Classification of the indicators proposed by the European Union to measure the circular economy development. Strategies inside brackets mean the indicator contains aspects of that measurement. Direct CE indicators are *Direct CE with Specific Strategies;*

Indicator	Sub-indicator	Strategy	Scope	Measurement type
1. Self-sufficiency for raw materials	-	[4]	0	Indirect CE
2. Green public procurement	-	Indicator not available		
3. Waste generation	Generation of municipal waste per capita	6	0	Direct CE
	Generation of waste per GDP	6	0	Direct CE
	Generation of waste per DMC	6	0	Direct CE
4. Food waste	-	Indicator not available		
5. Recycling rates	Recycling rate of municipal waste	4, [6]	0	Direct CE
	Recycling rate of all waste	4, [6]	0	Direct CE
6. Recycling / recovery for specific waste streams	Recycling rate of overall packaging	4, [6]	0	Direct CE
	Recycling rate of packaging waste by type	4, [6]	0	Direct CE
	Recycling rate of wooden packaging	4, [6]	0	Direct CE
	Recycling rate of e-waste	3, 4, [6]	1	Direct CE
	Recycling of biowaste	4, [6]	0	Direct CE
	Recovery rate of C&D waste	4, [6]	0	Direct CE
7. Contribution of recycled materials to raw materials demand	End-of-life recycling input rates	4	1	Direct CE
	Circular material use rate	4	1	Direct CE
8. Trade in recyclable raw materials	Imports from non-EU countries	[4]	2	Indirect CE
	Exports to non-EU countries	[4]	2	Indirect CE
	Imports from EU countries	[4]	2	Indirect CE
	Exports to EU countries	[4]	2	Indirect CE
9. Private investments, jobs and gross value added	Gross investment in tangible goods	[2, 3, 4, 6]	2	Indirect CE
	Number of persons employed	[2, 3, 4, 6]	2	Indirect CE
	Value added at factor cost	[2, 3, 4, 6]	2	Indirect CE
10. Patents related to recycling and secondary raw materials	Patents of recycling and secondary materials	[4]	2	Indirect CE

According to the classification framework, the eight available indicators from the ‘CE monitoring framework’ mainly focus on materials - strategy 4 (Table 2.1). Four indicators are Direct CE with Specific Strategies; the other four are Indirect CE indicators. The Indirect CE indicators may concern to specific strategies, but they measure CE with ancillary aspects – not fitting in the sensu strictu or latu definition. Within the four Direct CE indicators, nine sub-indicators measure only materials, one is measuring both materials and components, and three are measuring the reference scenario. All those Direct CE indicators measure mass properties.

The ‘Recycling rate’ and ‘Recycling and recovery for specific waste streams’ do not consider a LCT approach (scope 0) except by one ‘sub-indicator’ in scope 1 (Recycling rate of e-waste), because of the inclusion of market and end-of-life (EoL) information. Furthermore, the ‘Contribution of recycled materials to raw materials demand’ clusters two sub-indicators in the scope 1: ‘End-of-life recycling input rates’ (EoL-RIR) and ‘Circular material use rate.’ Finally, the indicator ‘Waste generation’ monitors the amount of waste as a reference scenario to close material loops.

2.4.2 Classification of the ‘CE monitoring framework’: analysis

2.4.2.1 Direct CE with Specific Strategies indicators

Our classification framework shows that the Direct CE indicators from the ‘CE monitoring framework’ focus mainly on measuring material and waste production. Material resources and waste are considered the primary focus of the European policy on CE (McDowall et al., 2017). The specific indicators show that the EU has an understanding of a CE similar to the sensu stricto definition but mostly restricted to the circulation of materials. The indicators ‘Recycling rate,’ ‘Recycling for specific waste streams,’ and ‘Contribution of recycled materials to raw materials demand’ monitor the loops of materials by measuring the quantity of recycling and secondary materials. Those indicators gauge closed loop and open loop systems without differentiation, meaning that recycling and downcycling are accounted in the same way. Some authors argue that the distinction between loops is unnecessary; closed loops do not always displace more primary material than open loops, and closed loops may promote dispersive applications (Geyer et al., 2016). In any case, the measurement of the quality, or how much the loop of materials displace the primary production is relevant to CE (EEA, 2016;

Moriguchi, 2007). The 'Contribution of recycled materials to raw materials demand' addresses the contribution of recycling to raw materials demand. Its sub-indicators do not consider quality, but the approach can indicate the displacement of materials in general mass terms. Another point of discussion is about cascading use. Cascading is 'the efficient utilisation of resources by using residues and recycled materials for material use to extend total biomass availability within a given system' (Vis et al., 2016). Although there is not a specific indicator for cascading use, the concept can be partially verified with the indicators recycling rate of wooden package and biowaste.

Waste generation is an inevitable outcome of any economic activity due to entropy creation (Georgescu-Roegen, 1973), but changes in waste generation may indicate changes in consumption patterns (EC, 2018a). However, those changes may also result from other structural variations rather than CE promotion (EEA, 2016). The indicator 'waste generation' introduces the idea of waste decoupling. In this context, decoupling refers to a decrease in waste generation per gross domestic product (GDP) or per domestic material consumption (DMC) unit. Remarkably, the idea of resource and environmental decoupling is not present in the 'CE monitoring framework'. Resource decoupling is an intermediate objective from the European CE (Ghisellini et al., 2016), and it is included in the Resource Efficiency scoreboard as the lead indicator.

Besides evaluating materials and waste, the 'CE monitoring framework' has one specific sub-indicator accounting for the reuse of components in waste of electrical and electronic equipment (WEEE). Unlike the other recycling and reuse rate sub-indicators that only measure the EoL phase, this sub-indicator also measures the quantity of electrical and electronic equipment (EEE) entering the market (EC, 2018a). Despite WEEE information being critical to recovering resources, WEEE policy usually promotes weight-based targets considering neither resource types, quality, nor production steps (e.g. metallurgy) (UNEP/IRP, 2013). Furthermore, the complexity in EEE products determines the possibility of recycling, but the current EEE design tends to complexity (Graedel and Reck, 2014), which difficult their preservation. In a general manner, the 'CE monitoring framework' does not capture the assessment of products or information on products design, but the indicator on WEEE is a step forward.

The EU recognises the design of products as a fundamental CE aspect (EC, 2018a). However, the 'CE monitoring framework' puts the indicators for self-sufficiency, green procurements, waste generation, and food waste under the categorisation of 'production and consumption;' those do not assess products or services in a specific way. According to the European Economic and Social Committee (EESC, 2018), the 'CE monitoring framework' misses relevant indicators in eco-design and CE business models. Products and services are central to slowing resources loops (Bocken et al., 2016), but the 'CE monitoring framework' does not capture the role of the consumer in the flow of resources (EESC, 2018). The EU has a strong focus on eco-design policy (e.g. Eco-Design Directive), but this experience is not yet present in their CE indicators. It is worth mentioning that macro-scale data on products do not exist for the EU context; the information on durability, lifetime, disassembly, repair, and reuse cannot be monitored at this moment (EEA, 2016). However, not always product-related strategies are a priority in the EU policy, e.g. the EU policy for plastics acknowledges the reuse of products as low importance because it 'is only an option for a limited number of waste streams' (EC, 2018b). Moreover, correlating micro and macro-scale indicators is not yet presented in this current version of the EU monitoring framework; however, this is a shortcoming of the current literature. Arnsperger and Bourg (2016) reflected that this lack of relation in micro/macro-scales could lead companies to become more circular but not the economy. All in all, the inclusion of CE requirements in green public procurements, is promising for increasing data availability. Despite not covering the whole economy, public procurements represent over 14% of the European GDP (EC, 2017); they might be the most accessible path to assess products and services.

2.4.2.2 Indirect CE indicators

Indirect CE indicators from the 'CE monitoring framework' mainly focus on materials or aspects from materials, strategy 4. The indirect indicators measure ancillary aspects of CE, showing awareness of relevant areas but not necessarily encompassing circularity. For example, the indicator for the number of patents related to recycling does not consider the quantity or quality of secondary materials being produced nor its effects. It uses registered patents as a 'proxy for technological progress' (EC, 2018a). Innovation and technology support CE progress but are not objectives of the sensu stricto or latu definitions. Equally, the indicator for the trade of secondary raw materials shows the fluxes of materials considering a country's border but not necessarily CE requirements. Despite evaluating materials, the cause-and-effect chain of how trade affects recycling increase may exist, but the 'CE monitoring framework' does not document it. Differently from the recycling rates, which show the EU commitment to increase the recycling potential from waste, trade supports the dynamicity of the EU market (EC, 2018a).

Primary motivations for international trade rely on recycling costs and advantages (e.g. countries with less restrictive environmental laws for recycling) (van Beukering et al., 2014). Trade explains the international demand and supply of secondary materials, but its impact on recycling is ambiguous and challenging to summarise across different materials (van Beukering, 2001). Additionally, assuming a positive correlation between international trade and CE, it is also necessary to understand how illegal trade, not tracked by the indicator, influences the result.

Another indicator, the 'self-sufficiency for raw materials' is linked with the security of supply of raw materials in critical sectors, indicating a pivotal role for recycling actions, in particular when self-sufficiency is very low (EC, 2018a). However, considering the indicator also accounts for primary production, a country may increase self-sufficiency with mining; then, self-sufficiency is also a measure for the linear economy. Indeed, Europe is self-sufficient in construction minerals and wood because of domestic extraction (JRC, 2016). Recycling is directly correlated with self-sufficiency (i.e. increasing materials recycling means increasing self-sufficiency); however, it is not an indicator measuring the circularity of materials (EESC, 2018). Moreover, decreasing self-sufficiency can indicate the increased risk of supply disruptions or that other-policy measures achieve the EU's decoupling at the expense of countries that export the raw materials to the EU.

Additionally, the 'private investments, jobs, and gross value added' related to CE sectors shows the effects of CE considering products, components, materials, and waste. The indicator and its sub-indicators account for investments, employment, and share of GDP in 24 NACE codes (Statistical Classification of Economic Activities in the European Community) identified by the EU as proxies for recycling, repair, and reuse (EC, 2018a). The selected NACE codes cover diverse sectors, i.e. waste collection and trade, scrap trade, second-hand retail, components retail, dismantling, and maintenance and repair of industrial and household equipment (EC, 2018a). However, the NACE classification was not created to bear or distinguish CE activities (Ketels and Protsiv, 2017). For example, lifespan increase can be virtually applied to any product, but it is not possible to include all industries with actions to increase lifespan using the NACE codes. Finally, ICE indicators do not encompass CE main objectives, but their track can help governmental responses to promote CE if results are critically analysed.

2.5 Closing discussion and conclusion

In this chapter, we presented a classification framework for CE indicators and used it to evaluate what quantitative indicators used to assess CE measure specifically, and how they do so. This section aims to present the strengths and weaknesses of the classification framework (2.5.1) and the conclusion of the framework illustration with contributions to policy-making (2.5.2). For the specific discussion of the framework illustration, check subsections 2.3.2 and 2.4.2.

2.5.1 Strengths and weakness of the classification framework

The framework classifies indicators by common CE strategies (what) and measurement scopes (how) according to Life Cycle Thinking (LCT) approach. The presented approach is a novel way to categorise indicators without being restricted to one specific definition. Hence, the framework highlights the inherent characteristic of CE as an umbrella concept. We argue that CE has different strategies distinguished in five preservation groups (function, product, component, material, and embodied energy) and one group to measure the linear economy as a reference scenario. CE as a buzzword creates confusion that entails challenges for the selection and development of appropriate CE indicators. Our proposal has the added value to differentiate CE indicators by the measurement approach independent of the definition of CE, either in sensu stricto or latu. At this point, the framework cannot differentiate indicators measuring inputs and outputs, e.g. indicators for the total amount of recycled material (output) and the total investment in recycling activities (input) are part of the same strategy group. Both input and output are necessary to evaluate CE transition, but the framework still needs refining to include input indicators consistently.

Additionally, the classification framework includes three scopes considering the LCT approach. The scopes present an initial proposal to differentiate the possible mechanisms behind the cause-and-effect chain in CE. CE includes at least the circularity of materials, components, and products, but CE may also affect the economy, environment, and society. The relation amongst all these concerns is complex. The relation type between CE and sustainable development varies as conditional, beneficial, and having trade-offs that may also lead to adverse outcomes (Geissdoerfer et al., 2017). The cause-and-effect chain of how CE affects sustainable development is not fully documented, wherein other areas this is clearer, e.g. climate change has a well-documented impact pathway from the pollutant emission to the impact on areas of protection. For the sake of simplicity, we address the problem detailing two scopes for technological cycles (one without LCT approach and one with LCT approach) and one for the effects of technological cycles over the other sustainability concerns. The added value of the mentioned approach is the ability to easily differentiate how indicators measure CE progress. Future work may include an extended definition of how LCT approach is treated in scope 2 – effect of the technological cycles. Moreover, Figure 2.2 summarised the rationale behind CE indicators. However, at the present point, CE indicators are too heterogeneous, and we do not have evidence of a pattern to identify interrelationship amongst the presented aspects (e.g. Equation Type vs Implementation Scale vs Measurement Scope); this could be better explained in future studies.

Additionally, some authors argued that CE assessment includes the use of renewable energy, water, and land (EEA, 2016; Elia et al., 2017; Geng et al., 2012). For example, Elia et al. (2017) called as CE requirement the 'increase share of renewable and recyclable resources,' including renewable energy; Ellen MacArthur Foundation quoted 'replacing fossil fuels with renewable energy' as an example of the principles behind CE (EMF, 2015b). The (lack of) consideration of non-material flows is one of the critiques from CE; not all authors engage in the same interpretation (Blomsma and Brennan, 2017). Notwithstanding, our classification framework is designed to include the measurement of non-material flows. Energy and water, for example, influence all CE strategies, and their indicators fit the framework under the sensu latu definition as specific or non-specific strategies.

2.5.2 Conclusion and contribution for policy-making

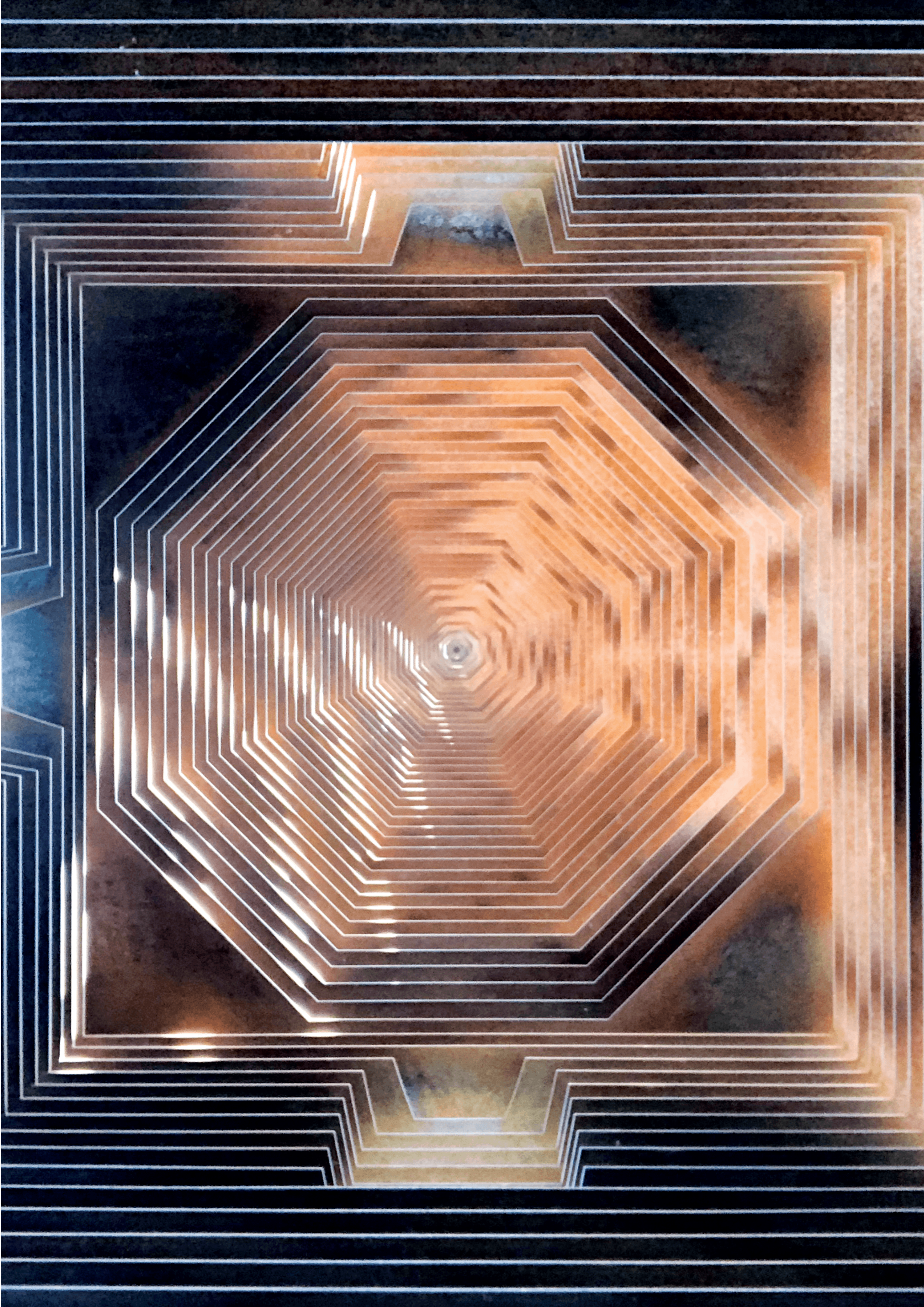
To illustrate the classification framework use, we applied it with micro-scale indicators (products, businesses, and companies) and macro-scale indicators (from the European 'CE monitoring framework'). From the analysed studies, it is possible to conclude that most indicators focus on preserving materials. Strategies focusing on materials, especially recycling, are well-developed actions, but they are limited: recycling, even being essential to the economy, is not the only aspect of a sustainable CE. According to the European Economic and Social Committee (EESC, 2018), all indicators from the 'CE monitoring framework' are 'heavily focused on waste' due to the reliability of waste data and lack of other options. Our classification framework can complement the EESC opinion. On the one hand, the indirect CE indicators contain aspects of waste and materials information. On the other hand, the direct CE indicators based on recycling rates use waste data to provide information on the possible preservation of materials[4]. The recycling rates from the 'CE monitoring framework' are a promise that a fraction of waste will be upgraded as a secondary resource. In this regard, what may be important in the materials side of the 'CE monitoring framework' is that only a fraction of the waste prepared for recycling will turn into a recycled material, whereas efficiency and quality of those materials and processes are not yet covered.

Additionally, none of the analysed indicators seems to focus on functions, such as multifunctionality or product sharing. Notwithstanding, we argue that existing methodologies, such as LCA and MFA, can provide a starting point for assessing functions. Those methodologies still need to deal with practical issues to evaluate CE. For example, diverse authors recommend LCA to evaluate CE (Elia et al., 2017; Lonca et al., 2018; Scheepens et al., 2016); but some CE strategies (such as recycling, reuse, repurposing, multifunctionality, or co-production) are in the scope of unsolved problems in the LCA methodology (Bobba et al., 2018; Reap et al., 2008). Furthermore, the evaluation of functions is challenging because it induces changes in consumer behaviour, e.g. sharing platforms may motivate a less-careful use of products when compared to ownership (Tukker, 2015). High-level CE strategies demand socio-institutional changes in the product chain, increasing the complexity of the evaluation (Potting et al., 2017b). Moreover, the definition of the specific strategies for preserving functions still needs clarification, e.g. which type of PSS promote CE. The classification framework shows the preservation of functions as an open question for CE indicators. Although the less clear boundary of functions preservation (compared to products or materials) may also increase uncertainty in CE evaluation.

Lastly, the framework's application seems to suggest that not one, but a set of indicators is necessary to assess CE. None of the analysed CE indicators measures all preservation strategies directly, i.e. CE includes many dimensions, and one indicator would hardly be able to summarise them all. In a similar sense, a set of indicators is promoted by CE monitoring systems on a macro-scale, e.g. Europe and China, and

[4] Recycling rate are the ratio of the waste prepared for recycling activities by the total amount of the waste stream.

micro-scale, e.g. Pauliuk (2018), and EMF (2015b) when the optional complementary indicators are considered for the second. Moreover, CE might promote sustainable development. Hopefully, future discussion and, in particular, the ISO technical committee for CE (ISO/TC 323) will shed light on a sustainable CE.



3 Circularity indicators based on the in-use occupation of materials

Redrafted from:

Title: Development of circularity indicators based on the in-use occupation of materials

Authors: Moraga, G., Huysveld, S., De Meester, S., Dewulf, J.

Journal: Journal of Cleaner Production

Date: January 2021

Title: Towards a circularity indicator to assess products' materials and lifetime: In-use occupation

Authors: Moraga, G., Huysveld, S., De Meester, S., Dewulf, J.

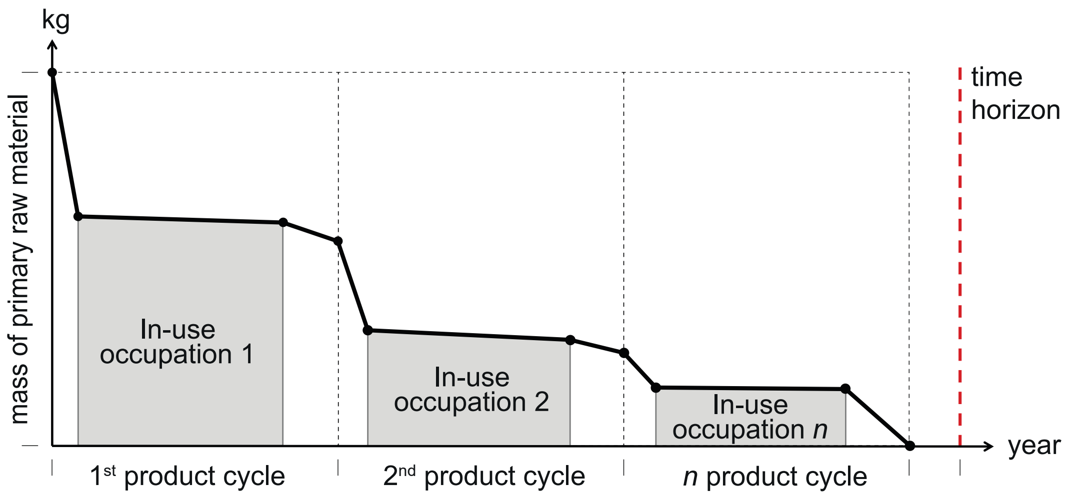
Journal: Procedia CIRP

Date: July 2020

Abstract

Circularity can help to prevent depletion of the earth's limited material resources, which are vital for human's modern society. Circular economy strategies of slowing and closing loops of resources have the ultimate goal of keeping materials useful (i.e. in-use) while avoiding losses (dissipation) and hibernation of materials; therefore, appropriate indicators that can measure these aspects are necessary. We propose a measurement of the circularity of materials by quantifying their in-use occupation, that is, the maintenance of materials in a useful state in products for as long as possible, avoiding dissipation or hibernation. Specifically, two indicators were developed: in-use occupation ratio (UOR) and final retention in society (FRS). These indicators were applied in two case studies (materials in laptops and wood products) with three scenarios each (linear, reuse, and recycling). The reuse scenarios generally presented a higher UOR (41–48% for laptop materials and 53% for wood) compared to recycling scenarios (29–45% for laptop materials and 52% for wood). Only two scenarios of wood products resulted in retaining materials for the next generation (FRS > 0%). We argue that the differentiation between supply, in-use, and hibernation phases is essential for a circular economy. The occupation of materials is shown in charts to facilitate understanding by non-experts and can provide a scientific basis for policies supporting technologies or products with increased in-use occupation and retention of materials.

Graphical abstract



3.1 Introduction

The current use of materials in the world economy threatens the earth's limited material resources. To address this issue, the circular economy (CE) concept aims to slow down natural resource use and waste production while boosting the economy. Understanding and measuring the effects of a CE is of paramount importance but at the same time a challenging effort (Alaerts et al., 2019; Moraga et al., 2019). Although at the micro-level, many indicators that assess the circularity of materials in single product cycles have been developed (e.g., Huysman et al., 2017; Linder et al., 2017), they miss the retention of the materials over different cycles. Moreover, although CE strategies can increase the conservation of materials embedded in products, the degree of conservation can vary from material to material (Ljunggren Söderman and André, 2019). Hence, the development of quantitative indicators measuring materials embedded in products is of particular interest for assessing the benefits of circularity over different product cycles.

A recent review of CE indicators by Saidani et al. (2019) pointed out that despite the growing number of quantitative and qualitative micro-level indicators, research on a more effective evaluation of CE strategies is still necessary. Elia et al. (2017) remarked that most quantitative indicators are related to the use and loss of materials but miss important aspects of the product's lifetime. Moraga et al. (2019) noted that most available quantitative indicators measure strategies related to end-of-life (EoL) and recycling and are less related to reuse or repair. Moreover, Pauliuk (2018) argued that natural resource depletion, in-use stock, and lifetime should be the foundation of CE indicators.

The issue of scarcity of materials is important, as the goal of a CE is 'to manage all natural resources efficiently and, above all, sustainably' (EEA, 2016). Managing resources is critical, especially for non-renewable materials (e.g. metals and minerals). The issue with such materials is that once extracted, they can be dissipated into either the natural environment or the human-made environment (technosphere) (Frischknecht, 2016). However, extraction (the mining process) does not mean that these materials will become unavailable; they are rather 'borrowed' in the technosphere and could be reused (Frischknecht, 2016; Zampori and Sala, 2017). In line with this, the United Nations Environment Programme's Life Cycle Initiative (UNEP/LCI) states that we should protect the value of resources in the technosphere as 'the damage [in mineral resources] is quantified as a reduction or loss of this potential [value] caused by human activity' (UNEP/LCI, 2019). Therefore, the inaccessibility of material resources is caused by anthropogenic compromising actions related to exploration, environmental dissipation, hibernation, and in-use occupation (van Oers et al., 2019). Of particular interest is the concept of in-use occupation, which considers materials in applications that are in-use, as the purpose of any extracted resource is to remain in a useful state (van Oers et al., 2019). Similarly to 'land occupation' expressed as ha × year, in-use occupation could be assessed by adding a time dimension, for instance, kg × year (van Oers et al., 2019).

The time dimension is indeed a key parameter for the CE. Bocken et al. (2016) distinguish between two major CE strategy groups: slowing and closing resource loops. The first one is mostly time-related and describes strategies intended to increase the lifetime of products, such as reuse, repair, or remanufacture, delaying the end-of-life (EoL) phase. This group also includes strategies that provide functionality without ownership of physical products (e.g. product-service systems), which require particular attention to user behaviour. The second group describes strategies related to the EoL of materials, such as recycling. Nonetheless, although time is a central aspect in at least one of these groups, it is often disregarded in many circularity indicators.

Notable examples of indicators that consider time with the use of materials in products include the Material Circularity Indicator (MCI) of the Ellen MacArthur Foundation (EMF, 2015b) and the Longevity indicator of Franklin-Johnson et al. (2016). These indicators are described in Appendix A (Table A3).

The MCI measures circularity by aggregating in a dimensionless index the multiplication of (i) the so-called linear flows (used primary raw materials and generated waste from upstream and downstream processes), (ii) the ratio of the product's lifetime to the industry's average, and (iii) the ratio of the product's number of EMF's functional units[5] to the industry's average. MCI results can be shown disaggregated per material or in the final index for the whole product, weighting the materials by their mass contribution. Because the MCI does not account for environmental impacts, the Ellen MacArthur

[5] MCI's functional unit should not be confused with the classical LCA's functional unit. In the case of the first, EMF (2015b) defined functional unit as 'a measure of the product's use. For example, it could be one kilometer driven for a car, or one wash cycle for a washing machine.' The given examples are similar to what the ISO 14040 defines as reference flow, that is a 'measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit.' To avoid confusion, EMF's functional unit herein are stated in genitive case, whereas other mentions to *functional unit* refers to LCA.

Foundation recommends complementing it with additional impact and risk indicators. Elia et al. (2017) found that the MCI is the only indicator that attempts to assess the loss of materials and product durability together. It aims to improve decision-making in the design phase of products in only one company's perspective, but it misses ways in which materials are saved in a broader time perspective than a single product cycle.

The Longevity indicator, initially proposed by Franklin-Johnson et al. (2016) and more recently extended by Figge et al. (2018), is an eco-efficiency indicator that measures the amount of time during which a resource is used. According to the authors, this is a value-oriented rather than a burden-oriented approach, such as the assessment of environmental impacts. The indicator calculates the sum of the time of a material's first use (as in a product), the time of its use when remanufactured, and the time of its use when recycled into a new product by the same company. In its adaptation by Figge et al. (2018), the Longevity indicator is coupled with an indicator measuring the number of times a resource is used, considering first use, refurbishment, and recycling into the same product. The Longevity indicator is focused on the analysis of the use of materials considering one company's perspective and does not include an analysis outside this company's boundary. Although it assesses the time value created by the in-use materials, it disregards the losses caused by the manufacturing process inside the company or along the supply chain.

Although indicators that assess strategies for the use of materials by one company can facilitate decisions within this particular boundary, they are less helpful for policy-making and society. Therefore, a perspective with a broader time horizon can consider pertinent information about previous or future use of materials in other cycles. Furthermore, Blomsma and Brennan (2017) suggest that we should move away from assessments of singular CE strategies, instead assessing them in sequential and parallel configurations. The latter could be done by considering a more comprehensive time horizon and boundary.

Moreover, the inclusion of different cycles of the use of materials could show details of their dissipation and hibernation along the supply chain. Dissipation is affected by the characteristics of the supply, use, and hibernation of materials. For example, it can occur in manufacturing or recycling (supply phase) in industrial and logistics steps or during use or hibernation due to chemical or physical processes (e.g. oxidation or abrasion). While hibernation does not keep the material in a useful state, supply is an unavoidable phase before its in-use occupation. Therefore, circularity indicators should encompass the supply, use, and hibernation phases, as well as dissipation in all these phases, over a longer time horizon. In this way, circularity indicators can assess how well materials are kept in a useful state in applications.

This study aimed to develop and apply a set of indicators that quantify the in-use occupation of materials in different product cycles. In this chapter, we present the framework for the indicators based on the material's in-use occupation (section 3.2), describe two case studies (section 3.3), report and discuss our results (sections 3.4 and 3.5), and present our conclusions (section 3.6).

3.2 Development of indicators based on materials' in-use occupation

3.2.1 Preamble: Definition of raw materials

Resources can be defined in several ways, such as biotic and abiotic; renewable and non-renewable; and funds, flows, and stocks (Swart et al., 2015). In this study, the framework proposed by Dewulf et al. (2015) was followed, in which natural resources at the primary production sector are transformed into primary raw materials or energy carriers, considering their future use, typically as market commodities. As commodities, materials exhibit standard characteristics that allow their (international) trade. As our focus is on the circularity of materials, we do not consider energy carriers. Primary raw materials can be divided into groups according to their origin: terrestrial and aquatic biomass, raw materials from water bodies and the atmosphere, metals, minerals, and raw materials from fossils (Dewulf et al., 2015). However, some of these materials can be used in products that are consumed, such as food/feed or pharmaceuticals. These applications are unsuitable for our analysis, as they cannot be considered for subsequent use. Moreover, primary raw materials can turn into secondary raw materials during the manufacturing or EoL phase. The first is known as post-industrial and the second as post-consumer; we focus mainly on the latter. In conclusion, the focus of our analysis is on raw materials that are embedded in products and that can potentially be recovered after the end-of-use. After recovery from a previous product cycle, materials may become secondary raw materials in a similar or different product.

3.2.2 Definition of parameters and time horizon

For a primary raw material retained in a cascade of products, typically three (occupation) phases can be discerned in each product cycle (Figure 3.1): supply, in-use, and hibernation. The supply phase includes the transformation of materials (primary or secondary) into products used by the final consumer (person or entity). In the life cycle assessment framework (ISO, 2006a, 2006b), this would closely relate to a system boundary that goes up to the market activities for the final consumer, such as retail. In the in-use phase, the product is used by the final consumer without further transformation; it starts after the retail process and ends before hibernation. During hibernation, products are not in use but waiting for EoL, for example, a PET bottle between discard (in the trash bin) and EoL collection for either final disposal or recycling (in a new supply phase). After hibernation, the materials can enter the following supply phase for a second product cycle.

The differentiation between these three phases with their respective occupations is essential. The longer the materials stay in the supply and hibernation phases, the lesser the materials can be useful in a CE. For example, it is no use having a non-functional battery hibernating in a shelf, waiting for a CE strategy, such as recycling.

Our analysis starts with the mass of primary raw material initially dedicated to incorporation into the first product and maintained in the economy embedded in other products. The supply phase begins with the raw material after its production (the defined types of raw materials are distinguished by Dewulf et al. (2015) and clarified in subsection 3.2.1). Each phase has an identifier for the input mass of materials (m_S, m_U, m_H); for the losses caused by the dissipation of materials (l_S, l_U, l_H); and for the associated time durations (Δt_S, Δt_U, Δt_H), measured considering their time occurrence (t_S, t_U, t_H).

