

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

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## <sup>1</sup> Environmental screening of novel

# <sup>2</sup> technologies to increase material

- <sup>3</sup> circularity: a case study on aluminium
- 4 cans
- 5
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## 15 Abstract

16 It is undisputed that the recycling of aluminium is desirable as long as the environmental and economic 17 implications of its reintegration do not exceed the burdens of its primary production. The efficiency of any 18 aluminium recycling system can be expressed by the total material losses throughout the entire process 19 chain, ideally reaching 0%, thus equivalent to 100% metal recovery. However, in most cases metals are 20 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 21 22 downcycling issues due to mixed alloy scrap streams. By means of Life Cycle Assessment (LCA) seven 23 scenarios, comprising specific systemic changes, are compared to the current recycling practice of the used 24 beverage can in the UK. The End-of-Life modelling of recycling is performed in accordance with the equal 25 share method to account for impacts both on the recyclability and the recycled content. The results confirm 26 the primary aluminium production and energy consumption in the ABC production as the hotspots in the 27 life cycle of the ABC. The toxicity and energy-related impact categories show the highest susceptibility to 28 increasing recycled content and recycling rate, while the technological novelties show little effect. In terms 29 of abiotic resource depletion the introduction of novel technologies could have the potential to retain 30 quality of the aluminium alloys by either establishing dedicated waste streams or upgrading the aluminium 31 scrap by dedicated sorting strategies.

#### 32 Keywords:

33 Aluminium beverage can, Life Cycle Assessment, Recycling, Solid state recycling, Laser induced breakdown

34 spectroscopy, Abiotic resource depletion

#### 35 **1 Introduction**

36 Aluminium has diffused modern times like no other metal next to steel, and its production continues to 37 grow with an average of 3.7% annually since 40 years (Bauxite Index, 2017). Its physical properties make it 38 an ideal candidate for a large range of industries, from packaging to aerospace, from building and 39 construction to automotive, among many others (EEA, 2017a). Both the primary and secondary production 40 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 41 42 manufacturing process), as well as the generation of large volumes of bauxite residue (red mud). The latter 43 faces issues with quality losses (when the purity-aluminium content of the produced material is lower than 44 the input material, e.g. by the addition of alloying elements during re-melting) and dilution losses (addition 45 of primary aluminium during re-melting to 'dilute' the concentration of the residual elements that cannot 46 be refined during re-melting) due to a combination of: i) the uncontrolled mixing of scrap streams, ii) 47 accumulation of impurities/tramp elements, and iii) limited melt purification options during re-melting 48 (Paraskevas et al., 2015a). Further, secondary aluminium production is also affected by the high variety in 49 the regional recycling rates (UNEP, 2011), negative social impacts depending on the geographical context 50 (UNEP, 2013), and a potential scrap surplus once the current in-use stock becomes available for recycling 51 (Modaresi and Müller, 2012). Several studies (e.g. Paraskevas et al., 2015b) highlight the fact that recycling 52 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. 53

54 The circularity of any aluminium recycling system can be expressed by the total material losses throughout 55 the entire process chain, ideally reaching 0%, thus equivalent to 100% material efficiency. Material 56 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 57 58 cans (ABC). Various factors contribute to material circularity in a recycling system (adapted from 59 Hagelüken, 2007). First, it depends on technical factors that determine the process capability (e.g. recovery 60 of specific alloy series) and installed capacity for material recovery. Second, societal and legislative factors 61 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 62 63 vital role by creating the incentive for recycling at the consumer level (e.g. deposit schemes) or scrap values 64 (e.g. informal recycling sector). Even though the ultimate target may be a closed material loop, it should be 65 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 66 67 many other aspects, such as thermodynamics, technology, human error, politics, theft and economics."

