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Environmental screening of novel technologies to increase material circularity: a case study on aluminium cans

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Abstract

It is undisputed that the recycling of aluminium is desirable as long as the environmental and economic implications of its reintegration do not exceed the burdens of its primary production. The efficiency of any aluminium recycling system can be expressed by the total material losses throughout the entire process chain, ideally reaching 0%, thus equivalent to 100% metal recovery. However, in most cases metals are recycled in open /cascade recycling loop where dilution and quality losses occur. Innovations in ABC design as well as in sorting and recycling technologies have the potential to increase recyclability and avoid downcycling issues due to mixed alloy scrap streams. By means of Life Cycle Assessment (LCA) seven scenarios, comprising specific systemic changes, are compared to the current recycling practice of the used beverage can in the UK. The End-of-Life modelling of recycling is performed in accordance with the equal share method to account for impacts both on the recyclability and the recycled content. The results confirm the primary aluminium production and energy consumption in the ABC production as the hotspots in the life cycle of the ABC. The toxicity and energy-related impact categories show the highest susceptibility to increasing recycled content and recycling rate, while the technological novelties show little effect. In terms of abiotic resource depletion the introduction of novel technologies could have the potential to retain quality of the aluminium alloys by either establishing dedicated waste streams or upgrading the aluminium scrap by dedicated sorting strategies.

Keywords:

Aluminium beverage can, Life Cycle Assessment, Recycling, Solid state recycling, Laser induced breakdown spectroscopy, Abiotic resource depletion

1 Introduction

Aluminium has diffused modern times like no other metal next to steel, and its production continues to grow with an average of 3.7% annually since 40 years (Bauxite Index, 2017). Its physical properties make it an ideal candidate for a large range of industries, from packaging to aerospace, from building and construction to automotive, among many others (EEA, 2017a). Both the primary and secondary production of aluminium is not uncritical. The former is associated with high energy consumption, resource depletion, and high material losses in the different life cycle stages (material production, semi-fabrication and part manufacturing process), as well as the generation of large volumes of bauxite residue (red mud). The latter faces issues with quality losses (when the purity-aluminium content of the produced material is lower than the input material, e.g. by the addition of alloying elements during re-melting) and dilution losses (addition of primary aluminium during re-melting to 'dilute' the concentration of the residual elements that cannot be refined during re-melting) due to a combination of: i) the uncontrolled mixing of scrap streams, ii) accumulation of impurities/tramp elements, and iii) limited melt purification options during re-melting (Paraskevas et al., 2015a). Further, secondary aluminium production is also affected by the high variety in the regional recycling rates (UNEP, 2011), negative social impacts depending on the geographical context (UNEP, 2013), and a potential scrap surplus once the current in-use stock becomes available for recycling (Modaresi and Müller, 2012). Several studies (e.g. Paraskevas et al., 2015b) highlight the fact that recycling of aluminium requires no more than 5% of the energy compared to primary production, hence presents a real opportunity to reduce environmental impacts, if managed in a sustainable way.

The circularity of any aluminium recycling system can be expressed by the total material losses throughout the entire process chain, ideally reaching 0%, thus equivalent to 100% material efficiency. Material circularity, as used in the context of this study, refers to a closed material loop, i.e. recycling of the material into the same product, e.g. re-melting of used beverage cans (UBC) to produce new aluminium beverage cans (ABC). Various factors contribute to material circularity in a recycling system (adapted from Hagelüken, 2007). First, it depends on technical factors that determine the process capability (e.g. recovery of specific alloy series) and installed capacity for material recovery. Second, societal and legislative factors motivate or oblige stakeholders to provide the necessary infrastructure or initiate public campaigns to stimulate a 'recycling culture' (i.e. consumer awareness and behaviour). Finally, economic factors play a vital role by creating the incentive for recycling at the consumer level (e.g. deposit schemes) or scrap values (e.g. informal recycling sector). Even though the ultimate target may be a closed material loop, it should be acknowledged that in reality a fully closed material loop is likely to be impossible to achieve. According to UNEP (2013, p.93) "*There will always be a slight loss of metals due to imperfections in the systems and many other aspects, such as thermodynamics, technology, human error, politics, theft and economics*."

Its physical characteristics make aluminium an ideal material for a range of packaging solutions. As a result, packaging industry absorbs nearly 17% of the aluminium output, ranking third behind the construction and transportation industries in Europe (EAA, 2017b). The ABC is one of the most widespread form of packaging in Europe, with an output exceeding 64 billion ABCs in 2015, to which the market within the United Kingdom of Great Britain and Northern Ireland (UK) contributes with an annual production of almost 10 billion ABCs (BCME, 2016).

Life Cycle Assessment (LCA) is a scientific methodology that has been successfully applied to quantify the potential environmental impacts of beverage packaging in general (van der Harst et al., 2016; Saleh, 2016; Simon et al., 2015), and the ABC in specific (Stichling and Nguyen-Ngoc, 2009; Niero et al., 2016; Niero and Olsen, 2016). Niero et al. (2016) have conducted a scenario-based LCA on the ABC in the UK market with varying recycled content and renewable energy consumption. Niero and Olsen (2016) performed a simulation of a closed loop scenario with reintegration of different sources and amounts of packaging scrap (mixed packaging scrap and UBC) in order to determine the effect on the alloying components. Main conclusion of the latter study was that the incorporation of alloying elements/composition of the metal streams into the LCA has a significant effect on the impact results and should consequently be considered (Paraskevas et al., 2013).

The present study investigates the potential increase of material circularity by employing novel sorting and recycling technologies. It considers mainly the conditions in Europe and focuses in particular on the UK, where the introduction of such novel technologies could lead to a substantial improvement of the purity of the waste stream.

1.1 Aluminium Beverage Cans in the UK context

The standard ABC is composed of a *body* (i.e. the container) and an *end*, in which the opening is punched and the tab riveted. The coil manufacturer supplies the respective aluminium sheets for the body (AA3004) and the end (AA5182). Production scrap is routed back to the coil supplier for recycling, hence is already managed in a closed loop (Stichling and Nguyen-Ngoc, 2009). Body and end are subsequently transported to the beverage producer, who fills and seams the ABC, and sells the product to the consumer through a distribution network of wholesalers and retailers.

Two individual collection schemes for used beverage cans (UBC) are implemented in the UK (Seyring et al., 2016). While any household may dispose of its UBC with a co-mingled waste stream (joint collection of plastic, metal and glass packaging), Every Can Counts, a UK-based partnership between drink can manufacturers and the recycling industry, has introduced bring-point solutions for a variety of organisations at which the UBC is collected separately (http://www.everycancounts.co.uk/). Mixed packaging scrap from households undergoes a sequence of sorting steps separating glass and plastic from the metal fraction, which is further sorted into ferrous and non-ferrous metals. The non-ferrous fraction is subject to additional sorting to separate heavy metals from aluminium (ALFED, 2017). The aluminium scrap at this point contains a mix of cast and wrought aluminium alloys, with high and low compositional tolerances in alloying element concentration respectively. This mixed alloy steam is mostly absorbed in the cast alloy production, which results in downcycling of the wrought scrap fraction to cast alloy. This form of recycling is commonly described in literature as “cascade recycling” or “downrecycling” or “open loop recycling”, as there is an accumulation of residual/alloying elements to lower purity alloy systems. Dilution losses on the other hand, occur when primary aluminium is added to reduce the concentration of residual elements in the scrap stream. Both dilution and quality losses during re-melting results in primary resource depletion (primary aluminium and alloying elements addition) and can be minimised by optimal material clustering prior re-melting (Paraskevas et al., 2015a).

1.2 Technological innovations in aluminium recycling

A wealth of research is dedicated to the improvement of aluminium recycling routes, and is primarily focussed on the pyrometallurgical re-melting route. Three main objectives can be derived from the state-of-the-art in recycling technologies: i) retention of the purity of the metal streams, ii) reduction of material losses in pre-processing (e.g. collection and sorting) and re-melting and further processing, iii) reduction in energy consumption in primary and secondary production. Main research topics are dross recycling (Bellqvist et al., 2015; Ingason and Sigfusson, 2014), refining/removal of specific alloying elements (Nakajima et al., 2011, 2012; Gesing et al., 2015), or sorting technologies and strategies (Gaustad et al., 2012; Nogueira et al., 2015; Takezawa et al., 2014). However, the reduction of material losses and energy consumption by incremental improvements seem to have reached a plateau after which only marginal savings are conceivable, hence opening the field for alternative technologies.

