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Environmental Trade-Offs of Downcycling in Circular Economy: Combining Life Cycle Assessment and Material Circularity Indicator to Inform Circularity Strategies for Alkaline Batteries

Edis Glogic [1,*], Guido Sonnemann [1] and Steven B. Young [2]

[1] Life Cycle Assessment and Sustainable Chemistry (ISM – CyVi), University of Bordeaux, 33405 CEDEX Talence, France; guido.sonnemann@u-bordeaux.fr

[2] Faculty of Environment, University of Waterloo, Waterloo, ON N2L 3G1, Canada; sb.young@uwaterloo.ca

* Correspondence: edis.glogic@u-bordeaux.fr

Abstract: The application of circularity strategies to improve resource use and recovery should be considered with their potential impacts on the environment. Their effectiveness could be evaluated by combining the material circularity indicator (MCI) and life cycle assessment (LCA) methods. Environmental trade-offs may be underestimated for some strategies given that the loss of material quality with recycling has not been captured within the methodological framework of MCI. The current study demonstrates how significantly this limitation may influence the trade-offs in a case study. The methods are applied to several scenarios for the circularity improvement of alkaline batteries. The joint interpretation of MCI and LCA scores is carried out using waterfall charts and normalized indicator scores. Results suggest that improving circularity generally reduces environmental impacts, although there is large variability among two sets of values. For example, an increase of MCI score by 14% for two recycling scenarios translates to a small reduction of impacts in one case (0.06–1.64%) and a large reduction in another (9.84–56.82%). Observations from the case study are used to discuss the design and scope of MCI use and its combining with LCA. Lastly, we draw on the opportunities of the new comparative approach.

Keywords: circularity; material circularity indicator; life cycle assessment; trade-offs; alkaline batteries

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1. Introduction

The circular economy requires various management strategies to be applied at different stages of the product life cycle, i.e., from the sourcing of raw materials, product manufacture, product use, and end-of-life. The implementation of circularity strategies presents an opportunity to conserve resources, but also a risk if they result in significant negative trade-offs to the environment [1–3]. To realize these trade-offs, the implementation of joint circularity and environmental impact assessment methods could be considered.

The most popular approach to guide circularity strategies is by using the material circularity indicator (MCI) developed by the Ellen MacArthur Foundation and Granta Design [4,5]. To investigate trade-offs of circularity strategies, MCI could be combined with life cycle assessment (LCA), an established methodology for the assessment of impacts on human health, ecosystems, and resources [6].

Combining LCA and MCI with the intention of combined circularity–environmental impact assessment, or assessment of trade-offs, has been explored in some previous works [7–9]. Neiro and Kalbar (2019) propose coupling MCI with LCA to compare hypothetical alternatives for beer packaging using multi-criteria decision analysis [7]. To enable coupling and resolve trade-offs, the authors evaluate the numerical distance of MCI to a positive ideal solution and LCA results to a negative ideal solution to achieve an integrated scoring of beer packaging alternatives [7]. Lonca et al. (2018) compared environmental impact categories and indicators with MCI to assess circularity strategies for the life-extension and manufacture of truck tires. Environmental trade-offs were noted for the life extension strategies, while the use of secondary material for tire manufacture was shown to be beneficial for both impact and circularity indicators. Combining and comparing indicators was facilitated by inverting the MCI value to calculate "material linearity" [8]. Lastly, Walker et al. (2018) compared several circularity indicators (including MCI) with a carbon footprinting indicator. Improvement of both MCI and greenhouse gas reductions were observed with improvement scenarios for a tidal turbine, considering scenarios incorporating additional energy recovery from end-of-life product, refurbishment, and extended product lifetime. A degree of correlation between MCI and LCA categories and indicators was observed, although the authors note that MCI was unable to recognize the true benefits of some scenarios that had a more significant impact on greenhouse gas reductions, in comparison to more moderate improvements in circularity [9]. In all three studies, it has been shown that the improvement of circularity can have trade-offs to the environment, especially for strategies for material recovery.

