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

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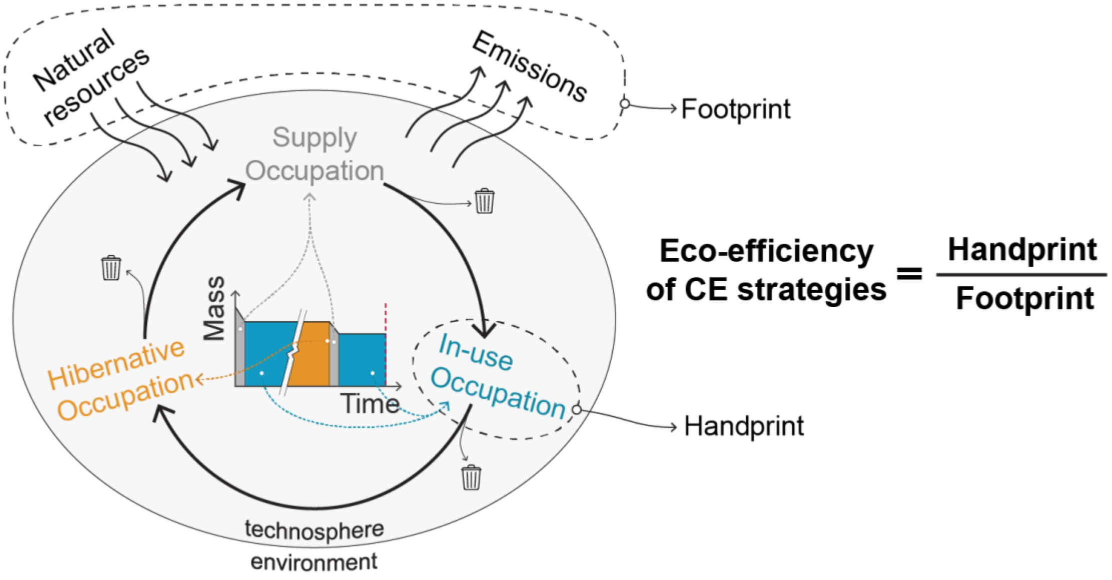
Abstract

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 loss of materials and environmental footprint in electronics. Resource efficiency indicators – typically defined as benefits (handprint) over burdens (footprint) – can measure materials' life cycle performance. This paper aims to develop 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, recycling, refurbishing, and reuse. The scenarios show the use of the materials in several cycles of laptops over a 25-year time horizon.

Generally, scenarios with cycles of refurbishment and reuse showed improved resource efficiency compared to recycling scenarios. Compared to energy recovery the improvement was up to 189% (refurbishment) and 157% (reuse) in the case of aluminium. Nonetheless, it is remarkable that the average resource efficiency results showed a preference for refurbishing over reuse during 25 years. The result is limited to a shorter functional in-use time of reused laptops. This analysis is relevant for a CE, where the value of materials should be kept for as long as possible. Our methodology expands the traditional one-cycle perspective by measuring the use of materials for 25 years. Policy-makers can use our indicators to assess CE strategies for several product cycles that keep materials in use lowering environmental impacts.

Keywords: circular economy, indicator, resource efficiency, life cycle thinking, raw material, WEEE

Graphical abstract



1. Introduction

The world population and its affluence is continuously growing, generating increased environmental impacts. Globally, the total material footprint per capita increased from 8.7 t in 2000 to 12.2 t in 2017 (Ritchie et al., 2018). By 2050, three planets could be needed to provide resources for our current lifestyle (UNEP, 2019). 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, 2015). 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 (several cycles) and their environmental benefit and burden.

A possible way to measure the progress towards a CE is using indicators. However, the CE is a debated concept with many definitions (Kirchherr et al., 2017), and despite the lack of agreement about CE, many indicators were proposed in a variety of scopes (Moraga et al., 2019). This variety can cause governments or companies to cherry-pick results that are suitable with a specific circularity message (Pauliuk, 2018). Moreover, as pointed out by several authors, CE does not necessarily show connections with sustainability (Geissdoerfer et al., 2017; Kirchherr et al., 2017; Kovacic et al., 2019). Hence, a key issue with the CE is the measurement of progress in consonance with sustainability and particularity with decreased environmental impacts. Hertwich et al. (2019) estimated that the absolute emissions related to the global material production were about 11 Gt CO_2-eq in 2015. 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, CE indicators can be related to resource efficiency.

1.1. Resource efficiency indicators for a circular economy

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).'

Huysman et al. (2015) ponder that the several different types of resource efficiency indicators can be expressed by Eq. (1) or Eq. (2), where resource efficiency with LCA is defined as a ratio of benefits divided by environmental impacts based on resource or emissions flows.

$$Efficiency\ 1 = \frac{benefits}{inventoried\ flows} \qquad (1)$$

$$Efficiency\ 2 = \frac{benefits}{environmental\ impacts} \qquad (2)$$

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) 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 (e.g. Efficiency 2 indicators). Eq. (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., 2015).