Two notable approaches promised to deliver great benefit, not only in decreasing material and quality losses, but also a reduction of greenhouse gas emissions. *Laser induced breakdown spectroscopy* (LIBS) is a sorting technology which has had its market introduction at Düsseldorf’s Aluminium Trade Fair in 2016 and has the capability to sort specific wrought alloys (Steinert, 2016; Hegazy et al., 2013; Takezawa et al., 2014). Several companies have developed prototypes that prove the concept with reliable and repeatable results. The LIBS technology can be applied as extension of the current sorting infrastructure to produce alloy-specific scrap streams, hence providing the re-melter with a high-quality feedstock that minimizes the input of alloying elements and primary aluminium to dilute impurities (see Gaustad et al., 2012 for a discussion on sorting technologies). *Solid state recycling* (SSR) has been in research since 1945 (Stern, 1945), but has recently attracted increased attention as various methods and studies have proven its potential to complement the traditional re-melting route by solid state scrap processing of light metal scrap (Paraskevas et al., 2014; Paraskevas et al., 2016; Behrens et al., 2016; Shamsudin et al., 2016). Current SSR prototypes are able to process 'new' or production scrap into near net semi-products and profiles by hot processing aluminium scrap below melting point in addition to exposure to severe plastic deformation (e.g. via hot extrusion) and/or by diffusion bonding (e.g. via Spark Plasma Sintering) (Paraskevas et al., 2014; Paraskevas et al., 2016). While all studies use machining chips, a relatively clean and high quality feedstock, for the recycling step, it has to be seen to which extent the technology is able to deal with varying scrap size and impurities. The major benefit of SSR is the avoidance of unrecoverable material losses due to oxidation during remelting (approx. 5% and at the levels of 15% for fine form scrap) (Duflou et al., 2015). However SSR does not offer the possibility to readjust the alloy composition (i.e. scrap input equals output alloy) and consequently requires well defined or single alloy stream. None of the SSR technologies has been introduced to market today (Paraskevas et al, 2013).

1.3 Aim of the study

This study provides insights on the environmental performance of novel technologies to increase material circularity in the ABC recycling industry. The UK market is chosen for a case study in order to perform a comparative LCA of eight scenarios including different sorting and recycling technologies, configurations of the can, and waste management options. The LCA concludes in a hotspot assessment of each scenario and defines the respective environmental impact abatement potential in comparison to the current practice. The study addresses challenges associated with the assessment of novel technologies and engages in the on-going debate on methodological choices in LCA such as End-of-Life (EoL) modelling and selection of life cycle impact assessment (LCIA) methods.

2 Methodology

2.1 Scenario development

A total of eight scenarios have been formulated for the comparative LCA. The first is considered as the baseline scenario (S1), describing the current ABC system in the UK in accordance with section 1.1. Its main characteristics are the co-mingled metal waste stream, subject to sorting prior to re-melting into cast alloy ingots, i.e. lower purity output. Scenario two (S2) introduces a uni-alloy can in the base scenario. As opposed to the standard dual-alloy ABC, it is produced out of the AA3004 sheet entirely. This type of can has been object of research (e.g. Novelis, 2012), but has never been introduced to the market. The uni-alloy can in itself is not interesting as a scenario due to its higher weight (and therefore resource consumption), but may benefit a closed material loop in combination with the other here considered technologies. Hence, scenario three (S3) combines SSR with the uni-alloy can, as the SSR route is only capable of handling one specific alloy at once, rendering it infeasible with the standard dual-alloy ABC. Diffusion bonding is chosen as the specific SSR technology, under the inclusion of the entire sintering cycle, assuming that the scrap preparation remains similar to the re-melting route. Scenario four and five introduce LIBS as an extension to the existing sorting infrastructure, once with the standard ABC (S4) and once with the uni-alloy type (S5). Scenario six and seven consider the introduction of a return system, as for example implemented in the Danish market. The return system is considered a closed material loop, as the UBC is directly sold to the coil manufacturer and is therefore reintegrated in the ABC system. The two scenarios simply differ in the applied recycling rates, which correspond to the ones reported by both countries in 2015, i.e. UK (S6) (Stanford, 2016) and DK (S7) (Dansk retursystem, 2016), respectively. Scenario eight (S8) represents an 'ideal system' in terms of recycling rate and recycled content of the input material. Both are considered to reach 100%, however material losses due to sorting and re-melting remain constant (approx. 10% cumulative for all scenarios except S3, see 2.2.2). Table 1 summarizes the main characteristics of each tested scenario.

Table 1 – Aluminium beverage can (ABC) scenario overview.

ID	Description
S1	Base scenario - Current practice in UK
S2	Base scenario but with uni-alloy can
S3	Solid state recycling & uni-alloy can
S4	Sorting by Laser Induced Breakdown Spectroscopy (LIBS)
S5	Uni-alloy can & sorting by Laser Induced Breakdown Spectroscopy (LIBS)

S6	Return system - closed material loop as current practice in DK, with UK recycling rate of 69% (2015)
S7	Return system - closed material loop as current practice in DK, with DK recycling rate of 90% (2015)
S8	'Ideal system' - closed material loop with 100% recycling rate and 100% recycled content

2.2 Life Cycle Assessment

The LCA study was conducted in adherence with the ISO 14040-44 standards (ISO, 2006a, 2006b) and ILCD Handbook requirements (EC-JRC-IES, 2011). The following sections present: the goal and scope definition (section 2.2.1), life cycle inventory (LCI) (section 2.2.2), life cycle impact assessment (LCIA) (section 2.2.3) and sensitivity analyses (section 2.2.4), as part of the life cycle interpretation. Scenario modelling is one of most used methods to estimate uncertainty propagation in LCA (Lyod and Ries, 2007) and has been considered in the present study.

2.2.1 Goal & scope definition

The goal of this comparative study is twofold: i) to provide an initial screening of the hotspots in the ABCs life cycle and ii) to establish the environmental impact abatement potential of each scenario in order to enable strategic decisions towards increased material circularity.

For the purpose of this study the functional unit (FU) has been defined as: "the production of 1000 pieces of 50cl ABCs". The FU reflects the focus on the can and is in line with other studies of similar scope, e.g. Stichling and Nguyen-Ngoc (2009). The 50cl size was selected due to data availability from a project partner and based on the argument that the size is secondary when performing an analysis on the materials' circularity i.e. no comparison to other sizes or container types.

A number of methods have emerged to model the recycling of materials in LCA, all of which emphasizing different stage in the life cycle, but little guidance is given in terms of standardisation (see van der Harst et al., 2016). Depending on the choice of method, credit is given to the recycled content, the recycling (avoided production) or a mix of both, its burdens and credits distributed respectively. The *equal share method* (ES) was chosen as the standard modelling approach, while the method *substitution with equal quality* (SEQ) has been introduced as part of the sensitivity analysis (section 2.2.4). ES distributes credits and burdens associated with resource consumption and EoL in equal shares. It therefore rewards both the increase in recycled content, as well as a high recyclability and is the recommended method in the Product Environmental Footprints (PEF) guide (EC, 2013). This approach is recommended for use in the context of policy support applications (Allacker et al., 2014) and has been used in recent LCAs on aluminium cans (e.g. van der Harst et al. 2016, Niero and Olsen 2016). As opposed to ES, SEQ assumes that the production of a product is based on 100% virgin materials, regardless whether secondary aluminium might be used in reality, and gives full credit to any recycled material in the EoL stage. The latter has been included based on its recommendation by the metals industry (Santero and Hendry, 2016) and the European Aluminium Association (EAA, 2013). Figure 1 illustrates the system boundaries of the ABC system considered in the study in accordance with the ES method. Upstream of the excluded distribution and use of the ABC, shared processes include the recycling of the scrap arising from the production of the ABC body and end, in addition to the recycled content fraction. Subsequent to the disposal of the UBC by the consumer, the non-recycled share is either landfilled or incinerated. The respective burdens and credits (heat and electricity recovery) are fully attributed to the current life cycle, whereas the burdens and credits arising from the recycling of the UBC are distributed in equal shares (50/50).