Trade-offs between environmental impacts and circularity are expected to occur as improving resource use is deemed separate from environmental protection [10]. However, other trade-offs arise due to specific limitations associated with the indicators. While MCI is considered the most complete circularity indicator, there are several methodological limitations associated, including the ability of the indicator to capture material quality loss with recycling [11,12]. The influence of this limitation on the trade-offs between circularity and environmental impacts can be better understood by exploring case studies for specific circularity strategies and scenarios that incorporate downcycling.

The current study builds on previous efforts by investigating how trade-offs occur for the case of single-use zinc-manganese alkaline batteries. In comparison with previous cases, this particular case study has characteristics that allow a deeper evaluation of trade-offs for strategies for recycling and explore how the specific limitation of MCI to account for material downcycling may influence the trade-offs. We investigate several strategies for their improved circularity based on the current and prospective best industry practices and policy targets, including strategies for the use of recycled content in product manufacture, end-of-life recycling, and adaptation of design for improved product performance.

Furthermore, to enhance joint interpretation between MCI and LCA results, the current study introduces a new approach to overcome the differences. The results of these methods are conventionally presented and in a more numerically and visually comparable manner. Thus, the correlations between the increase in MCI and the reduction or increase in environmental impact categories are described using waterfall charts and normalized values. The demonstrated approach and prospective findings are of interest to both industry and academia that require adequate methods to operationalize the circular economy and consider trade-offs in sustainability assessment.

2. Materials and Methods

2.1. Circularity Scenarios and Improvement Routes of Alkaline Batteries

Alkaline batteries are selected as a product with an obvious appeal in terms of analysis of the circular economy and impacts on the environment. Their intensive and widespread use, single-use life cycle, and small size all impose challenges to closing material loops. The disposal of spent alkaline batteries to the landfill is discouraged due to the potential toxicity of zinc and manganese [13,14], generally higher environmental impacts in comparison to recycling [15], and the indirect role of alkaline batteries in the recycling rates of other battery types [16]. The scope of battery production and use is of concern in the Canadian province Ontario, with approximately six million batteries used annually [17]. To improve the circularity of batteries, many strategies can be considered, ranging from recycling, the use of secondary materials in battery manufacture, and building more durable batteries. Given these possibilities, the current assessment considers several current and prospective scenarios for recycling, recycled content use, and design, described in the ensuing paragraphs.

A large portion of the 5000 metric tons of battery waste generated each year in Ontario [17] are processed by Inmetco (recycling route 1) and the Raw Materials Company (recycling route 2). Route 1 represents pyrometallurgical treatment to recover the main materials from battery electrodes and electrolyte. Steel and zinc are recovered following several process stages in an electric arc furnace (EAF) and Waelz kiln. Slag byproduct is valorized in the building industry as a substitute to clinker cement, although this is considered more as an effort to divert the waste product than to functionally recycle manganese and copper. Not accounting for the reuse of slag, the per-element recovery rate of this route is 48%, and if slag recovery is considered, the efficiency of the recycling process is 83%. Completing recycling route 2 involves hydrometallurgical treatment to recover materials from battery electrodes and electrolyte. This recycling scenario comprises more refined mechanical separation to segregate steel, wrapper, brass, and electrodes. Steel casings are sent to a steel smelter, where they are recovered in EAF. Wrapper and brass are sent to an energy recovery facility, but the recycler is not certain of their recovery and does not claim credit for the potential recovery of material and energy. Electrode material (black mass) consisting of zinc and manganese oxides, water, potassium hydroxide, and carbon is converted into agricultural fertilizer. The overall recovery efficiency of the base elements is 83%.

The use of recycled content in battery manufacture is presently considered for the needs of steel for steel casing and zinc for battery electrodes. Both metals could be recycled to high purity from the batteries themselves or sourced as recovered from other processes. As the open-loop option, secondary steel and zinc can be obtained from galvanized steel recycling. In the closed-loop, secondary zinc and steel are recovered from batteries employing recycling route 1. The current use of recycled content in the industry is observed for Energizer®. EcoAdvanced™ alkaline batteries currently contain 4% recycled content, and the company predicts that this might increase to up to 40% in the coming years.

Lastly, designing batteries with longer shelf life or higher capacity would lead to an increase in the discharge current delivered over the battery lifetime. Although the amount of energy that could be supplied through batteries depends on multiple factors such as the initial capacity, self-discharge, application, and environmental conditions, there are notable differences between how much energy can be supplied among the battery brands.