The denominator in Eq. (2) can be assessed using dedicated tools for the calculation of potential environmental impacts (footprint), such as Life Cycle Assessment (LCA). Moreover, the nominator *benefits* in both equations can embrace a multitude of uses. 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 value, created environmental benefit, the output of energy or exergy, and economic and social welfare (Huysman et al., 2015). More recently, on the quantification of (environmental) benefits, the handprint concept can be promising but not so easily captured with LCA (Alvarenga et al., 2020). In another perspective, the benefit of keeping materials useful (in the loop), minimising losses, could be done with the concept of in-use occupation (Moraga et al., 2021).

In-use occupation is the functional use of materials. This concept opposes to non-beneficial human actions with the use of materials, such as hibernation and dissipation to the environment or technosphere (Dewulf et al., 2021). With increased in-use occupation, the environment can benefit from less anthropogenic pressures to extract new materials. The concept could serve as a proxy of the societal benefit (handprint) of having materials in use over time. Hence, in-use occupation indicators could be used in Eq. (1) and (2) as a proxy for *benefit*.

In this sense, the development of CE indicators based on in-use occupation is of particular interest. The in-use occupation concept includes several aspects needed for the assessment of a CE. At least two aspects deserve attention in a CE that slows down and closes resources loops – quantity and quality (Moraga et al., 2019). With this reasoning, Moraga et al. (2021) 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 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 paper 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.

2. Methodology of the indicators

2.1. Indicators of the in-use occupation of materials

We distinguish three phases with the use of materials: supply, in-use, and hibernation. The supply phase is where the materials are being processed and manufactured in the economy. The materials are effectively used in the in-use phase. In the hibernation phase, materials are neither being used or being processed. The equation to calculate the in-use occupation of materials and the two derived indicators are modified from Moraga et al. (2021) in Eq. (3), (4), and (5), respectively. Eq. (3) measures the in-use occupation of a material group in product cycle *j* ($Occ_{U,j}$), that is, the mass of the material (minus dissipation) in the in-use occupation phase, in which products, embedding the materials, are effectively used. This equation considers the materials in a time horizon (TH) of 25 years. This TH is one of the temporal scopes proposed by the SUPRIM project (Sustainable Management of Primary Raw Materials) (Schulze et al., 2020), which was stated to be appropriate to encompass the use of materials within a similar technological boundary. The SUPRIM project analysed and proposed methodologies in search of cohesion for the assessment of abiotic resources. We use a TH of 25 years as an appropriate measure of one generation with less uncertainty regarding future technological development. Eq. (3) shows a set of formulae for three cases: when the occupation starts and ends before the TH; when it starts before but ends after the TH; and when it starts after the TH, which is not assessed.

$$Occ_{U,j} = \begin{cases} \left(\mathrm{m_{U,j}} - \mathrm{l_{U,j}}/2\right) \cdot \Delta \mathrm{t_{U,j}} & \text{when } \{\mathrm{t_{U,j}}, \mathrm{t_{H,j}}\} \leq \mathrm{TH} \\ \left(\mathrm{m_{U,j}} - \mathrm{l_{U,j}}/2\right) \cdot \left(\mathrm{TH} - \mathrm{t_{U,j}}\right) & \text{when } \mathrm{t_{U,j}} < \mathrm{TH} \text{ and } \mathrm{t_{H,j}} > \mathrm{TH} \\ \text{not assessed} & \text{when } \mathrm{t_{U,j}} > \mathrm{TH} \end{cases} \qquad (3)$$

where:

$Occ_{U,j}$: In-use occupation of a material in product cycle *j* [kg × year]

$m_{U,j}$: mass of a material in the in-use phase of product cycle *j* [kg], that is, material embedded in a consumer product

$l_{U,j}$: mass loss of a material during the in-use phase of product cycle *j* [kg]

$\Delta t_{U,j}$: in-use time of a product cycle *j* [year], that is, the time products are used not considering a possible hibernation phase

$t_{U,j}$: time occurrence at the start of the in-use phase of a product cycle *j* [years]

$t_{H,j}$: time occurrence at the start of the hibernation phase of a product cycle *j* [years]

TH : time horizon of 25 years

Eq. (2) shows the in-use occupation ratio (UOR), which is a performance measure of the occupation considering material losses and hibernation for products within the TH. Eq. (3) is the final retention in society (FRS) and shows the percentage of material still possible to recover after the TH (i.e. material not dissipated at or before the year 25). Following Moraga et al. (2021), we limit this article's scope by including hibernation only as of the hoarding of products; hence, other hibernation types (e.g. tailing, landfill, or abandoned infrastructure) are not assessed.

$$UOR_U = {}^{\sum_{j=1}^{n} Occ_{U,j}}\!\Big/_{Occ_{Umax}} \cdot 100\% \qquad (4)$$

where (symbols not previously introduced):

UOR_U : in-use occupation ratio of a material [%]

Occ_{Umax} : theoretical maximum in-use occupation of a material*,* which is the amount of material assessed without dissipation and hibernation [kg × year]

$$FRS_U = {}^{m_{U,TH}}\!/_{m_{S,1}} \cdot 100\% \qquad (5)$$

where (symbols not previously introduced):

FRS_U : final retention in society of a material [kg %]

$m_{U,TH}$: mass of a material that is still available at the year 25 (in *n* product cycle *j*) [kg]

$m_{S,1}$: mass of the primary raw material that is firstly used in the 1st product cycle *j* [kg] before losses of production or manufacturing

2.2. 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. (2015). This paper defines 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 (goods 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) 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 using products (e.g. electricity consumption during use).

$$Ftp = V_U + \sum_{j=1}^{n}\left(P_{U,j} + \frac{M_j + Rc_j + Rf_j + Re_j + Dis_j}{m_{U,j}} * AF_{U,j} \; - AVe_{U,j}\right) - AVm_{U,TH} \tag{6}$$

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*]. This parameter can be calculated by selecting a similar energy production pathway.