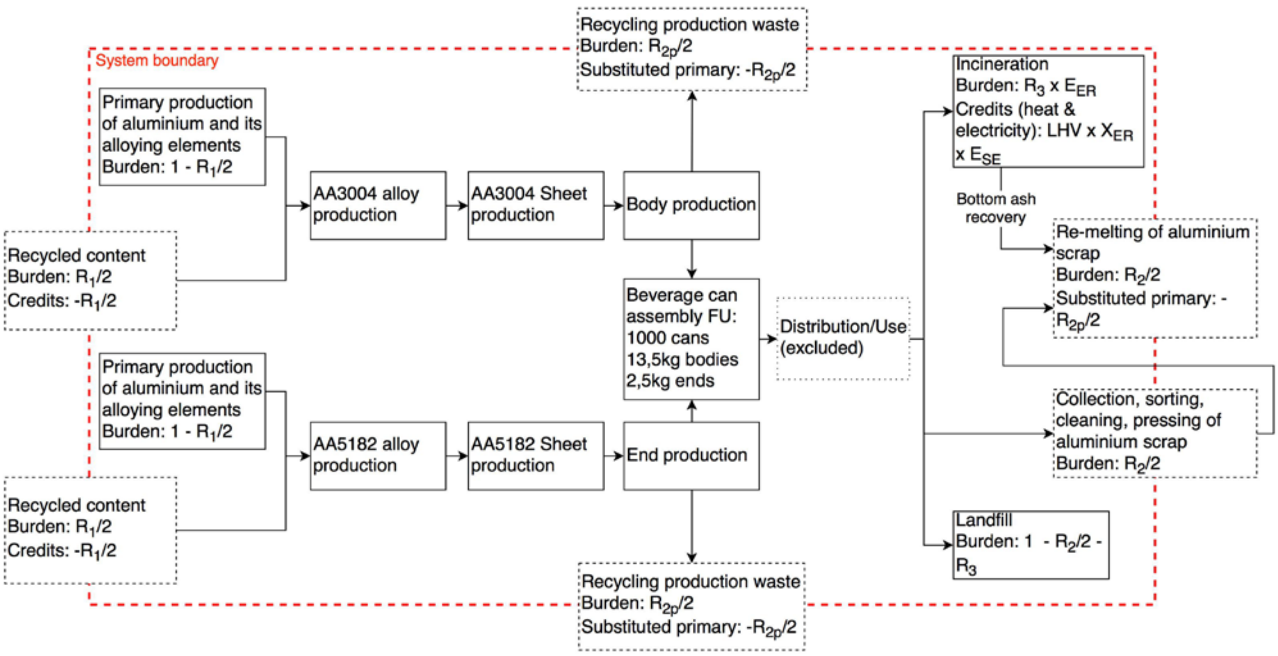

Figure 1 - System boundaries as modelled with the equal share method. The transport of the empty ABC/UBC is included, while the distribution and use of the filled ABC are excluded. The burdens and credits are calculated in accordance with the Product Environmental Footprint (PEF) baseline formula (EC, 2013). R_1: Recycled content; R_2: Recycling rate; R_{2p}: Recycling rate production scrap; R_3: Incinerated fraction; LHV: Lower Heating Value; X_{ER}: Efficiency of Energy recovery; E_{SE}: Avoided emissions and resource consumption of substituted energy source.

2.2.2 Life cycle inventory

To compile a complete inventory, a number of assumptions were made based various sources. While some assumptions are valid for all the scenarios (in italic, lower part of Table 2), others were deliberate choices to reflect systemic changes arising from the implementation of a given technological innovation or collection system (upper part of Table 2).

Table 2 - Scenario Assumptions, including assumptions valid for all the scenarios (in italic, lower part) and specific scenario-assumptions (upper part).

Assumptions	**Source**	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**
Average recycled content (RC) = 50%	EAA, 2013	X	X	X	X	X	X	X	
UK's recycling rate in 2015 = 69%	Stanford, 2016	X	X	X	X	X	X		
DK's recycling rate in 2015 = 90%	Dansk retursystem, 2016							X	
Weight of end (5182 alloy) = 2,5g	-	X			X		X	X	X
Weight of end (3004 alloy) = 3g (assumed weight increase of 20%)	-		X	X		X			
Average material losses during remelting = 5%	Duflou et al., 2015	X	X		X	X	X	X	X
Weight of Body (3004 alloy) = 13g	-	X	X	X	X	X	X	X	X
EoL treatment of the non-recycled fraction: landfill (88.8%) and incineration (11.2%)	DEFRA, 2016 p.11	X	X	X	X	X	X	X	X
Production scrap (equal to 15% of the aluminium coil) is recycled in a closed loop	Stichling and Nguyen-Ngoc, 2009	X	X	X	X	X	X	X	X
Material losses throughout scrap collection, sorting and preparation = 5%	Paraskevas et al., 2015a	X	X	X	X	X	X	X	X
30% recovery rate of aluminium fraction from bottom ash	Wernet et al., 2016	X	X	X	X	X	X	X	X

The LCA was performed considering the standard aluminium alloy composition, as suggested by Niero and Olsen (2016). The aluminium alloys were modelled with an average in-between the minimal and maximal tolerance regarding each alloying element (Table 3). UBC scrap has been used as a feedstock to model the recycled content, on which basis the primary elements have been calculated to match the target alloy's composition requirements.

Table 3 - Average composition in terms of mass fraction of alloying elements for the modelled aluminium alloys. All figures in %wt. Derived from DIN EN 573-3 (AA3004 & AA 5182) and EN 13920:2003 (UBC scrap)

	Mg	**Mn**	**Fe**	**Si**	**Cu**	**Zn**	**Cr**	**Ti**	**Al**
AA3004	1.05	1.25	0.7	0.3	0.25	0.25	-	-	96.2
AA5182	4.5	0.35	0.35	0.2	0.15	0.25	0.1	0.1	94

UBC scrap	1.3	1.1	0.5	0.3	0.2	0.05	-	0.05	96.5

The key figures regarding electricity consumption in the ABC production have been derived from two sources, of which both refer to primary data collection. Stichling and Nguyen-Ngoc (2009) reported the electricity consumption for the body and the end at 417.89 MJ/FU and 13.48 MJ/FU respectively. Niero and Olsen (2016) reported a 12.64MJ/FU for the filling and seaming processes. The electric power consumption is assumed to be similar for both types of ABC, i.e. uni-alloy and dual-alloy and is modelled as a high voltage market mix in the UK as documented in the ecoinvent datasets (45% hard coal, 15% natural gas, 18% nuclear, 14% combined heat and power, 8% wind; Wernet et al., 2016).

Transport intensity is subject to great variation throughout the entire ABC life cycle, depending on the locations of the individual processing factories. Hence, averaged distances, derived from van der Harst et al. (2016) and Niero et al. (2016) have been applied for the modelling (Table 4). The production scrap of both body and end are typically collected by the sheet producer and returned by the same means and modelled accordingly. Upstream transportation within primary resource production up until sheet rolling in addition to the EoL-transportation are adapted from the ecoinvent datasets (Wernet et al., 2016).

Table 4 - Distance in-between the various production stages as reported and modelled.