In reference to the described prospective circularity strategies for batteries, several scenarios are selected to be analyzed using MCI and LCA:

- Baseline (S0);
- Use of recycled content: secondary zinc and steel from galvanized steel production (open-loop recycling) (S1);
- Use of recycled content: secondary zinc and steel in a closed-loop (S1x);
- Recycling considering route 1 with the assumption of secondary application of metallurgical slag (S2);
- Recycling considering route 2 (S2x);
- Improved utility (S3).

The baseline scenario represents current the industrial practice. Among improvement scenarios, we investigate three complementary improvement scenarios (S1, S2, and S3) and two competing (or substitute) scenarios (S1x and S2x). Each improvement scenario represents a combination of the baseline scenario and a single described improvement in battery design or management.

The baseline scenario (S0) refers to average durability batteries (capacity of 2450 mAh per AA-type battery), produced entirely using virgin raw materials. Consistent with the present collection rates in Ontario [17], we assume that 40% of end-of-life batteries are collected for recycling and 60% disposed of at the municipal landfill. Recycling follows recycling route 1, without consideration of the reuse of metallurgical slag (48% material recovery). The use of recycled content scenarios (S1 and S1x) project the use of 10% of secondary material in the manufacture of batteries. This 10% is shared in equal parts by zinc and steel to substitute primary metals in the manufacture of battery casing and electrode. The assumed recycled content rate of 10% is higher than the current rates of recycled content use for the manufacture of alkaline batteries observed in the industry but still conservative given the aforementioned predictions by some of the battery producers. The supply of steel and zinc in a closed loop is practically viable given the current rates of collection and recycling of zinc and steel for recycling route 1. Recycling scenario (S2) considers recycling route 1, integrating the assumption that slag is used as a substitute for clinker cement, increasing the efficiency of recycling to 83%. The difference from the baseline scenario that employs the same recycling technology consists of a decision to include credit for avoiding the production of clinker cement. Scenario S2x considers recycling route 2. For the improved utility scenario (S3), we assume that the capacity of the batteries is increased by 20% in comparison to the default scenario.

The selected scenarios are organized within three improvement routes involving one of two scenarios for recycled content use and recycling and a single scenario for improvement of utility (Table 1 and Figure 1. Investigation of the closed-loop scenario by comparison of routes 1 and 2 considers only recycling 1 since closed-loop recycling is not possible for recycling 2, where recovered materials are used in a dissipative manner in agriculture.

Table 1. Organization of improvement scenarios for three improvement routes.

	Improvement Route 1	**Improvement Route 2**	**Improvement Route 3**
Recycled content	Open-loop—S1	Closed-loop—S1	Open-loop—S1x
Recycling	Recycling 1 (83% recovery rate)—S2	Recycling 1 (83% recovery rate)—S2	Recycling 2 (83% recovery rate)—S2x
Improved utility	Improved utility by 20% (S3)	Improved utility by 20% (S3)	Improved utility by 20% (S3)

