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Resource and environmental impacts of using second-hand laptop computers: A case study of commercial reuse

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ABSTRACT

The circular economy is proposed to reduce environmental impact, but as yet, there is limited empirical evidence of this sort from studying real, commercial circular economy business cases. This study investigates the environmental impacts of using second-hand laptops, mediated by a commercial reuse operation, instead of new ones. The method used is life cycle assessment (LCA) and special attention is given to laptops' metal resource use by using several complementary life cycle impact assessment methods. The results show that all activities required to enable reuse of laptops are negligible, despite the reuse company's large geographical scope. Two principal features of reuse reduce environmental impacts. Firstly, use extension reduces all impacts considerably since there are large embedded impacts in components. Secondly, the reuse company steers non-reusable laptops into state-of-the-art recycling. This provides additional impact reductions, especially with regards to toxicity and metal resource use. The results for metal resource use however diverge between LCIA methods in terms of highlighted metals which, in turn, affects the degree of impact reduction. LCIA methods that characterise functionally recycled metals as important, result in larger impact reduction, since these emphasise the merits of steering flows into state-of-the-art recycling. The study thus demonstrates how using second-hand laptops, mediated by a commercial reuse operation, compared to new ones, in practice, reduces different types of environmental impact through synergistic relationships between reuse and recycling. Moreover, it illustrates how the choice of LCIA method can influence interpretations of metal resource use impacts when applying circular economy measures to information and communication technologies (ICT).

1. Introduction

Production of information and communication technology (ICT), such as laptop and desktop computers, tablets and smartphones, draws on energy and resource intensive manufacturing processes (Williams et al., 2002) as well as a broad spectrum of metal resources (Graedel et al., 2015; NRC, 2008) that may be considered scarce due to different types of constraints, e.g., economic (Tilton, 2010), social (Fitzpatrick et al., 2014) geochemical (Skinner, 1979) and geopolitical (EC, 2014; NRC, 2008). ICT is also characterised by rapid technological development, for instance in terms of computational power and memory capacity. A downside of rapid technological development is the risk of premature obsolescence and underutilised lifetimes (Proske et al., 2016b). As a result of the omnipresence and often underutilised lifetimes of ICT, there are large volumes of ICT products that can be "mined" for extended use or material recovery in accordance with the vision of a circular economy (CE). CE can be described as a new framing around pre-existing concepts that aim to extend the productive life of resources through measures such as long-life design, reuse, repair, remanufacturing and recycling (Blomsma and Brennan, 2017; Ghisellini et al., 2016). Such measures as a collective will henceforth be referred to as CE measures. Considering the energy and resource intensive ICT production, applying CE measures could be a fruitful way of reducing the environmental impact of ICT use in general (Williams et al., 2002). With specific regards to contained metals, most are currently not functionally recycled at end-of-life (Graedel et al., 2011; Prakash et al., 2016). CE measures such as reuse and repair could thereby reduce the anthropogenic losses of scarce metals by extending the lifetimes of ICT.

Most environmental assessments of reuse, repair and remanufacturing of ICT have been limited to either energy or material efficiency (André et al., 2016; Ljunggren Söderman and André, 2018; Quariguasi-Frota-Neto and Bloemhof, 2012; Sahni et al., 2010;

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Williams and Sasaki, 2003). Only a few (e.g. Prakash et al., 2016; Proske et al., 2016a, Zink et al., 2014) have adopted a wider environmental scope, using for instance life cycle assessment (LCA). LCA is a methodology that aims to systematically account for all the relevant environmental impacts of a product or process occurring from the cradle to the grave (ISO, 2006). LCA has been argued a suitable tool for assessing environmental impact of CE measures as it may reveal burden shifting between life cycle stages and trade-offs between types of environmental impacts (Haupt and Zschokke, 2017). While the mentioned LCAs have for instance indicated environmental benefits of extending product lifetimes, this study elaborates on such findings by addressing two principal research gaps.

The first one concerns the impact of CE measures on metal resource use, which is relevant to address in more detail considering the diversity of metals in ICT. In LCA, there are differing perceptions on what constitutes the environmental problem with regards to metal resource use (Sonderegger et al., 2017; Steen, 2006). In short, it can be formulated as being a question of limits to availability caused by geological scarcity or exergy, economic cost or environmental impacts of extraction and production (Sonderegger et al., 2017; Steen, 2006). These different perceptions are represented in the rationales of commonly used life cycle impact assessment (LCIA) methods. As a consequence, LCIA methods for metal resource use result in highly different results (Finnveden et al., 2016; Peters and Weil, 2016; Rigamonti et al., 2016; Rørbech et al., 2014) emphasising specific resources or resource types (Klinglmair et al., 2014). Hence, a more comprehensive assessment of metal resource use could utilise several complementary LCIA-methods (Finnveden et al., 2016; Klinglmair et al., 2014). Rigamonti et al. (2016) studied the effect of LCIA-methods for the interpretation of toxicity and metal resource use impacts of recycling waste electrical and electronic equipment (WEEE). They conclude that utilising several complementary LCIA-methods to assess the effects of CE measures is useful as a sensitivity analysis of metal resource use and suggest that LCA may provide insights concerning the potential of CE to avoid different types of environmental impacts (Rigamonti et al., 2016). However, no study has used complementary methods on metal resource use in environmental assessments of CE measures other than recycling.

The second research gap relates to the absence of studies examining the environmental impact of real-world, commercially existing business cases of CE measures. Several researchers have argued a need for such studies (Blomsma and Brennan, 2017; Geissdoerfer et al., 2017; Korhonen et al., 2018). For instance, Blomsma and Brennan (2017) suggest that it could reveal potential synergies between cases' constituent CE measures, such as between reuse and recycling.

Motivated by these research gaps, this study uses LCA to investigate the environmental impact of a real-world business case of CE, primarily with regards to metal resource use. It is based on a Swedish company that sources discarded professional-use laptops from companies and resells them to other companies, public sector and private users. To our knowledge, this is the first study of ICT reuse based on real data for the inventory of preparation for reuse. The study thereby aims to serve as a useful illustration of the practical results of CE measures applied to ICT. More specifically, the study investigates how using second-hand laptops, mediated by a commercial reuse operation, compare to using new laptops in terms of environmental impact, in particular, metal resource use.

2. Literature on life cycle assessment of reuse of ICT

In order to compare the use of second-hand and new ICT, environmental impacts over the full life cycle from cradle to grave are relevant to consider. Also, the difference in functionality between new and second-hand ICT due to technological development, as well as the extension of lifetime made possible through reuse, are important aspects. This section provides a literature background to such aspects, serving as a basis for the scope, modelling, data collection and assumptions in the study.