$AVm_{U,TH}$: 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]. This parameter can be calculated by selecting a similar primary raw material production pathway.

Based on Eq. (4) for the footprint (*Ftp)*, the resource efficiency indicators of the in-use occupation and final retention in society can be derived in Eq. (5) and Eq. (6), respectively.

$$Eff_{Occ,U} = {\sum_{j=1}^{n} Occ_{U,j}} \Big/ {Ftp} \quad (7)$$

where (symbols not previously introduced):

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

$$Eff_{FRS,U} = {FRS} \Big/ {Ftp} \quad (8)$$

where (symbols not previously introduced):

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

3. Case study: four materials in a laptop

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. 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.

3.1. Description of the case study

The overview of the case study is presented in Figure 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 2), based on Babbitt et al. (2020) and Van Eygen et al. (2016). A detailed inventory can be found in the supporting information for dissipation (Tables S1–S6) and time of supply, in use, and hibernation phases (Tables S7–S8).

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.

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.

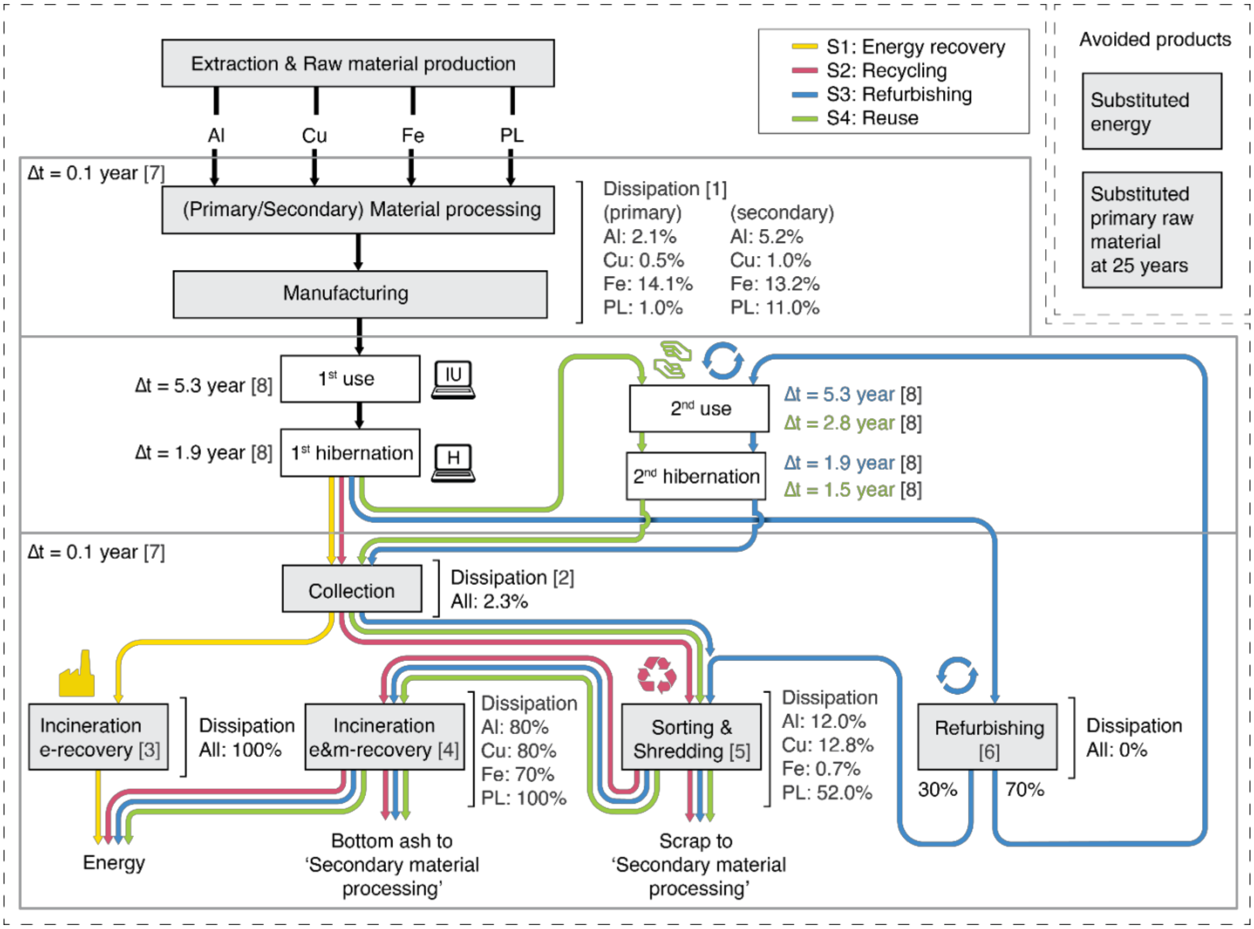


Figure 1: System boundaries of the four scenarios in analysis considering four materials: aluminium (Al), copper (Cu), iron (Fe), and plastics (PL). References between brackets: [1] Dissipation of primary/secondary material production and manufacturing is calculated based on (Cullen and Allwood, 2013) *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)

3.2. Goal and scope definition

The goal is to quantify the environmental impacts from the use of materials defined in Figure 1. The life cycle impact assessment (LCIA) results will be used in the footprint assessment (Eq. 4) to provide a specific in-use occupation. The scope is exemplified by the system boundary (Figure 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 these processes, 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, plastic materials 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.