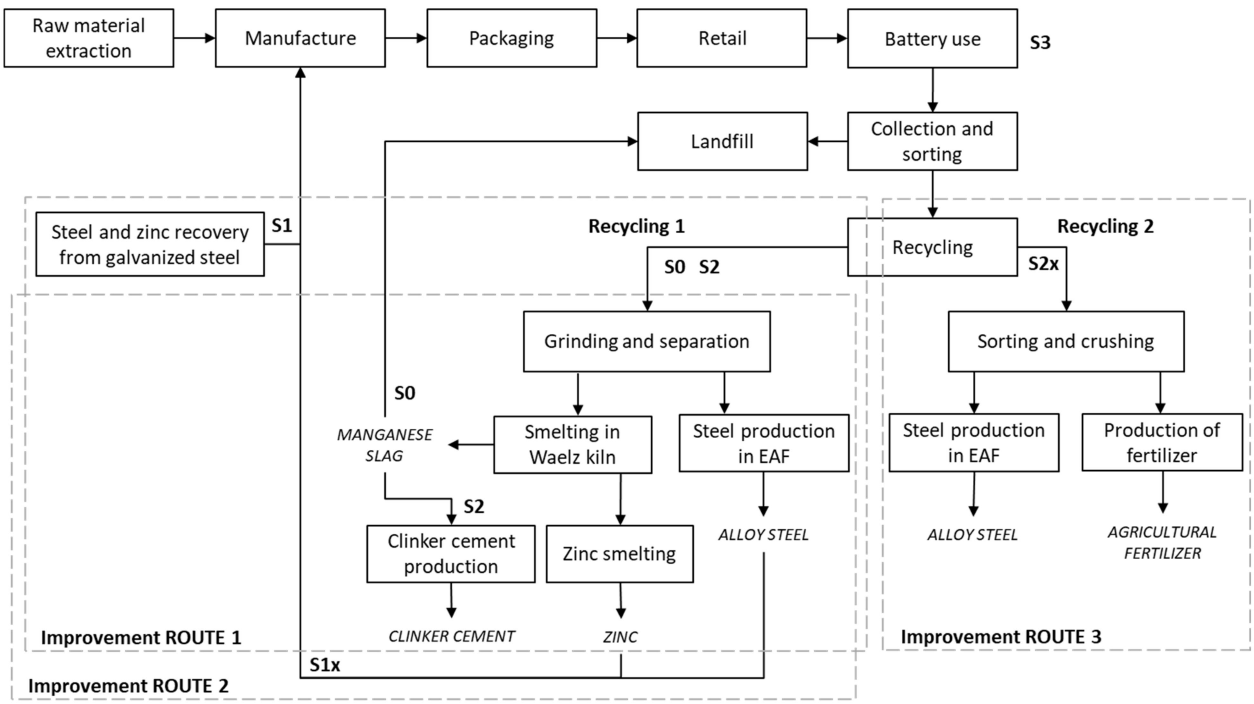

Figure 1. Flow diagram representing all processes in the life cycle of batteries, showing differences in system boundaries for improvement routes. Processes outside of depicted boundaries are shared among all scenarios.

2.2. Data Requirements and Modeling Assumptions for LCA

The LCA provides an evaluation of the environmental impacts of batteries by investigating their potential contribution to several environmental impact categories [6,18]. The environmental impact of batteries is evaluated based on environmental emissions and resource use associated with stages of manufacturing, use, and disposal of batteries including the transportation between these stages. The model is constructed based on alkaline batteries of AA type made of steel casing, brass connectors, zinc electrode, manganese electrolyte, copper connectors, and plastic-paper separator. The batteries are manufactured using either primary or secondary material (steel and zinc derived through the closed loop or open loop, from the recycling of galvanized steel), used, and disposed to the landfill or recycled through one of the two recycling routes. The model excludes impacts from capital goods such as infrastructure for recycling, buildings and transport vehicles, and plastic containers for collection and transportation of end-of-life batteries. The batteries are credited for the avoided impacts of materials recovered through recycling. Allocation of impacts between batteries and secondary materials used in battery manufacture and byproducts of recycling is made using the 50–50 method [19], consistent with how the allocation is dealt with in the computation of MCI [5]. Accordingly, impacts are allocated between upstream products and materials recovered through recycling in a 50/50 ratio. The functional unit for comparison for battery scenarios was to supply 1 Wh of electricity. Considering the average AA battery capacity of 2450 mAh operating at 1.5 V, approximately 0.27 wt% of a single AA battery is needed to supply 1 Wh of electricity.

All foreground data for battery manufacture, packaging, retail, collection, and recycling were adopted from previous publications and reports commissioned for one of the recyclers and summarized in Table 2. Data for battery manufacture, packaging, and retail, including transportation of batteries and materials, are adopted from previous work [20]. Transportation distances for waste batteries are taken from [21]. All data considering recycling routes are obtained from an industry report commissioned by the Raw Material Company [22]. The data for secondary steel and zinc used for battery manufacture in the recycled content and maximum circularity scenarios are assumed from galvanized steel scrap recycling carried out at 95% efficiency [23]. All background data of material, energy, and waste burdens are derived from Ecoinvent v3.3. Background data for chemicals and solid waste disposal are assumed as average global, and electricity is modeled for the province of Ontario.

Classification and characterization of material and energy inputs and waste outputs throughout the life cycle of batteries are carried out using OpenLCA v1.5.0 software and the ReCiPe endpoint (H) impact assessment method: human health, ecosystem, and resources [24].