LCAs of ICT indicate that production and use dominate life cycle impacts (Andrae and Andersen, 2010; Arushanyan et al., 2014; Subramanian and Yung, 2016). Recycling of ICT has generally small benefits in terms of climate change but more significant benefits for impacts related to resources and human health (Eugster et al., 2007; Proske et al., 2016a). Most LCAs of ICT tend to focus on climate change and energy use, deprioritizing relevant impacts related to e.g. toxicity (Arushanyan, 2013). Environmental impacts from ICT production are to a large extent linked to printed circuit boards (PCB) (Choi et al., 2006; Deng et al., 2011; Duan et al., 2009; Eugster et al., 2007) and to a lesser extent to liquid crystal display (LCD) (Andrae and Andersen, 2010; Ercan et al., 2016; Zgola, 2011). In particular, production of integrated circuits (IC) mounted on PCBs is consistently reported as environmentally burdensome (Andrae and Andersen, 2010; Eugster et al., 2007; Kasulaitis et al., 2015; Liu et al., 2011; Williams et al., 2002) due to use of cleanroom conditions, high-purity silicon and chemicals, and perfluorinated compounds which are highly potent greenhouse gases (Boyd, 2012; Liu et al., 2011). Energy consumption and climate change impacts of IC production have decreased steadily over time per basis of computational power, but on the other hand, computational power per laptop has increased (Boyd, 2012). However, literature does not provide any conclusions on what the net effect of efficiency and increased functionality is over time for environmental impact of laptop production.

ICs are a source of variation in ICT LCAs (Teehan and Kandlikar, 2012). This could partly relate to real variation but also modelling uncertainties. Modelling uncertainties regard, for instance, the use of chemicals (Boyd, 2012; Plepys, 2004a,b; Williams et al., 2002) and estimates of semiconductor area, which is the relevant parameter for ICs' environmental impact (Kasulaitis et al., 2015; Liu et al., 2011; Proske et al., 2016a; Teehan and Kandlikar, 2012). Another modelling challenge regards access to representative data. Because of the complexity of ICT and production processes, many LCAs of ICT depend on electronics data from large databases such as Ecoinvent database (Hischier et al., 2007; Wernet et al., 2016). The electronics data in Ecoinvent was collected in the beginning of the 2000 s. A board inspection indicates that a significant miniaturization of surface-mounted devices (SMDs) has occurred since the creation of respective Ecoinvent production processes (Nordelöf and Alatalo, 2017). On the other hand, the material content of key laptop components, such as motherboards, have been indicated to be fairly constant per laptop over time since efficiency gains and increased functionality tend to balance out (Kasulaitis et al., 2015).

An important modelling aspect regarding the use phase is the effect of technological development, predominantly in terms of energy-efficiency and functionality. While new products may be more energy-efficient per computational power (Boyd, 2012), they are also likely to be more energy consuming in total (Boyd, 2012; Deng et al., 2011; Schischke et al., 2003). Consequently, when it comes to reuse, assumptions on these parameters can lead to conclusions of both increased or decreased environmental impact (Bakker et al., 2014; Deng et al., 2011; Prakash et al., 2016; Sahni et al., 2010; Schischke et al., 2003). Moreover, it has been argued that environmental assessments of reuse may insufficiently assess displacement since second-hand products may not be functionally equivalent to new ones (Cooper and Gutowski, 2017). Products are functionally equivalent (Vadenbo et al., 2017) or approximately equivalent (Wolf et al., 2010), if they share a set of obligatory properties including the main function (Ekvall and Weidema, 2004). This may not be the case when technological development is rapid or when products are reused for new purposes (Zink et al., 2014). However, in an LCA of reuse of computers, Schischke et al. (2003) argued for functional equivalence between computer generations over a period of six years, since functionality is a matter of individual preferences and that many applications do not require the latest functionality improvements.

Another important aspect for studying environmental impact of reuse are total lifetimes, including first and second use. Observations of this sort vary in the literature, partly due to setting, and are therefore not directly transferable to the specific case of commercial reuse studied in this paper. Nevertheless, first uses of around 3–5 years are commonly reported (Prakash et al., 2016; Thiébaud et al., 2017; Williams and Hatanaka, 2005) and second uses have been suggested around 2–3 years (Prakash et al., 2016; Thiébaud et al., 2017).

Regarding end-of-life (EoL) fates of ICT, there are substantial uncertainties due to factors such as hibernation and informal pathways (Arushanyan, 2013; Buchert et al., 2012; Ongondo et al., 2011; Van Eygen et al., 2016; Williams et al., 2008) which makes it difficult to model the life cycles to the grave. In a north-European setting, about half of laptops are collected and sent to WEEE recycling but as yet, no reliable estimates on the other pathways exist (Buchert et al., 2012). Literature however suggests that it would be a clear overestimation to assume that all laptops are collected for WEEE recycling (Ongondo et al., 2011). LCAs of repair and refurbishment of smartphones, which have significantly lower collection rates (Navazo et al., 2014; Zink et al., 2014) have either assumed that all are collected (Proske et al., 2016a) or compared options for already collected smartphones (Zink et al., 2014).

3. Materials and methods

The real reuse operation studied was based on Inrego AB, a Swedish resale and refurbishment company that acquires discarded high-grade and professional-use ICT equipment, predominantly laptops (about two-hundred thousand per year), from companies and finds new users in the public sector, other companies and private users. This type of operation is described as "IT Asset management" in a typology of reuse operating models for electric and electronic equipment (Kissling et al., 2012). The supplying companies tend to replace entire fleets of laptops at a three-year interval due to aspects such as maintenance efficiency rather than functional obsolescence (Pettersson, 2017).[1] As a result, laptops have typically been used for three years and little more than two thirds can be reused without requiring any spare parts (Pettersson, 2017). These are sold in a condition that resembles "as-new" with a minimum warranty of one year (Pettersson, 2017). The principal activities of preparation for reuse consist of sourcing and transport from supplying companies, sorting, testing, data erasure, resale and transport to customers. Transportation utilises airplane, freight ship and light and heavy-duty trucks. Laptops are mostly sourced in Sweden while customers are predominantly located within Europe but also in Asia (Pettersson, 2017).