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 the supporting information for the four scenarios (tables S9–S26).

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.

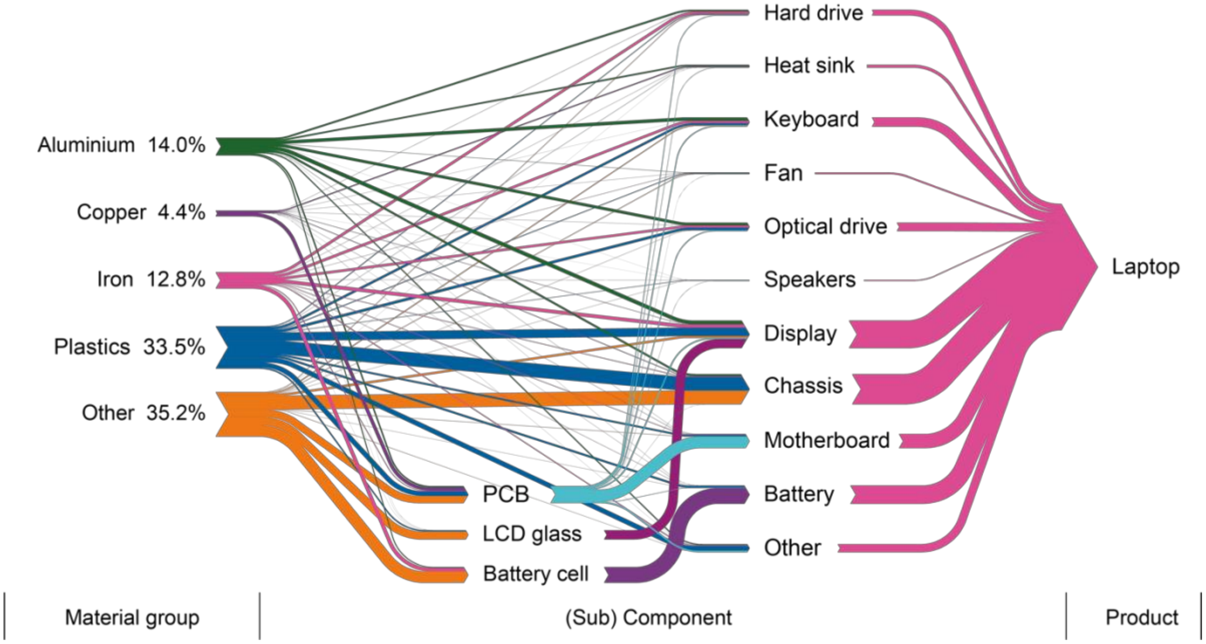


Figure 2: Share of the mass of materials in laptops' components. Based on the arithmetic mean values from Babbitt et al. (2020) *and Van Eygen et al.* (2016). *PCB (printed circuit board); LCD (liquid crystal display)*

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 S28). Inventory tables with the distribution of each parameter are provided in the supporting information (Tables S9–S26).

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 Nations 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. Results

4.1. In-use occupation and final retention in society

Figure 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 ($\Delta t = 0.1a$), but they contribute significantly to the dissipation of materials.