Origin	Destination	Distance (km)	Mode of Transport
Sheet production	Can manufacturer	400	Transport, lorry 16-32 metric ton, EURO4
Can manufacturer	Beverage producer	29	
Beverage producer	Warehouse	100	

The modelling of the life cycle was performed in Simapro 8.2.3.0 (Goedkoop et al., 2016), using the ecoinvent v3.1 database (Wernet et. al., 2016)

2.2.3 Life cycle impact assessment

The study follows the recommendations of Santero and Hendry (2016), who discussed the harmonization of LCA methodologies for the metal and mining industry. The impact categories are thus: global warming potential (GWP), acidification potential, eutrophication potential, smog potential, and ozone depletion potential (Santero and Hendry, 2016). Based on the recommendations of the European Commission's Joint Research Centre (Hauschild et al., 2013), the ILCD 2011 Midpoint+ V1.08 impact assessment method (EC-JRC-IES, 2012) was chosen as a reference. Therefore, the following ILCD recommended impact categories have been considered: climate change, ozone depletion, particulate matter, acidification, terrestrial, marine and freshwater eutrophication. In addition, the toxicity related impact categories human toxicity cancer and non-cancer, and freshwater ecotoxicity are included and assessed with the USEtox impact assessment method (Rosenbaum al., 2008; ILCD recommended). Further, the impact category abiotic resource depletion is included, although "*there is no scientifically correct method to derive characterization factors*" (Oers and Guinée, 2016, p.1). Drielsma et al. (2016a) argue that resource availability rather depends on markets, politics and technology, than a theoretical environmental constraint and attest LCA an inadequate performance in quantifying those dependencies. As a consequence, LCA studies with the potential to improve the current datasets in resource depletion, omit the inclusion of abiotic depletion altogether (Van Genderen et al., 2016) or even advise against its use (Santero and Hendry, 2016). However, in order to conform with the goal of this study to provide an initial screening, while still producing results that allow conclusions on material circularity, it was decided to include an additional three commonly applied methods for AD characterization. The CML baseline method (version 3.03; van Oers et al., 2002), an enhanced method compared to the ILCD recommended CML non-baseline method (version 3.02; Guinée et al., 2002), has been chosen as it differentiates between AD 'elements' and AD 'fossil fuels' (Oers and Guinée, 2016). Further, Impact 2002+ version 2.12 (Jolliet et al. 2003), based on the damage characterisation factors of the Eco-Indicator99 method as developed by Goedkoop and Spriensma (2001) and Recipe Midpoint version 1.08 (Goedkoop et al. 2013) have been applied.

Novel, more robust, impact assessment methods are under development. Schneider et al. (2015) reported characterisation factors for AD (anthropogenic stock extended abiotic depletion (AADP)), which have been included as a fifth and final impact assessment method.

Table 5 - Overview of recommend and applied impact categories and assessment methods

Impact categories	Recommended by	Applied impact assessment method
Climate change, ozone depletion, particulate matter, acidification, terrestrial, marine and freshwater eutrophication	- Santero and Hendry (2016) - Hauschild et al., 2013	- ILCD 2011 Midpoint+ V1.08 (EC-JRC-IES, 2012)
Human toxicity cancer and non-cancer, freshwater ecotoxicity	- Hauschild et al., 2013	- USEtox impact assessment method (Rosenbaum al., 2008)
Abiotic resource deletion	- Hauschild et al. (2013) (recommend the CML	- CML baseline method (version 3.03; van Oers et al., 2002)

	method (Guinée et al., 2002))	- CML non-baseline method (version 3.02; Guinée et al., 2002) - Eco-Indicator99 method (Goedkoop and Spriensma, 2001) - Recipe Midpoint version 1.08 (Goedkoop et al., 2013) - AADP (Schneider et al., 2015)

End-point modelling is deliberately not applied here, since it is considered relevant to establish, whether or not all impact categories follow the same trend (i.e. are sensitive to the same parameters) when comparing across scenarios.

2.2.4 Sensitivity analyses

The performed LCA included several sensitivity analyses. In terms of EoL modelling, the ABC life cycle has been modelled in accordance with SEQ and ES methods (see 2.2.1). At the LCI level, fluctuations in electricity consumption in production and transport intensity were tested. As the electricity consumption is modelled based on data from 2009, the analysis includes a reduction by 10 %. This is assumed to be a realistic reduction based on recent efforts of the aluminium industry to decrease its power consumption in production.

To analyse the results' sensitivity towards transport intensity, the cumulated distances have been increased until they start to affect the results significantly (i.e. 10% of total GWP impact in base scenario S1, see Humbert et al. (2009) for a discussion on significance). ABC weight, recycling rate and recycled content have not explicitly been included in the sensitivity analyses as they are subject to change within the individual scenarios. Finally, at the LCIA level, the impact on AD has been calculated with the five distinct impact assessment methods described in section 2.2.3.

3 Results

The LCIA results are characterized and normalized at midpoint with the exception of the AD impact category, which is evaluated independently and is consequently not normalized. Weighting and aggregation as optional steps are omitted, as they do not deliver any further information on the circularity of materials.

3.1 Energy related impact categories

Similar trends can be observed across all impact categories recommended by Santero and Hendry (2016) (Figure 2). This is consistent with the findings of Laurent et al. (2010), who established a positive correlation in-between those impact categories, provided the primary driver is the energy consumption. In the case of the ABC life cycle, the production of the primary aluminium and the ABC are the main contributors to the GWP, both being energy-intensive processes. Hence hereafter, the discussion focusses on the GWP.

In terms of GWP, all scenarios with 50% recycled content and 69% recycling rate show a similar performance across the entire life cycle (S1-6, Figure 3). The variations are with ±3% pretty narrow and may originate from the uncertainty of modelling, therefore they may be considered insignificant (i.e. below 10%). In any of these scenarios, the production of primary aluminium and the energy consumption in the ABC production are responsible for the greatest share of GWP with 42% and 46% respectively. Approximately 25% of the electric energy consumption can be attributed to the filling and seaming process, while the remaining share arises from the ABC body and end production.

Surprisingly, the scenarios with technological changes (S2 - S6) perform consistently worse than the base scenario. In case of S2 and S5, this is primarily due to the increased resource consumption due to the higher weight of the uni-alloy can. The LIBS and return system scenarios (S4 & S6) on the other hand, perform worse, due to the fact that they account for avoided production of the alloying elements, which have a lower carbon emission in their production compared to primary aluminium (i.e. GWP (100% aluminium) > GWP (94% aluminium + 6% alloying elements)). Overall, the SSR route (S3) shows the highest GWP as the final recycling step is energy intensive (included in remaining processes in Figure 3). Yet, it is important to highlight that the respective data is based on an experimental settings and the process expected to improve in efficiency once scaled up (Duflou et al. 2015). Additionally, S3 results in a slightly decreased aluminium primary production compared to S2 due to the reduced materials losses in the recycling step (from 72.2 to 71.4 kg CO_2 eq/FU).

The return system scenario with the UK recycling rate of 69% (S6) performs similar to the above discussed scenarios, while an increase of the recycling rate to 90% UBC (S7) shows a reduction of impacts in the range of 8% of the total GWP. This might be more conservative than in reality, as the model assumes the same 5% losses for scrap preparation as all other scenarios. Since the return system does not require any sorting prior to re-melting, these losses might actually be less significant. Conversely, the increased transport intensity associated with a return system might lead to an increase in GWP. However, it can be clearly shown that a combined increase of the recycling rate and the recycled content has the highest abatement potential (38% in S8 Figure 3), leaving the electricity consumption in the ABC production as the major contributor to the impact category.