3.1. Goal and scope

The goal of the LCA study was to compare using second-hand laptops, mediated by a commercial reuse operation, with using new laptops. The functional unit (FU) was "one year of access to a laptop computer". This FU was motivated since the second-hand laptops can be argued functionally equivalent to new laptops considering the high-quality, "as-new" condition and given warranties. In addition, it was motivated by the argument that functionality of laptops is a subjective matter (Schischke et al., 2003). Given that material contents of key laptop components such as the motherboard have been demonstrated to be fairly constant per laptop over time (Kasulaitis et al., 2015), one laptop configuration could be chosen to represent both new and second-hand laptops in the two alternatives. With regards to the aim of studying a real-world commercial reuse case it was purposive to base the study on a case-specific laptop, chosen by the company as representative for the typical laptop handled in their operation (see SI, section 1), rather than a generic laptop from e.g. Ecoinvent (Wernet et al., 2016). Based on the literature review regarding highly contributing components as well as the specific focus on metal resource use, the scope was delimited to the casing, LCD, light-emitting diode (LED) backlights, cables connected to the motherboard and a number of printed circuit boards (PCBs),[2] specifically the motherboard containing the central processing unit, the dynamic random-access memory, the LCD driver and the solid-state drive. The components included were deemed to represent the majority and diversity of life cycle environmental impacts of a complete laptop: some are characterised by high complexity and the use of multiple metals, e.g. integrated circuits (ICs) (Kasulaitis et al., 2015; Proske et al., 2016a), LED backlights (Deubzer et al., 2012) and LCDs (Andrae and Andersen, 2010; Deubzer et al., 2012), while components such as the casing consist of more abundant metals used in greater quantities (Kasulaitis et al., 2015). All other components, such as battery, loudspeakers, camera, power cord and keyboard were excluded.

In the second-hand laptop alternative, the lifetime of 70% of laptops discarded by first users can be doubled through resale as second-hand laptops (Fig. 1). In this way, the FU can be fulfilled using less laptops compared to the new laptop alternative, where they are used for three years (Pettersson, 2017, Prakash et al., 2016, Williams & Hatanaka, 2005). The assumption of a three-year use extension is indicated by the case company (Pettersson, 2017) but other studies also suggest similar figures (Prakash et al., 2016; Thiébaud et al., 2017). In this specific case, it is furthermore motivated by the high-quality, "as-new" condition and given warranties. This assumption, used as the baseline scenario (Sections 4.1 and 4.2), is tested in a sensitivity analysis (Section 4.3). Considering the assumed functional equivalence of laptops, the use phases are identical in both alternatives and do not affect the comparison. To highlight the differences between the alternatives, the results are presented without the contribution of the use phase. The alternatives also differ in the overall collection rate into recycling. In the new laptop alternative, the collection rate from users of 50% (Buchert et al., 2012) is applicable to all laptops. In the second-hand laptop alternative, the 50% collection rate is applicable after second-hand use. After the first use, 30% of laptops are deemed non-reusable by Inrego and sent to WEEE recycling at a collection rate of 100%. As a result, a larger share of laptops reaches WEEE recycling in the second-hand laptop alternative (65% compared to 50%). Other EoL fates are highly uncertain. Therefore, laptops not collected for recycling were modelled to be disposed in a controlled landfill, although this is not necessarily the actual pathway. Nevertheless, this reflects a characteristic difference between the compared alternatives, namely the collection rates into WEEE recycling. In addition, it avoids overestimating both positive and negative impacts, in terms of recycled materials, on the one hand, and highly impacting informal EoL handling, on the other hand.

[1] Personal communication, Inrego, Erik Pettersson, 2017.

[2] PCB and PCB panel refer to printed circuit board panel including and excluding SMDs, respectively.

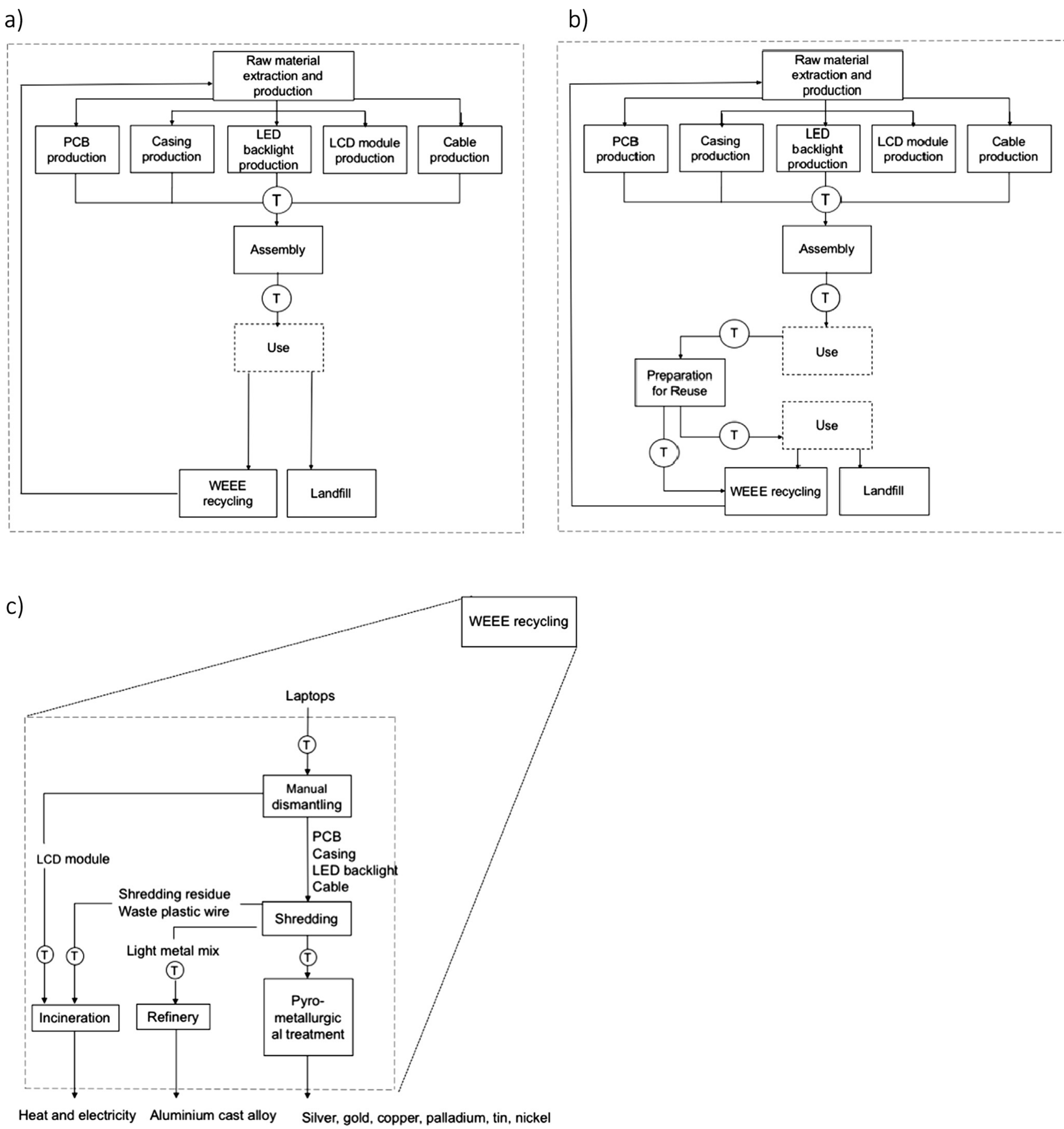

Fig. 1. Flowcharts of: (a) new laptop alternative, (b) second-hand laptop alternative and (c) WEEE recycling.