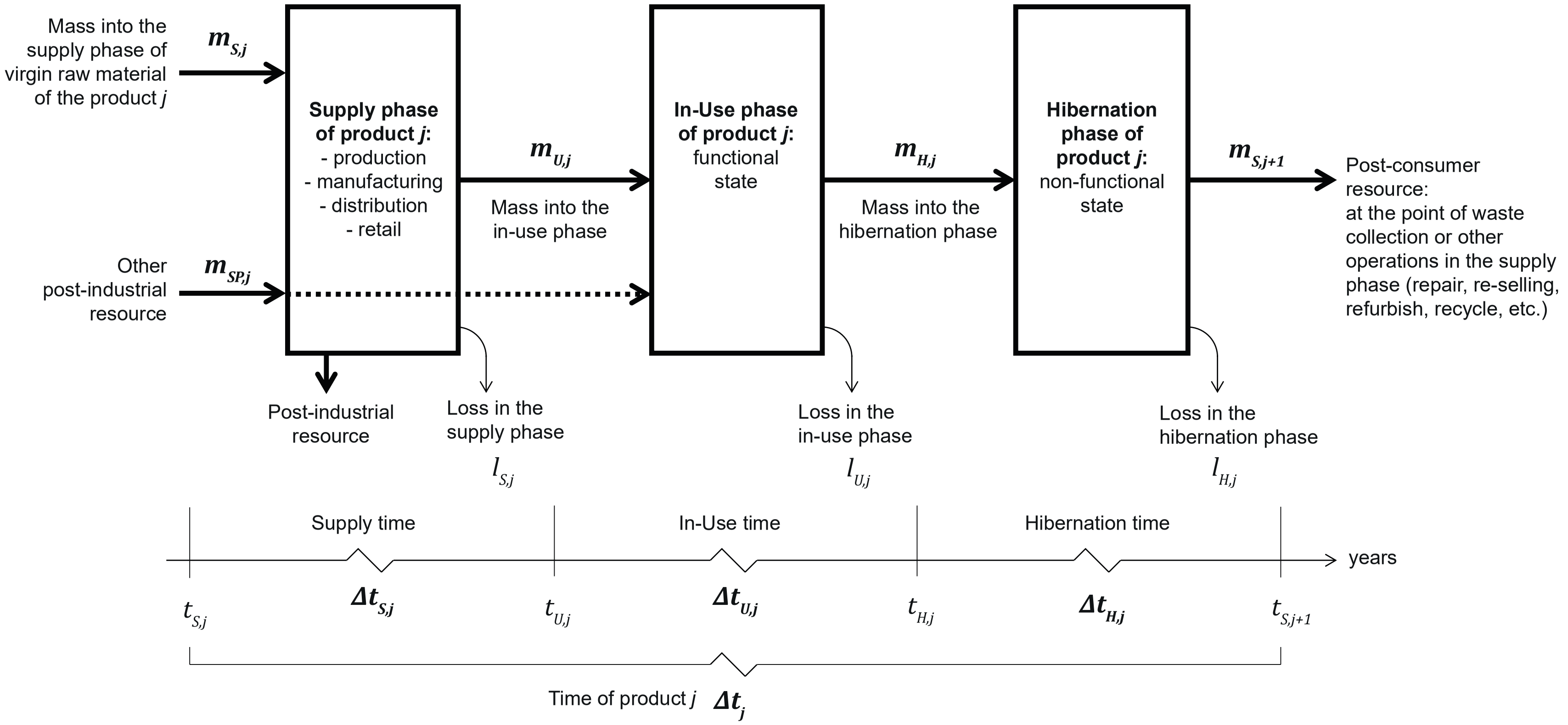


Figure 3.1: Input and output mass (mS, mU, mH, mSP), material losses (lS, lU, lH), time occurrence (tS, tU, tH), and duration (ΔtS, ΔtU, ΔtH, Δt) parameters for the first cycle of the material in a product cycle j

Many losses can occur in the supply phase due to industrial processing and manufacturing. As per definition, losses in the supply phase are those that resulted from dissipation to the environment and technosphere. Hence, residual materials that are not dissipated and are readily available for reuse are not accounted as losses (e.g. new metal scraps). However, residual materials that are not readily available for reuse, such as those sent to landfills and mine tailings, are accounted as losses. Furthermore, losses occurring in other phases need to be accounted for – these can be a result of a material degradation (e.g. oxidation and abrasion) or product design (e.g. dissipative uses).

Consecutive use of a raw material in different product cycles can increase its in-use occupation (e.g. by repair, refurbishment, and reuse of the same product) or by recycling to deliver a secondary raw material for the same (closed loop) or another product (open loop). Therefore, after the hibernation phase, a product can also go into the in-use phase without passing through the supply phase (e.g. a stockpiled mobile phone that is reused with no repair activity). In this way, the cascading use of materials in products allows consideration of the CE strategies in a sequential configuration.

Furthermore, the whole cascading is assessed through the definition of a time horizon (TH). The purpose of this TH is to provide a time perspective from which materials should be evaluated considering technological uncertainties and the interests of future generations. Technological uncertainties are related to forecasting future recovery of materials; for instance, a technology for recovering low concentrations of materials may not be feasible (or existent) in the next 25 years but may be in the next 500 years (van Oers et al., 2019). Circularity aims to keep resources in a useful state for as long as possible; however, the longer the TH is, the higher the uncertainties are in predicting future technologies. In this regard, the TH should be as long as necessary to safeguard the interests of future generations but also short enough to reduce the time interference of technological changes. Because this time duration is not fixed, we follow one of the temporal scopes proposed by the SUPRIM project (Sustainable Management of Primary Raw Materials). This project proposed three temporal scopes in search of cohesion for the assessment of abiotic resources: 5, 25, and >100 years (Schulze et al., 2020a). We use a TH of 25 years as appropriate for measuring the circularity of a material in-use in the current generation and that can be available for future generations, resulting in a dual set of indicators based on the material's in-use occupation. Indeed, at the end of this TH, it is possible to calculate the retention of material, that is, the mass of primary material initially dedicated to a first application that is transferred and made available for future generations.

3.2.3 Definition of indicators

Within the defined frame, it is possible to define indicators using a two-dimensional chart for the occupation of materials in a cascade of products (Figure 3.2). This area chart exemplifies the three types of occupation (kg × year), where the y- and x-axes represent mass (kg) and time (years), respectively. Each occupation is the result of the amount of material entering the phase minus half of the losses multiplied by time (area of a trapezium).

From this chart, we deduce two initial equations: in-use occupation (Eq. (3.1)) and theoretical maximum in-use occupation (Eq. (3.2)). These equations are summarised in two indicators: in-use occupation ratio (UOR – Eq. (3.3)) and final retention in society (FRS – Eq. (3.4)). Collectively, we call them indicators based on the in-use occupation of materials.

$$Occ_{U,j} = \begin{cases} (\mathrm{m_{U,j}} - \mathrm{l_{U,j}}/2) \cdot \Delta \mathrm{t_{U,j}} & \text{when } \{\mathrm{t_{U,j}}, \mathrm{t_{H,j}}\} \le \mathrm{TH} \\ (\mathrm{m_{U,j}} - \mathrm{l_{U,j}}/2) \cdot (\mathrm{TH} - \mathrm{t_{U,j}}) & \text{when } \mathrm{t_{U,j}} < \mathrm{TH} \text{ and } \mathrm{t_{H,j}} > \mathrm{TH} \\ \text{not assessed} & \text{when } \mathrm{t_{U,j}} > \mathrm{TH} \end{cases} \tag{3.1}$$

Where $Occ_{U,j}$ is the in-use occupation of a material in a product cycle j (kg × year), $m_{U,j}$ is the mass of a material going into the in-use phase of a product cycle j (kg), $l_{U,j}$ is the loss of material during the in-use phase (kg), and TH is the given time horizon (25 years). For cases where the in-use occupation occurs within the TH, then $\Delta \mathrm{t_{U,j}}$ is the in-use time of a product cycle j (years), the same as $(t_{H,j} - t_{U,j})$. For cases where the in-use occupation starts before the TH but ends after it, then $(TH - t_{U,j})$ accounts for the occupation only within the TH, where $t_{U,j}$ is the time occurrence at the start of the in-use phase of a product cycle j (years). Moreover, j is the natural number of product cycles, which ranges between 1 and n. Occupations beyond the TH are not part of the analysis.

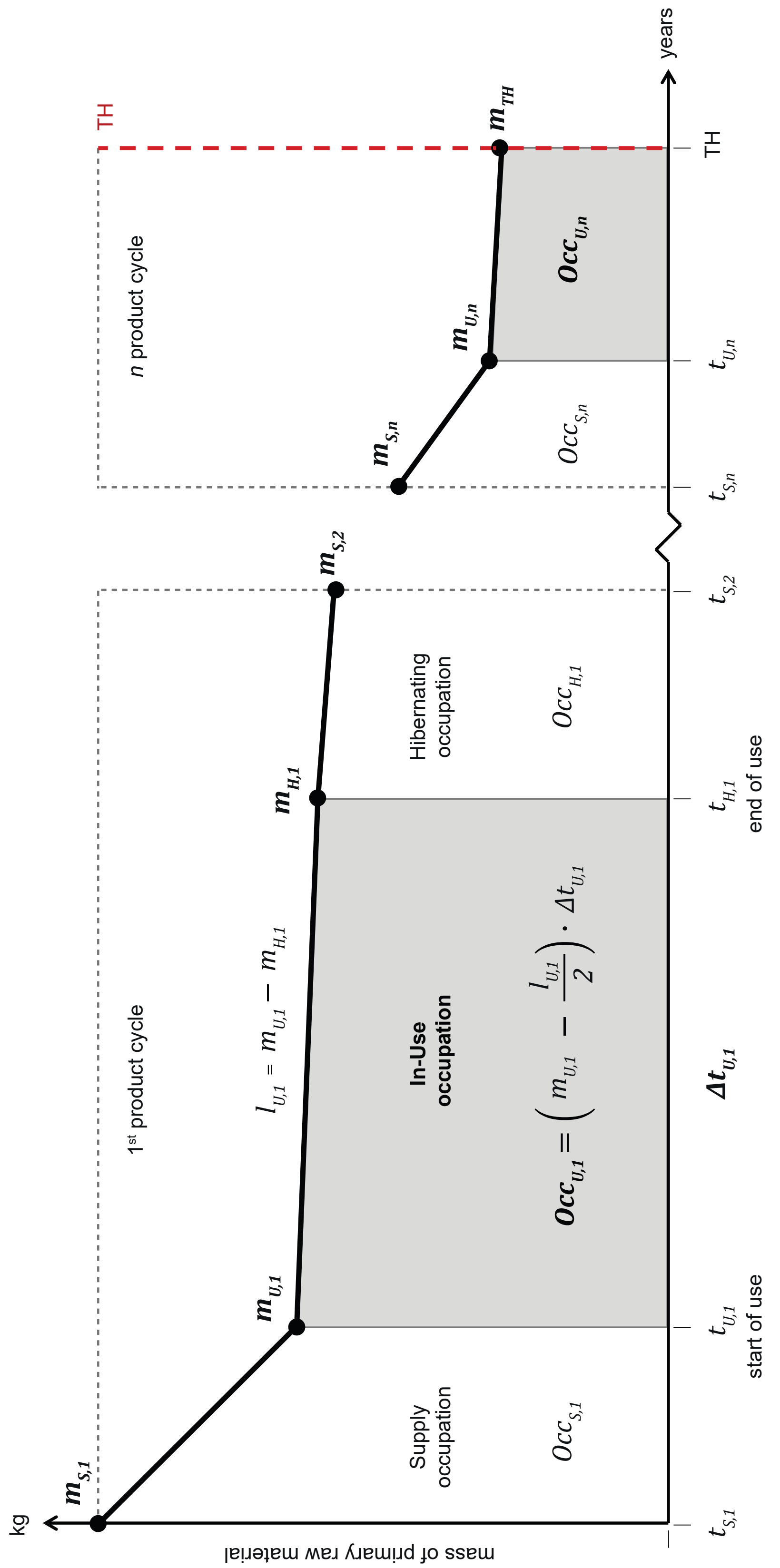


Figure 3.2: Area chart summarising the occupation phases and the equations for in-use and reference occupation for any product cycle j

The theoretical maximum in-use occupation equals the theoretical maximum use of materials within the TH of 25 years.

$$Occ_{Umax} = m_{S,1} \cdot TH \tag{3.2}$$

Where Occ_{Umax} is the theoretical maximum in-use occupation of a material in n product cycles during a given time horizon (kg × year), and $m_{S,1}$ is the mass of the material going into the in-use occupation of the first product cycle j (kg).

From the relation of those equations, two indicators are summarised. First, the UOR is the percentage ratio between the in-use occupation along the product cycles and the theoretical maximum in-use occupation, that is, the performance of the entire occupation for the use of the material within the TH. Additionally, the FRS shows the remaining percentage of the primary raw material at year 25.

$$UOR = {\sum_{j=1}^{n} Occ_{U,j}} \Big/ {Occ_{Umax}} \cdot 100\% \tag{3.3}$$

Where UOR is the in-use occupation ratio of a material in n product cycles during a given TH (%).

$$FRS = {m_{TH}} \big/ {m_{S,1}} \cdot 100\% \tag{3.4}$$

Where FRS is the final retention of a material in society (%), and m_{TH} is the mass of primary raw material initially used in the first product cycle j still available at year 25 (in n product cycles j).

3.3 Description of the case studies

To illustrate the application of the indicators described in section 3.2, we present two case studies, each with different scenarios in terms of the flow of materials in the product cycles. First, we present the inventory for four materials used in laptops: iron, aluminium, plastics, and precious metals (subsection 3.3.1). Second, we present the inventory for the use of wood in two different product applications (subsection 3.3.2). Details on references, assumptions, calculation methods, and allocation procedures are provided in the appendix B and mentioned along the text.

In all cases, we used as a reference 1 kg of primary raw material dedicated to the first product application and its conservation within the economy for the entire TH of 25 years. This reference allows comparisons between the results of each material in the different scenarios.

3.3.1 Iron, aluminium, plastics, and precious metals: EEE (laptop) case study

3.3.1.1 Description of the three scenarios

We developed three scenarios using published data when available, considering iron, aluminium, plastics, and precious metals used in laptops (Figure 3.3). Scenario 1 is a linear flow starting with the primary raw material production and ending after one product cycle. The justification for Scenario 1 is that a large share of the world's e-waste is undocumented and likely to be incinerated without any material recovery (Baldé et al., 2020). Scenarios 2 and 3 have two product cycles each. In both scenarios, the first product cycle is the same as in Scenario 1. For Scenario 2, the second product cycle comprises EoL collection and recycling of the materials and the production of another laptop with the recovered materials. For Scenario 3, the second product cycle involves the reuse of the existing laptop.

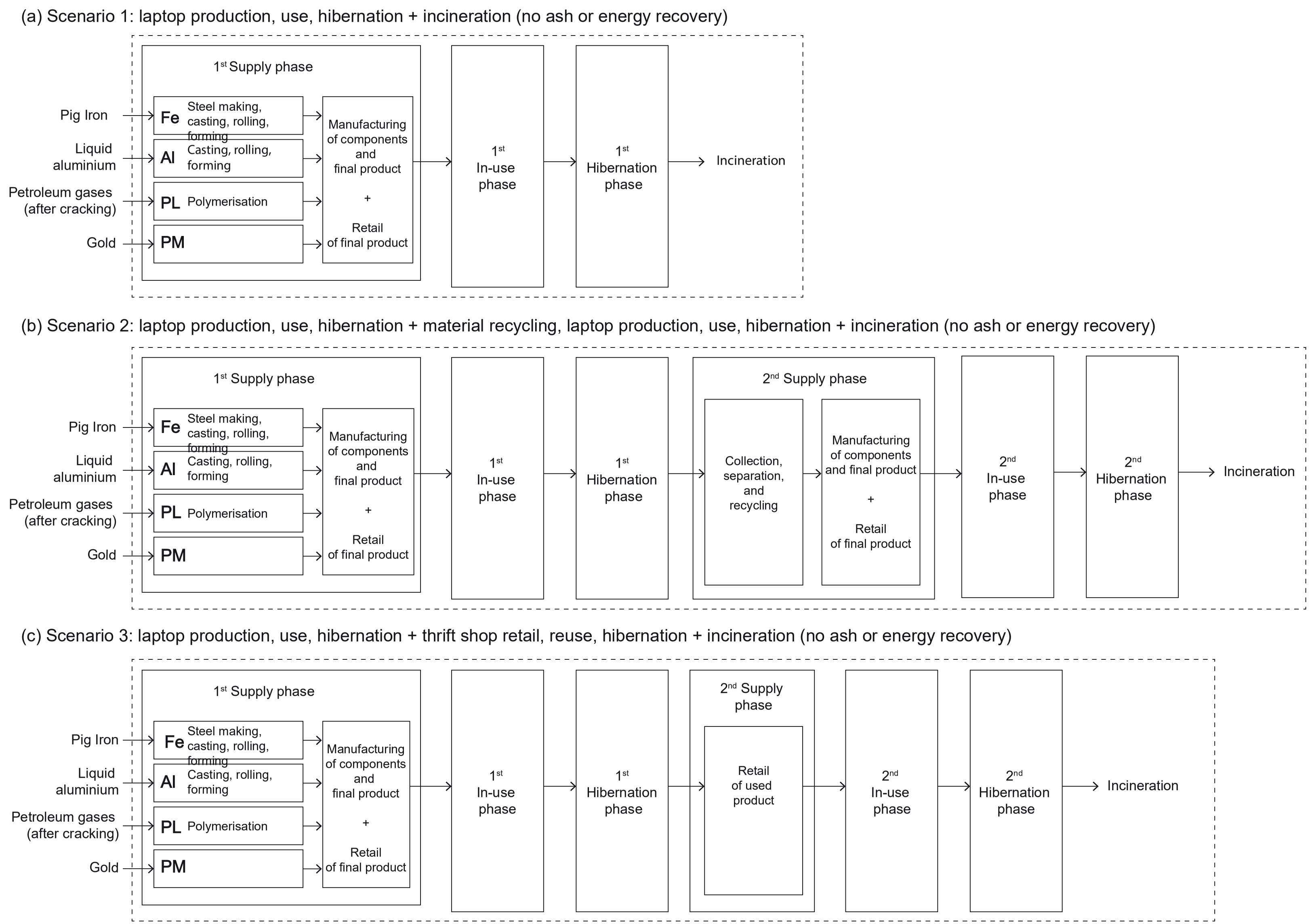


Figure 3.3: System boundaries of the three scenarios of four materials used in laptops: iron (Fe), aluminium (Al), plastics (PL), and precious metals (PM). The dashed lines represent the system boundaries of the analysis.

In all scenarios, the system boundary of the four materials starts after the primary raw material production. For iron, the analysis starts after pig iron production, while for aluminium, it starts after liquid aluminium production. For the sake of simplicity, the analysis for plastics starts with the bulk chemicals after the cracking process (e.g. ethylene and propylene). For precious metals, we consider the example of gold after the refining process. At the end of all scenarios, the four materials are incinerated as part of unsorted residual waste.

3.3.1.2 Description of the inventory

The inventory for the masses of the four materials entering the supply, in-use, and hibernation phases is displayed in Table 3.1. For the iron losses in the first product cycle, we consider the sum of all losses in the steelmaking, casting, rolling, forming, and manufacturing processes according to Cullen et al. (2012, Figure S2). In the case of aluminium in the first product cycle, although the yield ratio is 50% in the entire supply, there are no losses reported after the production of liquid aluminium, as all the unused material is processed as scrap (Cullen and Allwood, 2013). The material 'plastics' is composed of five polymers found by Van Eygen et al. (2016) in the production of one metric ton of laptop and desktop computers (polypropylene, polystyrene, polyethylene, acrylonitrile butadiene styrene, and polymethyl methacrylate). In the case of plastics production from primary raw materials, no losses are reported in the polymerisation process (Levi and Cullen, 2018), but small losses occur in the moulding process (Hischier, 2016). For gold, we assume no losses after the refining process, as manufacturing strives to recover and reuse gold because of the high associated value (Hewitt et al., 2015). Losses during recycling of the four materials (Scenario 2) are related to the recycling efficiency reported by Van Eygen et al. (2016, Table 1). In all scenarios, we assume no losses for the in-use and hibernation phases. Finally, we assume that no materials and energy are recovered after incineration. The information about losses is detailed in appendix (Tables B1–B3).

Table 3.1: Mass parameters for the three scenarios (S1, S2, S3) of iron (Fe), aluminium (Al), plastics, and precious metals (PM) used in laptop computers, where m_S, m_U, and m_H are the masses of materials entering the supply, in-use, and hibernation phases, respectively. The numbers in brackets are the sources regarding the losses in each phase: [1] calculation based on Cullen et al. (2012, Figure S2); [2] assumption of no losses related to use and hibernation; [3] value based on Van Eygen et al. (2016); [4] no losses reported after the production of liquid aluminium (Cullen and Allwood, 2013); [5] reference based on the moulding process (Hischier, 2016); and [6] assumption of no losses based on Hewitt et al. (2015).

	Fe (kg)			**Al (kg)**			**Plastics (kg)**			**PM (kg)**		
	S1	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**
$m_{S,1}$	1.00 [1]	1.00 [1]	1.00 [1]	1.00 [4]	1.00 [4]	1.00 [4]	1.00 [5]	1.00 [5]	1.00 [5]	1.00 [6]	1.00 [6]	1.00 [6]
$m_{U,1}$	0.85 [2]	0.85 [2]	0.85 [2]	1.00 [2]	1.00 [2]	1.00 [2]	0.99 [2]	0.99 [2]	0.99 [2]	1.00 [2]	1.00 [2]	1.00 [2]
$m_{H,1}$	0.85 [2]	0.85 [2]	0.85 [2]	1.00 [2]	1.00 [2]	1.00 [2]	0.99 [2]	0.99 [2]	0.99 [2]	1.00 [2]	1.00 [2]	1.00 [2]
$m_{S,2}$	-	0.85 [3]	0.85 [2]	-	1.00 [3]	1.00 [2]	-	0.99 [3]	0.99 [2]	-	1.00 [3]	1.00 [2]
$m_{U,2}$	-	0.73 [2]	0.85 [2]	-	0.75 [2]	1.00 [2]	-	0.13 [2]	0.99 [2]	-	0.63 [2]	1.00 [2]
$m_{H,2}$	-	0.73 [2]	0.85 [2]	-	0.75 [2]	1.00 [2]	-	0.13 [2]	0.99 [2]	-	0.63 [2]	1.00 [2]

The inventory for the time duration of the supply, in-use, and hibernation phases is presented in Table 3.2. Taking into account the manufacture of laptops and its components, the supply time ranges from 2–4 days to 24–26 days (lead time), according to Miyajima et al. (2019). For the duration of the overall supply phase, we assume 0.1 year to include both the industry and the retail activities. For the supply time of second-hand retail, we also assume 0.1 year. The durations of in-use and hibernation phases are retrieved from Thiébaud et al. (2018, Tables S14–S15) regarding the average use and reuse of laptops.

Table 3.2: Time parameters for the three scenarios (S1, S2, S3) of laptop computers. Δt_S, Δt_S, and Δt_H are the durations of supply, in-use, and hibernation phases, respectively. The sources in brackets refer to the following: [1] assumption based on Miyajima et al. (2019); [2] reference based on Thiébaud et al. (2018, Tables S1-S15); and [3] assumption for second-hand retail.

	S1	S2	S3
$\Delta t_{S,1}$	0.10 [1]	0.10 [1]	0.10 [1]
$\Delta t_{U,1}$	6.50 [2]	6.50 [2]	6.50 [2]
$\Delta t_{H,1}$	5.30 [2]	5.30 [2]	5.30 [2]
$\Delta t_{S,2}$	-	0.10 [1]	0.10 [3]
$\Delta t_{U,2}$	-	6.50 [2]	5.60 [2]
$\Delta t_{H,2}$	-	5.30 [2]	2.80 [2]

3.3.2 Wood: flooring and furniture case study

3.3.2.1 Description of the three scenarios

In this case study, we describe wood, a bio-based raw material, to be used as flooring for a building and particle board for furniture. We developed three scenarios using published data when available (Figure 3.4). Scenario 1 has a linear flow starting after wood production and ending after one product cycle of flooring in a building. Scenarios 2 and 3 have two product cycles each. In both scenarios, the first product cycle is the same as in Scenario 1. In Scenario 2, the second product cycle comprises EoL collection and recycling into particle board for furniture. In Scenario 3, the second product cycle considers the reuse of the existing wood flooring.

The system boundary of each scenario starts with the harvested roundwood and includes the sawmilling and flooring production processes. The recycling process (Scenario 2) includes chipping of the wood from flooring and particle board production. The reuse process (Scenario 3) involves repair of the flooring after the first use. At the end of all scenarios, the wood is incinerated.

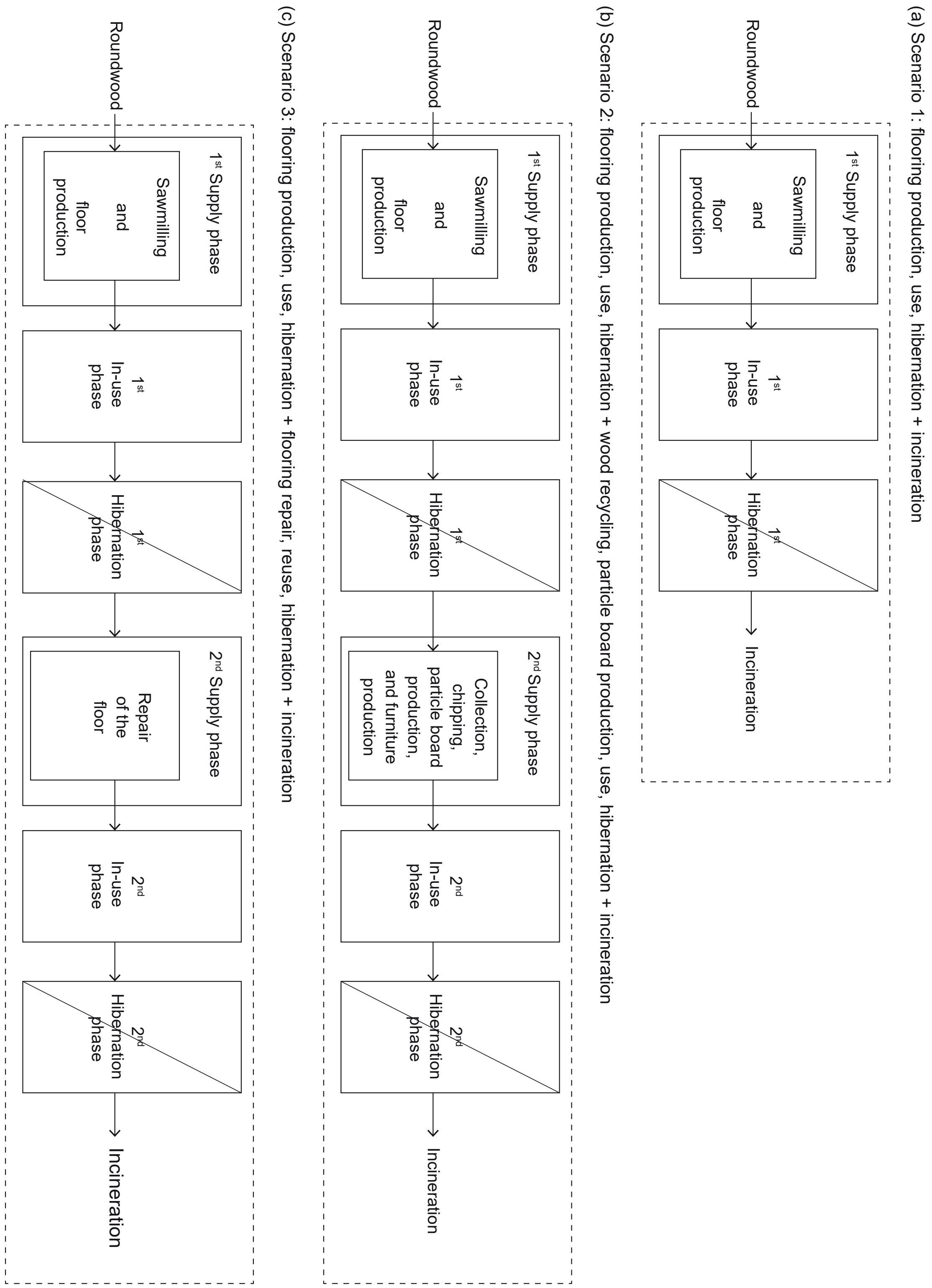


Figure 3.4: System boundaries of the three scenarios for the wood material. The dashed lines represent the system boundaries of the analysis.

3.3.2.2 Description of the inventory

The inventory for the mass of wood entering the supply, in-use, and hibernation phases is displayed in Table 3.3. For losses in the first product cycle, we calculate an allocation of mass losses between the co-products during sawmilling with values based on Garcia et al. (2020). Garcia et al. (2020) assumed that 50% of the roundwood becomes timber to be used in the flooring, while 10% is used in engineered wood, such as oriented strand board or particle board. The remaining 40% is incinerated and is therefore lost. We distribute these losses between the two co-products following the alignment of the life cycle assessment framework that prioritises allocation based on physical relationships (such as mass) when the subdivision of the process by co-product is not possible, or when it is not possible to expand the product system to include all co-products (ISO, 2006b). The allocation procedure is detailed in appendix (Tables B4–B6).

Table 3.3: Mass parameters for the three scenarios (S1, S2, S3) of wood products, where m_S, m_U, and m_H are the masses of material entering the supply, in-use, and hibernation phases, respectively. The numbers in brackets are the sources regarding the losses in each phase: [1] calculation based on Garcia et al. (2020); [2] assumption of no losses in the in-use and hibernation phases; [3] reference based on particle board production (Garcia et al., 2020); and [4] assumption based on the sanding process and replacement of damaged pieces.

	S1	S2	S3
$m_{S,1}$	1.00 [1]	1.00 [1]	1.00 [1]
$m_{U,1}$	0.58 [2]	0.58 [2]	0.58 [2]
$m_{H,1}$	0.58 [2]	0.58 [2]	0.58 [2]
$m_{S,2}$	-	0.58 [3]	0.58 [4]
$m_{U,2}$	-	0.50 [2]	0.55 [2]
$m_{H,2}$	-	0.50 [2]	0.55 [2]

For the particle board production, we use the losses from Garcia et al. (2020). For the reused flooring, we assume 5% losses in the sanding process and replacement of damaged planks. Like in the first case study, we assume no losses in the in-use and hibernation phases in any scenario and no ash or energy recovery upon incineration. The information about losses is detailed in appendix (Tables B7–B9).

The inventory for the duration of the supply, in-use, and hibernation phases is presented in Table 3.4. The supply time of the wood flooring in the first product cycle is assumed to be one year based on US data on the average construction duration of new residential buildings (US Census Bureau, 2018), according to which the supply duration ranges from four to six months for 51% of one-unit buildings and is longer than 13 months for 51% of multi-unit buildings. The supply phase of the other two products is assumed to be 0.5 years, considering the time for flooring repair or removal and particle board production. For the in-use phase, the data are based on the Lifespan database for Vehicles, Equipment, and Structures (LiVES). For the flooring in the first product cycle (all scenarios) and its reuse (Scenario 3), we considered half of the mean of the 'service lifespan' of a house (LiVES, 2008a), as the floor would have to be repaired or replaced once during the house's use. For the particle board furniture, in the second product cycle of Scenario 2, we use the mean 'service lifespan' for wooden furniture (LiVES, 2008b). In all scenarios, we assume no hibernation period.

Table 3.4: Time parameters for the three scenarios (S1, S2, S3) of wood products. Δt_S, Δt_U, and Δt_H are the durations of supply, in-use, and hibernation phases, respectively. The sources in brackets refer to the following: [1] assumption based on the average length of time from start to completion of new privately-owned residential buildings (US Census Bureau, 2018); [2] calculation based on LiVES (2008a); [3] assumption of no hibernation; [4] assumption for the repair/refurbishment time; and [5] reference based on LiVES (2008b)

	S1	S2	S3
$\Delta t_{S,1}$	1.00 [1]	1.00 [1]	1.00 [1]
$\Delta t_{U,1}$	17.10 [2]	17.10 [2]	17.10 [2]
$\Delta t_{H,1}$	0.00 [3]	0.00 [3]	0.00 [3]
$\Delta t_{S,2}$	-	0.50 [4]	0.50 [4]
$\Delta t_{U,2}$	-	12.40 [5]	17.10 [2]
$\Delta t_{H,2}$	-	0.00 [3]	0.00 [3]

3.4 Results

Figure 3.5(a) shows the area charts of the four materials' occupation within a 25-year TH in the three scenarios of the EEE (laptops) case study. The bold black lines represent the masses of primary raw materials still available to society at any point in time within the TH. The grey-coloured areas represent the in-use occupation of each product cycle. The white areas represent the supply and hibernation phases before and after each in-use occupation, respectively.