- 68 Its physical characteristics make aluminium an ideal material for a range of packaging solutions. As a result,
- 69 packaging industry absorbs nearly 17% of the aluminium output, ranking third behind the construction and
- 70 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
- 72 Kingdom of Great Britain and Northern Ireland (UK) contributes with an annual production of almost 10
- billion ABCs (BCME, 2016).
- 74 Life Cycle Assessment (LCA) is a scientific methodology that has been successfully applied to quantify the 75 potential environmental impacts of beverage packaging in general (van der Harst et al., 2016; Saleh, 2016; 76 Simon et al., 2015), and the ABC in specific (Stichling and Nguyen-Ngoc, 2009; Niero et al., 2016; Niero and 77 Olsen, 2016). Niero et al. (2016) have conducted a scenario-based LCA on the ABC in the UK market with 78 varying recycled content and renewable energy consumption. Niero and Olsen (2016) performed a 79 simulation of a closed loop scenario with reintegration of different sources and amounts of packaging scrap 80 (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 81
- 82 streams into the LCA has a significant effect on the impact results and should consequently be considered
- 83 (Paraskevas et al., 2013).
- 84 The present study investigates the potential increase of material circularity by employing novel sorting and
- 85 recycling technologies. It considers mainly the conditions in Europe and focuses in particular on the UK,
- 86 where the introduction of such novel technologies could lead to a substantial improvement of the purity of
- the waste stream.

#### 88 **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.
- 95 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
- 97 plastic, metal and glass packaging), Every Can Counts, a UK-based partnership between drink can
- 98 manufacturers and the recycling industry, has introduced bring-point solutions for a variety of
- 99 organisations at which the UBC is collected separately (<u>http://www.everycancounts.co.uk/</u>). Mixed

100 packaging scrap from households undergoes a sequence of sorting steps separating glass and plastic from 101 the metal fraction, which is further sorted into ferrous and non-ferrous metals. The non-ferrous fraction is 102 subject to additional sorting to separate heavy metals from aluminium (ALFED, 2017). The aluminium scrap 103 at this point contains a mix of cast and wrought aluminium alloys, with high and low compositional 104 tolerances in alloying element concentration respectively. This mixed alloy steam is mostly absorbed in the 105 cast alloy production, which results in downcycling of the wrought scrap fraction to cast alloy. This form of 106 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 107 108 losses on the other hand, occur when primary aluminium is added to reduce the concentration of residual 109 elements in the scrap stream. Both dilution and quality losses during re-melting results in primary resource 110 depletion (primary aluminium and alloying elements addition) and can be minimised by optimal material 111 clustering prior re-melting (Paraskevas et al., 2015a).

#### **112 1.2 Technological innovations in aluminium recycling**

A wealth of research is dedicated to the improvement of aluminium recycling routes, and is primarily 113 114 focussed on the pyrometallurgical re-melting route. Three main objectives can be derived from the stateof-the-art in recycling technologies: i) retention of the purity of the metal streams, ii) reduction of material 115 116 losses in pre-processing (e.g. collection and sorting) and re-melting and further processing, iii) reduction in 117 energy consumption in primary and secondary production. Main research topics are dross recycling 118 (Bellqvist et al., 2015; Ingason and Sigfusson, 2014), refining/removal of specific alloying elements 119 (Nakajima et al., 2011, 2012; Gesing et al., 2015), or sorting technologies and strategies (Gaustad et al., 120 2012; Nogueira et al., 2015; Takezawa et al., 2014). However, the reduction of material losses and energy 121 consumption by incremental improvements seem to have reached a plateau after which only marginal 122 savings are conceivable, hence opening the field for alternative technologies.

123 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 124 125 sorting technology which has had its market introduction at Düsseldorf's Aluminium Trade Fair in 2016 and 126 has the capability to sort specific wrought alloys (Steinert, 2016; Hegazy et al., 2013; Takezawa et al., 2014). 127 Several companies have developed prototypes that prove the concept with reliable and repeatable results. 128 The LIBS technology can be applied as extension of the current sorting infrastructure to produce alloy-129 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 130 on sorting technologies). Solid state recycling (SSR) has been in research since 1945 (Stern, 1945), but has 131 132 recently attracted increased attention as various methods and studies have proven its potential to

133 complement the traditional re-melting route by solid state scrap processing of light metal scrap (Paraskevas 134 et al., 2014; Paraskevas et al., 2016; Behrens et al., 2016; Shamsudin et al., 2016). Current SSR prototypes 135 are able to process 'new' or production scrap into near net semi-products and profiles by hot processing 136 aluminium scrap below melting point in addition to exposure to severe plastic deformation (e.g. via hot 137 extrusion) and/or by diffusion bonding (e.g. via Spark Plasma Sintering) (Paraskevas et al., 2014; Paraskevas 138 et al., 2016). While all studies use machining chips, a relatively clean and high quality feedstock, for the 139 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 140 during remelting (approx. 5% and at the levels of 15% for fine form scrap) (Duflou et al., 2015). However 141 142 SSR does not offer the possibility to readjust the alloy composition (i.e. scrap input equals output alloy) and 143 consequently requires well defined or single alloy stream. None of the SSR technologies has been 144 introduced to market today (Paraskevas et al, 2013).