In order to reflect the difference in collection rates, and the enabling of increased functional recycling of metals,[3] it was purposive to use the avoided burden approach for the EoL modelling, i.e. using a closed loop approximation for the recycling process. This accounts for that materials and energy recovered at EoL can displace primary resource materials and other energy production (Frischknecht, 2010). For methodological stringency, it would have been ideal to model the production as completely composed of primary production but this was not possible since a large extent of the modelling had to rely on the Ecoinvent database due to lack of other relevant data. The avoided burden approach is inconsistent with the Ecoinvent inventory data models (Guérin, 2017; Wernet et al., 2016), where the burden of producing secondary materials is either allocated to the first user of the material (cut-off), or shared between users of the material (allocation at the point of substitution). Therefore, in order to apply the avoided burden approach in a way that was consistent with the used Ecoinvent production data, the cut-off version of Ecoinvent processes was used and the avoided burden approach was applied only to the primary inputs of the laptop. This avoided double-counting the benefits of recycling and maintained mass balance (SI Section 1.4).

[3] Functional recycling refers to recycling where the elemental properties of metals (e.g. physical or chemical) are utilised in a new metal cycle (Graedel et al., 2011; Guinée et al., 1999).

Table 1
Inventory: laptop composition and process data sources.

Life cycle activity	Composition data source	Process data source
Printed circuit boards production		
Printed circuit board panel, surface mounted, incl. conformal coating	Disassembly	Nordelöf (2018), Hischier et al. (2007)
Solder paste, lead-free	Disassembly, Deubzer (2007); Nordelöf and Alatalo (2017)	Nordelöf (2018), Hischier et al. (2007)
Integrated circuit	Disassembly, Kasulaitis et al. (2015), [dataset] Infineon Technologies (2013a,b,d,e,f,g, 2014, 2015) Parade Technologies, personal communication (2017)	Boyd (2012), Andrae and Andersen (2011), Proske et al. (2016a), Schmidt et al. (2012)
Capacitors, SMD and tantalum types	Disassembly, Nordelöf and Alatalo (2017)	Hischier et al. (2007)
Electric connector	Disassembly, Nordelöf and Alatalo (2017)	Hischier et al. (2007)
Inductor	Disassembly Hischier et al. (2007)	Hischier et al. (2007)
Resistor	Disassembly, Nordelöf and Alatalo (2017)	Hischier et al. (2007)
Transistor	Nordelöf and Alatalo (2017), Infineon Technologies (2013c)	Boyd (2012)
Casing production	Disassembly, Kasulaitis et al. (2015), Kahhat et al. (2011), Hewlett-Packard (2016)	Hischier et al. (2007)
LED backlight production	Deubzer et al. (2012)	Scholand and Dillon (2012), Deubzer et al. (2012), Nordelöf et al. (2018), de Souza (2013), Tähkämö (2013), AUO (2013)
LCD module production	Disassembly	
Indium tin oxide	Disassembly, Arvidsson et al. (2016)	Hischier et al. (2007)
Cable to motherboard production	Disassembly	Hischier et al. (2007)
Transportation	Hewlett-Packard (2016)	
Assembly		Deng et al. (2011)
Preparation for reuse	Pettersson (2017)	Pettersson (2017)
End-of-life treatment		Hischier et al. (2007), Navazo et al. (2014) Stena Recycling, personal communication (2019)

Economic allocation was applied to multi-output metal production processes as provided by Ecoinvent (Classen et al., 2009; Wernet et al., 2016).

The LCA includes various environmental impact categories. In light of the identified research gaps and literature review, the most relevant impact categories to include in the article were considered to be climate change (100a) (Pachauri et al., 2014; Wolf et al., 2010), human toxicity (Rosenbaum et al., 2008) and metal resource use (see Section 3.3). A complete presentation of impact categories according to the International Reference Life Cycle Data (ILCD) Handbook recommendation (European Commission, 2012; Wolf et al., 2010) can be found in SI, Section 3.

3.2. Life cycle inventory

The life cycle inventory data was based on manual laptop disassembly conducted at Inrego combined with literature sources, the Ecoinvent database (Wernet et al., 2016) and information provided by Inrego (Pettersson, 2017). During the disassembly, components were identified, weighed and measured to establish the product composition of the representative laptop (Table 1). Each component was matched with cradle-to-gate data for its production, regarded as the best available data for representation of a modern laptop. For instance, this included accounting for the miniaturization of SMDs that has occurred since the creation of respective Ecoinvent production processes (Nordelöf, 2018; Nordelöf and Alatalo, 2017). Data and assumptions for component production, laptop assembly, transportation, preparation for reuse and EoL treatment are provided in SI Section 1.

3.3. Life cycle impact assessment of metal resource use

In the impact assessment step in an LCA, characterisation factors (CF) are used to calculate the contribution of emissions and resources to environmental impact categories, related to three Areas of Protection: human health, ecosystem quality and natural resources. There are different perceptions as to what constitutes the environmental problem of using non-renewable natural resources such as metals, and consequently, different types of indicators (Sonderegger et al., 2017; Steen, 2006). Four indicator approaches are commonly distinguished (Sonderegger et al., 2017; Steen, 2006):

- 1. Exergy or solar energy consumption.
- 2. Use in relation to resources (e.g. average crustal concentration or economic reserves).
- 3. Increased future environmental impacts or economic costs of mining and material production, due to decreasing ore grades.
- 4. Aggregated mass or energy consumption.

Several terms are used interchangeably in literature to describe what is assessed by these LCIA-methods, such as use, consumption and depletion (Finnveden et al., 2016). For consistency, the term *metal resource use* is used in this paper. Five LCIA-methods assessing metal resource use are used, each representing one of the first three approaches, while the fourth was omitted due to low support (Sonderegger et al., 2017; Steen, 2006). The purpose of this was to compare using second-hand laptops and new laptops from several perceptions on the environmental problem of metal resource use.