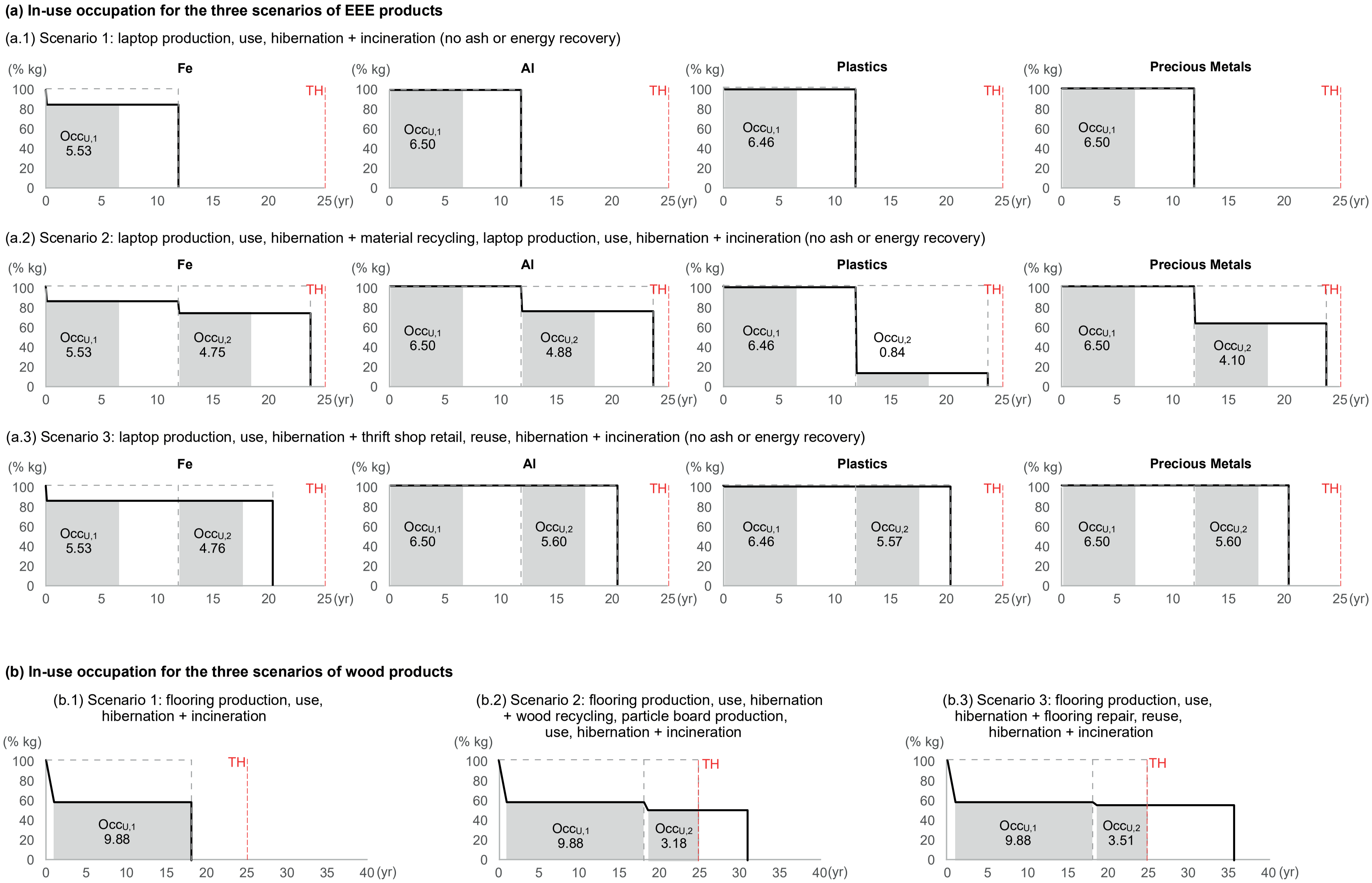


Figure 3.5: Results of in-use occupation (a) of the four materials in the three scenarios of the EEE product and (b) in the three scenarios of the wood products. The grey-coloured areas represent the in-use occupation of each product cycle. The y-axes represent the masses of primary raw materials initially used and still available in the economy at any point in time, normalised to 1 kg. The x-axes represent time in years. TH is the time horizon of 25 years.

Similarly, Figure 3.5(b) depicts the area charts with the occupation for the three scenarios in the wood products case study. In Scenarios 2 and 3, the total duration of the product cycles is longer than 25 years. For the calculation of the indicators, however, we only consider occupations within this TH. Nevertheless, the mass of material in the graph is shown beyond the TH for illustrative purposes of the full products' cycle.

As the reference mass of primary raw material going into the supply phase of the first product cycle ($m_{s,1}$) is 1 kg, and considering a 25-year TH, the theoretical maximum in-use occupation (Occ_{Umax}) is similar for all materials, that is, 25 kg × year. The UOR and FRS, which are complementary to the area charts of the four materials in the three scenarios of the laptop case study and wood in three scenarios of the flooring and furniture case study, are summarised in Table 3.5.

Table 3.5: Results of the UOR and FRS indicators for the three scenarios (S1, S2, S3) of the four materials in the laptop and wood in the flooring and furniture products

	Laptop case study												**Flooring and furniture case study**		
	Iron (%)			**Aluminium (%)**			**Plastics (%)**			**Precious metals (%)**			**Wood (%)**		
	S1	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**	**S1**	**S2**	**S3**
UOR	22.1	41.1	41.1	26.0	45.5	48.4	25.9	29.2	48.1	26.0	42.4	48.4	39.5	52.2	53.5
FRS	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	50.0	55.0

The UOR indicator ranges between 22.1% and 53.5%. These values are affected by the loss of materials and the duration of the in-use phase but also by the hibernation of products (laptop case), as shown in Figure 3.5. The lowest UOR values pertain to Scenario 1 (for both the laptop and wood product cases), as this scenario involves only one product cycle ending with incineration. The highest UOR values are observed in Scenario 3, except for iron, because of smaller losses in the reuse of the second-hand product. The UOR for iron is similar in Scenarios 2 and 3 (41.1%) because of (i) the higher recycling rate of iron (85%) among the analysed materials, and (ii) the shorter use (by 0.9 years) of the second-hand laptop compared to the one manufactured from recycled materials. The UOR for wood in Scenario 2 (recycling) is only 1.3% lower than that in Scenario 3. Of all materials, plastics present the lowest UOR in Scenario 2 (29.2%) because of losses related to the separation and recycling processes.

The FRS indicator is zero for all materials except for wood in Scenarios 2 and 3 (50% and 55%, respectively). This result suggests that none of the materials in the laptop case study is retained for longer than the defined TH of 25 years, which is related to the assumption of incineration after one or two product cycles. The FRS in wood can be explained by the longer duration of the in-use phase in building components and furniture. It should be noted, however, that the material's characteristics differ between the two scenarios. In the repair case (Scenario 3), the wood retains characteristics from the first product cycle, whereas in the recycle case (Scenario 2), it acquires new physical characteristics.

Additionally, we performed a two-way sensitivity analysis to understand how the results of the UOR and FRS would change with ±10% one-at-a-time variation of the main parameters (in appendix Figure B1 and B2). The parameters were the mass of material going into the in-use phase (m_U) and the duration of the in-use phase (Δt_U) per product cycle and scenario. Generally, the analysis showed that the UOR is more sensitive to input variation in Scenario 1 for all materials, as this is a scenario with greater possibilities for improvement. In the case of Scenarios 2 and 3, results for precious metals, aluminium and plastics could improve 5–10% with the variation in the first product cycle. For iron the chances are up to 15% because of the higher dissipation in the transformation of pig iron to steel in the first product cycle. In the case of wood, again the first product cycle can provide variations from 10% (Scenarios 2 and 3) to 20% (Scenario 1). However, the variations in the second product cycle for wood are around 2.5%. The smaller sensitivity of the second product cycle of wood is caused by the smaller contribution of this cycle's in-use occupation due to the analysed TH.

3.5 Discussion

3.5.1 UOR and FRS as new CE indicators

The ultimate goal of a CE is to keep materials in a useful state while avoiding losses and hibernation; therefore, appropriate indicators that can measure these aspects are necessary. This study proposes an innovative and gap-filling way to evaluate the conservation of materials in society. Given that higher circularity decreases the dissipation of materials and maintains them in a useful state for as long as possible, we propose that circularity can be measured in terms of the materials' in-use occupation using the new UOR and FRS indicators. Our case studies demonstrate that the higher the UOR, the longer the materials remain useful for society. Moreover, these indicators can be used to identify hotspots for material conservation in product design, manufacturing, use, hibernation, and EoL, especially between different product cycles.

The differentiation between supply, in-use, and hibernation occupations is essential for a CE, as increasing a product's lifetime (in the sense of durability or 'time of product', presented in Figure 3.1) would not necessarily increase the in-use phase of materials. Therefore, we argue that the lifetime of a product should be considered in terms of the duration of the in-use phase. As Corona et al. (2019) suggest, the lifetime of a product should be linked to its utility, as there is no benefit in hibernating products.

We underline that although a CE is beneficial for keeping materials in a useful state, there will always be dissipation in the course of either one or more product cycles. As illustrated by our case studies, recycling will likely lead to a certain extent of dissipation, and although reuse (second-hand products) can lead to smaller losses, it can also be associated with a shorter duration of use. Reuse is a way to extend the in-use occupation of materials by minimising their dissipation but requires a pre-existing product cycle. Furthermore, as it may not always be appealing from a consumer's point of view (EEA, 2019), actions coupling successive cycles of recycling and reusing could increase the conservation of the materials during and after the TH while minimising the effect of consumer's behaviour.

3.5.2 UOR and FRS vs existing indicators

Our in-use occupation-based indicators (UOR and FRS) expand the perspective of existing circularity indicators that take time into account, namely, the MCI (EMF, 2015b) and the Longevity indicator (Figge et al., 2018; Franklin-Johnson et al., 2016). Our proposal is not attached to a single product cycle, as is the case with the MCI, and it broadens the scope of using a time-based indicator in a one-company boundary, as is the case with the Longevity indicator. The MCI and the Longevity indicator are mainly focused on managerial decisions regarding production and business scale. Indicators based on in-use occupation, on the other hand, can be suitable for policy-making related to the development of less intensive material configurations of supply chains and technologies. Moreover, results based on in-use occupation can be presented in comprehensible graphical plots that can increase non-experts' understanding of material conservation.

Our proposal for the in-use occupation charts (Figure 3.5) shows the flow of materials over a continuing constraint of time, which differs from the usual circular flow diagrams of CE. Such diagrams (e.g. the Ellen MacArthur Foundation's butterfly diagram or mass-based Sankey diagrams) often show 'returning flows' to illustrate the closed loops of materials. However, we argue that time should be explicit in circularity diagrams, as materials are used in a sequential temporal dimension, as shown in the in-use occupation charts.

Improving the understanding of and ability to predict the hibernation phase is important for a CE, as its duration considerably affects the in-use occupation of resources; for example, the UOR of Scenario 2 in both case studies is lower with hibernation. Moreover, although perfect circularity would only be achievable with no losses, the latter are inherent in the industrial process. Nevertheless, if the industry and users increase the in-use occupation of products, society as a whole will be able to use materials more efficiently.

3.5.3 In-use occupation and paths for improvement

3.5.3.1 Quality of materials

Our proposed indicators do not take into consideration the quality of the material entering the different product cycles. This limitation is evident in the second product cycle of wood Scenarios 2 and 3. In Scenario 3, the quality of the material with regard to the product's function (flooring) is similar to that of the first cycle, but in Scenario 2, the material is cascaded for a product with a different economic value and function (furniture). Unlike mineral materials, wood cannot be reversed to its raw material state at EoL, which makes the differences in quality even more evident. However, quality is a controversial subject. Although a material's quality based on the functional requirements 'is the foremost critical factor' in the waste management system (Hahladakis and Iacovidou, 2018), there is no consensus in the scientific community on how to define quality (Huysman et al., 2017). In a CE, quality can be defined as a physical (Huysman et al., 2017), economic (EEA, 2019), or energy demand property (Steinmann et al., 2019). The problem with defining a quality factor is that functional requirements depend on the application type, and different quality factors will produce different results.

3.5.3.2 Material circularity and potential trade-offs

One of the most relevant (environmental) sustainability challenges is the potential trade-off between material circularity and the efforts related to resource use and emissions involved in each product cycle. Coupling indicators of in-use occupation with indicators of these trade-offs during the supply, in-use, and hibernation phases could increase the available information on materials' circularity with regard to energy and other resource policies.

Energy trade-offs should also be considered with respect to energy-using products. Our results show higher in-use occupation with reusing than with recycling, particularly for plastics and precious metals. However, these results concern material aspects and do not necessarily reflect environmental pressures from holistic perspectives. Richter et al. (2019) argued that shorter lifetimes are preferable from an environmental point of view if new products are more energy-efficient. This can be relevant for products with a considerable share of environmental impact in their in-use phase (e.g. diesel vehicles). For the time being, our indicators based on in-use occupation focus on the analysis of material retention; however, the assessment of complementary resource use and emissions is an important complement in a full sustainability analysis.

3.5.3.3 Occupation of materials in products

Like the issue of quality, in-use materials can be embedded in products that work on different functional levels. In the case of materials used in laptops, although a reused product yields a higher UOR in most cases, it also delivers different functionality compared to a new product. This limitation is particularly evident in the case of energy-using products, which can have higher energy efficiency when new while performing the same function. Coupling the UOR with an energy efficiency factor could increase its relevance in a CE context.

To illustrate our indicators, we conducted two case studies based on the literature. These examples have characteristics that allow the simplification of otherwise complex supply chains of materials and further manufacture and use of products. For example, for the system boundary of the materials in the laptop case study, we assumed that all recycled materials are used in the production of similar products. However, only a part of these recycled materials would be used in a closed loop, as assumed, while the rest would be used in different product applications. Although these simplifications in our case studies do not invalidate our proposed methodology, a path for improvement would be to extend the evaluation of different material streams and their further use in distinct product cycles.

Moreover, products can be composed of several materials. We chose to show results disaggregating the materials in each product cycle, but a cumulative in-use occupation index could show an aggregated result for all materials involved. The issue with such indexing is that materials do not have the same value in terms of quality or price; therefore, simply summing up the masses of materials can lead to misleading conclusions. A possible way to compensate for this shortcoming is to use a common unit that reflects both the amount and quality of the materials, such as joules of exergy (Huysveld et al., 2015), or to combine life cycle assessment with life cycle costing (Di Maria et al., 2018). The use of a common unit can also facilitate the evaluation of more complex product systems that include, for example, energy recovery with incineration or innovative processes, such as biomass pyrolysis (see e.g. Maroušek et al., 2019). Exergy or life cycle thinking could be combined with in-use occupation assessments to tackle the issues of trade-offs, quality, and aggregation of different materials.

3.6 Conclusion

For a transition to a CE, we should keep materials in a useful state for as long as possible, avoiding hibernation and dissipation to the environment or technosphere. Key strategies to avoid dissipation are increasing material efficiency and maintaining materials in the in-use state. The valuation of natural resource use is challenging and has proven to be debatable. We take the notion of the UNEP Life Cycle Initiative (UNEP/LCI, 2019) that mineral resources, excluding energy carriers, are not depleted but rather occupied for a certain time frame and extend it to renewable materials, such as wood. Our indicators based on in-use occupation that are presented herein represent an innovative way to calculate the circularity of materials in products while (i) taking into account the dissipation of materials, (ii) distinguishing their occupation in different phases of a product cycle (supply, in-use, and hibernation), and (iii) distinguishing their different product cycles. Of key importance to these indicators is the definition of a time horizon (TH) that includes the use of materials until the next generation. This TH balances a feasible prediction of technology development with our societal responsibility to future generations. Our indicators are complemented by charts to facilitate their comprehension by non-experts and can provide a scientific basis for policies to support technologies or products with increased in-use occupation and retention of materials. The inclusion of parameters balancing the trade-offs in terms of resources and emissions against materials' occupation could further increase the meaningfulness of our indicators from a CE perspective.

4 Resource efficiency indicators to assess circular economy strategies

A case study on four materials in laptops

Redrafted from:

Title: Resource efficiency indicators to assess circular economy strategies: a case study on four materials in laptops.

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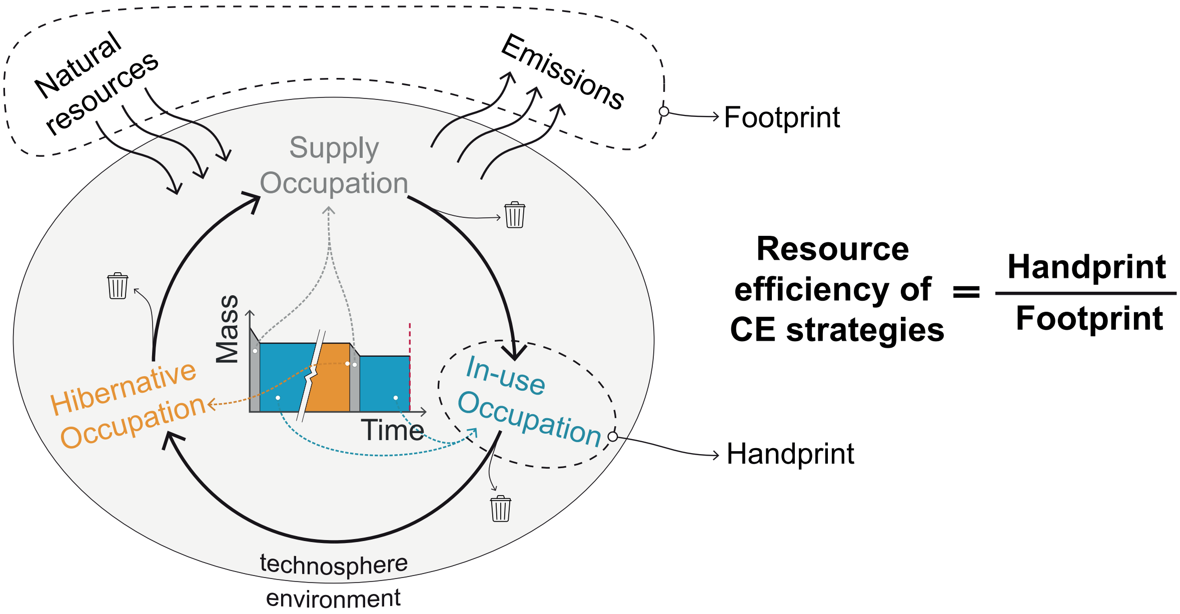
Manuscript submitted to A1 journal in March 2021

Abstract

Electronics require a complex composition and energy-intensive manufacturing. Yet, most of the world's waste electrical and electronic equipment is not collected and recycled. Circular economy (CE strategies can reduce the electronics' loss of materials and environmental footprint. Resource efficiency indicators – typically defined as benefits (handprint over burdens (footprint – can measure materials' life cycle performance. This chapter aims to develop (environmental and resource resource efficiency indicators that show the benefits and burdens of materials use. We illustrated the indicators with a case study of four materials (aluminium, copper, iron, and plastics embedded in laptops. The study includes scenarios with different CE strategies: energy recovery, only recycling, refurbishing with recycling, and reuse with recycling. The scenarios show the use of the materials in several cycles of laptops over a 25-year time horizon. Our methodology expands the traditional one-cycle perspective by measuring the use of materials for 25 years.

Scenarios with cycles of refurbishment and reuse showed a resource efficiency improvement up to 189% and 157% when compared to energy recovery, respectively. Nonetheless, it is remarkable that the average resource efficiency results showed a preference for refurbishing scenarios over reuse during 25 years. The result is limited to shorter functional periods for reused laptops than for refurbished or new laptops. Our analysis is relevant for a CE, where the value of materials should be kept for as long as possible. Policy-makers can use our resource efficiency indicators to assess CE strategies for several cycles of products that keep materials in use with lower environmental impacts.

Graphical abstract



4.1 Introduction

The world's climate is the direct subject of how the global economy manages natural resources, and resource efficiency will be vital to meet the Paris Agreement's temperature goals (UNEP/IRP, 2020). In this sense, the circular economy (CE) concept can profoundly influence how we manage resources. In a CE, 'the value of products, materials, and resources is maintained in the economy for as long as possible, and the generation of waste minimised' (EC, 2015a). In the European Union, the recent New CE Action Plan intends to achieve carbon neutrality and more efficiency in resources management (EC, 2020). It is evident within the definition above that CE management strategies of materials and products over time can be an asset in reducing environmental impacts.

Electric and electronic equipment (EEE) are critical products in the New CE Action Plan and worldwide. It is staggering that only about 17% of the world's waste electrical and electronic equipment (WEEE) generated in 2019 was properly collected and recycled (Baldé et al., 2020). Moreover, modern EEE are composed of complex components made with various energy-intensive extraction and processing steps (Althaf et al., 2019); hence, the loss of resources is even more alarming. Frequently, the potential recovery of materials is linked with their quantities in such products; smaller quantities are less likely to be recycled (Graedel and Reck, 2014). With technology miniaturisation and dematerialisation to provide similar or better functionality (Kasulaitis et al., 2015), EEE complexity tends to increase, which complicates even more the recycling of materials. On the other hand, CE strategies at the level of components and products can extend the lifetime of EEE, postponing the need for recycling. Often, CE strategies are evaluated at the level of products – reuse, remanufacture, or refurbish (e.g. André et al., 2019; Boldoczki et al., 2020; Tecchio et al., 2016) – or at the level of materials – recycling, or downcycling (e.g. Van Eygen et al., 2016; Wäger and Hischier, 2015). However, from the CE definition above, one should assess these strategies simultaneously considering materials functionality over time and their environmental benefit and burden.

Life cycle assessment (LCA) can evaluate the potential life cycle environmental impact of EEE for both products and materials. LCA can be used for product development, strategic planning, and policy-making (ISO, 2006a). In the case of LCA within CE's scope, assessing different product cycles can be particularly relevant. However, the current LCA guidelines are typically used to assess one life cycle of a product, that is, from materials extraction and production until the end-of-life (EoL) of one product with a single lifespan perspective (Suhariyanto et al., 2017). The assessment of multiple cycles presents challenges compared to a single cycle. For example, the definition of the functional unit, which is the ‘quantification of the identified functions of the product’ (ISO, 2006a), must ensure a focus on the function of the different cycles (Lu et al., 2015). If the product cycles are to be analysed simultaneously, the focus could be on a common aspect between them, for example, the used materials. To put it another way, it can be on the functions provided by the materials to deliver one or more products. Indeed, the ISO 14044 defines ‘product’ as any good or service, including, for example, processed materials (ISO, 2006a).

From a materials cycle perspective, Pauliuk (2018) argues that resource depletion, in-use stock, and lifetime should be the core of CE indicators. However, material depletion is a disputed topic, particularly for metals (Dewulf et al., 2021). It is clear that the natural stocks of abiotic material resources are finite; their mining – or the transfer of materials from the ecosphere to the technosphere – over time can lead to the depletion or a quality decrease of the natural reserves. This is the underlying paradigm behind some life cycle impact assessment methods characterising the use of resources, such as the Abiotic Depletion Potential (ADP) (van Oers et al., 2002). ADP takes into account the increasing impacts with the extraction based on a natural fixed-stock paradigm. There are, however, two issues with the assumption: (a) the quantification of natural stocks is scientifically challenging, and (b) the transfer of materials from the ecosphere to the technosphere does not necessarily mean resource depletion (Dewulf et al., 2021). When material resources are extracted, they are borrowed in society for some application and can become available for future use (Frischknecht, 2016). An alternative way to assess material resources is to identify and quantify the human actions that compromise their accessibility (Dewulf et al., 2021).

The actions that decrease the accessibility of materials are environmental dissipation, technosphere hibernation, and in-use occupation (van Oers et al., 2020). The first is related to the dispersion of materials to the environment to the point that their recovery is no longer technically or economically feasible. Technosphere hibernation comprises many actions that maintain non-functional materials stocks in society (i.e. landfills and tailings, abandoned infrastructure, dispersed stocks in the technosphere, and hoarded products). Here, it is important to acknowledge the time constraints of each action of technosphere hibernation. Dispersed stock in the technosphere, or technosphere dissipation, contribute to inaccessibility in a similar level as environmental dissipation – for any practical sense they are irreversible (Dewulf et al., 2021). However, landfills and tailings, and abandoned infrastructure may be reversible in a medium to long term (25–500 years), while hoarding may be reversible in a shorter term (1–5 years) (Dewulf et al., 2021).

The last action is in-use occupation – It can be understood as a cause of inaccessibility as it creates competition among consumers that can access the materials or not. However, the in-use occupation as a cause of inaccessibility may be questioned as materials serve their purpose of providing services (Dewulf et al., 2021). As the functional use of materials, the in-use occupation is the reason for materials extraction. Hence, if one takes into account that the assessment of resource depletion is impractical, we could redraft the Pauliuk's argument above as resource dissipation, in-use, and lifetime could be included in the core of CE indicators, alongside their environmental impacts.

With increased in-use occupation, the environment can benefit from less anthropogenic pressures to extract new materials. Increased material efficiency (through, e.g. less dissipation and hibernation) is key for environmental and economic sustainable use of materials (UNEP/IRP, 2017). Hence, the concept of in-use occupation is relevant as it can serve as a proxy indicator of the societal benefit (handprint) of having materials in use over time. Handprint refers to the beneficial actions towards sustainability. In LCA, Alvarenga et al. (2020) distinguish the beneficial effect (handprint) addressed to the intended user and to unintended affected subjects of a product. In this sense, in-use occupation can be a proxy of the benefits to the intended user, as users receive the benefit provided by the products, and to unintended affected subjects, as the environment can suffer less anthropogenic pressures for extraction when materials are not dissipated or hibernating.

Moreover, the development of indicators based on in-use occupation is of particular interest. Indeed, indicators are the backbone for monitoring a CE, but the field is still controversial. Many indicators were developed to measure a CE, but two aspects deserve attention to slow down and close resources loops – quantity and quality (Moraga et al., 2019). With this reasoning, Moraga et al. (2021), which is reproduced in chapter 3, expanded the rationale behind the in-use occupation of materials – as a measure of the initial use of primary raw materials (quantity) dedicated to an application in use for an amount of time (quality) – to develop a pair of indicators. One indicator, the in-use occupation ratio, shows the performance of the occupation of materials considering a 25-year time horizon; the other indicator, final retention in society, shows how much material can still be recovered at the end of this time horizon. These indicators (defined in subsection 3.2.3) measure the beneficial use of materials, factoring in the utilisation within a 25-year time horizon and the potential for utilisation beyond this time horizon, considering different CE strategies. However, these indicators miss the connection with environmental sustainability impacts. Thus, the measure of the environmental footprint with the use of materials could be used to develop resource efficiency indicators that assess different CE strategies.

Therefore, although the measure of the in-use occupation of materials is a valid indication of the useful retention of the materials in society, this occupation comes with a footprint in an environmental, economic, and social sense; we focused on the first. Hence, this chapter aims to measure the environmental footprint of strategies that can prolong the in-use occupation of materials and propose resource efficiency indicators based on this measure. We will consider global warming emissions and cumulative resource use as proxies of the environmental pillar of sustainability. We illustrate the analysis with a case study of four materials embedded in laptops.

4.2 Resource efficiency indicators of the in-use occupation of materials and final retention in society

This section proposes resource efficiency indicators that can quantify the handprint and footprint for a particular in-use occupation and final retention in society. The indicators are based on the framework of efficiency indicators from Huysman et al. (2015b), where resource efficiency with LCA is defined as a ratio of benefits divided by environmental impacts based on resource or emissions flows; these indicators are also called eco-efficiency. Herein, we define handprint as a proxy for the benefit of the in-use occupation of materials within the 25-year TH and the final material retention. In contrast, the footprint is defined as the environmental impact caused by such in-use occupation. Potential environmental impacts can be assessed through the LCA framework (ISO, 2006a, 2006b).

LCA is intended for the assessment of products (good or services) that include, for example, processed materials (ISO, 2006a). However, we assess materials that demand further processing before being used in final products, which usually include several materials, during the TH. Hence, Eq. (4.1) defines the environmental impacts of materials focusing on evaluating 1 kg of material used in j products cycles during the TH. This equation considers a physical (mass) allocation factor to partition impacts related to the whole product (e.g. manufacturing) among the different materials. As we assess the employment of materials, the equation does not account for the environmental impacts of the use phase (e.g. electricity consumption during use).

$$Ftp = V_U + \sum_{j=1}^{n} \left(P_{U,j} + \frac{M_j + Rc_j + Rf_j + Re_j + Dis_j}{m_{U,j}} * AF_{U,j} - AVe_{U,j} \right) - AVm_{U,TH} \tag{4.1}$$

where:

Ftp: Footprint associated with the employment of 1 kg of material during the TH ['impact unit' / kg material used during the TH] – 'impact unit' stands for the unit of a chosen life cycle impact category (e.g. kg CO_2-eq).

V_U : environmental impacts of the production of 1 kg of the virgin raw material ['impact unit' / kg material]

$P_{U,j}$: environmental impacts of the primary or secondary processing of the material in the product cycle j ['impact unit' / kg of material in cycle j]

M_j : environmental impacts of the manufacturing of a product in cycle j ['impact unit' / unit of product]

Rc_j : environmental impacts of recycling the materials embedded in the product from cycle j ['impact unit' / unit of product]

Rf_j : environmental impacts of refurbishing the product in cycle j ['impact unit' / unit of product]

Re_j : environmental impacts of reusing the product in cycle j ['impact unit' / unit of product]

Dis_j : environmental impacts of final disposal of the product in cycle j ['impact unit' / unit of product]

$m_{U,j}$: mass of the material in the product from cycle j [kg / unit of product]

$AF_{U,j}$: allocation factor to the material regarding the impacts of the product from cycle j [%] – in this case, the mass allocation is used following the ISO 14040 preference for a physical relationship.

$AVe_{U,j}$: avoided environmental impact of energy production due to energy recovery from the material in the product cycle j ['impact unit' / kg of material in cycle j]

$AVm_{U,TH}$: avoided environmental impact of the production of the virgin raw material due to its secondary recovery after the TH ['impact unit' / kg of material at the TH]

Based on Eq. (4.1) for the footprint (Ftp), the resource efficiency indicators of the in-use occupation and final retention in society can be derived in Eq. (4.2) and Eq. (4.3), respectively.

$$Eff_{Occ} = {\sum_{j=1}^{n} Occ_{U,j}}/{Ftp} \tag{4.2}$$

where (symbols not previously introduced):

Eff_{Occ} : resource efficiency of the in-use occupation of a material [kg × year / 'impact unit']

$$Eff_{FRS} = {FRS}/{Ftp} \tag{4.3}$$

where (symbols not previously introduced):

Eff_{FRS} : resource efficiency of the final retention in society [% / 'impact unit']

4.3 Case study: four materials in laptops

This section consists of four parts. First, the case study with four scenarios is described. Afterwards, the first three phases of an LCA are described to calculate the environmental footprint (section 3). The LCA phases are goal and scope definition, inventory analysis, and impact assessment (ISO, 2006b, 2006a). The final interpretation phase is covered in the results and discussion sections.

4.3.1 Description of the case study

The overview of the case study is presented in Figure 4.1. The case study has four scenarios – S1, S2, S3, and S4 – focusing on materials used in laptops. The considered groups of materials are aluminium (Al), copper (Cu), iron (Fe), and plastics (PL) that represent 65% of the laptops' composition (Figure 4.2), based on Babbitt et al. (2020) and Van Eygen et al. (2016). Detailed inventory tables can be found in the appendix for dissipation (Tables C1–C6) and time of supply, in use, and hibernation phases (Tables C7–C8).

In S1 (energy recovery, baseline), laptops are incinerated after one cycle of use, and the energy is recovered based on the lower heating value (LHV) of the materials. If energy can be recovered from a certain material, we consider it as avoided energy. The efficiencies of energy production were retrieved from De Meester et al. (2019).

S2 includes the recycling of laptops' materials. Initially, the laptops are separately collected and manually dismantled with the separation of scrap fractions. Afterwards, these scrap fractions are shredded and mechanically separated with magnetic and eddy current separators and others, as described by Van Eygen et al. (2016). Finally, the mechanically separated scrap fractions are sent to (secondary) material processing. The share of unrecoverable materials after separation is sent to incineration. The incineration process in the case of S2–S4 includes energy recovery and bottom ash recovery. The efficiencies of energy recovery and bottom ash treatment were retrieved from De Meester et al. (2019). If energy or metals from ash are recovered, we include them as avoided energy or raw material production, respectively.