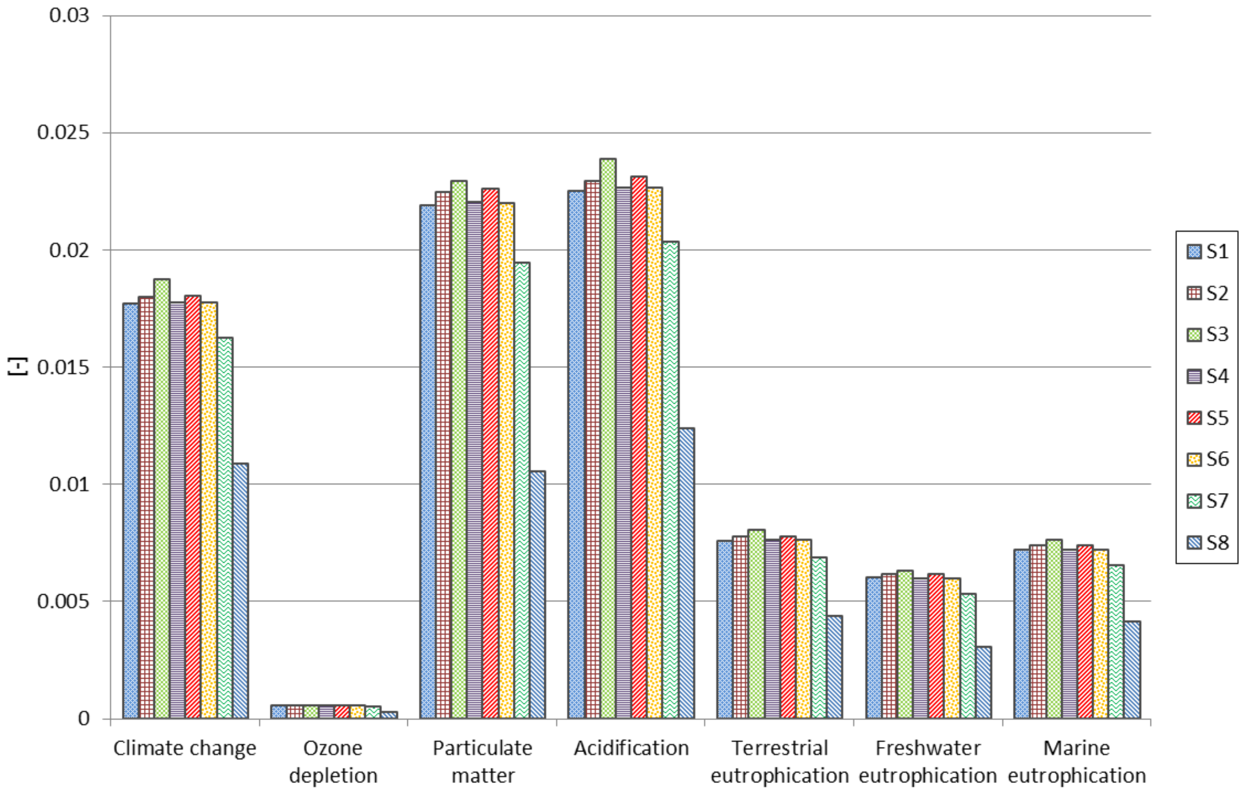

Figure 2 - Normalized impact results of all scenarios (S1-S8). See Table 1 for description of scenarios.

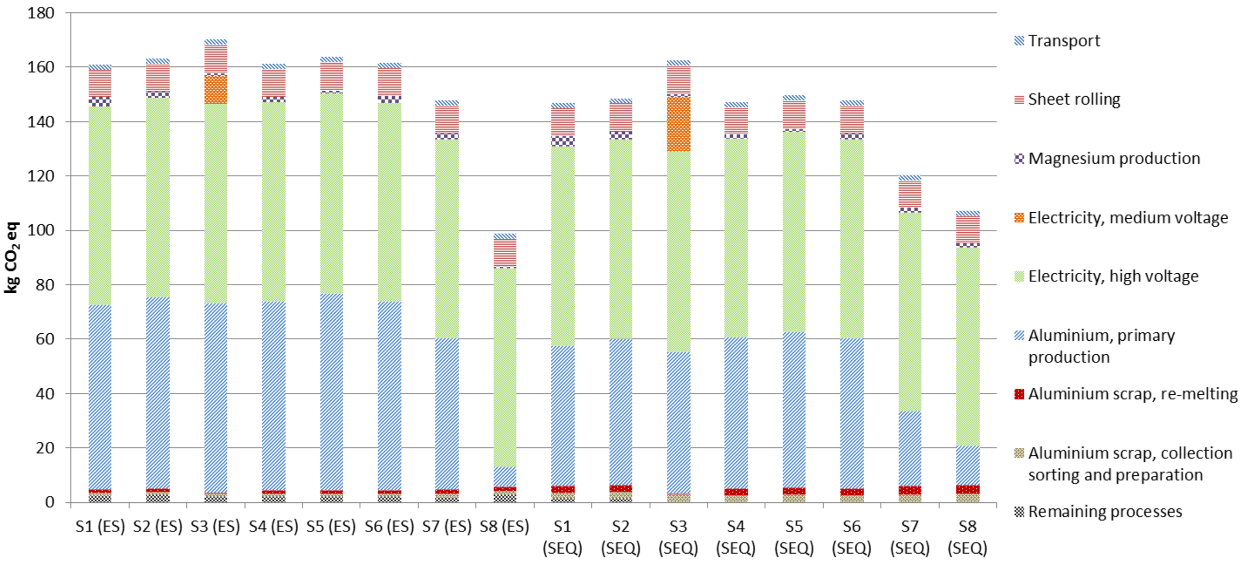

Figure 3 –Global Warming Potential (GWP) results of each scenario and the respective process contribution (1% cut-off). The scenarios have been modelled with the equal share (ES) and Substitution with equal quality (SEQ) method.

3.2 Toxicity related impact categories

The toxicity impact categories follow the same trends as observed for the ones driven by energy consumption. The primary aluminium production, respectively the deposition of red mud arising from the process, is the main contributor to the impacts. As a consequence, the impact categories respond well to an increase of the recycled content and recycling rate (Figure 4). Figure 5 illustrates the relative substance contribution in the three impact categories. For human toxicity (cancer) and freshwater ecotoxicity, chromium VI/water shows the highest contribution to the respective impacts. In both cases the emission arises from the primary production of aluminium (72%) and manganese (24%). The impact of human toxicity (non-cancer) is driven by arsenic/water and mercury/air emissions, yet again, they originate primarily from the aluminium production and to some extend from the energy production.

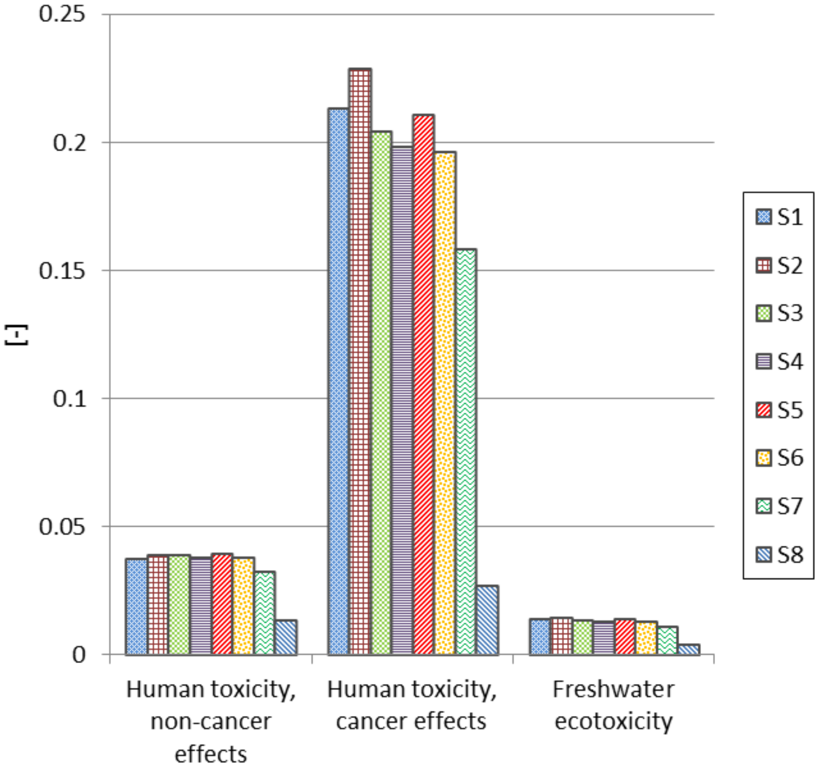

Figure 4 - Scenario comparison for toxicity impact categories, i.e. human toxicity (cancer and no-cancer) and freshwater ecotoxicity (Rosenbaum al. 2008).