The comparison between the results of MCI and LCA is carried out using a new comparative approach in order to overcome the differences between each set of results presented. Traditionally, benefit-type analysis of MCI is expressed in an absolute value range between 0 and 1, whereas the cost-type results of LCA are given in impact concentration or equivalence units and commonly normalized against the most impactful scenario (on 0–100 scale). To overcome these differences, the results are jointly interpreted and visualized using waterfall charts. LCA values of circularity strategy scenarios are shown in increment values relative to the impacts of the baseline scenario, and MCI scores are normalized to the net circularity value (Sn)—a sum of combined incremental values of improvement strategies and MCI scores of the baseline scenario.

Table 2. Inputs and outputs of two recycling routes considered in this study.

Recycling Route 1			**Recycling Route 2**		
INPUTS	Quantity	Units	INPUTS	Quantity	Units
Spent batteries	1000.0	kg	Spent batteries	1000.0	kg
Electricity (for separation)	38.0	kWh	Electricity (for separation)	38.0	kWh
Electricity (for EAF)	144.0	kWh	Electricity (for EAF)	144.0	kWh
Coke (for Waelz kiln)	88.0	kg	Sulfuric acid	271.0	kg
Oxygen (for Waelz kiln)	124.4	kg			
OUTPUTS			OUTPUTS		
Slag (from EAF)	2.6	kg	Carbon dioxide	16.6	kg
Carbon monoxide	268.8	kg	Slag (from EAF)	2.6	kg
Carbon dioxide	80.8	kg	Baghouse dust (to waste)	1.5	kg
Chlorine	12.4	kg	Steel low-alloyed (avoided)	190.1	kg
Baghouse dust (to waste)	31.2	kg	Zinc sulfate (avoided)	147.7	kg
Mn-Cu-Fe slag (avoided)	271.3	kg	Manganese sulfate (avoided)	229.0	kg
Crude zinc oxide (avoided)	231.3	kg	Potassium oxide (avoided)	31.6	kg
Steel low-alloyed (avoided)	190.1	kg	Zinc oxide (avoided)	149.0	kg
Baghouse dust (ZnO for recovery)	5.2	kg	Manganese oxide (avoided)	239.0	kg

2.3. Data Requirements for MCI

The material circularity indicator (MCI) measures "the extent at which linear flows of resources, used in the product have been minimized and restorative flows maximized, and how long and intensively the product is used compared to a similar industry-average product" [5]. Three main parameters are quantified to determine the MCI value: quantity of primary material used to manufacture a product, quantity of material that ends up as waste, and how long or intensively the product is used (product's "utility"). These parameters are responsive to a vast range of circularity strategies that can be implemented at different stages of the product's life cycle. Three main parameters of MCI are calculated by looking at the values of several sub-parameters listed in Table 2. For a full description and calculation of these parameters, refer to the published document [5].

Data for the calculation of MCI, with the exception of the calculation of the utility, are based on material and process efficiency rates. Data do not incorporate flows of energy and auxiliary emissions such as transportation and capital goods. Material efficiency rates for the recycled content use and recycling of batteries were measured based on the material flows of base elements, rather than the weight of recovery of materials in their oxidized state. This was necessary for the calculation to be viably applied as batteries' mass and volume increase by approximately 20% during their use and disposal [21]. The data used for each sub-parameter for the calculation of MCI are based on previously mentioned efficiencies and scenarios are detailed in Table 3. For the calculation of the MCI of batteries, we pursue the whole-product approach, meaning that MCI is calculated directly for the product (and not as a sum of the MCIs of the product's components).

Table 3. Material circularity indicator parameter values for all strategy scenarios.