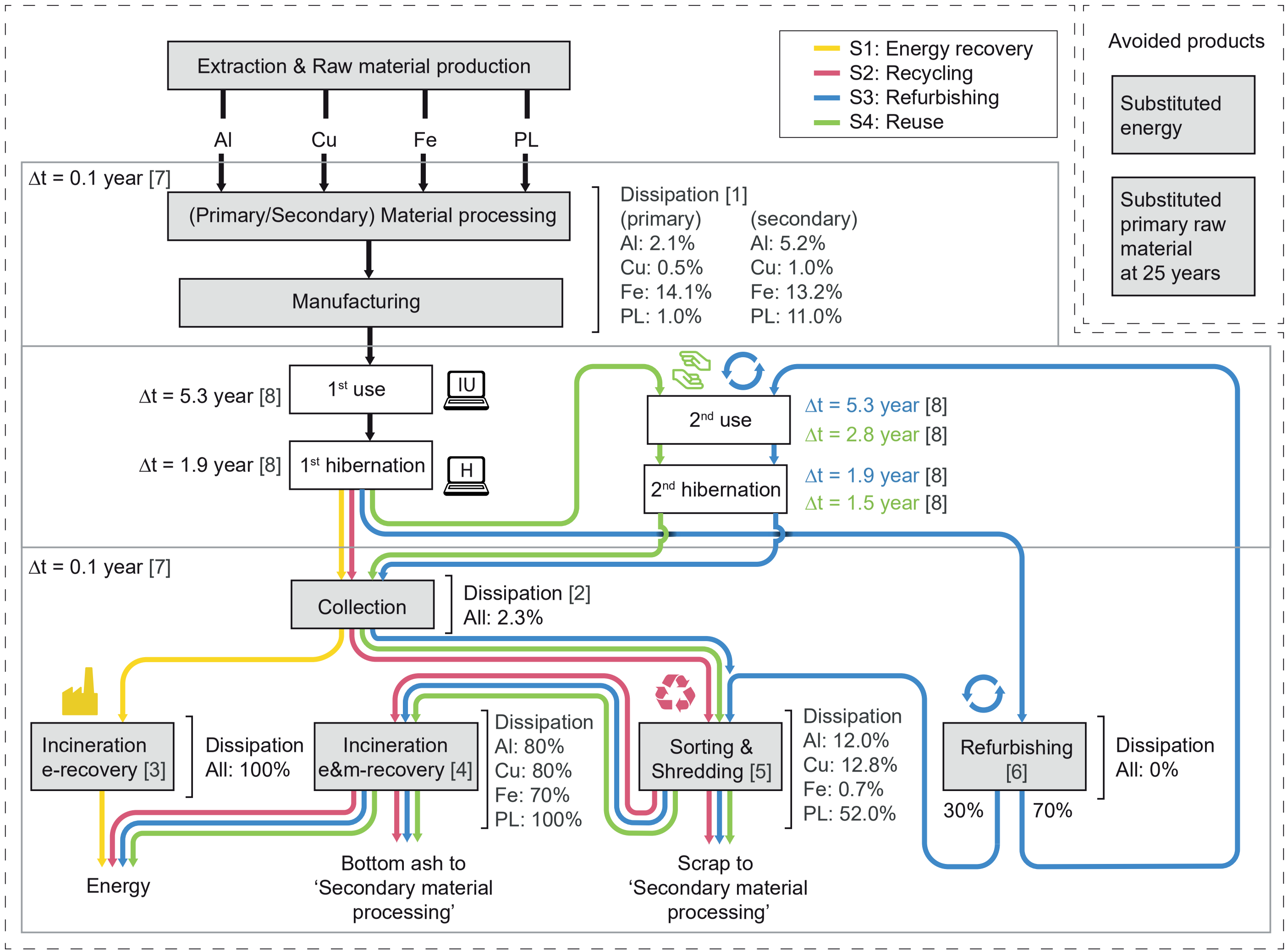


Figure 4.1: System boundaries of the four scenarios in analysis considering four materials: aluminium (Al), copper (Cu), iron (Fe), and plastics (PL). References between brackets: [1] Dissipation of primary/secondary material production and manufacturing is calculated based on (Cullen and Allwood, 2013) for Al, (Soulier et al., 2018) for Cu, (Cullen et al., 2012) for Fe, and ecoinvent for PL; [2] collection rate of WEEE is based on (Deloitte Consulting & Advisory, 2018); [3] energy recovery (e-recovery) rate is based on (De Meester et al., 2019); [4] energy recovery and material recovery from bottom ash (e&m-recovery) rate is based on (De Meester et al., 2019); [5] dissipation from sorting and shredding is based on (Van Eygen et al., 2016); [6] share of 70% of laptops refurbished is based on (André et al., 2019); we assumed no dissipation during refurbishment; [7] we assumed the time of supply phase; the time of in-use and hibernation phase is based on (Thiébaud et al., 2018)

S3 encompasses the refurbishing of laptops after the first use. In this case, we consider a share of laptops (70%) to be refurbished and commercialised as semi-new products, based on André et al. (2019). This refurbishing process includes sorting, testing, data erasure, and resale with a one-year warranty; hence, laptops will have a different user. The non-refurbished share (30%) follows the recycling pathway. In S3, the times for the in-use and hibernation phases of the second use are the same as for new laptops.

S4 comprises simple reuse – a second use of laptops by the same or a different user, but without a professional preparation for reuse (as in S3). In S4, the times of in-use and hibernation phase of the second use are shorter than for new products, as reported by Thiébaud et al. (2018). After one cycle of second use, the laptops follow the recycling pathway. In S4, the only impacts are related to transportation of the laptops for reuse.

4.3.2 Goal and scope definition

The goal is to quantify the environmental impacts from the use of materials defined in Figure 4.1. The life cycle impact assessment (LCIA) results will be used in the footprint assessment (4.1) to provide a specific in-use occupation. The scope is exemplified by the system boundary (Figure 4.1). In each step of the system boundary, there are dissipative losses associated with each of the materials. Therefore, we model the system boundary starting with 1 kg of the primary raw material as input to one of the scenario's pathway. The system boundary is time-constrained – it ends either when all the material dissipates or when the material reaches the 25-year TH.

The study starts with the acquisition and production of the four primary raw materials. After this phase, we account for the dissipation of materials and the time for production and manufacturing, use, and hibernation. Aluminium starts with the production of ingots from virgin bauxite through electrolysis. Copper starts with the production of casted copper anodes from copper sulphides. Iron, the scenarios start with the production of pig iron from iron ores and pellets through blast-furnace smelting. Finally, t plastics start with the production of naphtha. We expand the system boundary to include the avoided products used outside the system boundary (Figure 1), namely, avoided primary material production after year 25 and avoided energy (from incineration). Avoided materials do not include the materials recovered before the TH as the system boundary is time-constrained, and this inclusion would breach its boundaries.

In all cases, the functional unit is 1 kg of primary raw material (aluminium, copper, iron, or plastics) dedicated to the first product application (laptops), and its conservation in similar applications for the time horizon of 25 years.

4.3.3 Inventory analysis

We use data adapted from ecoinvent version 3.4 (cut-off model – OpenLCA Nexus) for the processes described in the system boundary. When adapting data, we used various scientific literature sources to improve the temporal scope quality of the ecoinvent data. Here the most important inventory information is described, but full inventory tables can be found in appendix for the four scenarios (tables C9–C26).

For laptops manufacturing, we use data of the mass amount of components and materials measured by Babbitt et al. (2020). These authors generated bills of materials of 16 laptops through product disassembly. Babbitt et al. (2020) presented the main components and their material composition but did not specify the composition of printed circuit boards (PCB), flat screens, and Li-ion batteries. We complement the material composition information with data from Van Eygen et al. (2016) for these three components.

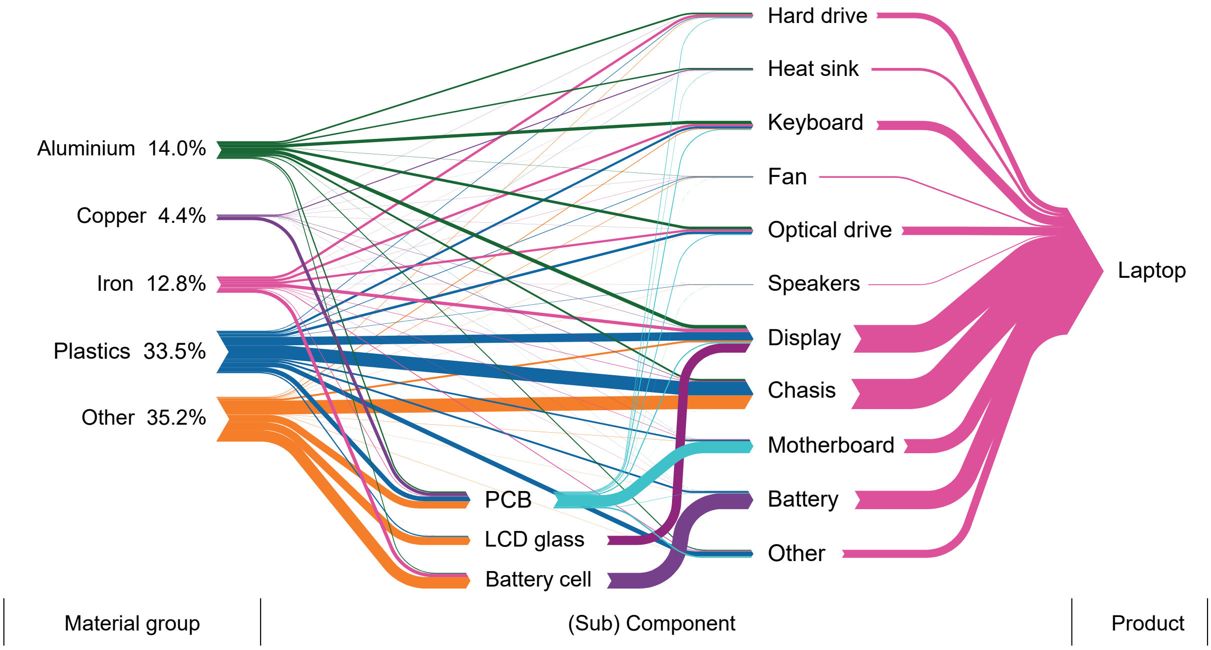


Figure 4.2: Share of the mass of materials in laptops; components. Based on the arithmetic mean values from Babbitt et al. (2020) and Van Eygen et al. (2016). PCB (printed circuit board); LCD (liquid crystal display)

Furthermore, to better understand the future 25-year effects of using those materials in laptops computers, we include the uncertainty of the laptops' composition based on disassembly data. As this data is based on computers produced between 1999 and 2011, and because of the fast effects of technology change in EEE, we use stochastic modelling with asymmetric triangular distributions for the mass contribution of laptops' components and their materials. We perform a Monte Carlo simulation with 1000 interactions in OpenLCA. For the choice of the number of interactions, we did a sensitivity analysis varying the interactions by a factor of 10 (i.e., 100 and 10.000). The arithmetic mean varied by 0.008% and 0.004%, respectively; hence, we choose 1000 calculations to save computation time (Table C28). Inventory tables with the distribution of each parameter are provided in appendix C (Tables C9–C26).

4.3.4 Impact assessment

Two midpoint LCIA methods are used for broader coverage of environmental impacts: one based on resource consumption and another based on emissions. For the first, we select the cumulative exergy extraction from the natural environment (CEENE) version 2013 as natural resource footprint (Alvarenga et al., 2013; Dewulf et al., 2007). CEENE assesses resources that are withdrawn from the ecosphere by quantifying the cumulative extracted exergy (Dewulf et al., 2007). The method differentiates eight midpoint categories in a single scale (MJ_{ex}): abiotic renewable resources, fossil fuels, nuclear energy, metal ores, minerals (and mineral aggregates), water resources, land resources, and atmospheric resources. This method was recently recommended by the United Nation Environment Program's Life Cycle Initiative to evaluate the environmental impacts of mineral resource use based on thermodynamics (Berger et al., 2020; Sonderegger et al., 2020; UNEP/LCI, 2019). As an emissions-based method, we use the method climate change (CC) – global warming potential 100a (GWP100 based on IPCC 2013 from ecoinvent 3.4 LCIA methods compiled by OpenLCA Nexus) as carbon footprint. This method was recommended by UNEP/LCI as the midpoint impact category to describe short-term environmental and human health consequences of climate change (UNEP/LCI, 2016).

4.4 Results

4.4.1 In-use occupation and final retention in society

Figure 4.3 shows the occupation of aluminium, copper, iron, and plastics used in laptops for scenarios S1, S2, S3, and S4. The coloured line in these charts accounts for the remaining mass of the initially produced input of primary raw material along the 25-year TH. The supply phases are brief and have almost negligible occupation (Δt = 0.1a), but they contribute significantly to the dissipation of materials.

In S1 (energy recovery), the overall in-use occupation is the lowest amongst all scenarios as all materials are dissipated after the first-product cycle in the incineration process. In S2 (recycling), the in-use occupation is similar to S3 (refurbishment) and S4 (reuse), except for plastics. There are considerable losses in the mechanical recycling of plastics, which makes the in-use occupation of this material group smaller than for the other materials when recycled. The share of materials' dissipation in S2 is similar for all cycles, and the absolute dissipation decreases over time (as less material is remaining after each cycle). In S3 (refurbishment), most of the dissipation happens in the recycling process for the non-refurbished share of laptops. In contrast to S2, the absolute dissipation of materials increases over time, except for plastics. This is because we considered that 70% of the laptops are refurbished, and in this case, the dissipation increases each cycle if losses of materials' EoL processing are below a 10–25% threshold (material dependent). Above this threshold, the 3rd cycle will present the highest absolute dissipation; the smallest absolute dissipation will be on the 2nd or 4th cycles, depending on the remaining amount of material from the previous cycle (in appendix, Figure C1–C3, a sensitivity analysis of the absolute dissipation amount per cycle is provided). Unlike the other scenarios, S4 (reuse) shows five in-use occupation phases within the TH (while the others show four or fewer). The reuse has shorter times of in-use and hibernation; hence, more product cycles. In S4, most dissipative losses happen in the supply phase from the 3rd and 5th product cycles, which are related to recycling of the materials after the reuse of laptops (in the 2nd and 4th cycles).

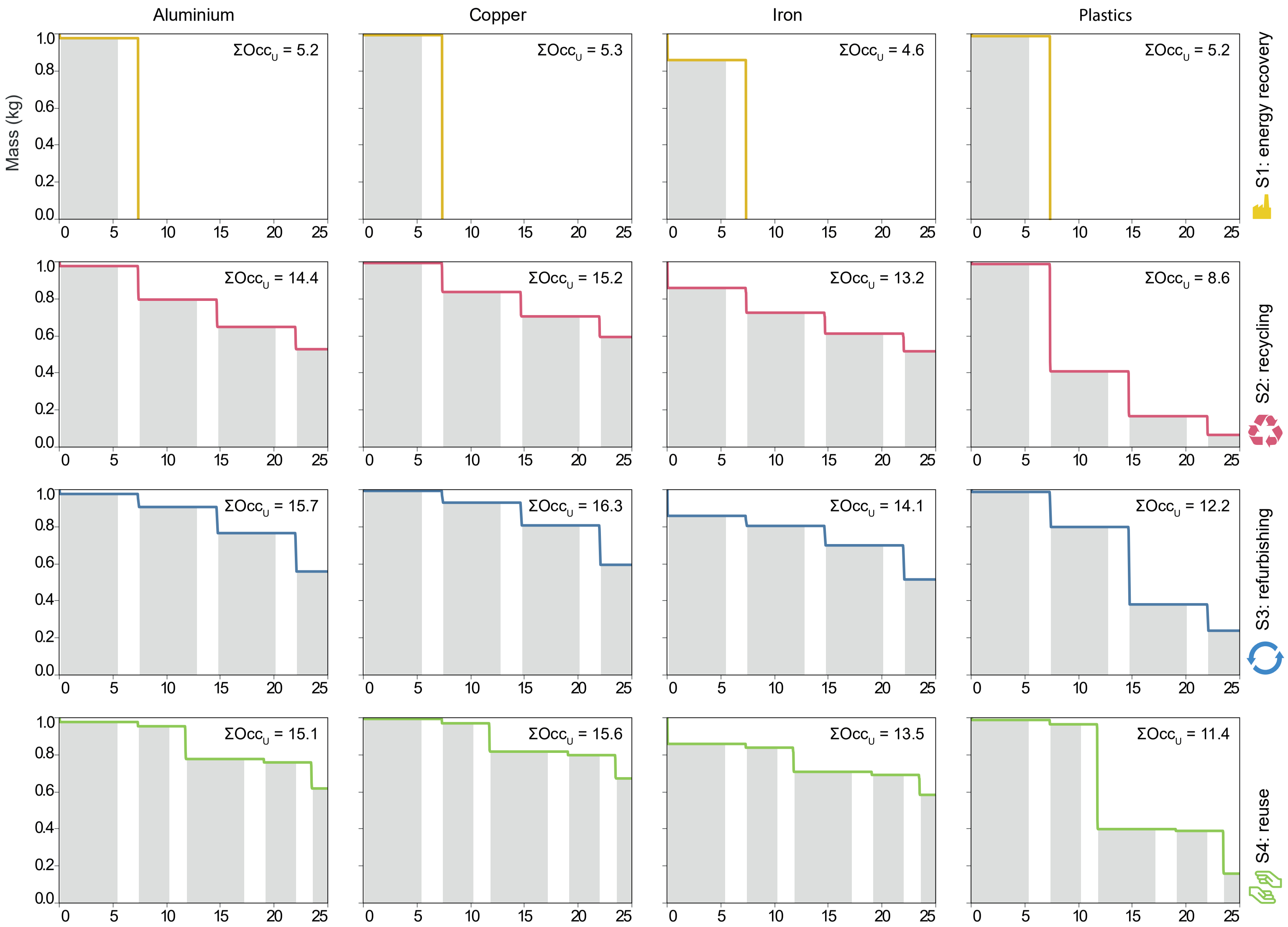


Figure 4.3: In-use occupation of four materials used in laptops considering four scenarios. Coloured lines account for the remaining mass of the initially produced primary raw material input along the TH. Grey areas identify the in-use occupation phases (kg × year). The supply (production, manufacturing, and EoL) and hibernation phases are white areas before and after the in-use, respectively.

4.4.2 Footprint of the in-use occupation of materials

Figure 4.4 and Figure 4.5 show the cumulative footprint with the occupation of aluminium, copper, iron, and plastics used in laptops for four scenarios regarding carbon footprint and natural resource footprint, respectively. In these graphs, we combined the coloured line – remaining mass of the initial input of primary raw material over the 25-year TH – with a box plot – footprint of the in-use occupation in each scenario. Figure 4.4 and Figure 4.5 zoom in specific years. Impacts are shown in box plots along time and refer to the supply phase of each cycle. The box plot's error refers to the uncertainty of mass variation of those materials in laptops' components. The first zoom-in section in each chart shows two box plots; the left one refers to the upstream primary production of the raw materials, while the right one concerns the material's downstream processing up to the manufacturing of the laptops in the 1st cycle. These figures only show the impacts within the system boundary, but not those from the avoided burdens.

The highest impacts of extraction and raw materials production are related to aluminium and copper. However, since most of the impacts are related to the manufacturing of the laptops (Figure 4.6), and those impacts are equally allocated based on their mass contribution, the cumulative impacts will decrease with a higher dissipation. For this reason, the cumulative impacts of S1 are lower than those from S2–S4. But in the latter scenarios, a share of the materials will be functional in society until year 25 and available beyond, while in the first, there is a demand for the extraction of the whole amount of materials initially used.

For both natural resource and carbon footprint, the cumulative impacts in each product cycle decrease with the decreasing remaining mass in S1 and S2. However, in S3 and S4, the cumulative impacts oscillate between product cycles. In S3, higher impacts are due to the laptops' manufacturing in the 1st cycle; in the 2nd cycle, 30% of the laptops are sent to recycling, while 70% is remanufactured for a second use. In the 3rd cycle of S3, the previously reused laptops are recycled, and new laptops are remanufactured – explaining the oscillation of the impacts. Whereas in S4, the cycles with second-use present the lower impacts among all product cycles from all scenarios as the reuse only accounts for transportation impacts.

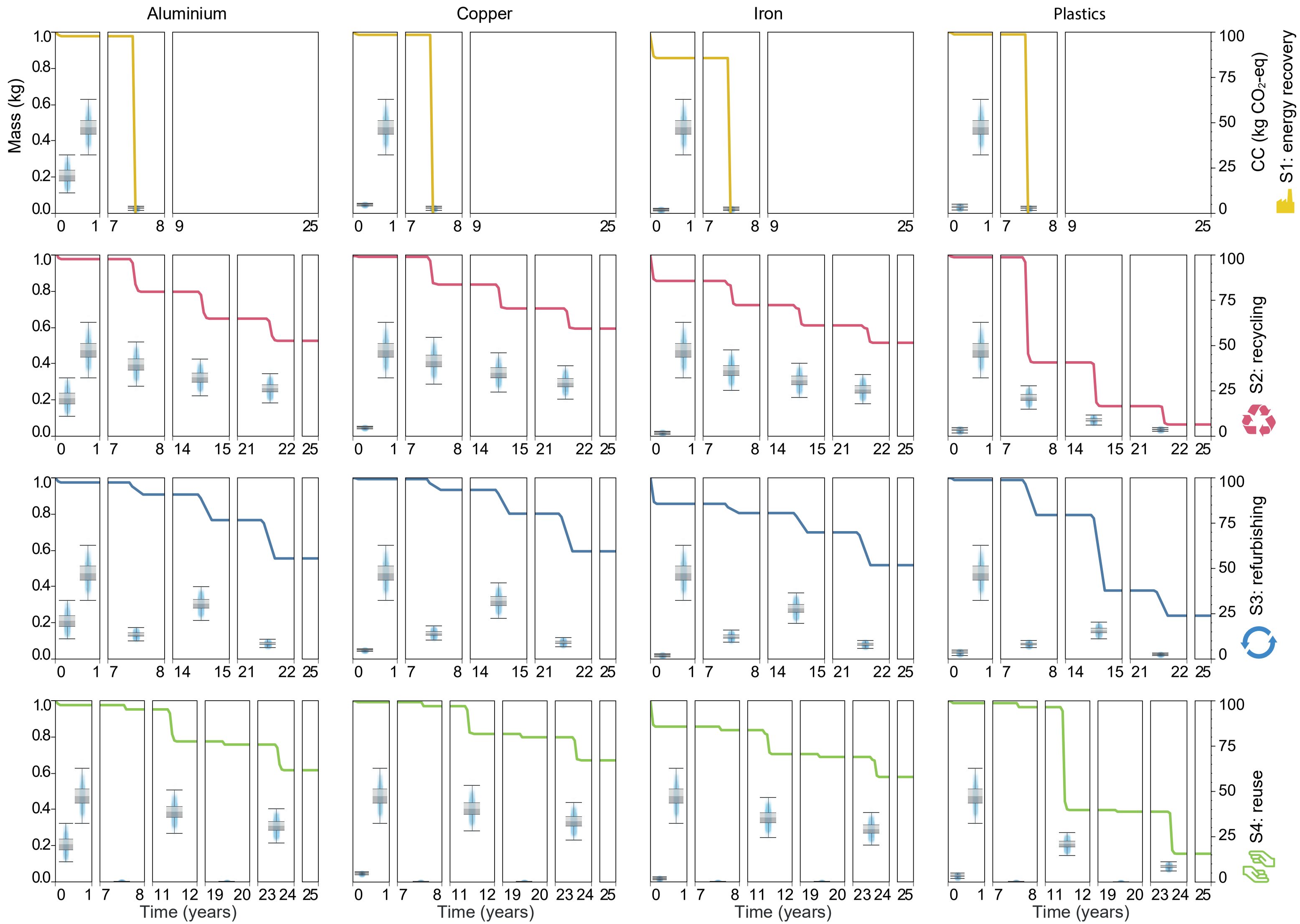


Figure 4.4: Footprint for the in-use occupation of four materials used in laptops considering four scenarios. Coloured lines account for the remaining mass of the raw material. Each supply cycle is shown in sections of one year. Box-plot shows the footprint of each supply phase along time. The dual box-plot group on the left side of each chart shows the extraction and raw material production for the first and processing and manufacturing for the second. The footprint is accounted as emissions responsible for CC (climate change – GWP100 in kg of CO2 equivalent).

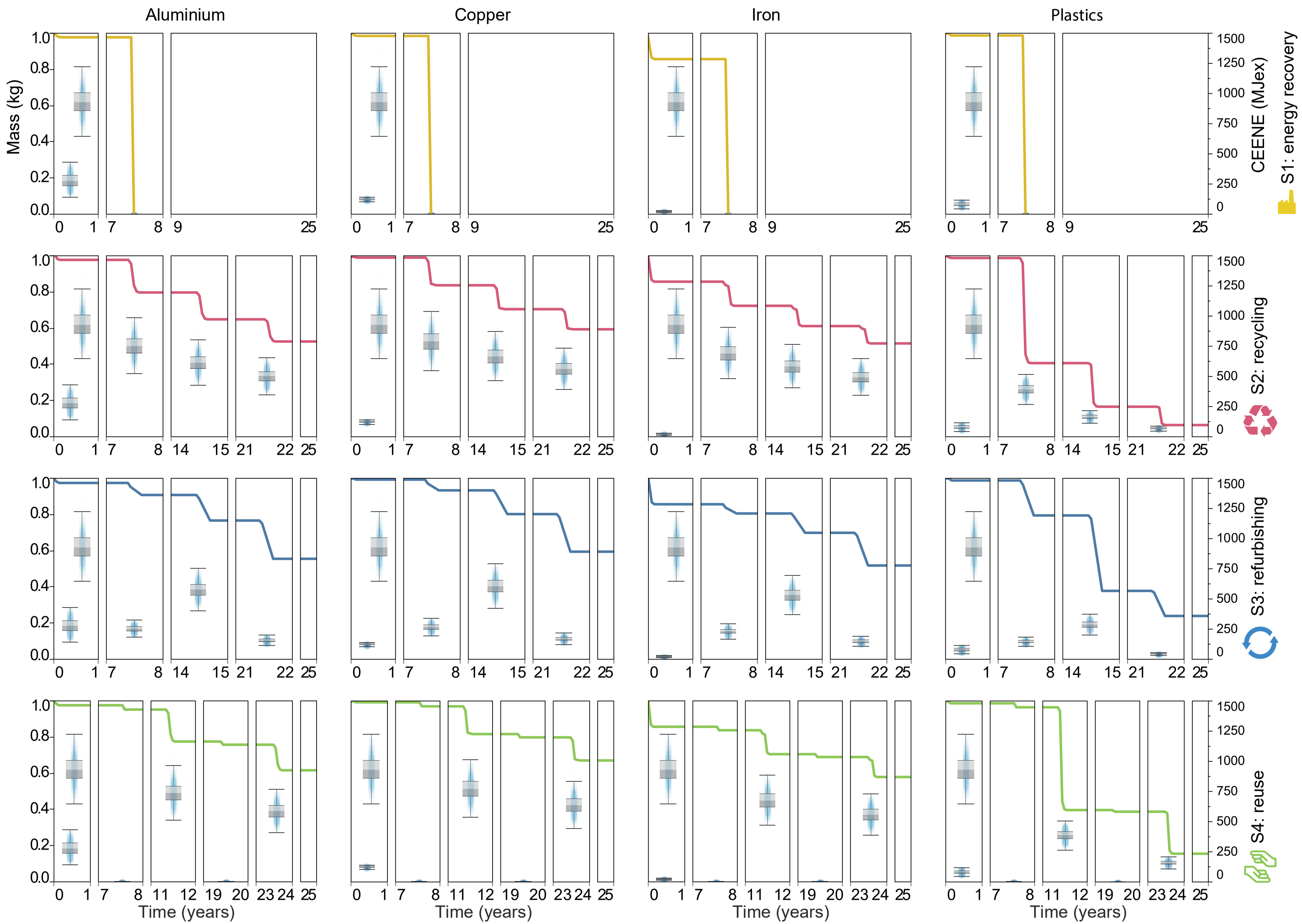


Figure 4.5: Footprint for the in-use occupation of four materials used in laptops considering four scenarios. Coloured lines account for the remaining mass of the raw material. Each supply cycle is shown in sections of one year. Box-plot shows the footprint of each supply phase along time. The dual box-plot group on the left side of each chart shows the extraction and raw material production for the first and processing and manufacturing for the second. The footprint is accounted as cumulative consumption of resources contributing to Cumulative Exergy Extraction from the Natural Environment (CEENE, summation of the resource categories in MJex).

4.4.3 Resource efficiency indicators for CE strategies

Figure 4.6 summarises the results for handprint – UOR and FRS indicators – and footprint – CC and CEENE – for the four materials and the four scenarios. UOR shows a slight preference for S3 (refurbishment), followed by S4 (reuse) and S2 (recycling) for the materials aluminium, copper, and iron. For plastics, there is a slight preference for S3 over S4, but a large preference over S2; this is because of the higher losses in the mechanical recycling process. A higher UOR means that materials have higher in-use occupation over 25 years (i.e. materials are embedded in functional products); hence, materials are more beneficial to society. In contrast, FRS shows a preference for S4 for all materials except plastics. The FRS for plastics shows a preference for S3. This is because most of the plastic materials are dissipated in the recycling process, which is delayed in the refurbishing case.

The cumulative impacts in CC (Figure 4.6c) show that the impacts are mostly related to the manufacture of laptops and components, which are energy-intensive processes, particularly for PCBs and integrated circuits. This can be complemented by information from the cumulative impacts in CEENE (Figure 4.6d) that shows fossil fuels as the source of >50% of the impacts for all materials in scenarios. The avoided impacts with the materials that are not dissipated before the TH are negligible for all materials except aluminium, which has a high contribution from impacts in the primary production.

Figure 4.7 shows the carbon-emission and natural resource efficiency indicators using CC and CEENE, respectively. The resource efficiency of the in-use occupation (Figure 4.7a and 7c) shows a clear preference for S3 followed by S4 with aluminium, copper, and iron. S3 is, on average, preferable to S4 for plastics, but considering the uncertainty, S4 can be more efficient than S3 depending on the amount the material used in the laptops. In the case of S1 and S2, although UOR shows a clear preference for the latter (Figure 6a), its resource efficiency shows similar results for both scenarios. This is because most of the impacts come from the manufacturing process. In S2, laptops are manufactured four times, whereas there is only one manufacturing in S1. The higher differences occur for aluminium, which is caused by the avoided impacts from virgin aluminium production at the year 25.

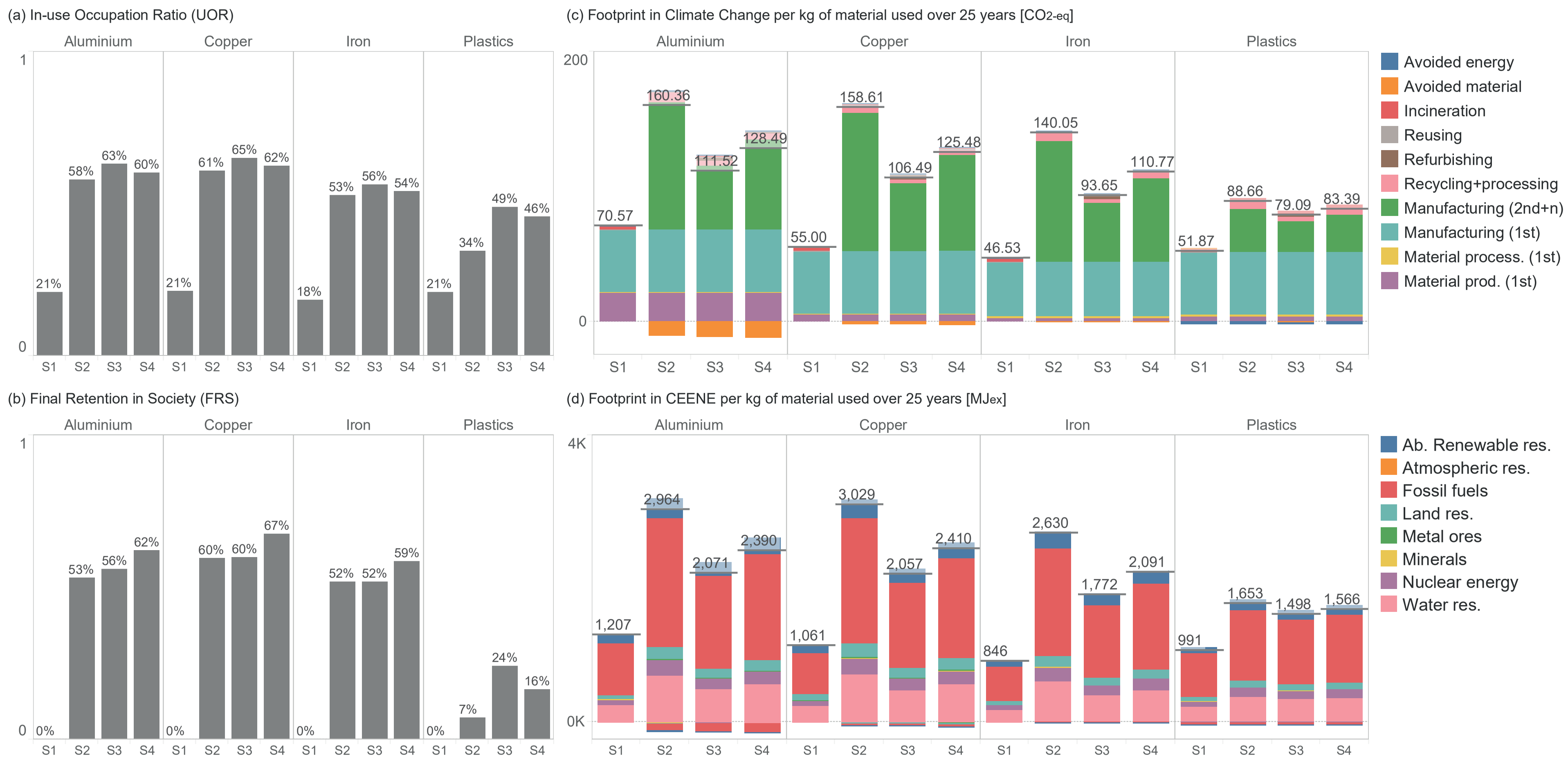


Figure 4.6: Result of indicators (a) in-use occupation ratio (UOR) and (b) final retention in society (FRS). Result of the cumulative impacts for 1 kg of material initially extracted and used over 25 years in (c) climate change (CC), which shows the phases related to the impact, and (d) Cumulative Exergy Extraction from the Natural Environment (CEENE), which shows the natural resource footprint

While FRS (Figure 4.6b) gives preference for S4 in the case of aluminium, copper, and iron, there is a slight preference on average for the resource efficiency of FRS of S3 compared to S4 (Figure 4.7b and 7d). This is because of the higher impacts in S4 – caused mainly by the energy use in the manufacturing of new laptops in the 3rd and 5th cycles – level out the benefits of retaining materials with the lower impacts of S3. This preference, however, is accentuated in the case of plastics, confirming the preference of S3 with this material. S2 presented the second-worst efficiency of FRS due to the higher cumulative impacts for all materials. In S1, the resource efficiency of FRS is equal to 0% per impact unit for all materials, following the same handprint result as in Figure 4.6b.