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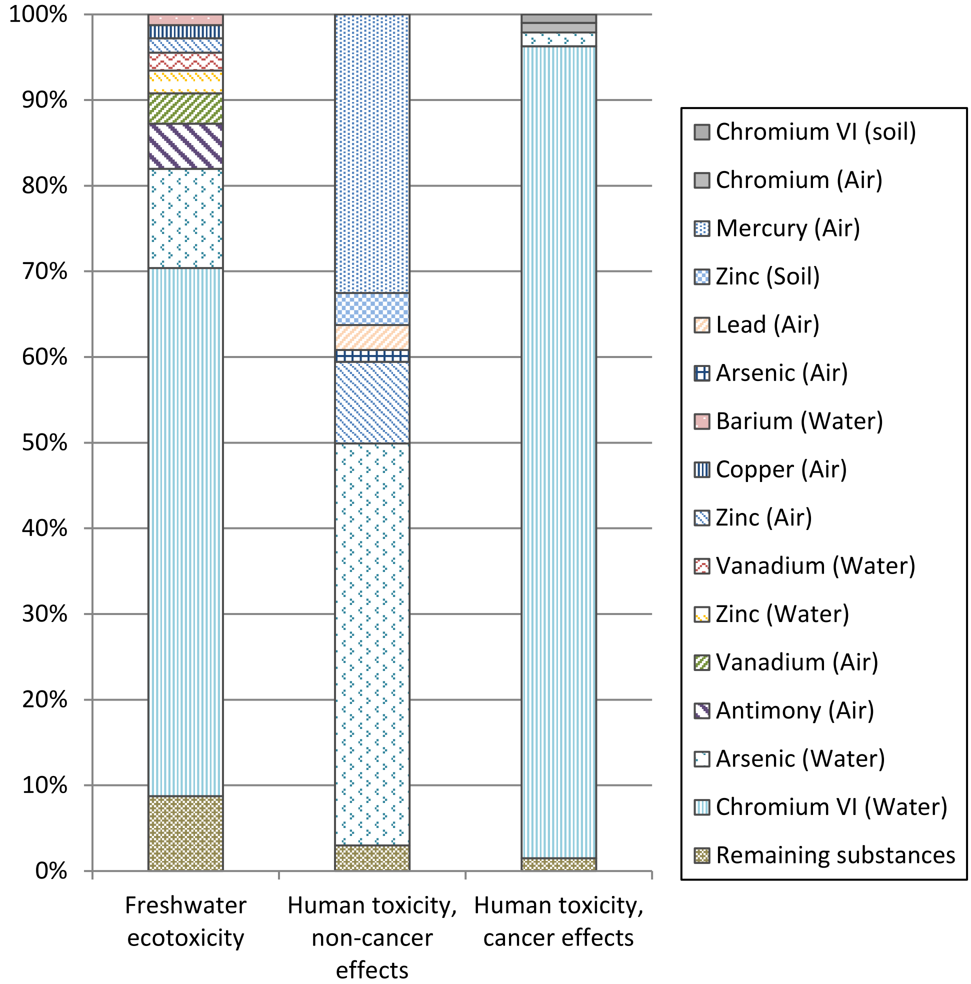

Figure 5 - Relative substance contribution in the toxicity related impact categories. In brackets the final emission compartment is reported.

3.3 Abiotic resource depletion

The five impact assessment methods applied for the abiotic depletion revealed inconsistent results in terms of relative substance contributions (Figure 6). The two CML methods (CML non-baseline as applied in ILCD 2011 Midpoint+ and CML baseline) show a high impact due to substance depletion of indium, lead and cadmium - each a by-product of the zinc mining process. Impact 2002+ assigns 85% of the impact to the depletion of aluminium resources, whereas in Recipe, the same 85% are allocated to manganese. AADP assigns 98% of the impact to cadmium, all other substances remain below the 1% cut-off criteria.

The differing focus of the methods can as well be illustrated by the abatement potentials of the individual scenarios. While the CML and AADP methods suggest an approximate impact reduction of 20% (worst (S2) to best (S6); at constant recycled content and recycling rate), Recipe suggests a 48% and Impact 2002+ merely a 4% abatement potential. The latter is a consequence of the emphasis on aluminium depletion,

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hence no benefit is derived from the recovery of the alloying elements i.e. the method is practically indifferent to the alloy composition of the recycled aluminium output (wrought vs. cast alloy).

However, the observed trend throughout the scenarios remains similar throughout each impact assessment method (Figure 7). The base scenarios with either ABC option (S1 and S2) have the highest potential impact due to the mixed scrap output and the resulting quality losses during re-melting (downcycling). The remaining scenarios with 69% recycling rate perform similarly, highlighting the benefit of either an aggressive sorting prior to re-melting or a closed product loop alternatively. Further, it can be concluded that both an increased recycled content and recycling rate make the biggest difference on the scenarios (S7 and S8). S7 performs similar to S1-S6 as the recycled content remains constant (50%) and the recycling rate is only improved by 21%. S8 however assumes 100% recycled content and recycling rate, de-facto resulting in an increase of 81% avoided production compared to S1-S6 (+50%RC & +31%RR).

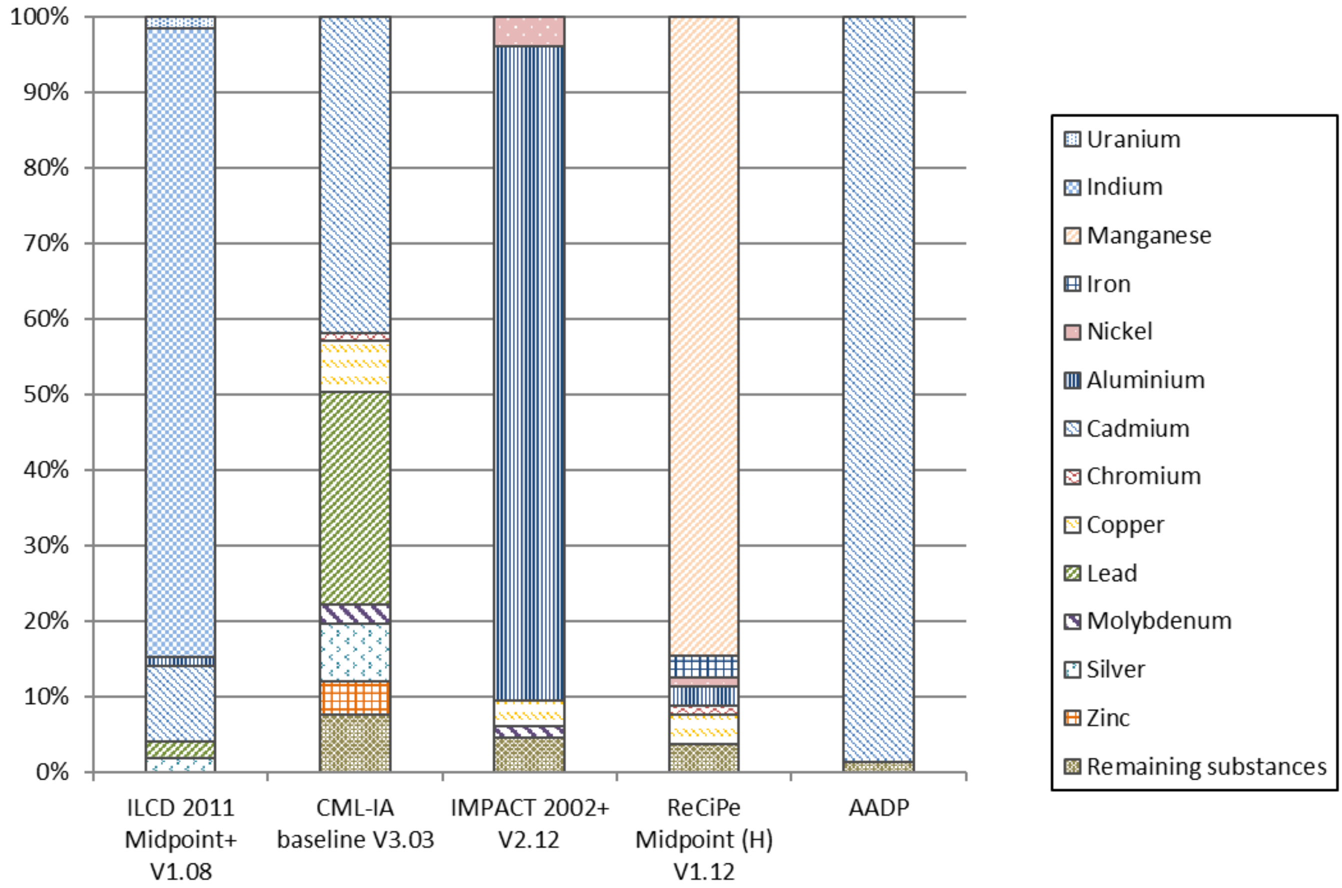

Figure 6 - Relative substance contribution to abiotic depletion considering 5 different impact assessment methods (Scenario S1).