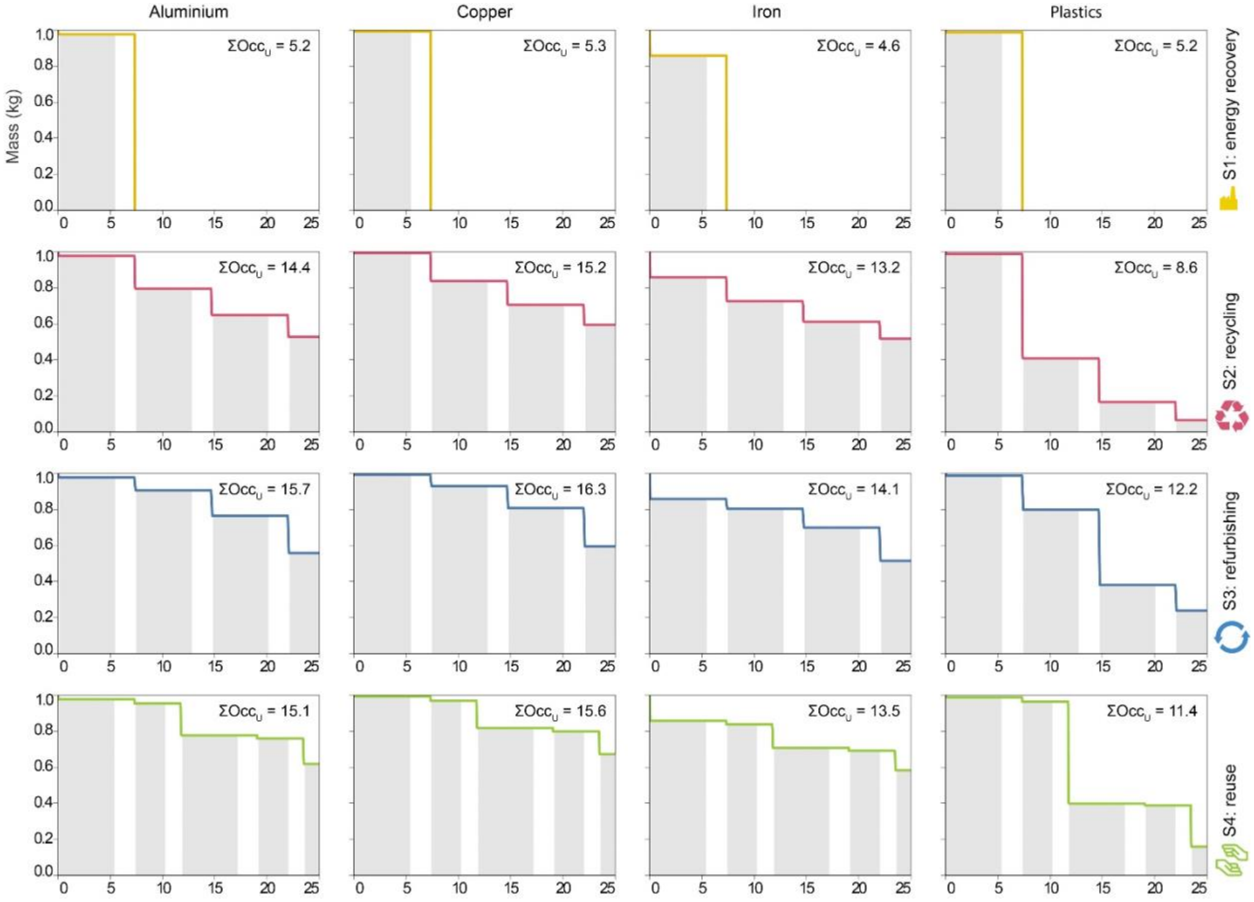


Figure 3: In-use occupation of four materials used in a laptop considering four scenarios. Coloured lines account for the remaining mass of the initially produced primary raw material input along the TH. Grey areas identify the in-use occupation phases (kg × yr). The supply (production, manufacturing, and EoL) and hibernation phases are white areas before and after the in-use, respectively.

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 remains 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 the supporting information (Figure S1–S3) 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).

4.2. Footprint of the in-use occupation of materials

Figure 4 and Figure 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 and Figure 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.