4.5 Discussion

Assessing material resources in a CE demands proper measurements so that we avoid depletion or dissipation. In the thermodynamic sense, abiotic materials are not destroyed in their mining process. However, they dissipate in the technosphere or natural environment, becoming economically or technologically unrecoverable. Dissipation can be avoided using CE strategies, which can maintain the value of products and materials for as long as possible. Although the extraction itself cannot lead to material's destruction, we can determine particular actions contributing to resources' inaccessibility (Dewulf et al., 2021). Among these actions, the lack of efficiency in the production and consumption systems contributes to increasing the dissipation and hibernation of materials. Moreover, the reason to extract materials is to create value via functional products that keep materials in use in society (van Oers et al., 2020). This functional use of materials is also responsible for inaccessibility, but unlike hibernation or dissipation, we benefit from extracted and manufactured materials during the in-use occupation. Hence, the better we advance in occupying materials in use, the better we take advantage of CE principles.

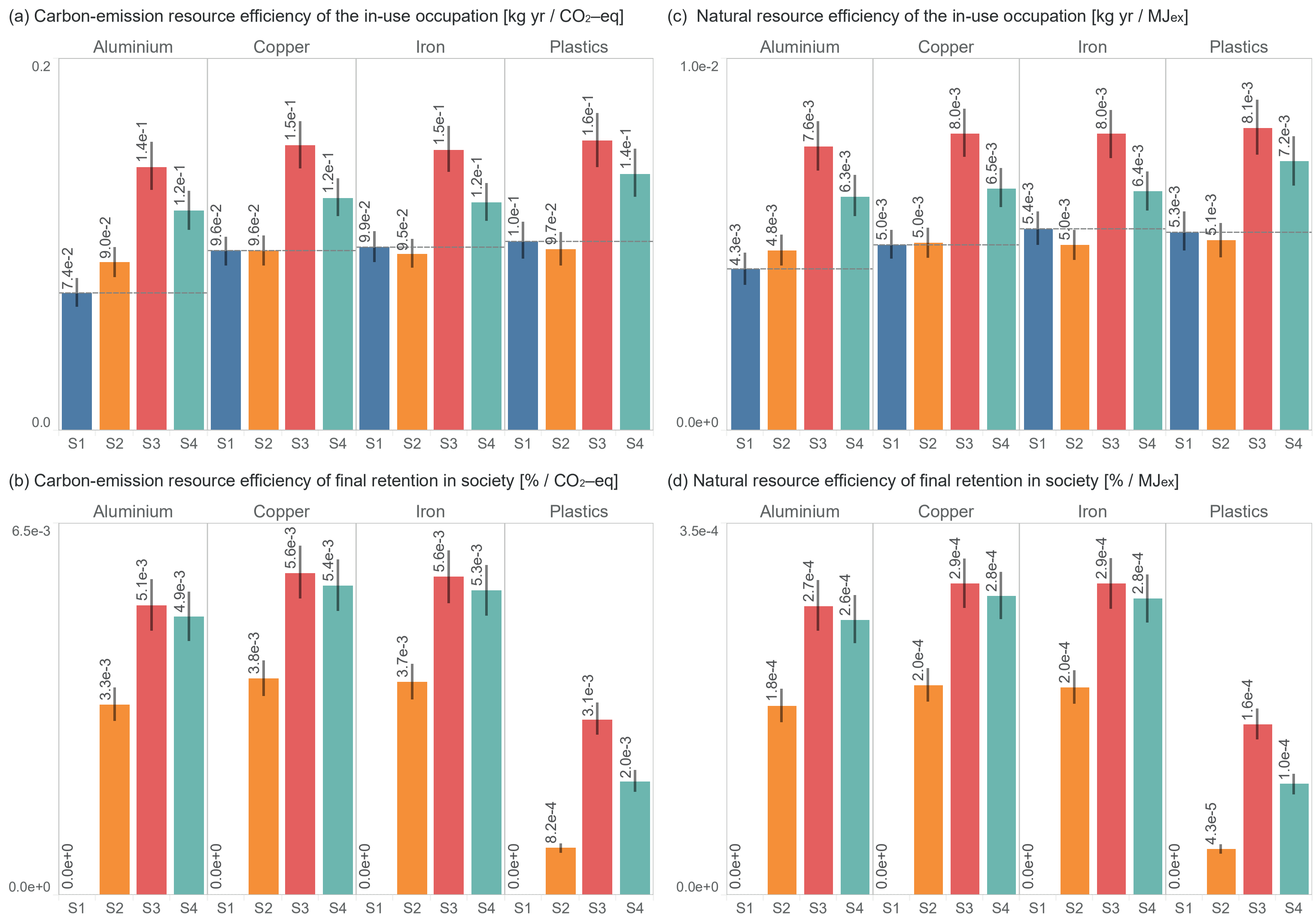


Figure 4.7: Result of the resource efficiency indicators of In-use Occupation using emission impacts as (a) climate change (CC) and use of resources impact as (c) Cumulative Exergy Extracted from the Natural Environment (CEENE). Result of the resource efficiency indicator of final retention in society as (b) CC and (d) CEENE

Nonetheless, occupying materials in use is a benefit (or proxy for a handprint) that presents challenges, such as managing the associated environmental footprints. The footprint is caused by the processing and manufacturing of materials into functional products and their recovery due to a CE strategy, e.g. recycling. In this chapter, we proposed a method and efficiency indicators to quantify the footprint and handprint of maintaining the in-use occupation of materials. We illustrated the method with four groups of materials – aluminium, copper, iron, and plastics – used in laptops with various CE strategies. From the illustration, it is remarkable that the cycles with refurbishment (S3) were on average more eco-efficient for the in-use occupation of the analysed materials compared with reusing (S4) and recycling (S2) in a 25-year time horizon. Still, both reusing and refurbishing were more resource efficient than recycling or energy recovery concerning materials' in-use occupation and final retention in society. Important to mention, however, is that the use phase was not part of the analysis. This phase is related to the use of the product itself, not the management of materials. Evidently, the use phase of energy-consuming products has environmental impacts. Thus, a possible improvement would be the development of product-specific indicators as further discussed.

The results found herein are in function of using materials in laptops; therefore, they cannot be expanded for materials used in other products. Furthermore, the results are dependent on the scenarios' assumptions, such as the in-use and hibernation time of products. However, these assumptions can be further investigated to include more specific data about products lifetime in different CE strategies. Likewise, the LCI data was adapted from ecoinvent inventories by using more recent data about the bill of materials in laptops and their components (Babbitt et al., 2020; Van Eygen et al., 2016). However, the energy and auxiliary requirements for their manufacturing and assembly were not modified – the original ecoinvent dataset for laptop manufacturing is from 2005. Nonetheless, the main source of impacts in computer products is related to the production of PCBs (André et al., 2019; Choi et al., 2006; Duan et al., 2009), mainly because of the energy requirements related to semiconductors. It is worth noticing, however, that the semiconductors' area in types of PCBs used in laptops remained constant from 1999–2011 due to miniaturisation and performance increase of integrated circuits (Kasulaitis et al., 2015). Although semiconductors were miniaturised over the years to provide the same functionality, their increased performance may have counterbalanced the gains with dematerialisation (Kasulaitis et al., 2015). Our results for the footprint of laptops' manufacturing and the relative contribution of PCBs are consistent with other authors (Table C29 – André et al., 2019; Liu et al., 2016; O'Connell and Stutz, 2010; Teehan and Kandlikar, 2013). Moreover, the LCI's geographical scope is 'global market', according to the ecoinvent nomenclature. As most of the footprint is related to the energy requirements globally, the impact results could decrease with the use of renewable energy in the manufacturing of laptops and components.

Similarly, the LCI's temporal scope does not include uncertainty due to changes in technological development. Future development of the case study could be about studying the effects of energy use and technology improvement. Time is also relevant for the environmental impact results – LCIA. Our methodology (Figure 4.4 and Figure 4.5) graphically shows the emissions in a specific time occurrence. However, we did not account for temporal aspects of the characterisation factors of environmental impacts, e.g. 100a or 500a climate change. For example, in the case of climate change 100a, we show results as if the emissions had occurred at the same moment. The temporal aspects could be improved in our methodology by calculating the specific characterisation factors for the year 0, year 25, and the interpolation between 0-25a.

Another point of discussion is the allocation approach to distributing the impacts of manufacturing among different materials. The manufacturing process is not related to only one material but to an assemblage of different materials that will constitute components and products providing different functions. However, to assess the materials individually, we proposed a simplification approach to distribute the impacts. In this regard, impacts were distributed according to the physical (mass) allocation recommended by ISO 14040/14044 in a multifunctional process that cannot be subdivided. The FU of the study is '1 kg of primary raw material dedicated to the first product application (laptops computers) and its conservation in similar applications for the time horizon of 25 years.' Hence, the allocated impacts of the manufacturing process were divided by the mass of material embedded in the laptops to provide the results per kg of material (Eq. (4.1)). However, as the allocation factor is also based on mass, the manufacturing (and other product-related processes, such as refurbishing or reuse) is the same for 1 kg of material and 1 kg of product. This could be questioned as materials are not valued socioeconomically by weight. Hence, allocation factors considering other characteristics (e.g., exergy or cost) could be more appropriate, but those factors still need to be developed.

Similar reasoning is valid for the 1:1 substitution of avoided materials production at the year 25. The societal and economic benefits of having materials in use are clear. However, although primary extraction may decrease with a better in-use occupation of materials, this may not always be the case. The 1:1 substituting assumption was criticised by Zink et al. (2018) because this substitution is market-driven and not based on the mass or quality of materials. In our case, avoided impacts were not as relevant as other impacts (e.g. manufacturing), so the footprint results would not be much affected. Avoided impacts could become more relevant in the future because of the quality decrease of the natural reserves. Our methodology could be improved with a substitution based on quality and market uptake factors, as proposed by Civancik-Uslu et al. (2021) for plastics.

Pathways for further research could be developing a product-specific indicator, which could be useful for industry in promoting products that have a more intensive in-use occupation. In this regard, in-use occupation could be explored as an LCIA method. Such a method could consider elementary flows of occupation in the function of the elements in materials used for a specific time (kg × year) and material transformation and restoration, taking, for example, the already established framework for land occupation. Van Oers et al. (2020) recently proposed a new LCIA method that couples the inaccessibility of materials (as environmental dissipation) with the traditional Abiotic Depletion Potential method. However, the method does not include other causes for inaccessibility, such as in-use occupation, because of the difficulty to operationalise a characterisation model that estimates the impact associated with future use of resources (van Oers et al., 2020). This type of LCIA method focuses on reducing negative impacts. We argue that the in-use occupation of materials also generates a benefit to the intended user of the products (as the classification proposed by Alvarenga et al. (2020)). For this reason, we considered in-use occupation as a proxy for the handprint provided by the use of materials. This benefit perspective does not need a reference for the future use of resources – the benefit will be higher with less dissipation and hibernation. Hence, assessing in-use occupation as a positive impact could be a way forward to operationalise an LCIA method, which could be used to assess products.

Additionally, the resource efficiency indicators of in-use occupation could be coupled with methodologies measuring the flow of materials to different products at a certain point in time. In this sense, the approach provided by the method MaTrace (Nakamura et al., 2014) and further explored with steel (Pauliuk et al., 2017) and cobalt (Godoy León et al., 2020) could be coupled with the method provided in this chapter, for in-use occupation and resource efficiency. Particularly important would be to expand the analysis of the occupation of critical raw materials in EEE.

4.5.1 Sensitivity analysis

All data compiled in this dissertation is based on literature. For the parameters of mass dissipation and time of in-use, hibernation, and supply, fixed values are used along the TH. Possibly, material production and recovery processes may be more efficient, and new technologies may be developed over time. However, these possible changes in efficiency were not considered, which evidently brings uncertainty to the presented results.

We assessed this uncertainty through sensitivity analysis. Considering UOR, a variation of ±10% was considered for the parameters dissipation and duration of in-use, hibernation, and supply. Results show two different trends – one for metals and one for plastics (Figure 4.8). Upon the dissipation changes in the scenarios of recycling, refurbishing, and reuse, the UOR variation was smaller (1–4%) for metals than for plastics (4–7%). Similarly, the variation of the in-use occupation resulted in a UOR variation of 2–3% for metals and 6–8% for plastics. The difference in these trends can be explained by the dissipation patterns between metals and plastics that happen in dismantling and secondary material processing. While metals presented a steady dissipation along time in the scenarios above, plastics presented much higher dissipation patterns in cycles after the first or second use. Ascertain the discussion in chapter 3, UOR is more sensitive to changes in the earlier cycles because of the higher mass quantity in those cycles.

Figure 4.8: Relative sensitivity of UOR results in scenarios S1–S4 with individual variation of ±10% of the parameters dissipation and time of in-use, hibernation, and supply

On the contrary of dissipation and in-use, the changes in hibernation show a UOR variation that is higher (2–3%) for metals than for plastics (0–1%). The changes in hibernation seem only to affect the last cycle of the in-use phase, which starts before and ends after the TH in the scenarios of recycling, refurbishing, and reuse. Again, the higher the mass of material, the higher the in-use occupation. The mass of metals is much higher than the mass of plastics in the last product cycle. Hence, this explains the smaller changes in UOR of plastics upon variation of hibernation. Moreover, for all scenarios and materials, changes in the supply phase duration did not present an important variation of the UOR results in the analysed scenarios. This is because the supply duration was considered to be a small share of the laptops' in-use or hibernation phase. Nonetheless, UOR results might become sensitive for the supply phase of short-lived products (such as packaging).

The higher sensitivity of UOR of plastics may be demonstrated by comparing the results from chapter 3 and 4. Chapter 3 analysed two product cycles in two of the scenarios (recycling and reuse). Chapter 4 analysed several product cycles (3–4 cycles) in three scenarios (recycling, refurbishing, and reuse). Logically, UOR results should be higher for the scenarios recycling and reuse in chapter 4. However, the reuse scenario is different for plastics: UOR was equal to 48% in chapter 3 and 46% in chapter 4. This can be explained by the data used for the in-use phase duration of the laptops. We used data from survey-based research about service lifetime and storage time of EEE (Thiébaud et al., 2018). Their research showed two values for service lifetime; one is about the interviewees' current devices and one about past devices. In the case of current devices, they added up the time that the devices were in use with interviewees' intention of future use of this device. In the case of past devices, they asked for how long the interviewees' used previous devices. For new laptops, the in-use duration is 6.5 year for current devices and 5.3 years for past devices. For reused laptops, the in-use duration is 5.6 years for current devices and 2.8 years for past devices. In chapter 3, we used the data of current devices under the assumption that it would be less subject to temporal bias. But later, we reasoned that the current-devices data might have been overestimated because of interviewees' positive bias (future use intention) while also being subject to temporal bias (beginning of use). Therefore, in chapter 4, we used the past-devices data. In the case of plastics, the combination of the higher dissipation and higher sensitivity of UOR with changes of in-use duration may explain the unexpected pattern in the results from chapter 3 and 4.

Furthermore, we performed a sensitivity analysis of FRS in the analysed scenarios (Figure 4.9). Results considering changes in dissipation indicate a smaller variation (3–7%) for metals than for plastics (16–42%). The highest variations may be explained by the decreasing dissipation in the recycling supply chain (because of the progressively decreasing mass in each cycle – explained in subsection 4.4.1). Hence, the FRS of scenarios with more recycling cycles showed the highest sensitivity for changes in dissipation – recycling, followed by reuse, and refurbishing. Although the high relative variations for plastics, the absolute FRS result seems never to be higher than 30% in all scenarios. Moreover, the FRS of analysed scenarios is not sensitive to any variation of the parameters supply, in-use, and hibernation time (as these parameters are not included in the indicator). Nonetheless, important to notice is the variation of time parameters in relation to TH. FRS results could be different if the variation in time modifies which product cycle is crossing the TH line.

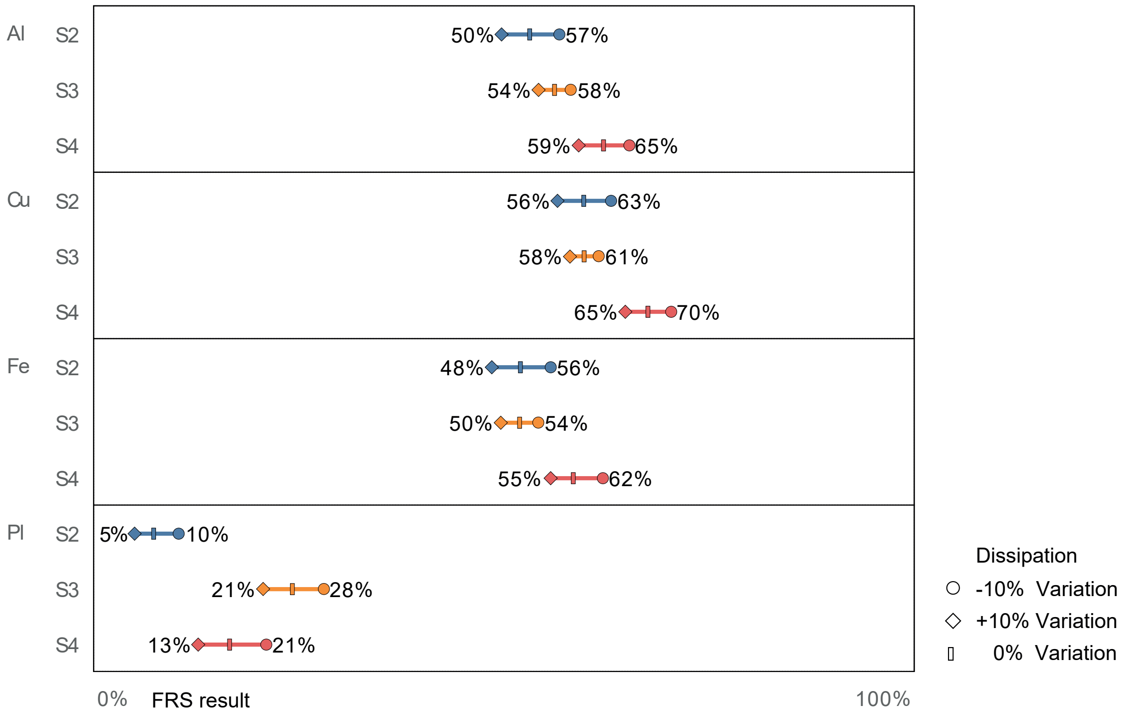


Figure 4.9: Absolute sensitivity of FRS results in scenarios S2-S4with a variation of ±10% dissipation. Higher values are related to dissipation decrease, while lower values are related to dissipation increase.

4.6 Conclusion

In a CE, materials should be kept functional for as long as possible and, in this way, minimising waste and environmental impacts. In this chapter, we have further developed the concept of in-use occupation as a handprint (i.e. materials are functional in society) by quantifying the environmental footprint caused by using materials. In this sense, we developed resource efficiency indicators that show the handprint and footprint for the in-use occupation and final retention of materials in society. Our methodology introduces resource efficiency indicators to assess and compare CE strategies that are difficult to compare (e.g. reuse of products and recycling or materials). We illustrated the indicators with four materials (aluminium, copper, iron, and plastics) used in the production of laptops over a 25-year time horizon. From the illustration, the highest resource efficiency of the in-use occupation was found for refurbishing scenarios of aluminium – an improvement of 189% for carbon-emission resource efficiency and 174% for natural resource efficiency in relation to energy recovery (baseline). Nonetheless, the reuse of laptops showed a carbon-emission resource efficiency improvement of the in-use occupation as high as 157% for aluminium in relation to the baseline. Overall, scenarios with cycles of refurbishment are preferable for most materials considering their resource efficiency of the in-use occupation and final retention in society. This result is because large shares of the impacts are from the laptops manufacturing, and refurbishment was the strategy that along the time horizon kept materials for a longer period, which delayed recycling and the manufacturing of new laptops. Our methodology expands LCA's traditional single-cycle perspective by measuring the cascaded use of materials during 25 years. This is particularly relevant for a CE, where the value of materials should be kept for as long as possible; hence, we should avoid analysing materials or products over only one or two cycles but over longer periods.

The methodology in this chapter can have two potential users. Firstly, it can be used in policy-making to analyse scenarios considering the promotion of different CE strategies or technologies to keep materials in use with a lower footprint. Secondly, in a research context, the methodology advances in at least two issues usually related to the LCA considering the assessment of materials in products. Firstly, LCA is often criticised for disregarding time constraints by considering that the emissions would occur not simultaneously but at different moments of the product's lifecycle. Our methodology graphically shows the emissions in specific time occurrence. Although we showed the footprint in cumulative sections along the life cycle (e.g. raw material production includes mining, which occurred previously), the information could be as disaggregated as needed for a particular LCA purpose. In this way, our methodology improves the communication about emissions' occurrence. Secondly, the methodology considered different cycles of products but avoided the allocation of the impacts among products. Our methodology proposes the analysis of materials not per product cycle but over a time horizon. In this way, we avoid the impact allocation problem in post-consumer activities (such as recycling, refurbishing, and reuse) between the previous and future product cycles. Moreover, we introduced carbon-emission and natural resource efficiency indicators capable of measuring multiple CE strategies that are not easily comparable, such as reusing products vs recycling materials.

5 Further analysis, perspectives, and concluding remarks

This chapter is structured in three sections. In section 5.1, we jointly analyse chapters 3–4 in two subsections, one about our results in terms of proposed indicators (5.1.1) and one about the results from the case studies (5.1.2). In section 5.2, we discuss the future development of the indicators and case studies in three subsections. In the first two subsections, we give a methodological panorama for the integration of the indicators in LCA (5.2.1) and policy-making (5.2.2); in the last subsection (5.2.3), we discuss the limitations of the case studies and modelling choices. Lastly, we present concluding remarks in section 5.3.

5.1 Further analysis of proposed indicators and their results

5.1.1 Proposed indicators vs the classification framework

Circular economy (CE) is not a steady-state business. In chapter 1, we labelled CE as a policy on the go. This is because CE feeds on several schools of thought, so it is difficult to find agreement about what it entails. Also, as society and technology advance, CE policies can be improved and complemented. Ground-breaking technology or disruptive business models can help to advance a CE. In this regard, it is difficult to find a systematically accepted definition of CE. Moreover, it is difficult, if not impossible, to say that a particular indicator is better than another in the CE context. As there is no agreement of what a CE entails, an agreement about 'better' indicators is challenging to find.

On the one hand, this lack of agreement is a disadvantage for the development of indicators – indicators for what exactly? On the other hand, CE as a policy on the go opens up possibilities for innovation. In chapter 2 of this doctoral dissertation, we took this disadvantage as a challenge to understand what CE indicators measure and how they do so without using a specific definition of a CE. We reviewed existing literature and indicators to establish a framework to classify indicators. CE is such a diverse area that many indicators may be needed. It was clear from our framework that no indicator could measure the different levels or strategies of a CE.

As proposed in the categorisation framework, indicators measuring the different strategies may be applied to functions, products, components, materials, and embedded energy (for energy recovery). No indicator was found for the assessment of function strategies,[6] and fewer indicators were found for the assessment of products/ component strategies than for material strategies. However, there is an interconnection among all these categories: materials are used in components that are part of products that provide functions. On the one hand, product-specific indicators can include the assessment of materials (e.g. Material Circularity Indicator (EMF, 2015c)). On the other hand, material-specific indicators could also assess component, products, and functions. The relevance of the indicators will rely on their implementation scale – for example, business, city, region, country, or world.

As proposed in the categorisation framework, different CE measurement scopes can be assessed with indicators. Higher scopes include life cycle thinking for either technological cycles (scope 1) or their cause-and-effect chain (scope 2) with respect to, for example, the environment. Within our definition, life cycle thinking includes the assessment of at least two – but often more – steps in a product chain (e.g. manufacturing and use, or recycling and manufacturing). These steps are implicitly or

[6] Considering that the research in chapter 2 was mainly conducted in 2018, more recent indicators that specifically tackle the preservation of functions may exist. One possible example is this methodology based on semi-quantitative information to measure the (financial and qualitative) value-in-use of service-based offerings (https://doi.org/10.1016/j.cirp.2019.04.084).

explicitly a time progression as they happen sequentially. Some indicators specifically measure time as the beneficial use of products (e.g. Longevity indicator (Figge et al., 2018; Franklin-Johnson et al., 2016)). However, many indicators considering life cycle thinking do not explicitly measure time. Yet, according to many definitions, the CE aims to keep the value of resources for as long as possible. All in all, the lessons from the classification framework shed light on the development of the indicators in chapter 3, In-use Occupation Ratio (UOR) and Final Retention in Society (FRS), and in chapter 4, Resource Efficiency of In-use Occupation (Eff_{Occ}) and Resource Efficiency of Final Retention in Society (Eff_{FRS}). For the purpose of this discussion, we refer to the former couple as handprint indicators and to the last couple as resource efficiency indicators.

To evaluate the proposed indicators of this dissertation, we assessed them against the classification framework (Figure 5.1). In this illustration, the indicators cover a wide range of CE strategies in scopes 1 and 2. We show the indicators in their usual notation for the cases that they were exemplified with case studies. And we show the indicators with notation between parentheses for the cases that they may be used, which were not exemplified. Both cases are discussed in this chapter.

The handprint indicators were tested with case studies of the use of materials in laptops and wood products (chapter 3 and 4). In chapter 3, the assessment included two whole life cycle of products and included strategies related to recycling (and cascading) of materials and reusing of products. Whereas in chapter 4, we analysed four materials in laptops for several cycles of products, with strategies that included reuse, refurbishment, recycling, and energy recovery. In the framework, we positioned the handprint indicators assessing CE strategies at product and material levels. Similarly, the proposed resource efficiency indicators were positioned as capable of measuring strategies at materials and product levels. But, differently, the resource efficiency indicators are capable of accounting for the effects of the energy recovery of materials, which is not the case for the handprint indicators. Any benefit from material-to-energy strategies causes a dissipative use of the materials, which is not captured by in-use occupation. On the other hand, the resource efficiency indicators can capture the effects of recovering energy with an avoided burden approach.

	WHAT DO INDICATORS MEASURE? **CE Strategies**	HOW DO INDICATORS MEASURE? **Measurement scopes** — Technological cycles with physical properties: Scope 0 Technological cycles without aspects of Life Cycle Thinking	Scope 1 Technological cycles with aspects of Life Cycle Thinking	Cause-and-effect modelling from Technological cycles: Scope 2 Cause-and-effect modelling with/without aspects of Life Cycle Thinking
Preservation strategies	1 Function e.g. refuse, rethink, reduce		(UOR) (FRS)	
	2 Product e.g. reuse, refurbish, remanufacture		UOR FRS	Eff_{Occ} Eff_{FRS}
	3 Component e.g. reuse, repurpose		(UOR) (FRS)	(Eff_{Occ}) (Eff_{FRS})
	4 Material e.g. recycle, downcycle		UOR FRS	Eff_{Occ} Eff_{FRS}
	5 Embodied Energy e.g. energy recovery, landfilling with energy recovery			Eff_{Occ} Eff_{FRS}
Linear	6 Reference e.g. waste generation, landfilling without energy recovery			

Figure 5.1: Proposed circularity indicators (chapters 3 and 4) sorted according to the classification framework for CE indicators (chapter 2). Indicators between parentheses were not tested with case studies, and its application needs further investigation. Proposed indicators: In-use Occupation Ratio (UOR), Final Retention in Society (FRS), Resource Efficiency of In-use Occupation (EffOcc), Resource Efficiency of Final Retention in Society (EffFRS)

The assessment of functions was not tested with either handprint or resource efficiency indicators. However, we reason that the handprint indicators could be used at least with the strategies related to some types of product-service systems (PSS). PSS are business instruments developed for a more resource-efficient and circular economy; they are a mix of product and service offering capable of fulfilling customer needs (Tukker, 2015). Generally, PSS are described in three types: product-oriented, use-oriented, and result-oriented services (Tukker, 2015). The first type is related to post-sale services, such as insurance or maintenance contracts; the product ownership is with the user. The use-oriented type is related to businesses (or parties) that share the use of a product with a number of users (e.g., leasing, sharing, or pooling); the product ownership is with the business (or party) provider. Lastly, result-oriented services are an agreement between provider and consumer on the delivery of a result (e.g., catering services or pay-per-copy printing); the ownership is not with the user and a product may not be needed.

The type of PPS that could be assessed with the handprint indicators are use-oriented services. This type deals with more efficient use of products to fulfil functions. Take, for example, transportation. Private cars can take a person from point A to point B, but in terms of their time in use, cars are not efficient. It is estimated that private cars spend around 95% of the time parked, therefore, not in use (Shoup, 2018). One of the proposed solutions for the more effective use of cars is car sharing. The idea behind car sharing is that idle time can be reduced while (possibly) preventing the users from buying new cars. If the idle time information is taken into account for the calculation of the in-use occupation, we could further define that in-use occupation has two subparts: effective in-use occupation and idle in-use occupation (Figure 5.2). These subparts could be factored in the in-use occupation calculation to indicate the benefits of more intensive use of materials. We understand that this could be measured in scope 1 – materials in a life cycle thinking approach. However, there may be trade-off consequences with the change of using private cars to using sharing cars (Chapman et al., 2020). Some of these consequences, such as environmental effects, could be captured with indicators that are positioned on the framework's scope 2. However, we do not consider that the proposed resource efficiency indicators can be helpful to measure these effects in the case of strategies to preserve functions. This is because these function-related strategies can cause important changes during the use phase. But in the current proposal of our resource efficiency indicators, adverse environmental effects during the use phase are not captured. Therefore, functions could be assessed in scope 1 with the handprint indicators but not in scope 2 with neither proposed indicators.

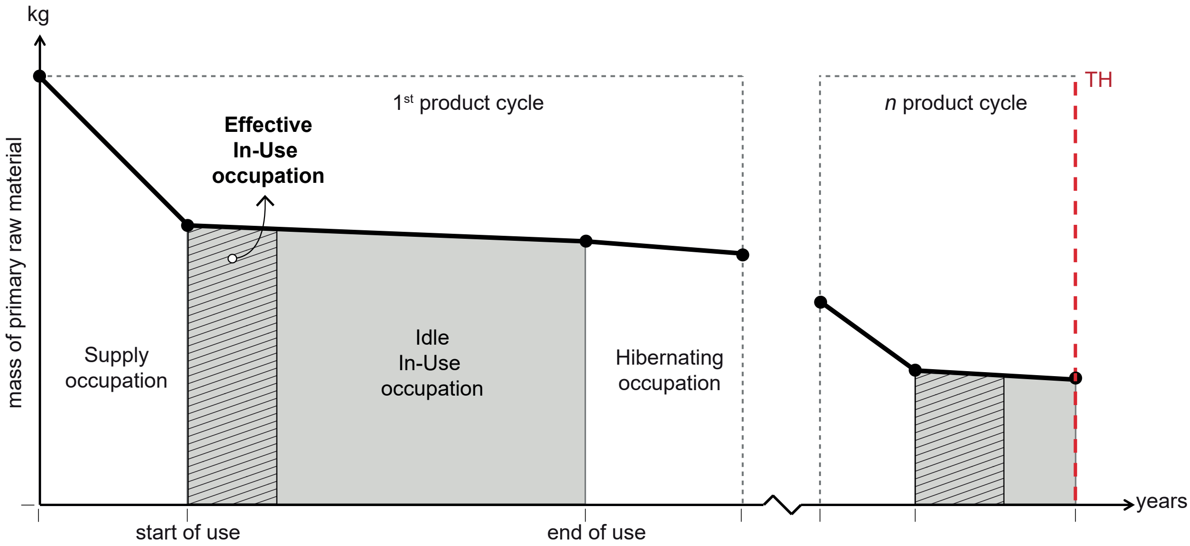


Figure 5.2: Illustration of the occupation chart with the distinction of effective and idle in-use occupation

The group of CE strategies to preserve components were also not analysed with case studies. These strategies include, for example, the reuse of existing parts in new or existing products, such as remanufacturing. This analysis could be based on the example of laptops refurbishing in chapter 4. In this case, three CE strategies were happening simultaneously. A share of 70% of the laptops was being refurbished, the remaining share of 30% was dismantled to materials recycling, and a (material-dependent) share was sent to incineration with energy and ash recovery. The non-dissipated share of materials from recycling and ash recovery was used in the production of new products. Similarly, a strategy to preserve components could be assessed. A more careful dismantling of used laptops, for example, could distinguish components that are reusable from the ones that should go to recycling. Reusable components could be then used in different products with specific in-use time. Hence, we argue that handprint and resource efficiency indicators could be used with strategies assessing components.