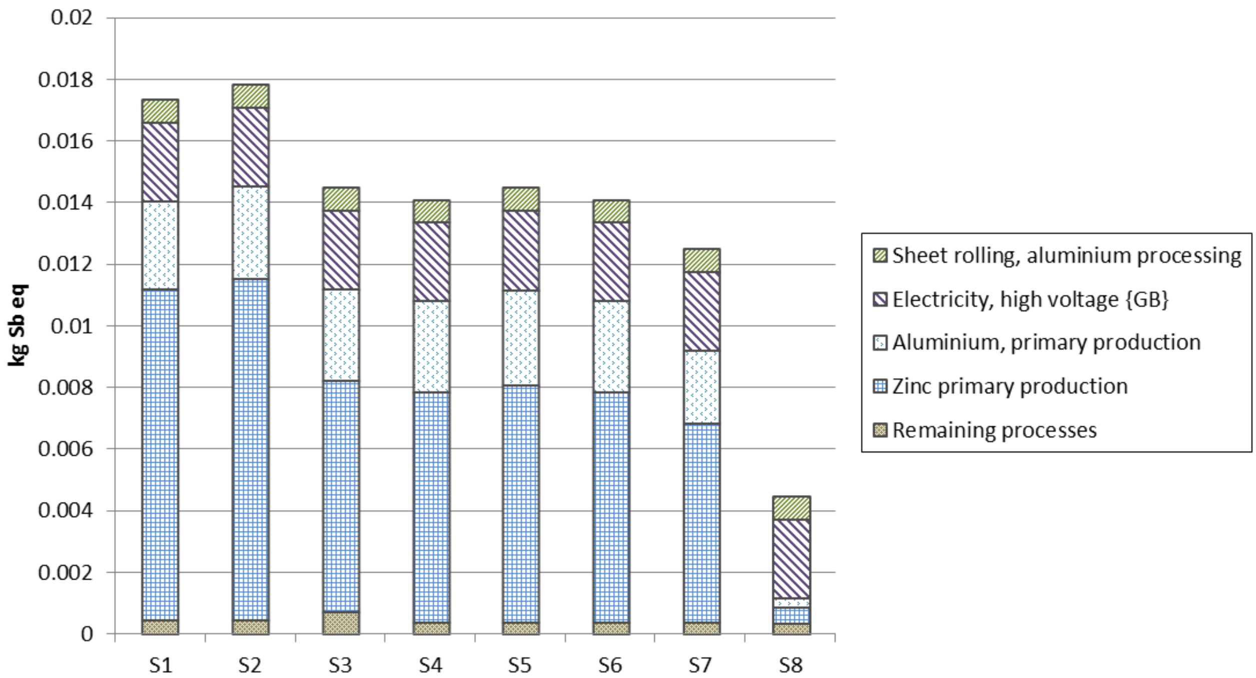

Figure 7 - Abiotic resource depletion results of each scenario, including contribution of the most significant processes (ILCD 2011 Midpoint+)

3.4 Results of the sensitivity analyses

3.4.1 Transport intensity and electricity consumption

Additional scenarios with variations in transport and electricity parameters were modelled. The sensitivity analyses indicated a higher susceptibility to changes in energy consumption rather than transport intensity. In terms of the latter, only a sixteen-fold increase of the total transport distance (i.e. 16 x 529km) led to a significant impact on the GWP impact category, indicating a low sensitivity to fluctuations in transport, which can be explained by the relative low weight of the ABC. This is consistent with the only marginal increase in transport intensity resulting from the slightly higher weight of the uni-alloy ABC.

The 10 % reduction in electricity consumption during the production and the filling of the ABC led to a 4.5% decrease in GWP. In the case of aluminium cans, where most of the environmental impacts come from raw material extraction and production, no significant differences in terms of potential environmental impacts were found when different % and sources of renewable energy are used in the manufacturing stage (Niero et al. 2017). Their research on 33 cl aluminium cans produced in the UK market concluded that only by increasing the % of renewable energy in primary aluminium production, it is possible to significantly reduce the environmental impacts of aluminium cans.

3.4.2 Effects of End-of-Life modelling approaches

The comparison of the two EoL modelling approaches delivered consistent results for all scenarios (Figure 3). While the SEQ method gives full credit to the recyclability of the material and assumes a 100% virgin materials for the production, the ES method gives credits to both recycled content and recycling rate and includes burdens arising from the respective processes. In line with other studies (e.g. van der Harst et al., 2016), the ES provides higher impact score values as only 50% of the benefits arising from recycling are attributed to this product life cycle. From Figure 3 it becomes obvious that the benefit originates from the decreased primary aluminium production, while the rest of the processes remain similar. For S3 it can be clearly shown that energy intensive EoL procedures are working diametrical to the benefits gained from accounting 100% of the recycling rate to this lifecycle. Consequently the difference arising from the modelling is in S3 not as significant as in the remaining scenarios. S8 is the only scenario that performs better in the ES compared to its SEQ version. This is due to the allocation of the burdens arising from collection, sorting and re-melting, respectively the connected material losses, which are to 100% attributed to this life cycle with SEQ method.

4 Discussion

4.1 Validation of LCA results

The scenario analysis confirmed the results of previous LCA studies that found the greatest environmental abatement potential for the ABC system in the increased recycling rate, recycled content and by reducing the energy consumption of the ABC production (Niero et al., 2016, Amienyo and Azapagic, 2016, van der Harst et al., 2016).

The screening of novel technologies (i.e. LIBS & SSR) did not result in significant environmental abatement potentials in terms of GWP. The results for AD however, indicate impact reductions, which range from trivial (4%, Impact 2002+) to significant (48%, Recipe). Besides the introduction of the uni-alloy can in the current UK market (S2), all scenarios highlight the importance to retain the materials quality by separating the waste streams. However, this study falls short of recommending how this separation may be achieved, as the LIBS (S4) and the return system (S6) perform similar in all impact categories. The level of detail in this study is not sufficient to differentiate the two, even though one can expect significant differences in terms of transport intensity, material losses and changes in infrastructure, which are not accounted for here.

The results of the SSR route (S3) are consistent with the findings of Duflou et al. (2015), which reflect the higher energy consumption of the diffusion bonding process compared to the re-melting route. As opposed to their study, the here performed assessment does not give any credit for the fact that the SSR route directly results in a near net shape, compared to the re-melting route which requires an additional hot-extrusion step to get to a similar output. However, the future development of the SSR technology will have to show, to which degree it is able to substitute the traditional re-melting route and consequently decrease material losses to a minimum.

The results of the alternate material composition, as assessed with the scenarios including the uni-alloy can (S2, S3 and S5), do not indicate any environmental benefits. Buffington and Peterson (2013) attest the uni-alloy potential to increase the materials reuse, but point out that the recycling rate has to reach a high level in order to generate the supply necessary for a high recycling content. In the current system, the UBC is re-molten and the melt diluted with primary aluminium and alloying elements to subsequently be reprocessed into body coils (Løvik and Mueller, 2014). Considering the cumulative material losses and the continuous growth of the industry, it is argued here, opposed to Buffington and Peterson (2013), that it is unlikely that the current ABC design will hinder a full reintegration of UBCs in subsequent life cycles.