As such, the selection of LCIA-methods was based on representation of different problem perceptions and is not exhaustive. These LCIA-methods and their basic principles are briefly presented below. Because of different problem perceptions, indicator approaches and data, CFs vary significantly between methods (see Supplementary information (SI), Section 2, Table S3[4]). Two of the selected methods, EcoSc and EPS, also include mineral and fossil resources, but such flows were excluded for the sake of comparability.

Cumulative Exergy demand (CExD) of metal resources is based on the chemical exergy consumed by a product or process (Bösch et al., 2006). The rationale for considering exergy consumption as a major problem of resource use departs from the second law of thermodynamics, making exergy the scarce resource as opposed to metals per se (Bösch et al., 2006; Finnveden et al., 2016).

The abiotic depletion potential method developed by Centrum voor Milieuwetenschappen Leiden (CML) relates extraction rates to three resource classes used as approximations of what might be ultimately extractable (Van Oers et al., 2002). CFs are derived by the ratio of extraction rates to the amount of a given resource squared, in relation to a reference resource. CML-ultimate reserves (CML-UR) uses the average crustal concentration (Guinée and Heijungs, 1995). CML reserve base[5] (CML-RB) and CML economic reserves[6] (CML-ER) use data from the US Geological Survey (USGS) for the reference year 1999 (Van Oers et al., 2002).

The ecological scarcity method (EcoSc 2013) (Frischknecht, 2013) is based on distance-to-target: the present situation in relation to environmental protection policy. Since there is no established political target for mineral and metal resource extraction, this has been temporarily set to maintenance of current extraction levels (Frischknecht, 2013; Peters and Weil, 2016). CFs are derived by the ratio of extraction rates to the available reserves squared, like the CML-RB approach, but with reference to USGS data from 2011.

Environmental Priority Strategies (EPS) (Steen, 1999a,b) bases CFs on the willingness-to-pay (WTP) for sustainably produced resources. It is based on a hypothetical market in which the current generation and all future ones bid on resources that are sustainably produced from large and dilute sources such as common bedrock or seawater. It is assumed that the WTP for resources will approximate sustainable production costs which, in turn, are assumed to be inversely proportional to average crustal concentrations.

ReCiPe midpoint (Goedkoop et al., 2009) is based on estimates of increased efforts for future resource extraction taking decreasing ore grades into account. CFs are calculated as the increased economic marginal cost of extracting resources from declining ore grades caused by extraction, derived from USGS deposit data from 1997.

4. Results

The results show clear benefits of using second-hand laptops for all included environmental impact categories (Figs. 2a–b and 3, Fig. S1a–m). Despite that Inrego's average resale distance extends beyond Europe, activities in preparation for reuse have almost no comparable impact. Consequently, the results for each impact category essentially depends on two key features: use extension which reduces the need for primary laptop production and steering of flows into recycling which reduces the need for primary metal production.

Since preparation for reuse is mostly negligible, the contribution of use extension to the total impact reduction for any impact category depends on the reuse efficiency (70%) and the duration of use extension. Therefore, use extension results in a quite constant reduction across all impact categories. In the baseline scenario this reduction is about 41% in comparison to using new laptops.

The feature of steering flows into recycling is a result of the perfect collection rate into WEEE recycling that applies to the laptops that are deemed non-reusable by the company. In absolute quantity, the displacement of primary metal production is smaller for the second-hand laptop alternative since somewhat less material is recycled per year of laptop use (noticeable for e.g. human toxicity, Fig. 2b). However, since a larger share of laptops reaches recycling in this alternative, a larger proportion of the production impacts are offset. The merits of this feature vary between impact categories. For impact categories where extraction and production of functionally recycled metals constitute a considerable share of total life cycle impacts, this feature is considerable. Such impact categories are human toxicity and metal resource use according to non-reserve based LCIA-methods. For other impact categories, such as climate change and metal resource use according to reserve-based LCIA-methods, the merits of steering flows into recycling are small. Altogether, environmental impacts of using second-hand laptops are between 39% and 50% lower than that of new laptops (Figs. 2a–b and 3, Fig. S1a–k).

As regards laptop component contributions, most impact categories are dominated by the production of PCBs and the casing, while those of the LED backlights and the LCD module are less contributing. The impact of cable is negligible.

4.1. Climate change and human toxicity

For climate change (Fig. 2a), the ICs on the PCBs are responsible for about a third of all impact. This is primarily caused by process electricity but also production of infrastructure and high-purity silicon (Boyd, 2012). For the casing, the contribution to climate change derives from the production of a relatively large mass of magnesium alloy. The smaller contribution of the LCD and backlight LEDs is mainly attributed to energy use. Overall, fossil carbon dioxide is the emission contributing most to climate change. The perfluorinated compound, SF_6, used as a protective gas in magnesium-alloy die casting, also contributes a few percent while its uses in LCD, LED and IC production are negligible. The benefit of EoL treatment is relatively small as regards climate change, since it does not offset the dominant impacts in IC production, nor the casing since magnesium is not functionally recycled.

Emissions causing human toxicity are emitted predominantly during primary production of metals (Fig. 2b). Production of gold used in ICs and connectors causes about 80% of human toxicity impacts and production of copper and magnesium contributes a few percent each. The most contributing emissions are zinc, arsenic and chromium to water. Due to the dominant contribution from production of functionally recyclable metals, in particular gold but also copper, impacts are greatly offset by recycling – half in the new laptop alternative and 75% in the second-hand laptop alternative.

4.2. Metal resource use

The value of using complementary LCIA-methods on a real case of commercial laptop reuse is manifested firstly in the contribution of different metals between methods (Fig. 3a). These differences in turn influence the importance of steering flows into WEEE recycling, and hence the comparative reduction of metal resource use. The reasons why individual metals are important in specific LCIA-methods are accounted for in more detail in Sections 4.2.1–

[4] All references to tables and figures in SI are denoted with S.

[5] Resources that have a reasonable potential to become economically and technically available (Van Oers et al., 2002).

[6] Part of the reserve base which could be economically extracted or produced at the time of determination (Van Oers et al., 2002).