A problem with the assessment of products with different lifetimes is that the proposed visualisation of the materials occupation (area charts in chapter 3 and 4) would be complex. In the presented case studies, for simplification of the examples, the assessed materials were assumed to be used in products of the same kind with similar in-use time. If multiple products were assessed, the number of in-use occupation charts would have to be multiplied by the number of assessed products. We argue that such a procedure would not help with the communication of results. Still, regardless of the visualisation, the calculation of our handprint and resource efficiency indicators could be possible with the assessment of multiple products.

All in all, we proposed a set of CE indicators (chapter 3 and 4). Assessing our indicators with the classification framework (chapter 2) helped us to understand their potentials and limitations. Our indicators can assess several CE strategies including a qualitative perspective of time in the analysis. More than that, the proposed indicators bring consistency in the measurement and comparability of several CE strategies analysed in sequential configurations. Consistency is also found for the measurement scopes 1 and 2 that consider life cycle thinking in the perspective. The indicators however are so far inadequate to measure negative effects during the in-use phase.

5.1.2 A cautionary tale about CE strategies

The results of the handprint indicators in chapter 4 showed that the in-use occupation is similar for recycling, refurbishing, and reuse. In contrast, the UOR of plastics showed a clear preference for reuse or refurbishing over mechanical recycling. With resource efficiency indicators, scenarios delaying recycling are across-the-board preferred. Reuse and refurbishing scenarios were 25–65% more carbon-emissions resource efficient and 30–60% more natural resource efficient than recycling. Nonetheless, the analysis of only handprint indicators showed that all scenarios could improve considerably (e.g. the FRS was < 25% in all scenarios with plastics). Hence, the resource efficiency for these materials can also improve. Another information from the resource efficiency indicators is that refurbishing scenarios performed better on average than reuse scenarios. This was surprising as CE R-ladders usually show another order of preference (a ladder of sorts is shown in Figure 5.3 (a)). To extend the meaning of the results, some explanations are necessary.

Firstly, the definition of either refurbishment or reuse varies. We used the definitions provided in the 10R-ladder by Potting et al. (2017b). Reuse is defined as the use of ‘discarded product which is in good condition and fulfil its original function.’ Refurbishing is defined as ‘restore an old product and bring it up to date.’ In the case of refurbishment, we used data about a company that buys used laptops and runs performance tests on the products and their components before selling. The company sells a share of these laptops with one year warranty, while the other share is sent to dismantling for materials recycling. In the case of reuse, we assumed that no repairs were made and that the laptops were reused by the same or another user. Hence, in our reuse or refurbishment scenarios, there is no change of parts or inclusion of new materials. However, in the refurbishment scenario, many more actors are involved in the supply chain of the laptops than in the case of reuse. Hence, one could assume that reuse should be preferable over refurbishing and especially over recycling. This, of course, can be true, particularly if strategies are analysed individually considering one product cycle.

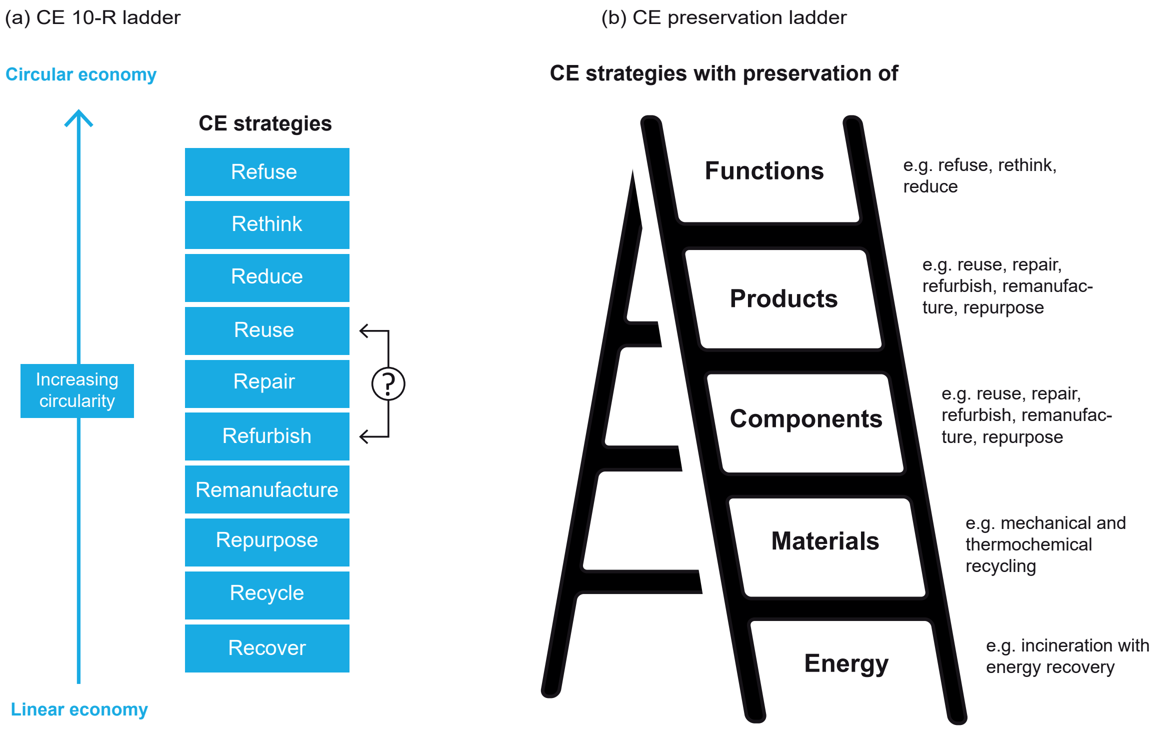


Figure 5.3: (a) CE 10-R ladder (modified from Potting et al. (2017b)). Note that our results do not support the order of the strategies Reuse and Refurbishing. (b) CE preservation ladder based on the results from chapter 2

Nonetheless, in the case studies, we showed CE strategies being used in parallel or in a sequential way. In chapter 3, this was implemented limitedly with the analysis of two product cycles. But in chapter 4, we assessed 4–5 cycles of materials used in products. Interestingly, the handprint indicator UOR showed similar results for recycling and reusing in chapter 3 for materials iron, aluminium, and wood. This was again the case in chapter 4 for the materials aluminium, copper, and iron. On the one hand, reuse having a similar result as recycling can be criticised because of the different resource requirements of both strategies, which will generate consequences to the environment. On the other hand, the UOR result shows that the beneficial use of these materials in the analysed products are comparable because of the shorter in-use time of reused products. Hence, in light of the more beneficial use of materials, the promotion of reusing needs to be coupled with longer use of reused products with shorter periods of hibernation. Moreover, when resource efficiency is considered, all strategies delaying recycling are preferred. It follows that policy should incentivise strategies to make products more durable while stimulating people to use them longer.

On the producers' side, this promotion could be achieved by stimulating businesses that help products use to be extended, such as repairing and refurbishing companies and retiling shops for second use. Potentially, companies that remanufacture their own products could increase the in-use occupation and resource efficiency of materials with, for example, more durable products. Evidently, recycling plays an important role in the CE, but its promotion as a main CE objective can hamper the possible gains with resource efficiency. The promotion of in-use time extension of second-hand products may create less economic value than the incentive for materials recycling. But the incentive to innovation with business models that extend the use of products may develop such value. This is particularly important for products embedding materials known for higher dissipation in the recycling process, as in the case of mechanical recycling of plastics, as we have shown.

On the consumers' side, policy could promote user-behaviour changes towards reusing. One of the main issues with pre-owned products is their perceived (or symbolic) status (Hood, 2016). With EEE, there is rampant consumerism possibly caused by a perceived social status that new devices can bring – a pattern that can be related to exclusivity. Hood (2016) explains that there is a desirability-induced pleasure one derives from using exclusive products, which is unrelated to products' functionality. Therefore, the society economically values newer and more exclusive products, which is problematic in a CE. Evidently, this issue is far beyond the scope of this doctoral dissertation, but policy towards user-behaviour changes of second-hand products needs to take this type of bias into consideration.

The promotion of longer or more intensive in-use of products (and therefore materials) should be promoted but carefully. Our results showed that the order of preference of CE strategies should be taken with a grain of salt. The results from other research lines from the CE Centre seem to point out a similar understanding. For example, the study about car sharing indicates that it can potentially reduce car use but only if users give up on car ownership (Chapman et al., 2020). Another research showed that reuse could stimulate consumers to buy and stock more, which can offset possible environmental impacts (Delanoeije and Bachus, 2020). Moreover, the assessment of CE strategies for energy-using products needs further investigation, mainly because of energy efficiency improvements of newer products. Energy efficiency increase may be a challenge with the promotion of more durable products. In this dissertation, we assessed energy-using products but disregarded the use phase environmental impacts. However, the energy consumption of laptops may have remained constant on average for the last decade (Deram, 2021), which may point out that energy improvements in mature technology are limited unless disruptive changes are implemented. Nonetheless, the impacts of the use phase can affect the order of preference of CE strategies.

Considering CE R-ladders, we argue that the one proposed in Figure 5.3(b) based on our classification framework (chapter 2) is an inclusive type. In our CE-preservation ladder, there is no prescription of strategies but an indication of what the strategies can preserve (i.e. materials, components, products, and functions). It does not give preference for strategies applied to the same subject (e.g. reuse vs refurbish vs remanufacture). However, our ladder indicates a possible preference for the preservation subjects. This follows the reasoning that higher strategies would require shorter loops and thus fewer resources – materials and energy – leading to fewer impacts. Indeed, reuse, remanufacturing, and refurbishing are preferable to recycling in light of thermodynamics (Korhonen et al., 2018a). In the end, while more inclusive, our ladder is not so different from others.

We do not intend to say that our CE-preservation ladder better represents the CE order of strategies. The cautionary tale is not about replacing one ladder with another but about being sceptic if a ladder can help us climb up towards a CE. It might be counterproductive to promote some CE strategies over others when there is no clear evidence that the order in such ladders is generally applicable. Our results suggest that the benefit and footprint of materials need to be studied case by case. And although CE ladders may present the order of preference of strategies that are true for some materials and products, this should not be generalised. The preference of strategies is subject to uncertainty and may not take into account unintended consequences. This may be particularly relevant for assessing CE strategies that are closer in such ladders – namely, reuse and refurbish, as we analysed.

5.2 Perspectives

Based on the results from this dissertation, there are challenges related to the further development of the proposed CE indicators and the shown case studies. In the next subsections, these challenges are discussed.

5.2.1 Materials in-use occupation and inaccessibility in LCA

The inaccessibility of materials was described in chapters 1, 3, and 4. Inaccessibility is caused by human actions that limit the use of materials for a period. Generally, the causes for inaccessibility are distinguished into three types: dissipation to the environment, hibernation, and in-use occupation (van Oers et al., 2020). Thus, we could describe the inaccessibility of mineral resources in Eq. (5.1) [kg × year].

$$\begin{matrix} Inaccessible \\ to\ use \end{matrix} = \begin{matrix} dissipation\ to \\ the\ environment \end{matrix} + \begin{matrix} hibernation\ in \\ the\ technosphere \end{matrix} + \begin{matrix} in-use \\ occupation \end{matrix} \tag{5.1}$$

The in-use occupation was widely discussed in the dissertation, but the other terms not so much. Dewulf et al. (2021) extensively discuss these terms and propose distinctions that include their duration and (ir)reversibility. Dissipation (or dispersion) to the environment is one action defined as the emission of diluted stocks from the technosphere to the environment. Hibernation, on the other hand, is composed of five actions: dispersion in the technosphere, landfilling, tailing, abandoning, and hoarding. Dispersion in the technosphere is the emission of materials to the point that they become economically or technologically inaccessible (e.g. trump elements in successive metal recycling). Dissipation (or dispersion) to the environment and to the technosphere are the least reversible actions of inaccessibility – they can last 500a to infinite (Dewulf et al., 2021). Other less reversible actions are abandoning (e.g. abandoned infrastructure), landfilling (i.e. disposal of industrial or households materials), and tailing (i.e. disposal of mining and metallurgical materials) – these actions can last 25–500a (Dewulf et al., 2021). The most reversible action is hoarding[7], which can last 0–5a (Dewulf et al.,

[7] The term hibernation was used as a synonym for hoarding in chapters 3 and 4, but here they are distinguished to deepen the discussion.

2021). Maintaining the time horizon (TH) used in this dissertation at 25a, we can extend Eq. (5.1) into Eq. (5.2) [kg × year].

$$\begin{aligned} \begin{matrix} \mathit{Inaccessible} \\ \mathit{to\ use} \end{matrix} = & \left(\begin{matrix} \mathit{stocks\ dissipated} \\ \mathit{in\ the\ environment} \end{matrix} + \begin{matrix} \mathit{stocks\ dissipated} \\ \mathit{in\ the\ technosphere} \end{matrix} + \begin{matrix} \mathit{stocks} \\ \mathit{abandoned} \end{matrix} \right. \\ & + \left. \begin{matrix} \mathit{stocks} \\ \mathit{landfilled} \end{matrix} + \begin{matrix} \mathit{tailing} \\ \mathit{stocks} \end{matrix} \right) \times TH + \begin{matrix} \mathit{hoarded} \\ \mathit{stocks} \end{matrix} \times \Delta t_h \\ & + \begin{matrix} \mathit{in\text{-}use} \\ \mathit{stocks} \end{matrix} \times \Delta t_u \end{aligned} \tag{5.2}$$

Note that for the less reversible actions, the stocks are multiplied by the TH. In the case of more reversible actions, the stock is multiplied by their specific time, that is, duration of the in-use phase (Δt_u) and hoarding phase (Δt_h).

Eq. (5.2) could describe the inaccessible use of materials as a negative effect of the use of materials, that is, a footprint. As discussed in chapter 4, in-use occupation can have double-edged effects. While in-use occupation can be a cause for materials inaccessibility, it is also the reason why materials are extracted in the first place. Indeed, Dewulf et al. (2021) proposed a conceptually similar equation as described by Eq. (5.2), but omitting the in-use occupation for the reason that its cause for inaccessibility can be questioned. In this dissertation, we took a similar position of not defining in-use occupation as inaccessibility as proposed in the indicators in chapters 3 and 4. Nonetheless, based on the framework proposed by Alvarenga et al. (2020), in-use occupation could be positioned as a proxy for either handprint or footprint. This is because a product life cycle can cause beneficial and adverse impacts to both intended and unintended users (Alvarenga et al., 2020). Hence, in-use occupation can be described as a proxy for the benefits to the intended user of products, while it can also be a proxy for the negative impacts of inaccessibility caused to unintended users. Thus, based on Eq. (5.2) the resource efficiency for in-use occupation (Eff_u) in chapter 4 could be adapted to Eq. (5.3) [dimensionless].

$$Eff_u = \frac{\mathit{in\text{-}use\ occupation}}{\mathit{inaccessible\ to\ use}} \tag{5.3}$$

Eq. (5.3) shows similarities with the UOR indicator proposed in chapter 3. Both equations have an upper limit, which means a theoretical maximum efficiency of the system. However, here, the denominator is not fixed as in UOR – the amount of material initially used multiplied by TH. In Eq. (5.3), the maximum efficiency is equal to 1, meaning an in-use occupation that does not promote more irreversible causes for inaccessibility of materials. Another difference is that the evaluation of UOR starts with the primary raw material as a commodity, which is a concentrated form of the elements present in metals. For consistency of the system boundary, this was also the choice of other materials illustrated with UOR – wood and plastics.

Eq. (5.3) could arguably be a powerful tool to evaluate resource efficiency with handprint and footprint in the same unit (kg × year). With the resource efficiency indicators proposed in chapter 4, the different units of nominator and denominator and lack of upper limit can be criticised in the sense of efficiency indicators. As pointed out in the introduction chapter, efficiency indicators have their origin in thermodynamics, in which outputs are results of the initial inputs; thus, the denominator is the upper limit while nominator and denominator have both the same unit. However, the operationalisation of the equations above is not straightforward. Although the calculation of dissipation of materials is often done in MFA studies, this is more complex in LCA.

In LCA, the life cycle impact assessment (LCIA) phase is where the life cycle inventory (LCI) is translated into potential environmental impacts. This is done by selecting the LCI's elementary flows related to a specific impact category and characterising these flows to the same unit considering an impact pathway. Elementary flows are 'material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation' (ISO, 2006a). In other words, elementary flows are the exchange of flows between the technosphere and the ecosphere as natural resource extraction (or harvesting, capture, etc.) and anthropogenic emissions. It is because of this exchange of flows that, for example, natural ecosystems or human health are potentially impacted (Bjørn et al., 2017). Elementary flows are usually in the form of elements or substances. For example, emissions of CH_4 can be assigned to global warming and one unit of CH_4 is characterised as 28–34 units of CO_2-eq.[8] In the case of global warming, natural ecosystems and human health can suffer damage; its impact pathway is widely studied and generally consensual.

Thus, the first point of attention to link Eq. (5.3) to LCA is the transition of measuring raw materials, as presented in this dissertation, to measuring elements or substances that can be translated in elementary flows. This is evidently feasible for some abiotic materials by measuring, for example, the elementary content of metals in products. But the assessment of other materials (e.g. plastics and wood as presented in chapter 3 and 4) would require more mature reasoning. For example, how to measure the dissipation of plastics or the cascading of wood? Different materials being (a)biotic and/or (non-)renewable have inherently different dissipation pathways. Hence, the assessment of different materials might require case-specific approaches.

In a CE context, differences of materials are somewhat acknowledged with the so-called butterfly diagram from Ellen MacArthur Foundation (EMF, 2015a), which was presented in chapter 1. This diagram differentiates biological and technical flows in which they defend that the former should be designed to serve as nutrients to biological cycles, while the latter should be designed to stay in the technosphere. This is, of course, overly simplified as a large share of materials contains composites and mixtures of both technical and biological flows. Such composites and mixtures can occur naturally (sedimentary rocks, metal ores, soils, living organisms) or by human intervention (e.g., car components, paint, sewage water, bioenergy residues) (Velenturf et al., 2019). Nonetheless, our choice of assessing materials starting with their commodity form is also a simplification. Much of the dissipation and technosphere hibernation happens before or within the production of commodities. Hence, for the integration of Eq. (5.3) to LCA, our approach needs to be reassessed with the definition of possible elementary flows per materials type.

Still, a starting point in this LCA integration could be with mineral resources. Mineral resources are natural or anthropogenic stocks of chemical elements (e.g., copper), minerals (e.g., gypsum), and aggregates (e.g., sand) (UNEP/LCI, 2019). Based on the SUPRIM project[9], mineral resources hold a potential value for humans in the

[8] According to the IPCC Fifth Assessment Report's Global Warming Potentials with 100 year time horizon – https://www.ipcc.ch/site/assets/uploads/2018/02/WG1AR5_Chapter08_FINAL.pdf

[9] SUPRIM – Sustainable Management of Primary Raw Materials through a better approach in Life Cycle Sustainability Assessment – https://eitrawmaterials.eu/project/suprim/

technosphere, and its damage is quantified as the reduction of this value by human actions (UNEP/LCI, 2019). With this definition, revisions of existing LCIA methods characterising mineral resources were made (Sonderegger et al., 2020) to provide recommendations for future development efforts in LCA (Berger et al., 2020). The quantification of the inaccessibility of resources is one such effort. However, considering Eq. (5.1), only dissipation to the environment is (partially) quantified in LCI databases. Indeed, this type of dissipation can be reported as an elementary flow and therefore classified and characterised with LCIA methods. One such method is the Environmental Dissipation proposed by van Oers et al. (2020), which classifies and characterises dissipative emissions per functional unit. However, this method depends on the inclusion of dissipative elementary flows in LCI data. This is an important issue as LCI databases are far from complete when reporting emissions, and the reported mass that comes in seldom match the mass that comes out (Charpentier Poncelet et al., 2019; van Oers et al., 2020).

Another issue for LCIA methods characterising dissipation based on emissions is that LCI databases do not distinguish between dissipative and non-dissipative emissions. Only emissions of resources that 'have held a function and/or an instrumental value along the system life cycle' can be dissipated (Beylot et al., 2020). However, this information is currently not present in LCI databases (Beylot et al., 2020; Charpentier Poncelet et al., 2021). An example is the dissipation of limestone in Portland cement production. The main binder in Portland cement (clinker) is produced by the calcination of limestone and clay at temperatures up to 1400 ºC. The calcination process releases a relevant amount of CO_2 emission. This emission is partially due to fuels combustion used in the calcination and partially due to the decarbonisation reaction of limestone (i.e. $CaCO_3 \rightarrow CaO + CO_2$). Thus, considering that limestone has other usages than clinker production, decarbonisation is arguably an environmental dissipation of limestone. However, LCI data of clinker production does not distinguish between fuel and decarbonisation emissions.[10] Thus, a given LCIA that includes the selection of CO_2 from clinker will give the same characterisation for these emissions regardless of the source.

Therefore, the inclusion of elementary flows that specifically characterise environmental dissipation in LCI is possible but not feasible in the short term (Beylot et al., 2020). To

[10] See, for example, 'clinker production|clinker' in Ecoinvent v3.4 and 'Portland cement (CEM I)' in ELCD v3.2

circumvent this problem, Charpentier Poncelet et al. (2021) proposed a workaround LCIA method that quantifies the dissipation in the LCIA method without the need for a dissipative elementary flow description in LCI. Such a method uses the quantification of metals dissipation from MFA that is then applied to the amount of extracted metals in a product system. Evidently, this approach cannot show detailed dissipation information that will change from different products supply chains, neither can it show technological improvements to avoid environmental dissipation. Nonetheless, it is a functional approach to deal with dissipation in LCA in the short term.

Apart from environmental dissipation, the LCA integration of the other compromising actions for inaccessibility is far more complex. Firstly, it is difficult to describe them as elementary flows. This is because technosphere hibernation and in-use occupation are not exchanges with the environment but exchanges within the technosphere. Yet, some types of hibernation are modelled as product flows in LCI databases, such as flows to landfilling and tailings. Product flows are flows within the technosphere and between product systems (ISO, 2006a). Interestingly, in the past, some LCIA methodologies proposed the inclusion of an elementary flow of sorts to describe waste in LCI[11]. Their calculation depended on elementary flows such as ‘metal waste’ or ‘packaging waste’. However, such flows are currently not reported by major LCA databases, such as ecoinvent, making the LCIA selection and characterisation of flows for hibernation less straightforward.

A way to include in-use occupation and technosphere hibernation in LCA could be in a similar workaround as presented by Charpentier Poncelet et al. (2021) with environmental dissipation. Indeed, in-use stocks are reported in MFA studies; the quantification of the in-use occupation would depend on the description of the products and their in-use time. On the other hand, the quantification of technosphere hibernation would require specific MFA studies that distinguish the types of hibernation. One such study was proposed by Dewulf et al. (2021), where they identify five types of technological hibernation and present a case study for the flows of cobalt used in the EU. Yet, the proposal for an LCIA method that aggregates all causes for inaccessibility for natural resources is still to be made.

[11] See, for instance, CML 1992 (Heijungs et al., 1992) and Eco-indicator 95 (Goedkoop, 1995)

5.2.2 Integration of the proposed indicators into policy-making

The use of indicators to measure a CE or circularity was the main topic of this dissertation. The implementation of a CE demands the use of indicators to measure the progress of specific goals. At the EU level, a monitoring framework was proposed with the EC's CE Action Plan (EC, 2018a). In chapter 2, this monitoring framework was analysed as an example of macro-level indicators with our classification framework. The results showed that most of the direct CE indicators do not take life cycle thinking into consideration (scope 0). Also, they mostly measure strategies related to materials (such as recycling). Apart from indicators, the EC uses other tools such as MFA to measure the use of materials within the EU economy.

Furthermore, EU member states and regions are developing additional CE monitoring systems. One example is the proposal for the Flemish CE monitor (Alaerts et al., 2019). The CE monitor is composed with the aim to fill the gap between micro-level and macro-level indicators. This gap is proposed to be filled with an intermediate (meso) level that monitors societal needs (e.g. mobility, housing, and nutrition). In this sense, MFA could be used to monitor the amount of material needed for the so-called societal needs. Indeed, MFA within the scope of housing and nutrition needs exist in varied geographical levels, such as EU level (Caldeira et al., 2019), country-level (Bergsdal et al., 2007) and city level (Condeixa et al., 2017).

MFA could be integrated into the handprint indicators to consider 1 kg of material that is initially used in a regional economy (e.g. Flanders) and its maintenance in the economy throughout a TH. Although much of the data used to calculate the indicators based on in-use occupation came from MFA studies, there is a clear distinction between the two, the time constraint. MFA is a static visualisation of the past state of the materials in a period (e.g. one year). On the other hand, an extended tool is the dynamic MFA (dMFA), which can assess past and future stocks and flows of materials cycles (Müller et al., 2014). Based on dMFA, the MaTrace model is particularly interesting for the case of metals as it can track their fate across different products and CE strategies (see, e.g. Figure 5.4).

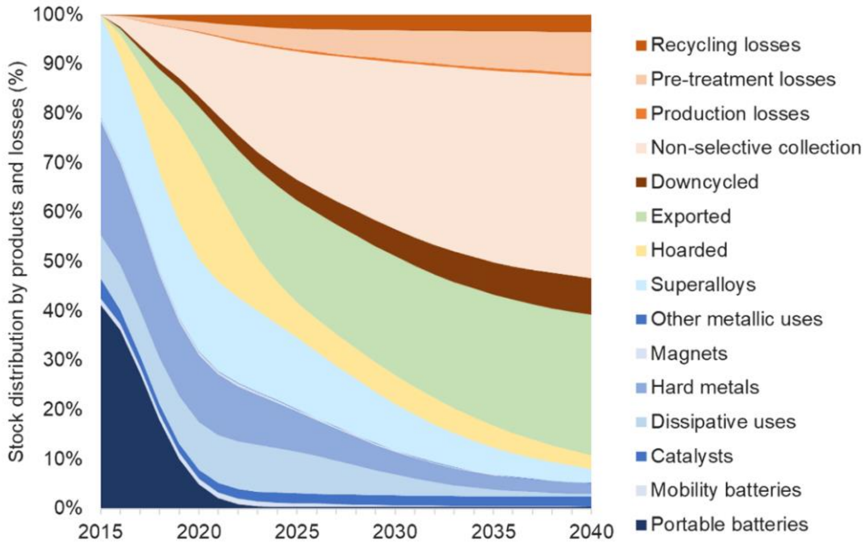


Figure 5.4: Example of the application of MaTrace with the measurement of cobalt used in the EU (Godoy León et al., 2020). Note that the method measures the use of one material in several product applications.

Both in chapters 3 and 4, MaTrace was mentioned to extend the capabilities of the handprint and resource efficiency indicators proposed in this dissertation. Indeed, in the STEN group, the master student Faizan Muneer is working with integrating the handprint indicators and MaTrace (Muneer, 2021). His method proposes the integration of the blue-coloured area in Figure 5.4 to calculate the in-use occupation. A challenge with this integration is how to deal with exports. In the example, exports are not in use in the EU, but they might be in use, dissipated, or hibernating elsewhere. The exports are a good example of the in-use occupation as both a benefit and a cause for inaccessibility in a geographic region. The exports solutions could be done with MaTrace in a global scope (see, e.g. Pauliuk et al., 2017), but if several products are taken into account, the data collection might be a challenging effort.

Concerning the resource efficiency indicators in this dissertation, a further improvement can be the footprint estimation approach. In chapter 3, we used a conventional process-based LCA to estimate the footprint. With this type of LCA, results can be shown in a detailed resolution for each step in the supply chain of materials. However, process-based LCAs are already time demanding to estimate one product's footprint, let alone several products, as presented in Figure 5.4. Another approach would be an input-output LCA (IO-LCA). IO-LCA are based on Input-Output Tables, compiled by statistical authorities as part of national accounts in most industrialised countries (Finnveden et al., 2009). The databases of IO-LCA provide environmental estimations of a sector's product group (e.g. the footprint of one euro amount in copper products). One such database is called Exiobase – they provide LCI of product groups from more than 40 countries. For the case of laptops use, as we presented, the closest proxy data in Exiobase would be 'office machinery and computers' and 'computer and related services.'[12] An analysis with IO-LCA can be a powerful way to estimate the footprint of product's supply chains in reduced time, but it can miss the detailed information about materials management provided from process-based LCAs. Still, the integration of IO-LCA and our resource efficiency indicators could be a path for future research in the context of policy-making.

All in all, the indicators proposed in this thesis can be used with policy to indicate the circularity of materials used in products and their resource efficiency. These indicators can be further adapted with powerful tools such as dMFA, MaTrace, and IO-LCA.

5.2.3 Limitations and further development of case studies

Besides the suggestions for improving the proposed indicators, future research may also be related to the case studies, particularly considering the data and modelling approach.

First, as presented in the sensitivity analysis in chapter 4, although some parameters can result in a high relative variation of UOR and FRS, the change of the absolute results did not modify the order of preference among the analysed scenarios. A limitation of this dissertation is that all data is based on literature, and more research could be done with the collection of primary data for the more sensitive parameters – dissipation (UOR and FRS) and in-use time (UOR). Furthermore, both UOR and FRS are sensitive to the occurrence of the time horizon (TH). In this dissertation, we selected a TH of 25 years based on the results from the SUPRIM project. They also mentioned THs of 5 and 500 years. In a CE context, the longest the TH perspective, the better. However, as were reasoned in chapter 3, a longer TH can bring even more uncertainty to the results. As a future work suggestion, intermediate TH between 5–25 years and 25–500 years could be tested to understand how the handprint and resource efficiency indicators from this dissertation would change.

For the calculation of the footprints used with the resource efficiency indicators, we took the life cycle inventory (LCI) from the ecoinvent database. An important share of the footprint was related to the manufacturing of the laptops, but this dataset in the ecoinvent

[12] Exiobase monetary version 3.4 as compiled by OpenLCA Nexus – https://nexus.openlca.org/

was made with data from a typical computer in 2005. Hence, this data was improved with more recent values about material composition. Nonetheless, the footprint (CEENE and climate change) is not as much related to the materials composition as they are related to energy use for the production of some components, namely integrated circuits on PCBs. As shown in chapter 4, PCBs are presented in almost all laptop components and roughly account for more than 50% of laptops footprint. As shown by Kasulaitis et al. (2015), the quantification of integrated circuits in different PCB types is highly demanding and uncertain. Still, we used the best available data that we had access to for integrated circuits production.

In the LCI adaptation of laptops production, we used data from laptops dismantling by Babbitt et al. (2020). Their data is based on the dismantling of 16 laptops produced in the period 1999–2011. As the focus of this dissertation is about the use of materials, we used the distribution of materials in each component and the distribution of each component in laptops. For the footprint calculations, the uncertainty was propagated with Monte Carlo modelling with random sampling. The mass variation could have been characterised in several ways in the used calculation software (i.e. in OpenLCA: uniform, triangular, and (log)-normal distributions), but we chose triangular distributions. This type of distribution is often used with more uncertain information (Lloyd and Ries, 2007) with the specification of lower, upper, and median values. We chose this distribution for two reasons. Firstly, the laptops evaluated in Babbitt et al. are between 10–21 years old; thus, in temporal scope, the mass information can be classified as more uncertain. Secondly, although their data points out some degree of dematerialisation in more recent laptops, one could not ascertain that this dematerialisation would follow the same trend after 2011. Indeed, minimal dematerialisation at the product level can be seen in more mature products, such as laptops, unless there is a shift to smaller forms (e.g. with smaller screen size) (Kasulaitis et al., 2015). Hence, triangular distribution seemed to be a good fit for this case, as its limits are bounded (differently from (log)-normal distributions), but it indicates a median value towards the dematerialisation of newer laptops (differently from uniform distributions). Nonetheless, a more in-depth study about laptops dematerialisation could increase the relevance of our results considering the TH.