Both, S7 and S8 highlight the importance to re-integrate the UBC in the production of new ABCs, since there is a clear reduction of impact across all impact categories. Yet, technological innovation alone will not be able to increase the efficiency of the materials' circularity. While technological innovations (assessed in

S3, S4, S5) certainly have the ability to increase the quality of the scrap stream and reduce materials losses, other factors determine the full exploitability of the ABC life cycle (derived from UNEP, 2013). First, the recycling rates depend on customer behaviour, embedded in a 'recycling culture'. Second, infrastructure needs to be in place to enable the consumer to recycle UBC in a waste stream that retains its quality. Third, increasing the recycled content depends on the availability of a high quality scrap feedstock in order to avoid downcycling and an industry that promotes the use of aluminium with a high recycled content.

The sensitivity analysis confirmed that the modelling approach does influence the LCA results (van der Harst et al. 2016). For the goal of this study the ES approach, as included in the PEF guide, was the correct choice, as it allows to alternate recycled content and recycling rate simultaneously. This approach distributes the environmental impacts of virgin production, recycling processes and disposal amongst the different products of the cascade system. The EoL formula included in the PEF guide is aligned with three key criteria: physical realism (i.e. conformity with the as-is situation), distribution of burdens and benefits in a product cascade system and applicability (Allacker et al., 2017). A formula taking into account the number of recycling cycles of a material would be preferred to reach physical realism and to allocate burdens and benefits of repeatedly recycling of a material over the different products in a product cascade system. However, data on the number of recycling cycles is currently not available for all products on the market and hence fails the criterion of applicability (Allacker et al., 2017). Therefore, such an approach is suited to model material circularity in case of a single life cycle, as in the current study, but also in case of multiple loops (Niero and Olsen, 2016).

A major challenge in the interpretation of results of AD is the dominance of specific elements in the results of the different LCIA results, such as cadmium and lead (CML baseline), indium (CML non-baseline), manganese (Recipe), aluminium (Impact 2002+) and cadmium (AADP). While manganese and aluminium are an integral part of the alloy, cadmium, lead and indium are not. The three elements occur as by-products of a zinc mine and are therefore modelled as such in the underlying ecoinvent dataset, which was used in this study. However, the economic allocation applied in the ecoinvent dataset does, to date, not include resource correction (ecoinvent support, 2017). The lacking transparency of the assumptions behind the available datasets is therefore a key aspect that needs to be improved in order to be able to provide reliable results, as also pointed out by other authors (van Genderen et al., 2016; Brogaard et al., 2014).

4.2 Assessment of material circularity

Considering the lacking consensus regarding AD characterisation, complementary methods to LCA might be considered for the interpretation of results instead. Rigamonti et al. (2016) tested the influence of the selection of the LCIA method for the resource depletion impact category in the case of recovery of electric and electronic waste. A sensitivity analysis has been performed, adopting different sets of characterization factors based on existing models for minerals and metals as well as recently proposed sets accounting for critical raw materials and the results showed misalignment in terms of contribution analysis at the substance level among the different methods. When confronted with the choice of impact assessment methods for AD, the practitioner is well advised to choose more than one method to ensure consistency of the observed trends. Further, it is advisable to simply compare scenarios that concern the same product on the basis of their total score in AD. Otherwise the huge discrepancy in characterisation factors might lead to misleading results and make a proper interpretation difficult. In cases where the primary production of resources is energy intensive (such as aluminium), GWP can be used as an approximation to model material circularity, as it clearly shows the benefit of avoided production. However, as shown in the results above, materials with a low mass fraction and energy consumption during production might get underestimated (e.g. manganese, silicon).

LCA is regularly applied with complementary methods to assess material circularity. Niero et al (2017) defined a framework combining LCA and the Cradle to Cradle® (C2C) certification program to identify which actions should be prioritized to achieve a continuous material loop for beverage packaging, both from an environmental and an economic point of view. Recent studies combined LCA with Material Flow Analysis (MFA) (Turner et al., 2016; Sevigné-Itoiz et al., 2014) or the Cumulative Exergy Extraction from the Natural Environment (CEENE) method (Huysman et al., 2017; Van Eygen et al., 2016), each describing an integrated assessment for specific waste stream scenarios. Turner et al. (2016) applied LCA and MFA in order to support local solid waste management decision making by assessing the performance of different waste policy measures in terms of archived recycling rates and greenhouse gas reduction. Sevigné-Itoiz et al. (2014) used the same MFA methodology to map global streams of aluminium scrap and applied LCA to assess consequences of changes in the system. Both studies conclude independently that the combination of MFA and LCA is a '*prerequisite to consistent development from a linear towards a circular economy*' (Sevigné-Itoiz et al., 2014, p. 94). Huysman et al. (2017) propose for this purpose a *circular economy performance indicator* (CPI) based on the CEENE method to expresses the quality of recycled material to its virgin counterpart. Van Eygen et al. (2016) analyse the efficiency of recycling streams by a MFA and subsequently apply CEENE to express resource consumption of the recycling scheme.

Material circularity has also been discussed as a key prerequisite in the context of circular economy. The Material Circularity Indicator (MCI), developed by the Ellen MacArthur Foundation and Granta (2015), allows measuring how well a product performs in the context of a circular economy. The inputs used to calculate the *MCI* refer to the following four aspects: i) material input in the production process, i.e. the recycled content; ii) utility during use stage, i.e. how long and intensely the product is used; iii) destination after use, i.e. the recycling rate and iv) efficiency of recycling, i.e. the yield of the recycling process. However, as the present case study demonstrates, such information is not sufficient to identify the best option to close material loops. The inclusion of an assessment of the potential environmental impacts in terms of climate change proved sufficient to perform a screening assessment.

The number of different approaches suggested to assess material circularity, respectively their variations, highlights the fact that it is hardly sufficient to consider only a single parameter, but a holistic approach is required, which considers the implications of market forces and policy development on a given scenario. The here discussed screening was able to shed some light on novel technologies that might boost materials' circularity from an environmental perspective, but the results will have to be assessed from an economic and social perspective as well, in order to assure a solution with minimal trade-offs.

5 Conclusion

In order to assess whether novel technologies have the potential to increase material circularity, we performed a screening LCA under the inclusion of most recent methodological developments. The results demonstrate the importance of novel technologies to improve the waste stream's quality/purity and a consequently reduced impact on AD. However, the results show a high sensitivity regarding the choice of impact assessment method for abiotic depletion. The most significant changes showed to be based on the increase of recycled content and recycling rate, emphasising the need to expand the scenario analysis to economic and social aspects in order to capture and understand the implications of a systemic change, i.e. the implications and consequences of changes in e.g. consumer behaviour, infrastructural conditions, and legal instruments.

Besides AD, all other impact categories show only insignificant differences for the scenarios in which recycling rate and recycled content are constant (S1 –S6). This indicates that the environmental influence related to the included alloying elements is similar and hence results in negligible differences, making it impossible to recommend either of the technological novelties. This is contrasted by the fact that a large scale implementation has effects on factors such as transport intensity and energy consumption in the recycling process, which have not been accounted for and may lead to significant different environmental impacts not captured by this study.

In general, the results confirm the hotspots found in the current life cycle of the ABC. While strategies exist to reduce the pressure on resource consumption and to increase the retention of material quality, no evidence for such actions targeting the production has been identified (beyond conventional resource-efficiency related improvements). With 46% of GWP at current conditions, the production is the second largest contributor to life cycle GWP of ABCs after the primary aluminium production. As recycling rates continuously increase, production may soon replace materials extraction and production as the major contributor to the ABC's environmental profile.

We reflected on the methodological choices within LCA and discussed various approaches and indicators proposed in recent studies that illustrate the trend towards combined methods to holistically assess the circularity of materials. These combined methods are especially valuable when considering that the effort to reach consensus on how to characterize abiotic resource depletion (AD) has only just started (Drielsma, 2016b). Such a further development in impact assessment methodology would increase LCA's capabilities to not only assess scenarios based on their quantity (e.g. material mass recovered), but would allow illustrating quality losses during recycling and would hence provide a real added value in the determination of sustainable strategies for the management of specific waste streams.

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