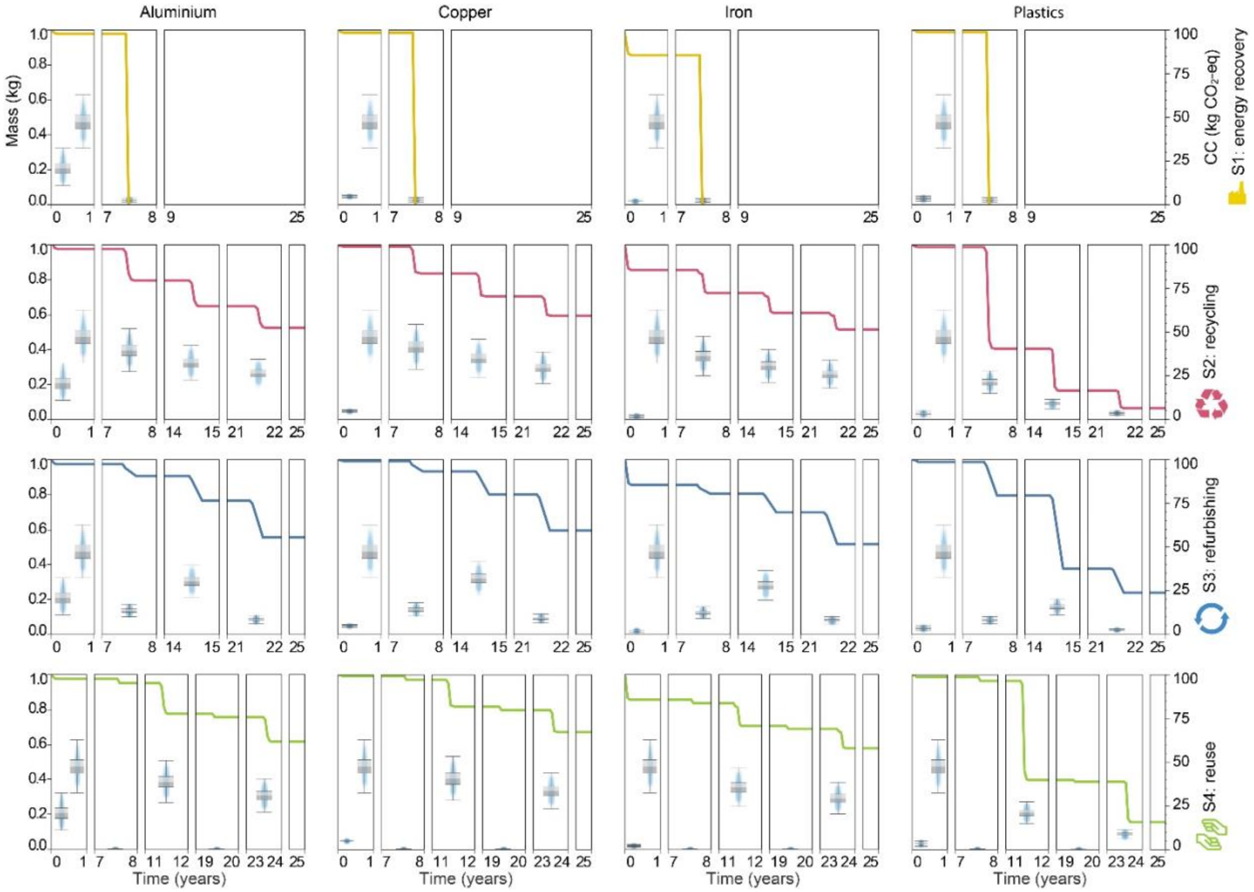


Figure 4: Footprint for the in-use occupation of four materials used in a laptop considering four scenarios. Coloured lines account for the remaining mass of the raw material. Each supply cycle is shown in sections of one year. Box-plot shows the footprint of each supply phase along time. The dual box-plot 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 CO_2 equivalent).

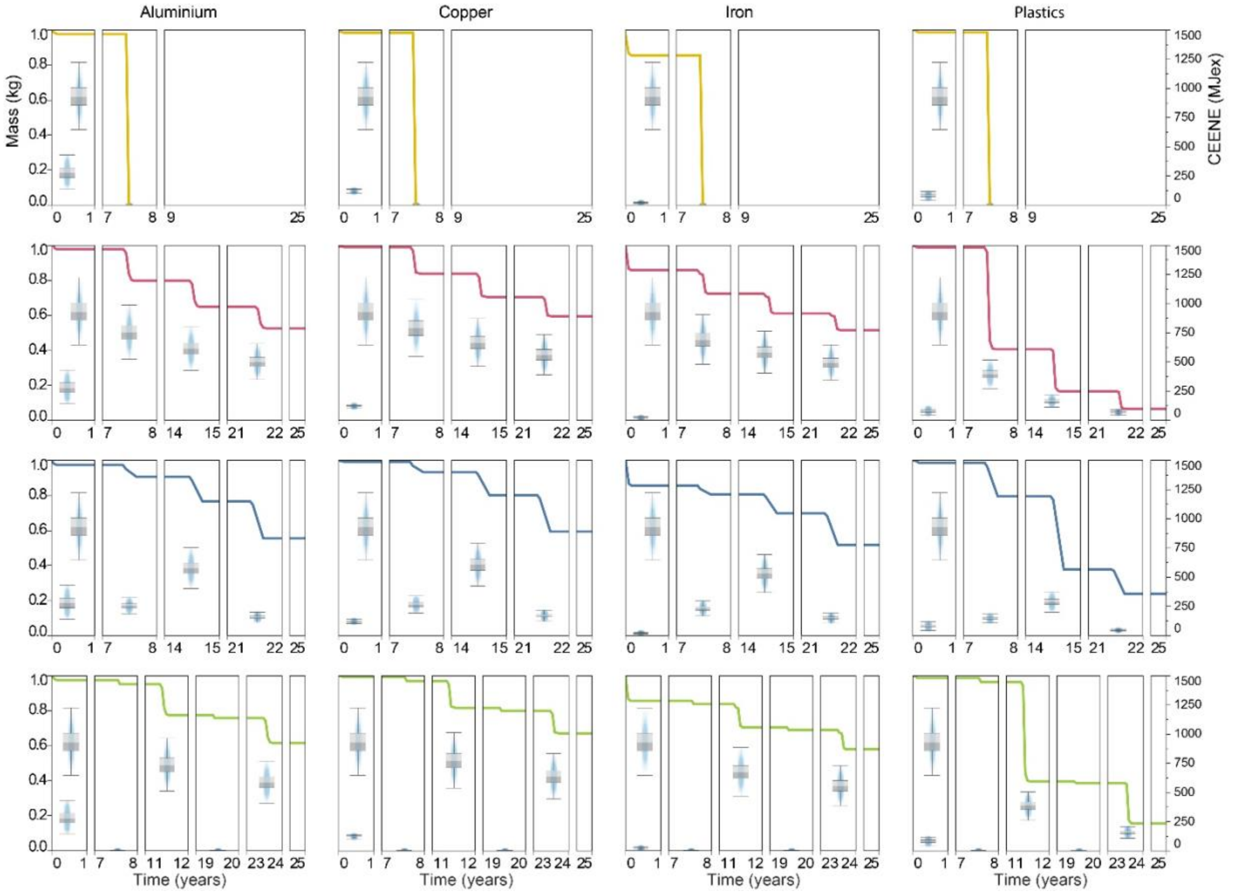


Figure 5: Footprint for the in-use occupation of four materials used in a laptop considering four scenarios. Coloured lines account for the remaining mass of the raw material. Each supply cycle is shown in sections of one year. Box-plot shows the footprint of each supply phase along time. The dual box-plot 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).