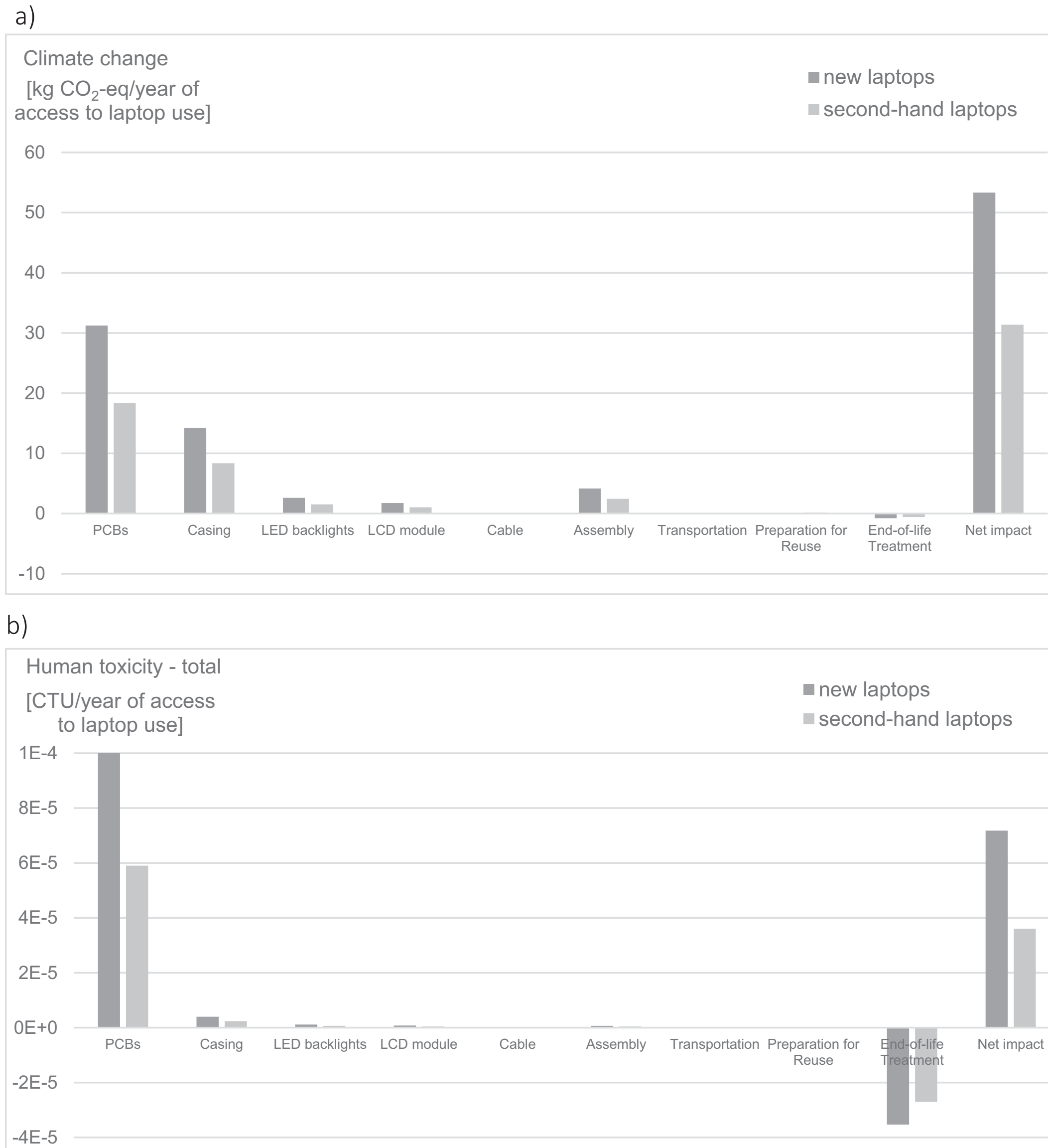

Fig. 2. Dominance analysis of activities in the two alternatives: (a) climate change [kg CO_2-eq.], (b) human toxicity – total impact [comparative toxic units (CTU)]. Components = from raw material extraction to finished component. Assembly = from finished components to finished laptop. Transportation = transports in production, distribution of new products and at EoL. Preparation for reuse = Inrego's activities as listed in Section 3. EoL = treatment and displaced primary material and energy production.

4.2.4. The insights from the entire Section 4.2 and their implications are discussed in Section 5.2 with regards to assessment of metal resource use of CE measures applied to ICT.

In Fig. 3a, it can be observed that some metals are consistently important regardless of problem perception, indicator and data, while other metals are important only according to some aspect. Ten metals contribute notably to the results in at least one method: gold, silver, palladium, platinum, indium, cadmium, lead, tantalum, tin and copper. Their shares differ significantly between LCIA-methods, but gold clearly contributes in all. Tin and copper also contribute notably in all methods, but to a lesser extent.

Analogous to the other environmental impact categories, use extension results in a constant reduction of metal resource use across LCIA-methods (41%), while additional reductions (1–9%) derive from the feature of steering of flows into recycling (Fig. 3a) resulting in a total reduction of 42–50% (Fig. 3a). The merits of steering flows into recycling depend on the LCIA-methods' characterisation of individual metals, and specifically, functionally recycled metals. LCIA-methods such as CML-UR, EPS and ReCiPe that give more importance to gold in particular, but also copper, tin, silver and palladium, credit this feature more (Fig. 3a and b). Furthermore, Fig. 3b demonstrates that as a result of this feature, a greater share of metal resource use is being recycled per FU in the second-hand laptop alternative. With these LCIA-methods specifically, 38–44% of laptops' metal resource use is functionally recycled in the second-hand laptop alternative (Fig. 3b). With other LCIA-methods, that give high importance to metals such as tantalum and indium, which are not functionally recycled, the merits of this feature are not as significant. In other words, the metals that benefit most from this reuse operation are functionally recycled metals, as the company's routine of steering flows into recycling decreases their losses in addition to use extension.

Fig. 3. (a) Metal resource use per FU: the second-hand laptop alternative (right) compared to the new laptop alternative (left) [%] with five impact assessment methods of which one in three versions. Metals with >1.5% contribution with at least two methods or >4% with at least one method are displayed individually. Others include 20 metals such as iron, magnesium, nickel and rare earth elements. (b) Avoided metal resource use from functionally recycled metals per FU: the second-hand laptop alternative (right) compared to the new laptop alternative (left) [%] with five impact assessment methods of which one in three versions.

4.2.1. Differences in impact of precious metals

Gold is significant in all methods, silver in some, while palladium and platinum are only shown to be important in one method. Gold is rare in terms of average crustal concentrations but has deposits with significantly higher concentrations (Ayres and Peiró, 2013). Consequently, gold is given much weight in CML-UR and EPS, which base CFs on average crustal concentrations. Conversely, gold is given comparably less weight with methods based on exergy and reserves, CExD, CML-RB, CML-ER and EcoSc, since concentrated deposits are extractable at low exergy costs making resources economically extractable, thus, creating reserves. In contrast to gold, silver is more important in reserve-based methods than in others. Compared to other methods, ReCiPe has lower disparity between precious metals and more abundant and highly extracted resources such as copper and tin. Silver even has a lower CF than tin (SI, S1). Palladium and platinum only contribute noticeably with EPS, due to low average crustal concentrations. CML-UR differs from EPS both because of the dependence on the extraction rate and an estimated average crustal palladium concentration that is twenty-five times higher than in EPS (Guinée, 1995; Steen, 1999b).