#### 145 **1.3** Aim of the study

146 This study provides insights on the environmental performance of novel technologies to increase material 147 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 148 149 the can, and waste management options. The LCA concludes in a hotspot assessment of each scenario and 150 defines the respective environmental impact abatement potential in comparison to the current practice. 151 The study addresses challenges associated with the assessment of novel technologies and engages in the 152 on-going debate on methodological choices in LCA such as End-of-Life (EoL) modelling and selection of life 153 cycle impact assessment (LCIA) methods.

### 155 **2 Methodology**

#### 156 2.1 Scenario development

157 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 158 characteristics are the co-mingled metal waste stream, subject to sorting prior to re-melting into cast alloy 159 160 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 161 162 has been object of research (e.g. Novelis, 2012), but has never been introduced to the market. The uni-alloy 163 can in itself is not interesting as a scenario due to its higher weight (and therefore resource consumption), 164 but may benefit a closed material loop in combination with the other here considered technologies. Hence, 165 scenario three (S3) combines SSR with the uni-alloy can, as the SSR route is only capable of handling one 166 specific alloy at once, rendering it infeasible with the standard dual-alloy ABC. Diffusion bonding is chosen 167 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 168 169 to the existing sorting infrastructure, once with the standard ABC (S4) and once with the uni-alloy type (S5). 170 Scenario six and seven consider the introduction of a return system, as for example implemented in the 171 Danish market. The return system is considered a closed material loop, as the UBC is directly sold to the coil 172 manufacturer and is therefore reintegrated in the ABC system. The two scenarios simply differ in the 173 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 174 'ideal system' in terms of recycling rate and recycled content of the input material. Both are considered to 175 176 reach 100%, however material losses due to sorting and re-melting remain constant (approx. 10% 177 cumulative for all scenarios except S3, see 2.2.2). Table 1 summarizes the main characteristics of each 178 tested scenario.

179

#### 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

180

#### 181 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.

#### 188 **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.

197 A number of methods have emerged to model the recycling of materials in LCA, all of which emphasizing 198 different stage in the life cycle, but little guidance is given in terms of standardisation (see van der Harst et 199 al., 2016). Depending on the choice of method, credit is given to the recycled content, the recycling 200 (avoided production) or a mix of both, its burdens and credits distributed respectively. The equal share 201 method (ES) was chosen as the standard modelling approach, while the method substitution with equal 202 quality (SEQ) has been introduced as part of the sensitivity analysis (section 2.2.4). ES distributes credits 203 and burdens associated with resource consumption and EoL in equal shares. It therefore rewards both the 204 increase in recycled content, as well as a high recyclability and is the recommended method in the Product 205 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. 206 207 van der Harst et al. 2016, Niero and Olsen 2016). As opposed to ES, SEQ assumes that the production of a 208 product is based on 100% virgin materials, regardless whether secondary aluminium might be used in

209 reality, and gives full credit to any recycled material in the EoL stage. The latter has been included based on 210 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 211 212 study in accordance with the ES method. Upstream of the excluded distribution and use of the ABC, shared 213 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-214 215 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 216 recycling of the UBC are distributed in equal shares (50/50). 217

- 218
- 219



220

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). R1: Recycled content; R2:
- 224 Recycling rate; R<sub>2p</sub>: Recycling rate production scrap; R<sub>3</sub>: Incinerated fraction; LHV: Lower Heating Value; X<sub>ER</sub>:
- 225 Efficiency of Energy recovery; Ese: Avoided emissions and resource consumption of substituted energy source.