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 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 resources 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.

4.3. Resource efficiency indicators for CE strategies

Figure 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.

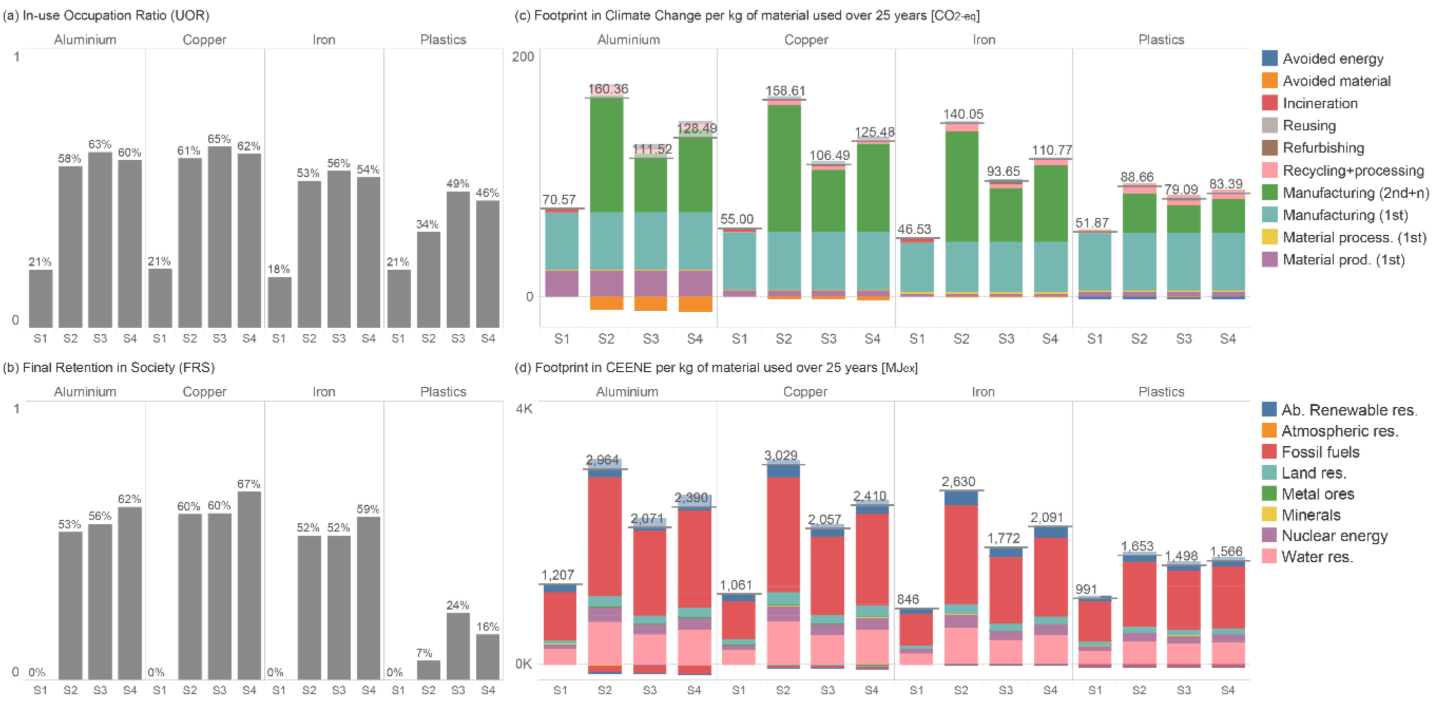


Figure 6: Result of indicators (a) in-use occupation ratio (UOR) and (b) final retention in society (FRS). Result of the cumulative impacts for 1 kg of material initially extracted and used over 25 years in (c) climate change (CC), which

shows the phases related to the impact, and (d) Cumulative Exergy Extraction from the Natural Environment (CEENE), which shows the natural resource footprint

The cumulative impacts in CC (Figure 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 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 7 shows the carbon-emission and natural resource efficiency indicators using CC and CEENE, respectively. The resource efficiency of the in-use occupation (Figure 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.