4.2.2. Differences in impact of zinc by-products

Indium is given much weight with reserve-based methods such as CML-RB, CML-ER and EcoSc. Cadmium is also of importance according to the same methods as well as CML-UR and EPS. Lead has a similar pattern, but smaller contributions.

The contributions of these three metals depend on economic allocation of multi-output extraction processes as applied in Ecoinvent (Wernet et al., 2016), i.e. a large share of all environmental impacts including resource use are assigned to zinc. Interestingly, it is therefore not the indium content in the LCD screen that causes the main input of indium in the studied life cycles but zinc in various components.

4.2.3. Differences in impact of tantalum and tin

Tantalum is the major contributor to metal resource use according to the CExD method and is also important using CML-ER, CML-RB and EcoSc. The latter methods give very similar results because of analogous approaches but slightly differing data with regard to the resource and the year represented. The main differences are the contributions of tantalum and tin which is likely explained by a lower tantalum extraction rate (Sverdrup et al., 2017) in the more recent data used in EcoSc. The dominance of tantalum in CExD reflects high exergy demand for metals with low deposit concentrations (Bösch et al., 2006). In contrast, tantalum contributes very little with CML-UR and EPS, due to a quite ordinary average crustal concentration (Guinée, 1995; Steen, 1999b). ReCiPe lacks a CF for tantalum, but instead gives much weight to tin (Goedkoop et al., 2009).

4.2.4. Differences in impact of copper

Copper use has a noticeable impact according to all methods, and especially with ReCiPe as this method places higher value on more abundant and highly extracted metals. On the other hand, large copper reserves counterweigh a fairly high extraction rate (Sverdrup et al., 2017) making copper less important in reserve-based methods (CML-RB, CML-ER and EcoSc). The contribution with EPS relating copper use to average crustal concentration, indicates that copper could be a concern in a longer time frame.

4.3. Sensitivity analysis

Considering the uncertainty of the duration of the first and second use, as discussed in Sections 2 and 3.1, the sensitivity of the resulting use extension is analysed, defined as the relation between first and second use. Fig. 4 demonstrates how the observed features for environmental impact reduction of using second-hand laptops compared to using new ones depend on the use extension. Naturally, the benefits of use extension increase with greater use extensions, almost reducing half of the impacts from laptop production if, for instance, first use is three years and second use is four years. However, there is a diminishing marginal benefit of use extension since each extra year of second use constitutes a smaller share of the total product lifetime. Moreover, with increasing use extension the benefits of steering flows into recycling decrease per FU since they are divided over increasing time periods. With no use extension at all, the feature of steering flows into recycling reduces impacts up to 15% for impacts such as human toxicity and metal resource use according to CML-UR. In the case of a moderate use extension, for instance if the second use is one third of the first use, the two features for impact reduction contribute with comparably similar shares to the total reduction of the same impacts. Altogether, the figure demonstrates that the observed importance of the two features for impact reduction is robust.

As regards the sensitivity of the assumption of functional equivalence, an alternative to this assumption could be to account for some measurable aspect of functionality, such as computational power. As described in Section 2, this could affect the inventory of production and use phases, possibly influencing the comparison in both favour of and against the second-hand laptop alternative. However, if functional equivalence cannot be assumed but all inventory is equal, Fig. 4 could delineate the sensitivity of this assumption. For instance, assuming that second-hand laptops only provide 50% of the function "access to laptop computer" compared to new ones, a second-hand laptop being used for three years could conceivably be credited for 1.5 years, which is still favourable compared to the new laptop alternative.

5. Discussion

5.1. Validity of results

Results and conclusions of this commercial reuse case cannot be generalised to all cases of comparing the use of second-hand and new laptops. For example, lower-grade laptops involving shorter second use and mediation through channels with less efficient transportation and collection to recycling could be involved. Also, functionality between second-hand and new laptops may not be comparable. Regarding the scope of the study, excluding some

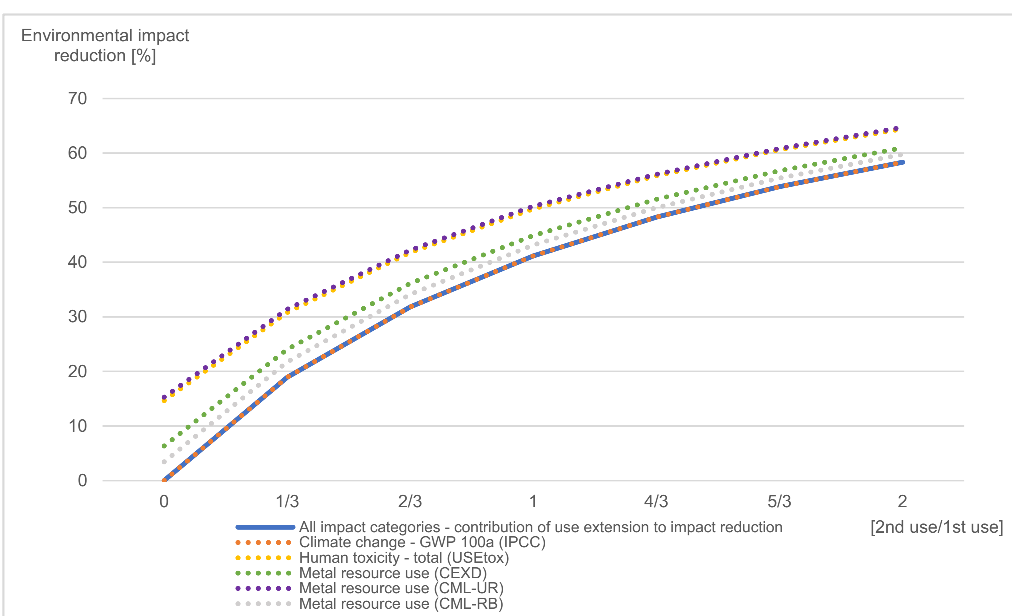

Fig. 4. Sensitivity analysis of use extension, i.e. relation between duration of second and first use, for relative environmental impact reduction from using second-hand laptops compared to new laptops. For simplicity, a few metal resource use impacts are selected but excluded ones have similar patterns. For all impact categories, the contribution from use extension is the same and thus represented by the blue line. Total reduction is represented by dashed coloured lines for each impact category. Consequently, the distance between each dashed coloured line and use extension (blue line) represents the contribution from steering flows into recycling. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

laptop components, is not expected to influence largely on the comparison, but absolute values may be affected. On the other hand, the most environmentally burdensome components according to literature were included. With regards to the modelled EoL fate of laptops not collected for recycling, other assumptions could influence results in both ways. On one hand, the modelled fate probably neglects occurrences of informal reuse occurring after collection to end-of-life treatment, which if accounted for, could benefit the new laptop alternative. On the other hand, it also neglects informal recycling, which if accounted for may increase the impact of this alternative, for example in terms of eco and human toxicity. Increased availability of information on real EoL pathways could enable studying such potential impacts.