#### 226 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

- 229 reflect systemic changes arising from the implementation of a given technological innovation or collection
- 230 system (upper part of Table 2).
- 231 232
- Table 2 Scenario Assumptions, including assumptions valid for all the scenarios (in italic, lower part) andspecific scenario-assumptions (upper part).
- 233

Assumptions	Source	<b>S1</b>	S2	<b>S</b> 3	<b>S4</b>	<b>S5</b>	<b>S6</b>	S	<b>S8</b>
								7	
Average recycled content (RC) = 50%	EAA, 2013	Х	х	Х	Х	Х	Х	Х	
UK's recycling rate in 2015 = 69%	Stanford, 2016	Х	х	Х	Х	Х	Х		
DK's recycling rate in 2015 = 90%	Dansk retursystem,							Х	
	2016								
Weight of end (5182 alloy) = 2,5g	-	Х			Х		Х	Х	Х
Weight of end (3004 alloy) = 3g (assumed weight	-		Х	Х		Х			
increase of 20%)									
Average material losses during remelting = 5%	Duflou et al., 2015	Х	Х		Х	Х	Х	Х	х
Weight of Body (3004 alloy) = 13g	-	Х	Х	Х	Х	Х	Х	Х	Х
EoL treatment of the non-recycled fraction: landfill	DEFRA, 2016 p.11	Х	Х	Х	Х	Х	Х	Х	х
(88.8%) and incineration (11.2%)									
Production scrap (equal to 15% of the aluminium	Stichling and Nguyen-	Х	Х	Х	Х	Х	Х	Х	Х
coil) is recycled in a closed loop	Ngoc, 2009								
Material losses throughout scrap collection, sorting	Paraskevas et al., 2015a	Х	Х	Х	Х	Х	Х	Х	Х
and preparation = 5%									
30% recovery rate of aluminium fraction from	Wernet et al., 2016	Х	Х	Х	Х	Х	Х	Х	Х
bottom ash									

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	Ті	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
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241

242 The key figures regarding electricity consumption in the ABC production have been derived from two 243 sources, of which both refer to primary data collection. Stichling and Nguyen-Ngoc (2009) reported the 244 electricity consumption for the body and the end at 417.89 MJ/FU and 13.48 MJ/FU respectively. Niero and 245 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 246 247 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). 248 249 Transport intensity is subject to great variation throughout the entire ABC life cycle, depending on the 250 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 251 252 body and end are typically collected by the sheet producer and returned by the same means and modelled 253 accordingly. Upstream transportation within primary resource production up until sheet rolling in addition 254 to the EoL-transportation are adapted from the ecoinvent datasets (Wernet et al., 2016).

255

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	
Can manufacturer	Beverage producer	29	Transport, lorry 16-32 metric ton, EURO4
Beverage producer	Warehouse	100	

256

257 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)

#### 259 **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

- 262 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
- 264 Research Centre (Hauschild et al., 2013), the ILCD 2011 Midpoint+ V1.08 impact assessment method (EC-
- 265 JRC-IES, 2012) was chosen as a reference. Therefore, the following ILCD recommended impact categories
- 266 have been considered: climate change, ozone depletion, particulate matter, acidification, terrestrial,
- 267 marine and freshwater eutrophication. In addition, the toxicity related impact categories human toxicity

268 cancer and non-cancer, and freshwater ecotoxicity are included and assessed with the USEtox impact 269 assessment method (Rosenbaum al., 2008; ILCD recommended). Further, the impact category abiotic 270 resource depletion is included, although "there is no scientifically correct method to derive characterization 271 factors" (Oers and Guinée, 2016, p.1). Drielsma et al. (2016a) argue that resource availability rather 272 depends on markets, politics and technology, than a theoretical environmental constraint and attest LCA an 273 inadequate performance in quantifying those dependencies. As a consequence, LCA studies with the 274 potential to improve the current datasets in resource depletion, omit the inclusion of abiotic depletion 275 altogether (Van Genderen et al., 2016) or even advise against its use (Santero and Hendry, 2016). However, 276 in order to conform with the goal of this study to provide an initial screening, while still producing results 277 that allow conclusions on material circularity, it was decided to include an additional three commonly 278 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 279 280 al., 2002), has been chosen as it differentiates between AD 'elements' and AD 'fossil fuels' (Oers and 281 Guinée, 2016). Further, Impact 2002+ version 2.12 (Jolliet et al. 2003), based on the damage 282 characterisation factors of the Eco-Indicator99 method as developed by Goedkoop and Spriensma (2001) 283 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.