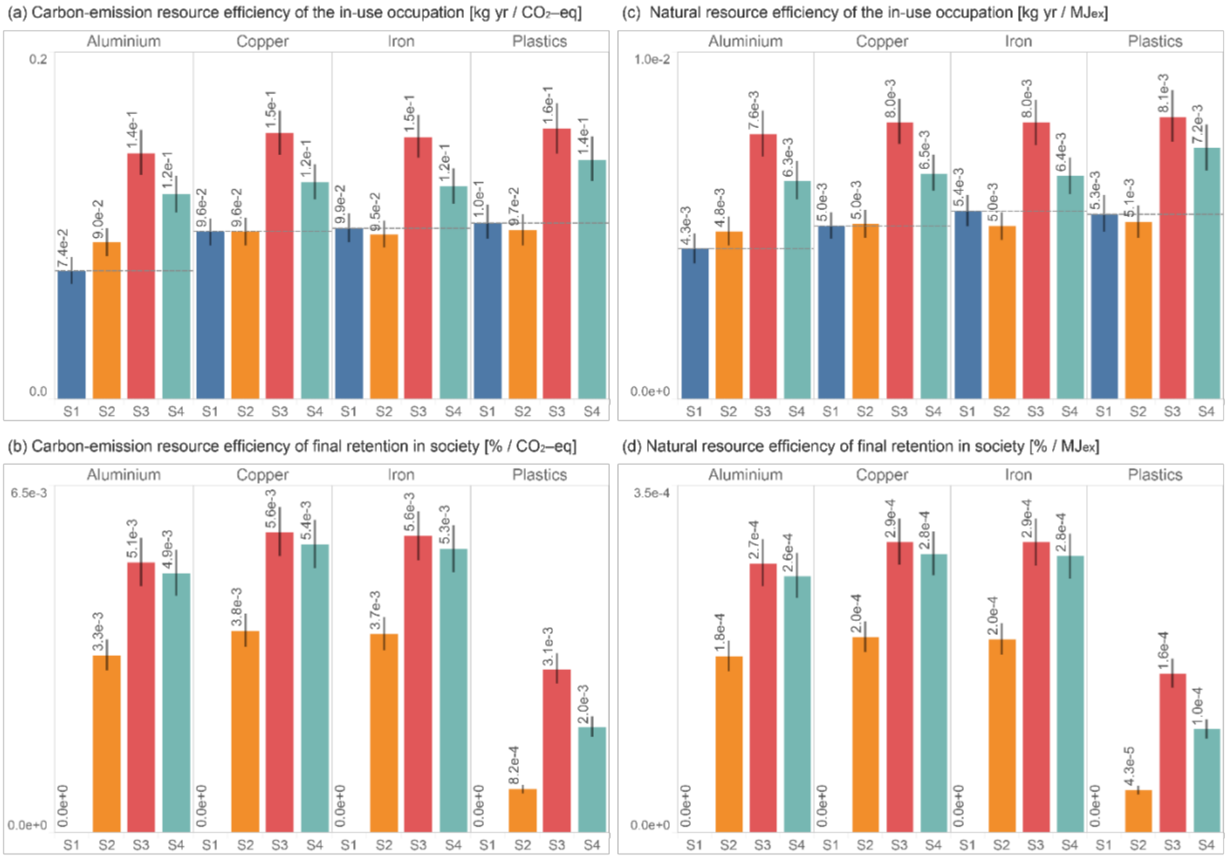


Figure 7: Result of the resource efficiency indicators of In-use Occupation using emission impacts as (a) climate change (CC) and use of resources impact as (c) Cumulative Exergy Extracted from the Natural Environment (CEENE). Result of the resource efficiency indicator of final retention in society as (b) CC and (d) CEENE

While FRS (Figure 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 7b and 7d). This is because 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 6b.

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.

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 article, 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 footprint during the use of the laptops was not part of the analysis – this footprint is related to the use of the product itself, not the management of materials. Evidently, the operation of energy-consuming products is linked to environmental impacts. Thus, a possible improvement would be the development of product-specific indicators, as further discussed.

Also, from the illustration, it is interesting the little difference of Eff_{Occ} results in S1 and S2. Although they have similar Eff_{Occ} for most materials, Eff_{FRS} result shows a clear preference for S2. This pinpoints the complementarity of these two indicators as they were designed to be used jointly. However, the evaluation of more complex scenarios may present challenges in assessing the results with different indicators. A similar challenge is well known in the LCA community regarding the prioritisation of different environmental impacts. In this sense, multi-criteria decision analysis was demonstrated to aid interpretation of complex results in LCA (Zanghelini et al., 2018) and could also be used with our indicators, particularly if other LCIA methods are used for the footprint evaluation

The results found in this article 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 S29 – 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.

Our methodology advances some aspects in communicating time aspects of the environmental impact results – LCIA. Figure 4 and Figure 5 graphically show the emissions in a specific time occurrence. We showed the footprint in cumulative sections along the life cycle (e.g. raw material production includes mining, which occurred previously) – this information could be as disaggregated as needed for a particular LCA purpose. LCA studies often disregard the dynamic occurrence of emissions along the life cycle of products. This dynamic information of emission may be particularly relevant for measuring the transition to a CE in a policy-making context. Future development in technology (e.g. green energy) is gradual, and our methodology potentially allows communication about this gradual development. The visualisation of the emissions in several steps allows benchmarking the reduction of the emissions over time. Nonetheless, the LCI's temporal scope of the case studies did not include the mentioned changes in technological development; hence, future development of the case study could be about studying the effects of energy use and technology improvement. Moreover, although we show the occurrence of emissions, we did not account for temporal aspects of the environmental impacts' characterisation factors (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). 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 Huysveld 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 article, for in-use occupation and resource efficiency. Particularly important would be to expand the analysis of the occupation of critical raw materials in EEE.

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 article, 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 over 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 and instead analysing more cycles over longer periods.

The methodology in this paper 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. (a) 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; we did not account, however, for temporal aspects of the environmental impacts, e.g. 100a or 500a climate change. (b) , 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. By taking this approach, our methodology tones down the discussion about who should be responsible for the impacts (e.g. waste producer vs waste recycler). 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.

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