5.3 Concluding remarks

A circular economy can promote the more efficient use of materials, decreasing waste generation. Nevertheless, a circular economy may not necessarily lead to sustainable development. In this context, there is a need for measuring a circular use of materials in light of environmental sustainability. In this doctoral dissertation, we looked at the problem of measuring circularity with a well-known policy approach of resource efficiency indicators. Simply put, resource efficiency indicators are a measure of beneficial effects over negative effects. This dissertation contributes to knowledge by developing an innovative way for measuring the beneficial effects (handprint) with the use of materials considering different circular economy strategies. The handprint indicators In-use Occupation Ratio (UOR) and Final Retention in Society (FRS) measure the use of materials in functional products over time. For the development of the resource efficiency indicators (Eff_{Occ} and Eff_{FRS}), we coupled the UOR and FRS with the environmental footprint of the use of materials. This dissertation is relevant because it shows that material circularity should be analysed in a longer time horizon than one product cycle.

The general objective of this dissertation was to develop circular economy indicators that evaluate the beneficial and adverse-environmental effects of the circular use of materials. The general objective was answered in chapter 4 with the proposal of the resource efficiency indicators (Eff_{Occ} and Eff_{FRS}) that measure the benefit with the functional use of materials based on their in-use occupation and final retention in society. To achieve the general objective, we formulated four intermediate specific objectives. (1) To understand the gaps with existing circular economy indicators – this was addressed in chapter 2 with the classification framework for such indicators. (2) To develop indicators that measure the benefit, or handprint, of the use of materials in functional products following different circular economy strategies – this was answered by developing the indicators UOR and FRS in chapter 3. (3) To measure the negative environmental effects of the use of materials – this was done in chapter 4 through the use of life cycle assessment to quantify materials footprint in terms of climate change and cumulative resource use. (4) To illustrate the indicators use with case studies – this was done in chapter 3 with the measurement of aluminium, iron, precious metals, and plastics in laptops and wood in flooring and furniture; and in chapter 4 with the measurement of aluminium, copper, iron, and plastics in laptops.

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Zink, T., Geyer, R., Startz, R., 2016. A Market-Based Framework for Quantifying Displaced Production from Recycling or Reuse. J. Ind. Ecol. 20, 719–729. https://doi.org/10.1111/jiec.12317

Appendix A

This appendix presents supporting information of chapter 2 – Circular economy indicators: What do they measure?

Table A1: Selected documents based on the literature review

Selected documents
Franklin-Johnson E, Figge F, Canning L. Resource duration as a managerial indicator for Circular Economy performance. J Clean Prod 2016;133:589–98. doi:10.1016/j.jclepro.2016.05.023.
Linder M, Sarasini S, van Loon P. A Metric for Quantifying Product-Level Circularity. J Ind Ecol 2017;21:545–58. doi:10.1111/jiec.12552.
Huysman S, De Schaepmeester J, Ragaert K, Dewulf J, De Meester S. Performance indicators for a circular economy: A case study on post-industrial plastic waste. Resour Conserv Recycl 2017;120:46–54. doi:10.1016/j.resconrec.2017.01.013.
Di Maio F, Rem PC. A Robust Indicator for Promoting Circular Economy through Recycling. J Environ Prot (Irvine, Calif) 2015;06:1095–104. doi:10.4236/jep.2015.610096.
Di Maio F, Rem PC, Baldé K, Polder M. Measuring resource efficiency and circular economy: A market value approach. Resour Conserv Recycl 2017;122:163–71. doi:10.1016/j.resconrec.2017.02.009.
Azevedo S, Godina R, Matias J. Proposal of a Sustainable Circular Index for Manufacturing Companies. Resources 2017;6:63. doi:10.3390/resources6040063.
Scheepens AE, Vogtländer JG, Brezet JC. Two life cycle assessment (LCA) based methods to analyse and design complex (regional) circular economy systems. Case: making water tourism more sustainable. J Clean Prod 2016;114:257–68. doi:10.1016/j.jclepro.2015.05.075.
Pauliuk S. Critical appraisal of the circular economy standard BS 8001:2017 and a dashboard of quantitative system indicators for its implementation in organizations. Resour Conserv Recycl 2018;129:81–92. doi:10.1016/j.resconrec.2017.10.019.
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Vanegas P, Peeters JR, Cattrysse D, Tecchio P, Ardente F, Mathieux F, et al. Ease of disassembly of products to support circular economy strategies. Resour Conserv Recycl 2017. doi:10.1016/J.RESCONREC.2017.06.022.
Adibi N, Lafhaj Z, Yehya M, Payet J. Global Resource Indicator for life cycle impact assessment: Applied in wind turbine case study. J Clean Prod 2017;165:1517–28. doi:10.1016/j.jclepro.2017.07.226.

Table A2: Selected documents based on grey literature and additional scientific literature

Selected documents
EMF - Ellen MacArthur Foundation. Circular Indicators: An approach to measuring circularity. Methodology. vol. 23. 2015b. doi:10.1016/j.giq.2006.04.004.
Graedel TE, Allwood J, Birat J-P, Buchert M, Hagelüken C, Reck BK, et al. What Do We Know About Metal Recycling Rates? J Ind Ecol 2011;15:355–66. doi:10.1111/j.1530-9290.2011.00342.x.
Matsuno Y, Daigo I, Adachi Y. Application of Markov Chain Model to Calculate the Average Number of Times of Use of a Material in Society. An Allocation Methodology for Open loop Recycling. Part 2: Case Study for Steel (6 pp). Int J Life Cycle Assess 2007;12:34–9. doi:10.1065/lca2006.05.246.2.

Table A3: List of micro-scale indicators (and its parameters) assessed. Some indicators are indexed by a main indicator (e.g. MCI = LFI + F(x)). Only the main indicators are presented in chapter 2 (e.g. MCI but not LFI or F(x)). The notations used here are adapted from original reference for clearness.

Indicator's full name	Indicator	Equation	Use	Reference
Material Circularity indicator	MCI	MCI=1 - LFI * F(x)	General	(EMF, 2015b)
Linear Flow Index	LFI	LFI=(Amount of virgin Material + Amount of unrecoverable waste) / 2 * Amount of total mass	-	(EMF, 2015b)
Utility factor	F(x)	F(x)=(Lifetime/Lifetime average) * ("EMF's Functional Unity" (i.e. No. of times product is used to its full capacity) / "EMF's Functional Unity" average)	-	(EMF, 2015b)
Old scrap collection rate	CR	CR=recycling input (i.e. Collection output)/total metal collected	Metals	(Graedel et al., 2011)
Recycling rate	RR	RR=recycling output / recycling input	Designed for metals, but used in general	(Graedel et al., 2011)
EoL recycling rate	EOL-RR	EOL-RR=total metal collected / recycling output (i.e. OLD scrap)	Metals	(Graedel et al., 2011)
Recycling input rate	RIR	RIR=(OLD+NEW scrap) / total demand of metal	Metals	(Graedel et al., 2011)
Old scrap ratio	OSR	OSR=OLD scrap/total secondary (i.e. OLD+NEW)	Metals	(Graedel et al., 2011)
Longevity	Longevity	Longevity = Li + Rf + Rc	General	(Franklin-Johnson et al., 2016)
Lifetime	Li	Li= initial lifetime of the product (e.g. number of months) of new use	-	(Franklin-Johnson et al., 2016)

Refurbished	Rf	Rf = % returned products * % refurbished products * Lifetime of refurbished product	-	(Franklin-Johnson et al., 2016)
Recycled	Rc	Rc= (Product lifetime + Ref) *[(% returned products * % Recycled products * % Unrecovered material from product) / (1 - % returned products * % Recycled products * % Unrecovered material from product)]	-	(Franklin-Johnson et al., 2016)
Product-Level Circularity Metric	PLCM	PLCM = (economic value of recirculated parts / economic value of all parts)	General	(Linder et al., 2017)
Circular economy performance indicator	CPI	CPI = actual benefit / ideal benefit according quality	General, tested with plastics	(Huysman et al., 2017)
Circular Economy Index	CEI	CEI=(Material value recycled from EOL product) / (value of material entering the recycling facility)	General	(Di Maio and Rem, 2015)
Value-based resource efficiency	VRE	VRE = [Gross value of outputs - (Input value of Energy, Materials, Services)] / [(Input value of Energy) + (Input value of Materials)]	General	(Di Maio et al., 2017)
Sustainable Circular Index	SCI	SCI=Weight set of indicators (economical, social, environmental, circularity) Circularity (LFI, F(x), Recycling efficiency)	Company	(Azevedo et al., 2017)
Eco-cost value ratio	EVR	EVR=eco-costs/value (i.e.. Value is the sum of the perceived product- & service- quality, and the image)	General	(Scheepens et al., 2016)
Total Restored Products	TRP	R=C1(Reuse)+C2(Refill or refurbish)+C3(redistribution)+C4(remanufacture)	General	(Pauliuk, 2018)
Lifetime of Materials on Anthropospher e	LMA	LMA=scenario simulation based on Markov chain model	General, tested with metals	(Pauliuk, 2018)
Material Circularity Indicator CIRC	CIRC	CIRC=(actual cumulative service)/(maximal theoretical cumulative service)	General, tested with metals	(Pauliuk, 2018)
Displacement	Displacem ent	market interactions using partial equilibrium analysis	General, for recycled materials	(Zink et al., 2016)
ease of Disassembly Metric	eDiM	eDiM = Σ (Tool change + Identifying+ Manipulation + Positioning + Disconnection + Removing)	General, for complex products	(Vanegas et al., 2018)
Global Resource Indicator	GRI	GRI = Scarcity / (Recyclability + Geopolitical availability)	General	(Adibi et al., 2017)

Average Number of Times of Use of a Material in Society	NTUM	NTUM= Σ (Tu Nsu) (i.e. average residence time in society Tu in each state u, the average residence time of elements of iron in society from initial state s until ultimately being landfilled after an unlimited number of transitions)	General, tested with metals	(Matsuno et al., 2007)

Appendix B

This appendix presents supporting information of chapter 3 – Development of circularity indicators based on the in-use occupation of materials

Inventory of losses in laptops case:

Table B1: Losses in the first-product cycle – materials in laptops

	Fe (%)	Source	Al (%)	Source	PL(%)*	Source	PM(%)**	Source
$l_{S,1}$	15.0%	[BT1.1]	0.0%	[BT1.2]	0.6%	[BT1.3]	0.0%	[BT1.4]
$l_{U,1}$	0.0%	[BT1.5]	0.0%	[BT1.5]	0.0%	[BT1.5]	0.0%	[BT1.5]
$l_{H,1}$	0.0%	[BT1.5]	0.0%	[BT1.5]	0.0%	[BT1.5]	0.0%	[BT1.5]

*Plastics (starting in the polymerization process, %)

**Precious metals (Gold as proxy)

[BT1.1] – Considering the sum of all losses from pig iron to final products (Figure S2 in Cullen, J.M., Allwood, J.M., Bambach, M.D., 2012. Mapping the Global Flow of Steel: From Steelmaking to End-Use Goods. Environ.)

[BT1.2] – Most of the losses happen before the system boundary in the electrolysis processes. Although the material efficiency of aluminium is 60% (manufacturing yield) the remaining material returns to industry in form of scrap. Cullen, J.M., Allwood, J.M., 2013. Mapping the Global Flow of Aluminum: From Liquid Aluminum to End-Use Goods. Environ. Sci. Technol. 47, 3057–3064. https://doi.org/10.1021/es304256s

[BT1.3] – Considering losses in the moulding process. Polymerisation is excluded. Loss reported in Ecoinvent v3.4 process 'injection moulding | injection moulding | Cut-off, U - RER'

[BT1.4] – Assumption of no losses based on Hewitt, A., Keel, T., Tauber, M., Le-Fiedler, T., 2015. The Ups and Downs of Gold Recycling [WWW Document]. Bost. Consult. Gr. URL https://www.bcg.com/en-be/publications/2015/metals-mining-cost-efficiency-ups-and-downs-of-gold-recycling.aspx (accessed 11.19.20).

[BT1.5] – Assumption

Table B 2: Losses in case of recycling – materials in laptops

	Fe (%)	Source	Al (%)	Source	PL(%)*	Source	PM(%)**	Source
$l_{S,1}$	15.0%	[BT2.1]	0.0%	[BT2.2]	0.6%	[BT2.3]	0.0%	[BT2.4]
$l_{U,1}$	0.0%	[BT2.5]	0.0%	[BT2.5]	0.0%	[BT2.5]	0.0%	[BT2.5]
$l_{H,1}$	0.0%	[BT2.5]	0.0%	[BT2.5]	0.0%	[BT2.5]	0.0%	[BT2.5]
$l_{S,2}$	14.0%	[BT2.6]	25.0%	[BT2.6]	87.0%	[BT2.6]	37.0%	[BT2.6]

$I_{U,2}$	0.0%	[BT2.5]	0.0%	[BT2.5]	0.0%	[BT2.5]	0.0%	[BT2.5]
$I_{H,2}$	0.0%	[BT2.5]	0.0%	[BT2.5]	0.0%	[BT2.5]	0.0%	[BT2.5]

*Plastics (starting in the polymerization process, %)

**Precious metals (Gold as proxy)

[BT2.1] – Considering the sum of all losses from pig iron to final products (Figure S2). Cullen, J.M., Allwood, J.M., Bambach, M.D., 2012. Mapping the Global Flow of Steel: From Steelmaking to End-Use Goods. Environ.

[BT2.2] – Most of the losses happen before the system boundary in the electrolysis processes. Although the material efficiency of aluminium is 60% (manufacturing yield) the remaining material returns to industry in form of scrap. Cullen, J.M., Allwood, J.M., 2013. Mapping the Global Flow of Aluminum: From Liquid Aluminum to End-Use Goods. Environ. Sci. Technol. 47, 3057–3064. https://doi.org/10.1021/es304256s

[BT2.3] – Considering losses in the moulding process. Polymerisation is excluded. Loss reported in Ecoinvent v3.4 process 'injection moulding | injection moulding | Cut-off, U - RER'

[BT2.4] – Assumption of no losses based on Hewitt, A., Keel, T., Tauber, M., Le-Fiedler, T., 2015. The Ups and Downs of Gold Recycling [WWW Document]. Bost. Consult. Gr. URL https://www.bcg.com/en-be/publications/2015/metals-mining-cost-efficiency-ups-and-downs-of-gold-recycling.aspx (accessed 11.19.20).

[BT2.5] – Assumption

[BT2.6] – Van Eygen E, De Meester S, Tran HP, Dewulf J. Resource savings by urban mining: The case of desktop and laptop computers in Belgium. Resour Conserv Recycl 2016;107:53–64. doi:10.1016/J.RESCONREC.2015.10.032.

Table B3: Losses in case of reusing – materials in laptops

	Fe (%)	**Source**	**Al (%)**	**Source**	**PL(%)***	**Source**	**PM(%)****	**Source**
$I_{S,1}$	15.0%	[BT3.1]	0.0%	[BT3.2]	0.6%	[BT3.3]	0.0%	[BT3.4]
$I_{U,1}$	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]
$I_{H,1}$	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]
$I_{S,2}$	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]
$I_{U,2}$	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]
$I_{H,2}$	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]	0.0%	[BT3.5]

*Plastics (starting in the polymerization process, %)

**Precious metals (Gold as proxy)

[BT3.1] – Considering the sum of all losses from pig iron to final products (Figure S2). Cullen, J.M., Allwood, J.M., Bambach, M.D., 2012. Mapping the Global Flow of Steel: From Steelmaking to End-Use Goods. Environ.

[BT3.2] – Most of the losses happen before the system boundary in the electrolysis processes. Although the material efficiency of aluminium is 60% (manufacturing yield) the remaining material returns to industry in form of scrap. Cullen, J.M., Allwood, J.M., 2013. Mapping the Global Flow of Aluminum: From Liquid Aluminum to End-Use Goods. Environ. Sci. Technol. 47, 3057–3064. https://doi.org/10.1021/es304256s

[BT3.3] – Considering losses in the moulding process. Polymerisation is excluded. Loss reported in Ecoinvent v3.4 process 'injection moulding | injection moulding | Cut-off, U - RER'

[BT3.4] – Assumption of no losses based on Hewitt, A., Keel, T., Tauber, M., Le-Fiedler, T., 2015. The Ups and Downs of Gold Recycling [WWW Document]. Bost. Consult. Gr. URL https://www.bcg.com/en-

be/publications/2015/metals-mining-cost-efficiency-ups-and-downs-of-gold-recycling.aspx (accessed 11.19.20).

[BT3.5] – Assumption

Allocation procedure of the wood products case:

Data based on: Garcia, R., Alvarenga, R.A.F., Huysveld, S., Dewulf, J., Allacker, K., 2020. Accounting for biogenic carbon and end-of-life allocation in life cycle assessment of multi-output wood cascade systems. J. Clean. Prod. 275, 122795. https://doi.org/10.1016/j.jclepro.2020.122795.

Table B4: Input and Output mass before allocation – wood case

Input	mass (kg)	Output	mass (kg)
Roundwood	399.4	Product 1: Timber wood	199.8
		Product 2: residue to particle board production	43.0
		Loss: dissipation or residue to energy	156.6
		total mass of products (not considering loss)	242.8

Table B5: Input and Output mass after allocation on mass basis: Product 1 – wood case

Input	mass (kg)	Output	mass (kg)
Roundwood	328.7	Product 1: Timber wood	199.8
		Loss: dissipation or residue to energy	128.9

Table B6: Input and Output mass after allocation on mass basis: Product 2 – wood case

Input	mass (kg)	Output	mass (kg)
Roundwood	70.7	Product 2: residue to particle board production	43.0
		Loss: dissipation or residue to energy	27.7

Inventory of losses in wood products case:

Table B7: Losses in the first-product cycle – wood products case

	Wood (%)	Source
$l_{S,1}$	42.2%	[BT7.1]

$I_{U,1}$	0.0%	[BT7.2]
$I_{H,1}$	0.0%	[BT7.2]

[BT7.1] – Considering the allocated losses on the timber production and additional 5% losses for the transformation of timber wood into flooring wood.

[BT7.2] – Assumption

Table B8: Losses in case of particle board production from flooring – wood products case

	Wood (%)	Source
$I_{S,1}$	42.2%	[BT8.1]
$I_{U,1}$	0.0%	[BT8.2]
$I_{H,1}$	0.0%	[BT8.2]
$I_{S,2}$	14.0%	[BT8.3]
$I_{U,2}$	0.0%	[BT8.2]
$I_{H,2}$	0.0%	[BT8.2]

[BT8.1] – Considering the allocated losses on the timber production and additional 5% losses for the transformation of timber wood into flooring wood.

[BT8.2] – Assumption

[BT8.3] – Garcia, R., Alvarenga, R.A.F., Huysveld, S., Dewulf, J., Allacker, K., 2020. Accounting for biogenic carbon and end-of-life allocation in life cycle assessment of multi-output wood cascade systems. J. Clean. Prod. 275, 122795. https://doi.org/10.1016/j.jclepro.2020.122795.

Table B9: Losses in case of reuse and repair – wood products case

	Wood (%)	Source
$I_{S,1}$	42.2%	[BT9.1]
$I_{U,1}$	0.0%	[BT9.2]
$I_{H,1}$	0.0%	[BT9.2]
$I_{S,2}$	5.0%	[BT9.3]
$I_{U,2}$	0.0%	[BT9.2]
$I_{H,2}$	0.0%	[BT9.2]

[BT9.1] – Considering the allocated losses on the timber production and additional 5% losses for the transformation of timber wood into flooring wood.

[BT9.2] – Assumption

[BT9.3] – Losses in the sanding process and changing of damaged pieces. Assumption

Sensitivity analysis materials in laptops case:

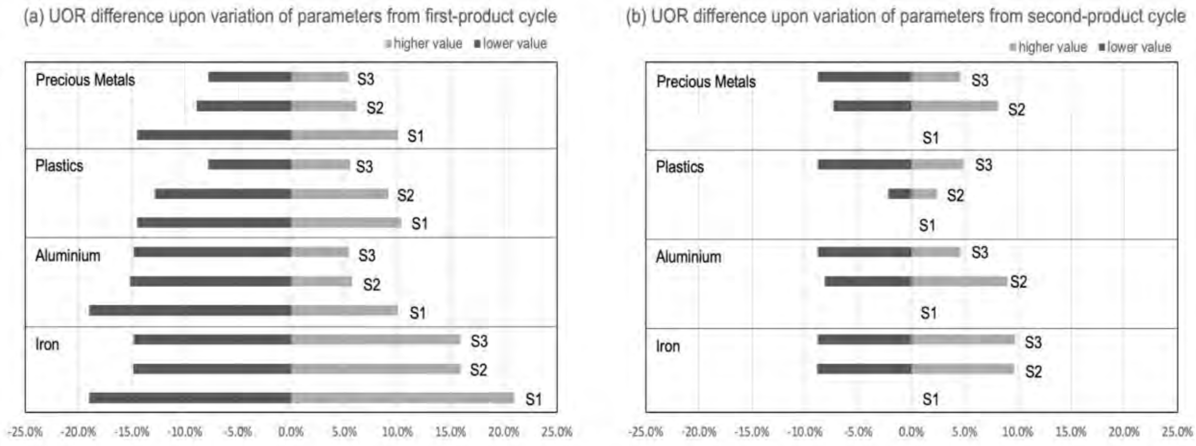

Figure B1: Sensitivity of UOR results upon two-way variation (±10%) of parameters m_{uj} and Δt_{Uj}

Sensitivity analysis materials in wood products case:

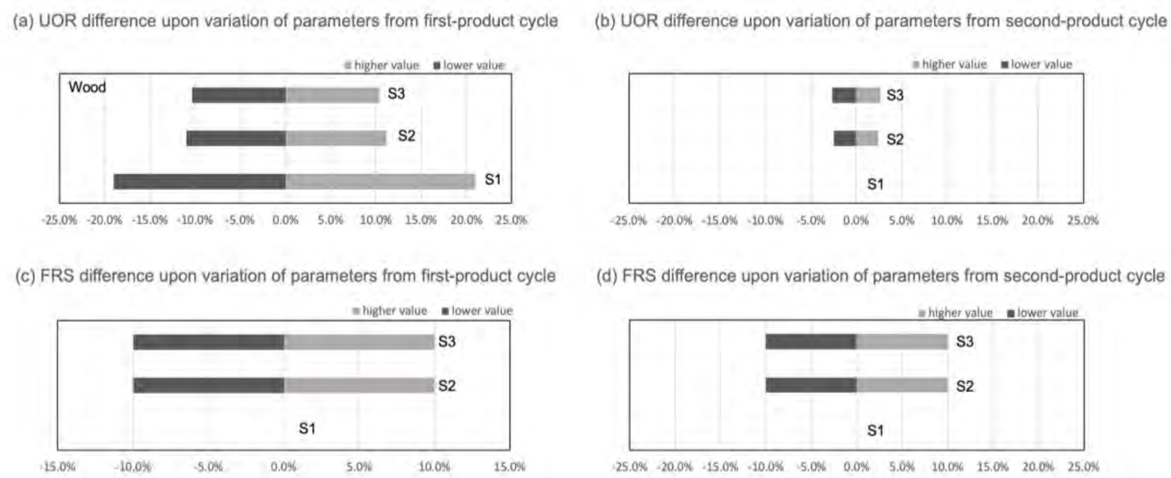

Figure B2: Sensitivity of UOR and FRS results upon two-way variation (±10%) of parameters m_{uj} and Δt_{Uj}

Appendix C

This appendix presents supporting information of chapter 4 – Resource efficiency indicators to assess circular economy strategies: a case study on four materials in laptops

Inventory for in-use occupation

Dissipation of aluminium

Table C1: Dissipation of aluminium in each production step of the system boundaries

Virgin aluminium	**loss**	**Source**
Remelting/refining	1.43%	calculation 'Liquid Al - virgin' based on Cullen and Allwood (2013)
Rolling/forming/casting	0.69%	calculation 'Total to processing' based on Cullen and Allwood (2013)
Manufacturing	0.00%	Based on ecoinvent datasets, remaining materials are processed as primary scrap
Used aluminium	loss	Source
Collection	2.35%	WEEE in municipal waste as percentage of small electronic appliances POM (Deloitte Consulting & Advisory, 2018)
Dismantling, shredding & sorting	12.00%	Calculation based on De Meester et al 2019 (=(1-eficiency of process)-(ash recovery*what goes to incineration); considering that waste is incinerated with ash treatment)
Remelting/refining	4.48%	calculation 'Total scrap (old+new)' based on Cullen and Allwood (2013)
Rolling/forming/casting	0.69%	calculation 'Total to processing' based on ecoinvent
Manufacturing	0.00%	Based on ecoinvent datasets, remaining materials are processed as primary scrap
Incineration	loss	Source
Ash treatment efficiency (inverse)	80%	De Meester et al (2019)

Table C2: Auxiliary calculation table for dissipation of aluminium

process	**amount**	**unit**	**losses**	**Dissipation (amount*losses)**	**Dissipation (total %)**	**Source dissipation**

virgin to remelter	0.9	mil t	0.58%	0.01		Cullen, J.M., Allwood, J.M., 2013. Mapping the Global Flow of Aluminum: From Liquid Aluminum to End-Use Goods. Environ. Sci. Technol. 47, 3057–3064. https://doi.org/10.1021/es304256s
virgin to refiner	7.1	mil t	7.53%	0.53		
virgin to recasting	29.8	mil t	0.00%	0.00		
Liquid Al - virgin	37.8	mil t		0.54	1.43%	
O_scrap to remelter	1.8	mil t	0.58%	0.01		Cullen, J.M., Allwood, J.M., 2013. Mapping the Global Flow of Aluminum: From Liquid Aluminum to End-Use Goods. Environ. Sci. Technol. 47, 3057–3064. https://doi.org/10.1021/es304256s
O_scrap to refiner	6.5	mil t	7.53%	0.49		
Old scrap	8.3	mil t		0.50	6.03%	
N_scrap to remelter	15.5	mil t	0.58%	0.09		Cullen, J.M., Allwood, J.M., 2013. Mapping the Global Flow of Aluminum: From Liquid Aluminum to End-Use Goods. Environ. Sci. Technol. 47, 3057–3064. https://doi.org/10.1021/es304256s
O_scrap to refiner	15.6	mil t	7.53%	1.18		
New scrap	31.1	mil t		1.26	4.07%	
Total scrap (old+new)	39.4	mil t		1.77	4.48%	
						Losses from ecoinvent (Losses = Aluminium input – scrap); amounts from Cullen, J.M., Allwood, J.M., (2013)
Rolling	29.4	mil t	0.30%	0.09		ecoinvent 3.4: sheet rolling, aluminium | sheet rolling, aluminium | Cutoff, U
Extrusion	11.9	mil t	1.20%	0.14		ecoinvent 3.4: section bar extrusion, aluminium | section bar extrusion, aluminium | Cutoff, U
Drawing	5.7	mil t	0.79%	0.04		ecoinvent 3.4: average from different drawing processes
Shape casting	27	mil t	0.86%	0.23		ecoinvent 3.4: casting, aluminium, lost-wax | casting, aluminium, lost-wax | Cutoff, U
Total to processing	74	mil t		0.51	0.69%	

Dissipation of copper

Table C3: Dissipation of copper in each production step of the system boundaries

Virgin copper	**loss**	**Source**
Refining	0.00%	No dissipation is reported in the refining process, based on Soulier et al. (2018).

Semi-finished goods production	0.47%	There was a dissipation of 20 kt for the semi-finishing of 4290 kt of cooper containing products. Calculation based on Soulier et al. (2018).
Manufacturing	0.00%	Based on ecoinvent datasets, remaining materials are processed as primary scrap

Used copper	loss	Source
Collection	2.35%	WEEE in municipal waste as percentage of small electronic appliances POM (Deloitte Consulting & Advisory, 2018)
Dismantling	12.80%	Calculation based on De Meester et al 2019 (=(1-eficiency of process)-(ash recovery*what goes to incineration); considering that waste is incinerated with ash treatment)
Refining (low grade)	0.54%	For the total collected copper scrap in EU28 2014, a share of 42.7% was recovered by smelting and refining. There was a dissipation of 10 kt for the refining of 790 kt. Hence, 42.7%*10/790. Calculation based on Soulier et al. (2018)
Semi-finished goods production	0.47%	There was a dissipation of 20 kt for the semi-finishing of 4290 kt of cooper containing products. Calculation based on Soulier et al. (2018).
Manufacturing	0.00%	Based on ecoinvent datasets, remaining materials are processed as primary scrap.

Incineration	loss	Source
Ash treatment efficiency	80.00%	De Meester et al 2019

Dissipation of iron

Table C4: Dissipation of iron in each production step of the system boundaries

Virgin Iron	**loss**	**Source**
Steelmaking	12.83%	calculation ‘total virgin to steelmaking’ based on Cullen and Allwood (2012)
Rolling/forming	1.25%	calculation ‘to rolling/forming’ based on Cullen and Allwood (2012)
Manufacturing	0.00%	Based on ecoinvent datasets, remaining materials are processed as primary scrap.

Used Iron	loss	Source
Collection	2.35%	WEEE in municipal waste as percentage of small electronic appliances POM (Deloitte Consulting & Advisory, 2018)
Dismantling, shredding & sorting	0.70%	Calculation based on De Meester et al 2019 (=(1-eficiency of process)-(ash recovery*what goes to incineration); considering that waste is incinerated with ash treatment)
Remelting/refining	11.77%	calculation ‘total scrap to steelmaking’ based on Cullen and Allwood (2012)
Rolling/forming/casting	1.25%	calculation ‘to rolling/forming’ based on Cullen and Allwood (2012)
Manufacturing	0.00%	Based on ecoinvent datasets, remaining materials are processed as primary scrap.

Incineration	loss	Source

Ash treatment efficiency (inverse)	70%	De Meester et al 2019

Table C5: Auxiliary calculation table for dissipation for iron

process	amount	unit	losses	Dissipation (amount*losses)	Dissipation (total %)	Source dissipation
to steelmaking_OBC	826.20	mil t	12.91%	106.70		Cullen, J.M., Allwood, J.M., Bambach, M.D., 2012. Mapping the Global Flow of Steel: From Steelmaking to End-Use Goods. Environ. Sci. Technol. 46, 13048–13055. https://doi.org/10.1021/es302433p
to steelmaking_EF	44.60	mil t	11.09%	4.95		
to steelmaking_OHF	34.00	mil t	12.94%	4.40		
total virgin to steelmaking	904.80	mil t		116.04	12.83%	
to scrap preparation	574.40	mil t	0.99%	5.70	0.99%	
to steelmaking_OBC	206.00	mil t	12.91%	26.60		Cullen, J.M., Allwood, J.M., Bambach, M.D., 2012. Mapping the Global Flow of Steel: From Steelmaking to End-Use Goods. Environ. Sci. Technol. 46, 13048–13055. https://doi.org/10.1021/es302433p
to steelmaking_EF	351.00	mil t	11.09%	38.94		
total scrap to steelmaking	557.00	mil t		65.54	11.77%	
to casting	892.00	mil t	0.48%	4.27		Cullen, J.M., Allwood, J.M., Bambach, M.D., 2012. Mapping the Global Flow of Steel: From Steelmaking to End-Use Goods. Environ. Sci. Technol. 46, 13048–13055. https://doi.org/10.1021/es302433p
to primary mill	89.00	mil t	0.98%	0.87		
total Fe to casting	981.00			5.14	0.52%	
to rolling/forming	1221.00	mil t	1.25%	15.32	1.25%	

Dissipation of plastics

Table C6: Dissipation of plastics in each production step of the system boundaries

Virgin plastics	loss	Source
Cracking and polymerisation	0.00%	Losses are not reported by Levi and Cullen (2018)
Moulding	1.01%	Average waste % reported in the ecoinvent processes ‘injection moulding | injection moulding | Cut-off, U – RER’, ‘blow moulding production | blow moulding | Cutoff, U’,

Manufacturing	0.00%	'stretch blow moulding production | stretch blow moulding | Cutoff, U' Based on ecoinvent datasets, remaining materials are processed as primary scrap.

Used plastics	loss	Source
Collection	2.35%	WEEE in municipal waste as percentage of small electronic appliances POM (Deloitte Consulting & Advisory, 2018)
Separation	52.00%	Calculation based on De Meester et al 2019 (=(1-eficiency of process)-(ash recovery*what goes to incineration); considering that waste is incinerated with ash treatment)
Secondary processing efficiency	10.0%	Calculation based on De Meester et al 2019 (=(1-eficiency of process); considering that waste is incinerated)
Moulding	1.01%	Average waste % reported in the ecoinvent processes 'injection moulding | injection moulding | Cut-off, U – RER', 'blow moulding production | blow moulding | Cutoff, U', 'stretch blow moulding production | stretch blow moulding | Cutoff, U'
Manufacturing	0.00%	Based on ecoinvent datasets, remaining materials are processed as primary scrap.

Timespan for all materials

Table C7: Time of supply, in-use, and hibernation for all j product cycles in scenarios 1, 2, and 3

	time (yr)	Source
$\Delta t_{S,j}$	0.1	The time needed to order the main components and manufacture laptops varies from 2-4 days to 24-26 days (lead time), according to Miyajima et al. (2019). We assumed an increased time to include the production of materials and retail activities.
$\Delta t_{U,j}$	5.3	Based on survey from Thiébaud et al (2018), average service lifetime of previously owned new devices ('stored/disposed of' on Table C14).
$\Delta t_{H,j}$	1.9	Based on survey from Thiébaud et al (2018), average storage lifetime of previously owned new devices (Table C16).