287

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

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

method (Guinée et al.,	- CML non-baseline method (version 3.02;
2002))	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)

288

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.

#### 292 2.2.4 Sensitivity analyses

- 293 The performed LCA included several sensitivity analyses. In terms of EoL modelling, the ABC life cycle has 294 been modelled in accordance with SEQ and ES methods (see 2.2.1). At the LCI level, fluctuations in 295 electricity consumption in production and transport intensity were tested. As the electricity consumption is 296 modelled based on data from 2009, the analysis includes a reduction by 10 %. This is assumed to be a 297 realistic reduction based on recent efforts of the aluminium industry to decrease its power consumption in 298 production. 299 To analyse the results' sensitivity towards transport intensity, the cumulated distances have been increased 300 until they start to affect the results significantly (i.e. 10% of total GWP impact in base scenario S1, see 301 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 302 303 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.

### 306 **3 Results**

- 307 The LCIA results are characterized and normalized at midpoint with the exception of the AD impact
- 308 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.

#### 310 **3.1 Energy related impact categories**

- 311 Similar trends can be observed across all impact categories recommended by Santero and Hendry (2016)
- 312 (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.
- 316 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.
- 321 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.

323 Surprisingly, the scenarios with technological changes (S2 - S6) perform consistently worse than the base 324 scenario. In case of S2 and S5, this is primarily due to the increased resource consumption due to the higher 325 weight of the uni-alloy can. The LIBS and return system scenarios (S4 & S6) on the other hand, perform 326 worse, due to the fact that they account for avoided production of the alloying elements, which have a 327 lower carbon emission in their production compared to primary aluminium (i.e. GWP (100% aluminium) > 328 GWP (94% aluminium + 6% alloying elements)). Overall, the SSR route (S3) shows the highest GWP as the 329 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 330 331 improve in efficiency once scaled up (Duflou et al. 2015). Additionally, S3 results in a slightly decreased 332 aluminium primary production compared to S2 due to the reduced materials losses in the recycling step 333 (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
- 340 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
- 342 contributor to the impact category.



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

345



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.

350

346

#### 351 3.2 Toxicity related impact categories

352 The toxicity impact categories follow the same trends as observed for the ones driven by energy 353 consumption. The primary aluminium production, respectively the deposition of red mud arising from the 354 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 355 contribution in the three impact categories. For human toxicity (cancer) and freshwater ecotoxicity, 356 357 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 358 359 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. 360



362

363 Figure 4 - Scenario comparison for toxicity impact categories, i.e. human toxicity (cancer and no-cancer) and

364 freshwater ecotoxicity (Rosenbaum al. 2008).



366

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

## 369 3.3 Abiotic resource depletion

370 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 371 372 2011 Midpoint+ and CML baseline) show a high impact due to substance depletion of indium, lead and 373 cadmium - each a by-product of the zinc mining process. Impact 2002+ assigns 85% of the impact to the 374 depletion of aluminium resources, whereas in Recipe, the same 85% are allocated to manganese. AADP 375 assigns 98% of the impact to cadmium, all other substances remain below the 1% cut-off criteria. 376 The differing focus of the methods can as well be illustrated by the abatement potentials of the individual 377 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+
- 379 merely a 4% abatement potential. The latter is a consequence of the emphasis on aluminium depletion,

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

382 However, the observed trend throughout the scenarios remains similar throughout each impact 383 assessment method (Figure 7). The base scenarios with either ABC option (S1 and S2) have the highest 384 potential impact due to the mixed scrap output and the resulting quality losses during re-melting 385 (downcycling). The remaining scenarios with 69% recycling rate perform similarly, highlighting the benefit 386 of either an aggressive sorting prior to re-melting or a closed product loop alternatively. Further, it can be 387 concluded that both an increased recycled content and recycling rate make the biggest difference on the 388 scenarios (S7 and S8). S7 performs similar to S1-S6 as the recycled content remains constant (50%) and the 389 recycling rate is only improved by 21%. S8 however assumes 100% recycled content and recycling rate, de-390 facto resulting in an increase of 81% avoided production compared to S1-S6 (+50%RC & +31%RR).