5.2. Reduction of metal resource use in a more CE

Using several LCIA-methods does indeed provide complementary perspectives on the metal resource of using second-hand laptops, mediated by a commercial ICT reuse operation, compared to using new laptops. In addition to highlighting different metals in laptops as important, using complementary LCIA-methods is valuable for comprehensively examining the benefits of reuse, since it influences both the distribution of impact reduction over metals and the reduction of impacts.

Considering that significant shares of laptops' metal resource use are recycled in WEEE treatment (Fig. 3b), the focus on gold, silver, copper and tin appears justified not only for economic reasons, but also for reduction of metal resource use (Fig. 3b). This could however resemble that economic value is both a prerequisite for functional recycling and also incorporated in some LCIA-methods to varying extent, for instance through the use of deposit data, even though they are claimed to be independent of economic and technological developments. Analysing correlations between market conditions and characterization factors could thus deserve attention in future research.

Nevertheless, functionally recyclable metals still contribute considerably to metal resource use due to, primarily, low collection rates. Thus, increasing collection rates into WEEE recycling could greatly increase the share of metal resource use that gets functionally recycled. The observed feature of steering flows into recycling may be prevalent in other cases of commercial CE measures. As such, increased implementation of CE measures similar to this case could potentially serve to increase collection rates and thereby further reduce metal resource use. It needs to be stressed that this observed feature is only valid for actors who are serious with handling flows in a controlled and efficient manner.

However, there are also metals with significant impacts, e.g. indium and tantalum, that are not currently functionally recycled in WEEE treatment, for which use extension is currently the only mitigating feature. Considering that there are inevitably limits to product lifetimes it is necessary for a CE to be adapted to the material diversity of ICT. The possibility of improving recycling practices to correspond to the material diversity of ICT products is however limited for several reasons and may require a more flexible and product-centric recycling system (Reuter et al., 2013). The possibility to substitute scarce metals for functionally recyclable or more abundant ones could therefore be interesting. While the often specific properties of metals make this challenging (Graedel et al., 2014; Sverdrup et al., 2017), substitution by carbon nanomaterials could be possible e.g. for indium in LCD screens and tantalum in capacitors (Arvidsson et al., 2016; Arvidsson and Sandén, 2017). On the other hand, such efforts may not be effective even in terms of metal resource use depending on the used LCIA-method. Using EPS, substituting indium with graphene in LCD screens resulted in higher metal resource use due to copper use in graphene production (Arvidsson et al., 2016). This illustrates that choice of LCIA-method can influence conclusions about CE measures.

Another example where the choice of LCIA-method is potentially relevant for the conclusions is a case of a modular smartphone design. Using CML-UR, this design was not beneficial in terms of metal resource use compared to a non-modular smartphone, due to an increased use of gold connectors between modules (Proske et al., 2016a). Considering that CML-UR has a particularly high CF for gold however, it cannot be excluded that it could have been preferable using other LCIA-methods. The influential role of LCIA-method for interpretations and conclusions regarding CE measures, as exemplified in both this case and the above discussed studies (Arvidsson et al., 2016; Proske et al., 2016a), strengthens the case for selecting purposive and complementary LCIA-methods so that assessments of CE measures applied to ICT are comprehensive with respect to relevant aspects of metal resource use.

6. Conclusions

This study solidifies the indicated environmental benefits of laptop reuse (Bakker et al., 2014; Deng et al., 2011; Prakash et al., 2016) and expands on them by demonstrating how environmental impacts are reduced in practice, by using second-hand laptops mediated by a real commercial reuse company, instead of new ones.

Despite the case company's handling of non-reusable laptops and large geographical scope, all transportation and other activities required to enable reuse of laptops were environmentally negligible. The risk of CE measures to cause burden-shifting (Haupt and Zschokke, 2017) can thus be concluded minor in this and similar cases of commercial ICT reuse. However, key conditions for such conclusions cannot be assumed for reuse in general, e.g. high-grade laptops, efficient transportation and responsible handling of non-reusable products into WEEE recycling.

Moreover, studying a real implementation of CE in practice did indeed reveal synergies between reuse and recycling, as hypothesised by Blomsma and Brennan (2017) and shown by Ljunggren Söderman and André (2018). Environmental impact reduction of commercial reuse is principally attributed to two key features: use extension and steering product flows into WEEE recycling. Because activities in preparation for reuse were negligible in almost all impact categories, the feature of use extension reduces environmental impacts to the same extent. This feature considerably reduces impacts, and constitutes the majority of total impact reduction even if second uses are relatively short. In addition, the feature of steering flows into WEEE recycling further reduces environmental impacts. This feature was found to be particularly important for reducing the environmental impacts to which primary production of functionally recycled metals contribute considerably. As such, environmental impacts such as human toxicity and metal resource use (depending on LCIA-method) benefit most from commercial ICT reuse. For the majority of environmental impacts however, the potential of recycling to offset impacts is limited because dominant contributions derive from component production as opposed to primary material production.

Furthermore, the study demonstrated how the differences between LCIA-methods (Finnveden et al., 2016; Peters and Weil, 2016; Rigamonti et al., 2016; Rørbech et al., 2014) are relevant for the interpretation of commercial ICT reuse. Most of the used LCIA-methods give considerable weight to metals that are functionally recycled in WEEE treatment while others, especially reserve-based methods, do not. Thereby, the choice of LCIA-method affects the comparative reduction of metal resource use since differences in characterisation of functionally recycled metals give varying credit to the reuse company's routine of steering flows into WEEE recycling. By using several LCIA-methods it could also be observed that despite that laptops contain a broad spectrum of metals, of which only a few are functionally recycled, a significant share of metal resource use is functionally recycled. However, the results also suggest that significant additional impact reduction could be achieved by increasing collection rates. In this respect, it is argued that increased implementation of CE measures similar to this reuse operation could be valuable. Nevertheless, the majority of impact reduction derives from use extension. Using complementary LCIA-methods demonstrates in terms of which metal resources the use extension is most beneficial. Thus, the results could also point to components that are beneficial to reuse if entire laptops cannot be reused. In sum, in order to take additional steps towards a more circular economy in terms of metals, the study points to the value of using complementary LCIA-methods for metal resource use in environmental assessments of CE measures applied to ICT.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2019.03.050.

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