Table C8: Time of supply, in-use, and hibernation for all j product cycles in scenarios 4

	time (yr)	Source
$\Delta t_{S,j}$	0.1	The time needed to order the main components and manufacture laptops varies from 2-4 days to 24-26 days (lead time), according to Miyajima et al. (2019). We assumed an increased time to include the production of materials and retail activities.
$\Delta t_{U,j}$	5.3	Based on survey from Thiébaud et al (2018), average service lifetime of previously owned new devices ('stored/disposed of' on Table C14).
$\Delta t_{H,j}$	1.9	Based on survey from Thiébaud et al (2018), average storage lifetime of previously owned new devices (Table C16).

$\Delta t_{U,j+1}$	2.8	Based on survey from Thiébaud et al (2018), average second service lifetime of previously owned used devices (‘stored/disposed of’ on Table C15).
$\Delta t_{H,j+1}$	1.5	Based on survey from Thiébaud et al (2018), average second storage lifetime of previously owned devices (Table C17).

Life cycle inventory analysis (LCI)

Raw materials

Table C9: Primary production of aluminium. Amounts are from the original share for global-region market multiplied by the amount of aluminium in laptops

Input Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
aluminium, primary, ingot	71.19693	g	triangular: min=35.8591 mode=71.1969 max=125.542	market for aluminium, primary, ingot | aluminium, primary, ingot | Cutoff, S - LCI - IAI Area, EU27 & EFTA
aluminium, primary, ingot	340.3112	g	triangular: min=171.401 mode=340.311 max=600.072	market for aluminium, primary, ingot | aluminium, primary, ingot | Cutoff, S - LCI - RoW
Output Flow	Amount	Unit	Uncertainty	Provider
Al in laptop	1	Item(s)	none	

Table C10: Primary production of copper. Amounts are from the original share for global-region market multiplied by the amount of copper in laptops

Input Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
copper	29.57908	g	triangular: min=19.4535 mode=29.5791 max=37.1577	copper production, primary | copper | Cutoff, S - LCI - RAS
copper	14.0831	g	triangular: min=9.26212 mode=14.0831 max=17.6914	copper production, primary | copper | Cutoff, S - RER
copper	29.92771	g	triangular: min=19.6827 mode=29.9277 max=37.5956	copper production, primary | copper | Cutoff, S - RoW
copper	30.09519	g	triangular: min=19.7929 mode=30.0952 max=37.8060	copper production, primary | copper | Cutoff, S - LCI - RLA
copper	4.669018	g	triangular: min=3.07070 mode=4.66902 max=5.86529	copper production, primary | copper | Cutoff, S - AU

copper	20.93092	g	triangular: min=13.7658 mode=20.9309 max=26.2937	copper production, primary | copper | Cutoff, S - RNA
Output Flow	Amount	Unit	Uncertainty	Provider
Cu in laptop	1	Item(s)	none	

Table C11: Primary production of iron

Input Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
pig iron	377.0766	g	triangular: min=219.911 mode=377.077 max=525.342	market for pig iron | pig iron | Cutoff, S - GLO
Output Flow	Amount	Unit	Uncertainty	Provider
Fe in laptop	1	Item(s)	none	

Table C12: Primary production of plastics. The dataset 'market for thermoplastics and thermosets' is detailed in the table below

Input Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
Plastics	984.4388	g	triangular: min=469.023 mode=984.439 max=1351.48	market for thermoplastics and thermosets
Output Flow	Amount	Unit	Uncertainty	Provider
PL in laptop	1	Item(s)	none	

Table C13: Dataset 'market for thermoplastics and thermosets' used in the primary production of plastics. Amounts are the global market average based on Levi and Cullen (2018)

Input Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
isophthalic acid based unsaturated polyester resin	0.002	kg	none	isophthalic acid based unsaturated polyester resin production | isophthalic acid based unsaturated polyester resin | Cutoff, S - RER
methylene diphenyl diisocyanate	0.016	kg	none	market for methylene diphenyl diisocyanate | methylene diphenyl diisocyanate | Cutoff, S - LCI - GLO
phenol	0.013	kg	none	phenol production, from cumene | phenol | Cutoff, S - RER
polycarbonate	0.043*0.5	kg	none	market for polycarbonate | polycarbonate | Cutoff, S - GLO
polyethylene terephthalate, granulate, amorphous	0.231	kg	none	polyethylene terephthalate production, granulate, amorphous | polyethylene terephthalate, granulate, amorphous | Cutoff, S - RER
polyethylene, high density, granulate	0.294	kg	none	market for polyethylene, high density, granulate | polyethylene, high density, granulate | Cutoff, S - GLO
polypropylene, granulate	0.204	kg	none	market for polypropylene, granulate | polypropylene, granulate | Cutoff, S - GLO

polystyrene, general purpose	0.055	kg	none	polystyrene production, general purpose | polystyrene, general purpose | Cutoff, S - RER
polyvinylchloride, bulk polymerised	0.111	kg	none	market for polyvinylchloride, bulk polymerised | polyvinylchloride, bulk polymerised | Cutoff, S - GLO
styrene-acrylonitrile copolymer	0.043*0.5	kg	none	styrene-acrylonitrile copolymer production | styrene-acrylonitrile copolymer | Cutoff, S - RER
urea formaldehyde foam slab, hard	0.031	kg	none	market for urea formaldehyde foam slab, hard | urea formaldehyde foam slab, hard | Cutoff, S - LCI - GLO
Output Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
Plastics	1	kg	none	

Primary materials processing

Table C14: Processing of primary aluminium. The material content of aluminium in upstream datasets was removed to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
sheet rolling, aluminium	411.5081	g	triangular: min=207.261 mode=411.508 max=725.613	market for sheet rolling, aluminium | sheet rolling, aluminium | Cutoff, S - LCI - GLO
Output Flow	Amount	Unit	Uncertainty	Provider
Al in laptop	1	Item(s)	none	

Table C15: Processing of primary copper. The material content of copper in upstream datasets was removed to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
sheet rolling, copper	129.285	g	triangular: min=85.0277 mode=129.285 max=162.410	market for sheet rolling, copper | sheet rolling, copper | Cutoff, S - LCI - GLO
Output Flow	Amount	Unit	Uncertainty	Provider
Cu in laptop	1	Item(s)	none	

Table C16: Steel production and processing from primary pig iron. The material content of pig iron in upstream datasets was removed to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
hot rolling, steel	377.0766	g	triangular: min=219.911 mode=377.077 max=525.342	market for hot rolling, steel | hot rolling, steel | Cutoff, S - GLO

steel, low-alloyed	58.57834	g	triangular: min=34.1629 mode=58.5783 max=81.6111	steel production, converter, low-alloyed | steel, low-alloyed | Cutoff, S (REMOVED PIG IRON) - LCI - RER
steel, low-alloyed	318.4982	g	triangular: min=185.748 mode=318.498 max=443.731	steel production, converter, low-alloyed | steel, low-alloyed | Cutoff, S (REMOVED PIG IRON) - LCI - RoW
Output Flow	Amount	Unit	Uncertainty	Provider
Fe in laptop	1	Item(s)	none	

Table C17: Plastics processing from primary production. The material content of plastics in upstream datasets was removed to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
injection moulding	984.4388	g	triangular: min=469.023 mode=984.439 max=1351.48	market for injection moulding | injection moulding | Cutoff, S - GLO
Output Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
PL in laptop	1	Item(s)	none	

Secondary materials processing

Table C18: Secondary processing of aluminium. The material content of aluminium in upstream datasets was removed to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
aluminium, wrought alloy	411.5081	g	triangular: min=207.261 mode=411.508 max=725.613	treatment of aluminium scrap, post-consumer, prepared for recycling, at remelter | aluminium, wrought alloy | Cutoff, S - RER
Output Flow	Amount	Unit	Uncertainty	Provider
Al in laptop	1	Item(s)	none	

Table C19: Secondary processing of copper. The material content of copper in upstream datasets was removed to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
copper	1.945551	g	triangular: min=1.27954 mode=1.94555 max=2.44403	treatment of metal part of electronics scrap, in blister-copper, by electrolytic refining | copper | Cutoff, S - LCI - SE
copper	127.3395	g	triangular: min=83.7481 mode=127.339 max=159.966	treatment of metal part of electronics scrap, in blister-copper, by electrolytic refining | copper | Cutoff, S - LCI - RoW

Output Flow	Amount	Unit	Uncertainty	Provider
Cu in laptop	1	Item(s)	none	

Table C20: Secondary processing of steel. The material content of pig iron in upstream datasets was removed to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
steel, low-alloyed	377.0766	g	triangular: min=219.911 mode=377.077 max=525.342	steel production, electric, low-alloyed | steel, low-alloyed | Cutoff, S - RER
Output Flow	Amount	Unit	Uncertainty	Provider
Fe in laptop	1	Item(s)	none	

Table C21: Secondary processing of plastics. The material content of plastics in upstream datasets was removed to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
polyethylene, high density, granulate, recycled	984.4388	g	triangular: min=469.023 mode=984.439 max=1351.48	market for polyethylene, high density, granulate, recycled | polyethylene, high density, granulate, recycled | Cutoff, S - Europe without Switzerland
Output Flow	Amount	Unit	Uncertainty	Provider
PL in laptop	1	Item(s)	none	

Laptop manufacturing

Table C22: Laptop manufacturing. The amounts were modified with data from Babbitt et al. (2020) and van Eygen (2016). This dataset and upstream datasets only include energy, auxiliaries, and infrastructure. The content of all materials was removed from this and upstream datasets to avoid double counting with the primary production.

Input Flow	Amount	Unit	Uncertainty	Provider
battery, Li-ion, rechargeable, prismatic	423.8	g	triangular: min=321.200 mode=423.800 max=495.300	market for battery, Li-ion, rechargeable, prismatic | battery, Li-ion, rechargeable, prismatic | Cutoff, U - MOD_NOmat - GLO
disk drive, CD/DVD, ROM, for laptop computer (mass)	194.6	g	triangular: min=147.400 mode=194.600 max=268.400	disk drive production, CD/DVD, ROM, for laptop computer | disk drive, CD/DVD, ROM, for laptop computer | Cutoff, U (mass) - MOD_NOmat - GLO
electricity, medium voltage	1.6667	kWh	lognormal: gmean=1.66670 gsigma=1.15191	market group for electricity, medium voltage | electricity, medium voltage | Cutoff, S - GLO
extrusion, plastic pipes	604.032	g	none	market for extrusion, plastic pipes | extrusion, plastic pipes | Cutoff, S - GLO

hard disk drive, for laptop computer (mass)	130.5	g	triangular: min=83.7000 mode=130.500 max=177.300	hard disk drive production, for laptop computer | hard disk drive, for laptop computer | Cutoff, U (mass) - MOD_NOmat - GLO
liquid crystal display, unmounted	640.6	g	triangular: min=383.100 mode=640.600 max=892.300	liquid crystal display production, unmounted | liquid crystal display, unmounted | Cutoff, U (Babbitt et al) - MOD_NOmat - GLO
photovoltaic cell factory	3.04E-08	Item(s)	none	market for photovoltaic cell factory | photovoltaic cell factory | Cutoff, S - GLO
printed wiring board, mounted mainboard, laptop computer, Pb free	327.3	g	triangular: min=200.600 mode=327.300 max=404.700	market for printed wiring board, mounted mainboard, laptop computer, Pb free | printed wiring board, mounted mainboard, laptop computer, Pb free | Cutoff, U - MOD_NOmat - GLO
printed wiring board, surface mounted, unspecified, Pb containing	18.7395	g	none	market for printed wiring board, surface mounted, unspecified, Pb containing | printed wiring board, surface mounted, unspecified, Pb containing | Cutoff, U - MOD_NOmat - GLO
printed wiring board, surface mounted, unspecified, Pb free	43.7255	g	none	market for printed wiring board, surface mounted, unspecified, Pb free | printed wiring board, surface mounted, unspecified, Pb free | Cutoff, U - MOD_NOmat - GLO
tap water	1620	kg	lognormal: gmean=1620.00 gsigma=1.15191	market group for tap water | tap water | Cutoff, S - GLO
wastewater, unpolluted	-1.62	m3	lognormal: gmean=-1.62000 gsigma=1.15191	market for wastewater, unpolluted | wastewater, unpolluted | Cutoff, U - RoW

Output Flow	Amount	Unit	Uncertainty	Provider
computer, laptop	1	Item(s)	none	
Water	98.82	kg	lognormal: gmean=0.0988200 gsigma=1.22140	

Incineration

Table C23: Incineration used for energy recovery

Input Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
residue from mechanical treatment, laptop computer	-2938.13	g	triangular: min=-4096.49 mode=-2938.13 max=-1621.33	treatment of residue from mechanical treatment, laptop computer, municipal waste incineration | residue from mechanical treatment, laptop computer | Cutoff, S - CH

transport, freight, light commercial vehicle	50.0*M	kg*km	M = triangular: min=1.62133 mode=2.93813 max=4.09649	market for transport, freight, light commercial vehicle | transport, freight, light commercial vehicle | Cutoff, S - LCI - GLO

Output Flow	Amount	Unit	Uncertainty	Provider
Incineration	1	Item(s)	none	

Recycling

Table C24: Sorting and shredding processes

Input Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
aluminium scrap, post-consumer, prepared for melting	-294.98	g	none	aluminium scrap, post-consumer, prepared for melting, Recycled Content cut-off | aluminium scrap, post-consumer, prepared for melting | Cutoff, U - GLO
copper scrap, sorted, pressed	-109.892	g	none	copper scrap, sorted, pressed, Recycled Content cut-off | copper scrap, sorted, pressed | Cutoff, U - GLO
electronics scrap	-366.261	g	triangular: min=-496.699 mode=-366.261 max=-149.976	electronics scrap, Recycled Content cut-off | electronics scrap | Cutoff, S - GLO
iron scrap, sorted, pressed	-255.411	g	none	iron scrap, sorted, pressed, Recycled Content cut-off | iron scrap, sorted, pressed | Cutoff, U - GLO
mechanical treatment facility, waste electric and electronic equipment	1.60E-08	Item(s)	uniform: min=1.60000E-08 max=1.60000E-08	market for mechanical treatment facility, waste electric and electronic equipment | mechanical treatment facility, waste electric and electronic equipment | Cutoff, S - GLO
residue from mechanical treatment, laptop computer	-563.112	g	triangular: min=-792.698 mode=-563.112 max=-372.988	treatment of residue from mechanical treatment, laptop computer, municipal incineration with fly ash extraction | residue from mechanical treatment, laptop computer | Cutoff, S - CH
used Li-ion battery	-364.033	g	triangular: min=-415.825 mode=-364.033 max=-274.800	market for used Li-ion battery | used Li-ion battery | Cutoff, S - GLO

waste electric and electronic equipment	-2938.13	g	triangular: min=-4096.49 mode=-2938.13 max=-1621.33	treatment of waste electric and electronic equipment, shredding | waste electric and electronic equipment | Cutoff, S - GLO
waste plastic, consumer electronics	-856.462	g	triangular: min=-1175.79 mode=-856.462 max=-408.050	treatment of waste plastic, consumer electronics, municipal incineration with fly ash extraction | waste plastic, consumer electronics | Cutoff, S - CH
waste polyethylene, for recycling, unsorted	-127.977	g	uniform: min=-127.977 max=-127.977	waste polyethylene, for recycling, unsorted, Recycled Content cut-off | waste polyethylene, for recycling, unsorted | Cutoff, U - GLO
transport, freight, light commercial vehicle	50.0*M	kg*km	M = triangular: min=1.62133 mode=2.93813 max=4.09649	market for transport, freight, light commercial vehicle | transport, freight, light commercial vehicle | Cutoff, S - LCI - GLO
Output Flow	Amount	Unit	Uncertainty	Provider
S2_recycling	1	Item(s)	none	

Remanufacturing

Table C25: Remanufacturing process

Input Flow	**Amount**	**Unit**	**Uncertainty**	**Provider**
electricity, medium voltage	2.5	kWh	triangular: min=1.40000 mode=2.50000 max=3.50000	market group for electricity, medium voltage | electricity, medium voltage | Cutoff, S - LCI - GLO
transport, freight, light commercial vehicle	50.0*M	kg*km	M = triangular: min=1.62133 mode=2.93813 max=4.09649	market for transport, freight, light commercial vehicle | transport, freight, light commercial vehicle | Cutoff, S - LCI - GLO
Output Flow	Amount	Unit	Uncertainty	Provider
S3_remanufacturing	1	Item(s)	none	

Reuse

Table C26: Reuse – only transportation is accounted for. Parameter 'M' is described in with the uncertainty.

Input Flow	Amount	Unit	Uncertainty	Provider
transport, freight, light commercial vehicle	50.0*M	kg*km	M = triangular: min=1.62133 mode=2.93813 max=4.09649	market for transport, freight, light commercial vehicle | transport, freight, light commercial vehicle | Cutoff, S - LCI - GLO
Output Flow	Amount	Unit	Uncertainty	Provider
S4_transport reuse	1	Item(s)	none	

Sensitivity Analysis

Monte Carlo

We performed three calculation runs with Monte Carlo in openLCA with the LCIA climate change (GWP100) using the dataset ‘computer production, laptop | computer, laptop | Cutoff, U (modified version with materials)’

Table C27: Difference in the LCIA results for climate change (GWP100) using the dataset ‘computer production, laptop | computer, laptop | Cutoff, U (modified version)’

Number runs	Mean	Standard deviation	Minimum	Maximum	Median	5% Percentile	95% Percentile
100	190.9595	19.43103	148.4263	233.5051	192.3183	157.4588	221.5544
1000	189.0039	18.26221	139.0376	244.3624	189.1295	159.1763	217.5861
10000	188.0931	18.70953	132.4563	263.9715	187.8974	157.2816	218.8344

Absolute dissipation per product cycle

We performed a sensitivity analysis to know the changes of the dissipation per product cycle in S2–S4 upon variation of the losses during the dismantling, shredding & sorting for secondary material processing. The figures S1–S2 are shown per scenario with the four analyses materials. The dashed line in each chart shows the used value ‘dismantling, shredding & sorting’ for each material as described in subsections 1.1–1.4. Note that for some cycles, the dissipation (y-axis) decreases with higher losses (x-axis). This decrease is because the remaining material reaching this cycle is smaller due to higher losses in the previous cycles; hence, the absolute dissipation of the reference unit 1 kg of material decreases.

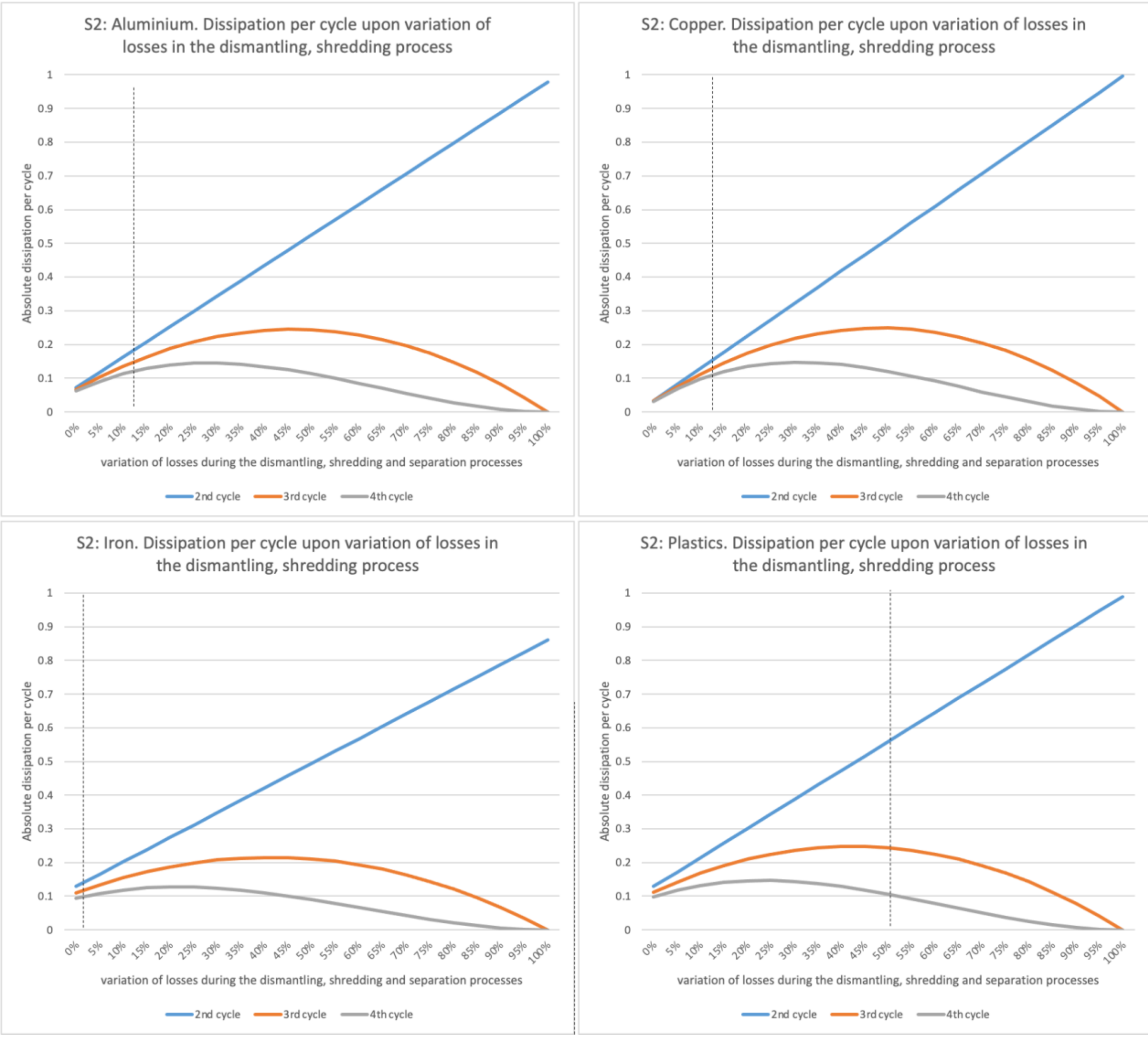

Figure C3: Scenario 2 (recycling) – Sensitivity of dissipation per product cycles upon variation of the losses during the dismantling, shredding & sorting for secondary material processing. The dashed line in each chart shows the used value for each material as described in subsections 1.1–1.4.

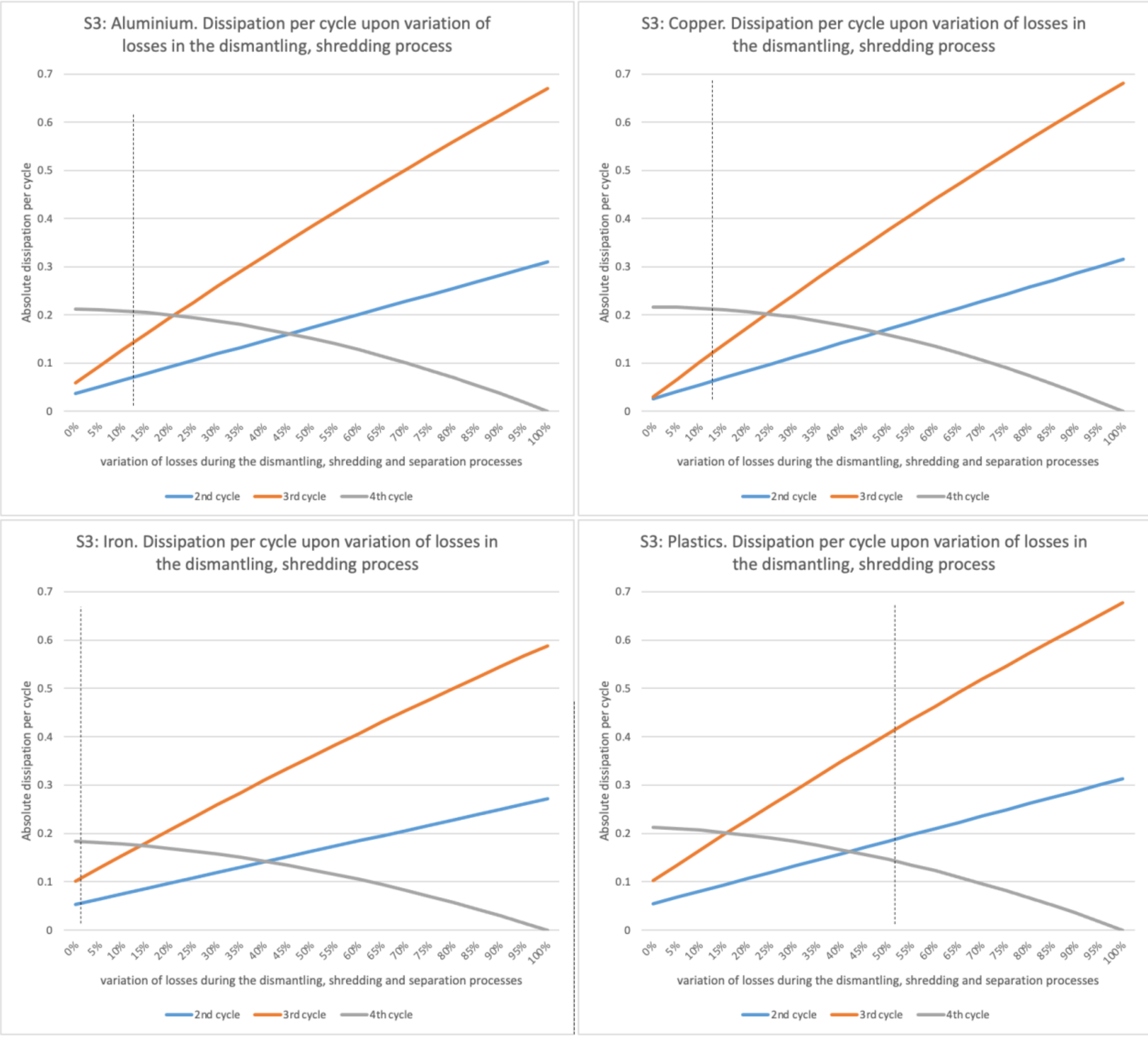

Figure C4: Scenario 3 (refurbishing) – Sensitivity of dissipation per product cycles upon variation of the losses during the dismantling, shredding & sorting for secondary material processing. The dashed line in each chart shows the used value for each material as described in subsections 1.1–1.4.

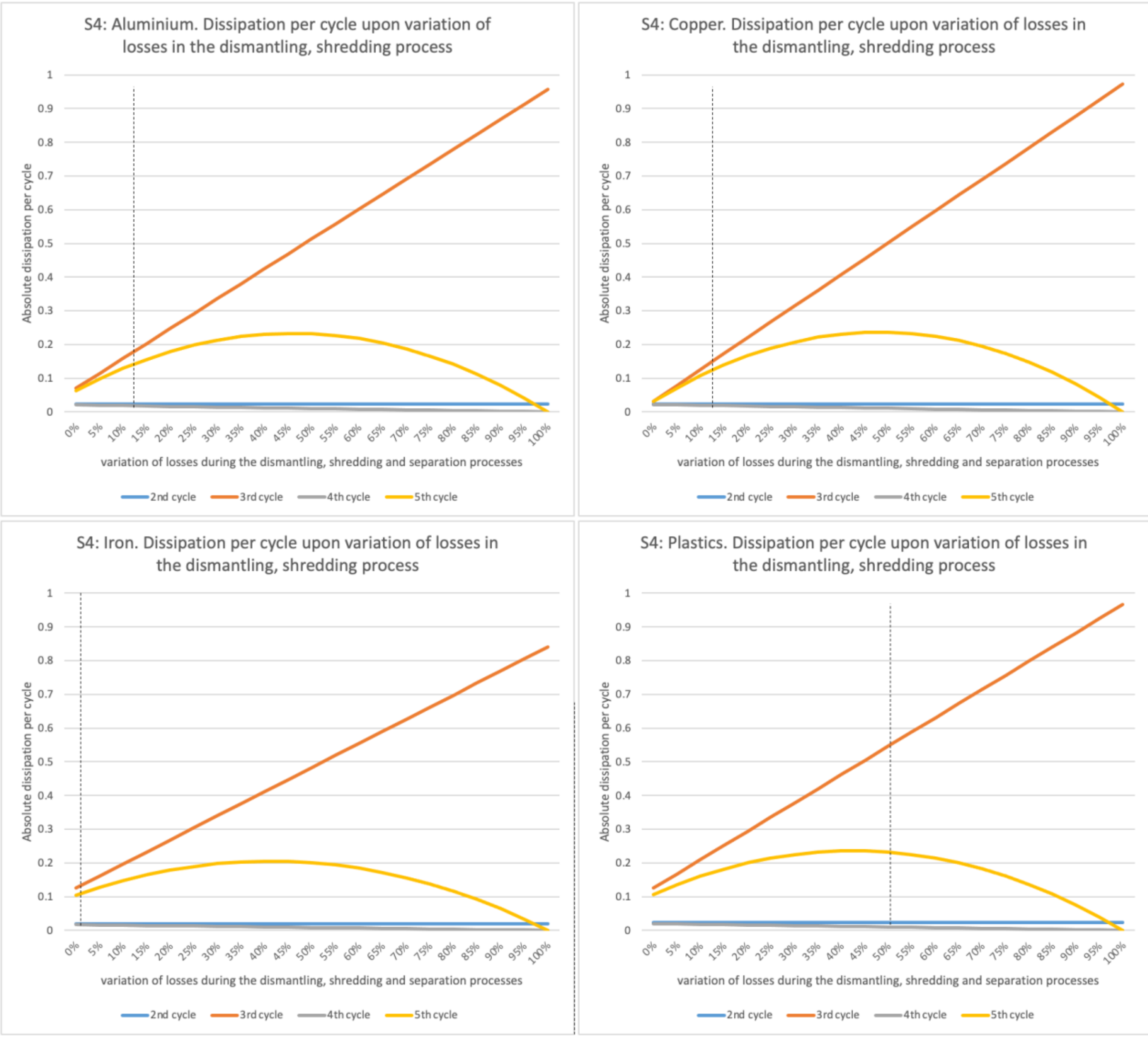

Figure C5: Scenario 4 (reuse) – Sensitivity of dissipation per product cycles upon variation of the losses during the dismantling, shredding & sorting for secondary material processing. The dashed line in each chart shows the used value for each material as described in subsections 1.1–1.4.

Comparison of laptop results with other authors

We made a comparison of laptop impact results in terms of CO_2-eq with other authors. Note that here, we include producing and processing all the materials used in the laptop from cradle to laptop's industry gate. The share of materials in our study was 51 kg CO_2-eq, while the manufacturing used in the calculation of indicators was 138 kg CO_2-eq.

Table C28: Comparison of laptop impact results in terms of CO2-eq with other authors. Note that here we include all the materials used in the laptop.

Reference unit	Weight (kg)	Screen size	Year	Database	Impact category (as described by the authors)	Impacts laptop [kg CO2-eq]	Impacts from PCBs (motherboard, RAM, inc IC)	Share PCB/total production [%]	Source
Dell Latitude E6400	2.52	14"	2008	GaBi	Carbon footprint - [kg CO2-eq]	150	72	48.0%	(O'Connell and Stutz, 2010)
14-inch Hewlett–Packard (HP) laptop	2	14"	2016	Simapro and RCEES LCI databases	GHG emissions [kg CO2-eq]	189	84.2	44.6%	(Liu et al., 2016)
12.1" HP Omnibook with dock	3.3	12.1"	2003	Ecoinvent v2.2	GHG - GWP100	256	158.7	62.0%	(Teehan and Kandlikar, 2013)
HP 530 laptop, 16"	2.8	16"	2009	Ecoinvent v2.3	GHG - GWP100	108	52.5	48.6%	(Teehan and Kandlikar, 2013)
HP Mini 110–1030 CA Netbook, 10"	1.3	10"	2009	Ecoinvent v2.4	GHG - GWP100	62.2	41	65.9%	(Teehan and Kandlikar, 2013)
one year of access to a laptop computer (use time is 3 years)	1.55	14"	2015	Ecoinvent v3.x	Climate change	55	32	58.2%	(André et al., 2019)
1 laptop	average=2.9 (min=1.6 max=4.09)	12.1–17"	1999-2011	Ecoinvent v3.4	Climate change GWP100 [kg CO2-eq]	189	98	51.9%	This study

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EDUCATION

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2006 _ 2012 – BSc in Architecture and Urbanism (Hons.)
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2018 _ 2019 – Tutor of student for the Master's thesis:
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PUBLICATIONS
Peer Reviewed Journals

MORAGA, G., HUYSVELD, S., MATHIEUX, F., BLENGINI, G.A., ALAERTS, L., VAN ACKER, K., DE MEESTER, S., DEWULF, J., 2019. **Circular economy indicators: What do they measure?** Resources, Conservation and Recycling. 146, 452–461. https://doi.org/10.1016/j.resconrec.2019.03.045

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CONFERENCE PROCEEDINGS

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