391

392Figure 6 - Relative substance contribution to abiotic depletion considering 5 different impact assessment methods393(Scenario S1).



395 Figure 7 - Abiotic resource depletion results of each scenario, including contribution of the most significant

- 396 processes (ILCD 2011 Midpoint+)
- 397

- 398
- 399

#### 400 **3.4 Results of the sensitivity analyses**

#### 401 **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
- 410 material extraction and production, no significant differences in terms of potential environmental impacts
- 411 were found when different % and sources of renewable energy are used in the manufacturing stage (Niero
- 412 et al. 2017). Their research on 33 cl aluminium cans produced in the UK market concluded that only by
- 413 increasing the % of renewable energy in primary aluminium production, it is possible to significantly reduce
- 414 the environmental impacts of aluminium cans.

#### 415 **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 416 3). While the SEQ method gives full credit to the recyclability of the material and assumes a 100% virgin 417 418 materials for the production, the ES method gives credits to both recycled content and recycling rate and 419 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 420 421 attributed to this product life cycle. From Figure 3 it becomes obvious that the benefit originates from the 422 decreased primary aluminium production, while the rest of the processes remain similar. For S3 it can be 423 clearly shown that energy intensive EoL procedures are working diametrical to the benefits gained from 424 accounting 100% of the recycling rate to this lifecycle. Consequently the difference arising from the 425 modelling is in S3 not as significant as in the remaining scenarios. S8 is the only scenario that performs 426 better in the ES compared to its SEQ version. This is due to the allocation of the burdens arising from 427 collection, sorting and re-melting, respectively the connected material losses, which are to 100% attributed 428 to this life cycle with SEQ method.

#### 429 4 Discussion

#### 430 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).

435 The screening of novel technologies (i.e. LIBS & SSR) did not result in significant environmental abatement 436 potentials in terms of GWP. The results for AD however, indicate impact reductions, which range from 437 trivial (4%, Impact 2002+) to significant (48%, Recipe). Besides the introduction of the uni-alloy can in the 438 current UK market (S2), all scenarios highlight the importance to retain the materials quality by separating 439 the waste streams. However, this study falls short of recommending how this separation may be achieved, 440 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 441 442 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 443

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 hotextrusion 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.

450 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-451 452 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-453 454 molten and the melt diluted with primary aluminium and alloying elements to subsequently be reprocessed 455 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 456 457 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

461 S3, S4, S5) certainly have the ability to increase the quality of the scrap stream and reduce materials losses, 462 other factors determine the full exploitability of the ABC life cycle (derived from UNEP, 2013). First, the 463 recycling rates depend on customer behaviour, embedded in a 'recycling culture'. Second, infrastructure 464 needs to be in place to enable the consumer to recycle UBC in a waste stream that retains its quality. Third, 465 increasing the recycled content depends on the availability of a high quality scrap feedstock in order to 466 avoid downcycling and an industry that promotes the use of aluminium with a high recycled content.

467 The sensitivity analysis confirmed that the modelling approach does influence the LCA results (van der 468 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 469 470 distributes the environmental impacts of virgin production, recycling processes and disposal amongst the 471 different products of the cascade system. The EoL formula included in the PEF guide is aligned with three 472 key criteria: physical realism (i.e. conformity with the as-is situation), distribution of burdens and benefits in 473 a product cascade system and applicability (Allacker et al., 2017). A formula taking into account the number 474 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. 475 476 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 477 478 model material circularity in case of a single life cycle, as in the current study, but also in case of multiple 479 loops (Niero and Olsen, 2016).

480 A major challenge in the interpretation of results of AD is the dominance of specific elements in the results 481 of the different LCIA results, such as cadmium and lead (CML baseline), indium (CML non-baseline), 482 manganese (Recipe), aluminium (Impact 2002+) and cadmium (AADP). While manganese and aluminium 483 are an integral part of the alloy, cadmium, lead and indium are not. The three elements occur as by-484 products of a zinc mine and are therefore modelled as such in the underlying ecoinvent dataset, which was 485 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 486 487 the available datasets is therefore a key aspect that needs to be improved in order to be able to provide 488 reliable results, as also pointed out by other authors (van Genderen et al., 2016; Brogaard et al., 2014).

#### 489 **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

493 and electronic waste. A sensitivity analysis has been performed, adopting different sets of characterization 494 factors based on existing models for minerals and metals as well as recently proposed sets accounting for 495 critical raw materials and the results showed misalignment in terms of contribution analysis at the 496 substance level among the different methods. When confronted with the choice of impact assessment 497 methods for AD, the practitioner is well advised to choose more than one method to ensure consistency of 498 the observed trends. Further, it is advisable to simply compare scenarios that concern the same product on 499 the basis of their total score in AD. Otherwise the huge discrepancy in characterisation factors might lead to 500 misleading results and make a proper interpretation difficult. In cases where the primary production of 501 resources is energy intensive (such as aluminium), GWP can be used as an approximation to model material 502 circularity, as it clearly shows the benefit of avoided production. However, as shown in the results above, 503 materials with a low mass fraction and energy consumption during production might get underestimated 504 (e.g. manganese, silicon).

505 LCA is regularly applied with complementary methods to assess material circularity. Niero et al (2017) 506 defined a framework combining LCA and the Cradle to Cradle<sup>®</sup> (C2C) certification program to identify which 507 actions should be prioritized to achieve a continuous material loop for beverage packaging, both from an 508 environmental and an economic point of view. Recent studies combined LCA with Material Flow Analysis 509 (MFA) (Turner et al., 2016; Sevigné-Itoiz et al., 2014) or the Cumulative Exergy Extraction from the Natural 510 Environment (CEENE) method (Huysman et al., 2017; Van Eygen et al., 2016), each describing an integrated 511 assessment for specific waste stream scenarios. Turner et al. (2016) applied LCA and MFA in order to 512 support local solid waste management decision making by assessing the performance of different waste 513 policy measures in terms of archived recycling rates and greenhouse gas reduction. Sevigné-Itoiz et al. 514 (2014) used the same MFA methodology to map global streams of aluminium scrap and applied LCA to 515 assess consequences of changes in the system. Both studies conclude independently that the combination 516 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 517 518 performance indicator (CPI) based on the CEENE method to expresses the quality of recycled material to its 519 virgin counterpart. Van Eygen et al. (2016) analyse the efficiency of recycling streams by a MFA and 520 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 521

522 Material Circularity Indicator (MCI), developed by the Ellen MacArthur Foundation and Granta (2015), 523 allows measuring how well a product performs in the context of a circular economy. The inputs used to

524 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

- 526 after use, i.e. the recycling rate and iv) efficiency of recycling, i.e. the yield of the recycling process.
- 527 However, as the present case study demonstrates, such information is not sufficient to identify the best
- 528 option to close material loops. The inclusion of an assessment of the potential environmental impacts in
- 529 terms of climate change proved sufficient to perform a screening assessment.
- 530 The number of different approaches suggested to assess material circularity, respectively their variations,
- 531 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.
- 533 The here discussed screening was able to shed some light on novel technologies that might boost materials'
- 534 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.

#### 536 **5 Conclusion**

537 In order to assess whether novel technologies have the potential to increase material circularity, we 538 performed a screening LCA under the inclusion of most recent methodological developments. The results 539 demonstrate the importance of novel technologies to improve the waste stream's quality/purity and a 540 consequently reduced impact on AD. However, the results show a high sensitivity regarding the choice of 541 impact assessment method for abiotic depletion. The most significant changes showed to be based on the 542 increase of recycled content and recycling rate, emphasising the need to expand the scenario analysis to 543 economic and social aspects in order to capture and understand the implications of a systemic change, i.e. 544 the implications and consequences of changes in e.g. consumer behaviour, infrastructural conditions, and 545 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 resourceefficiency 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.

560 We reflected on the methodological choices within LCA and discussed various approaches and indicators 561 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 562 563 to reach consensus on how to characterize abiotic resource depletion (AD) has only just started (Drielsma, 564 2016b). Such a further development in impact assessment methodology would increase LCA's capabilities 565 to not only assess scenarios based on their quantity (e.g. material mass recovered), but would allow 566 illustrating quality losses during recycling and would hence provide a real added value in the determination 567 of sustainable strategies for the management of specific waste streams.

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