MCI Parameters	S0	S1	S2	S3	S1x	S2x
Mass of product	1	1	1	1	1	1
Fraction of feedstock from recyclable resources	0	0.1	0	0	0.1	0
Fraction of feedstock from reused resources	0	0	0	0	0	0
Fraction of product collected for recycling	0.4	0.4	0.4	0.4	0.4	0.4
Fraction of product going for component reuse	0	0	0	0	0	0
Efficiency of recycling at product end-of-life	0	0.95	0	0	0.48	0
Efficiency of recycling to produce recycled content	0.48	0.48	0.83	0.48	0.48	0.83
Product’s lifetime	1	1	1	1.2	1	1
Average industries’ product lifetime	1	1	1	1	1	1
Product’s intensity of use	1	1	1	1	1	1
Average industries’ product intensity of use	1	1	1	1	1	1

3. Results and Discussion

3.1. Environmental Trade-Offs of Circularity Scenarios

Figures 2–4 show how the increase in the circularity corresponds to the reduction or increase in impacts on endpoint categories of the ReCiPe impact assessment method in the context of the three improvement routes. Endpoint category values of each improvement scenario are calculated relative to the baseline scenario, and MCI scores are normalized to the net circularity value. The absolute values of each scenario for endpoint categories and MCI are shown in Table 4.

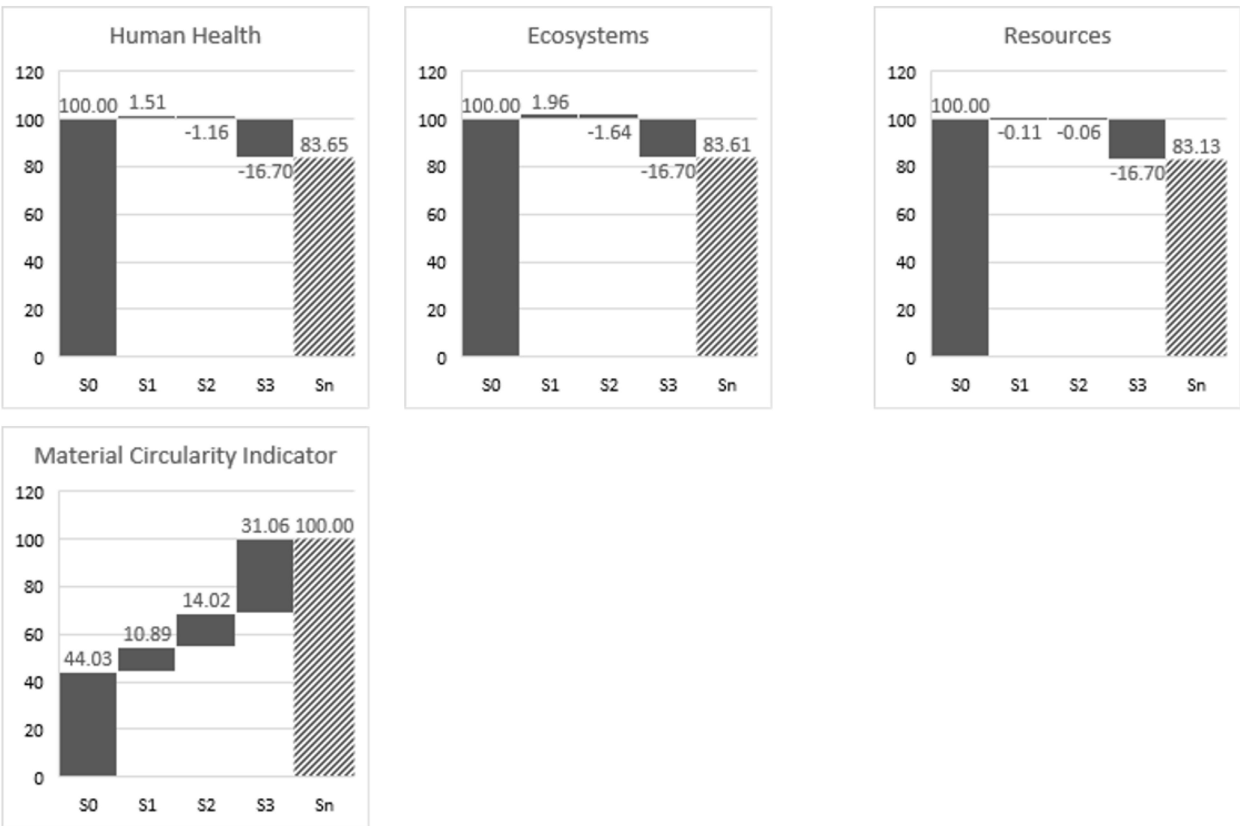

Figure 2. Relative performance of endpoint impact categories and MCI for circularity strategies of improvement route 1.

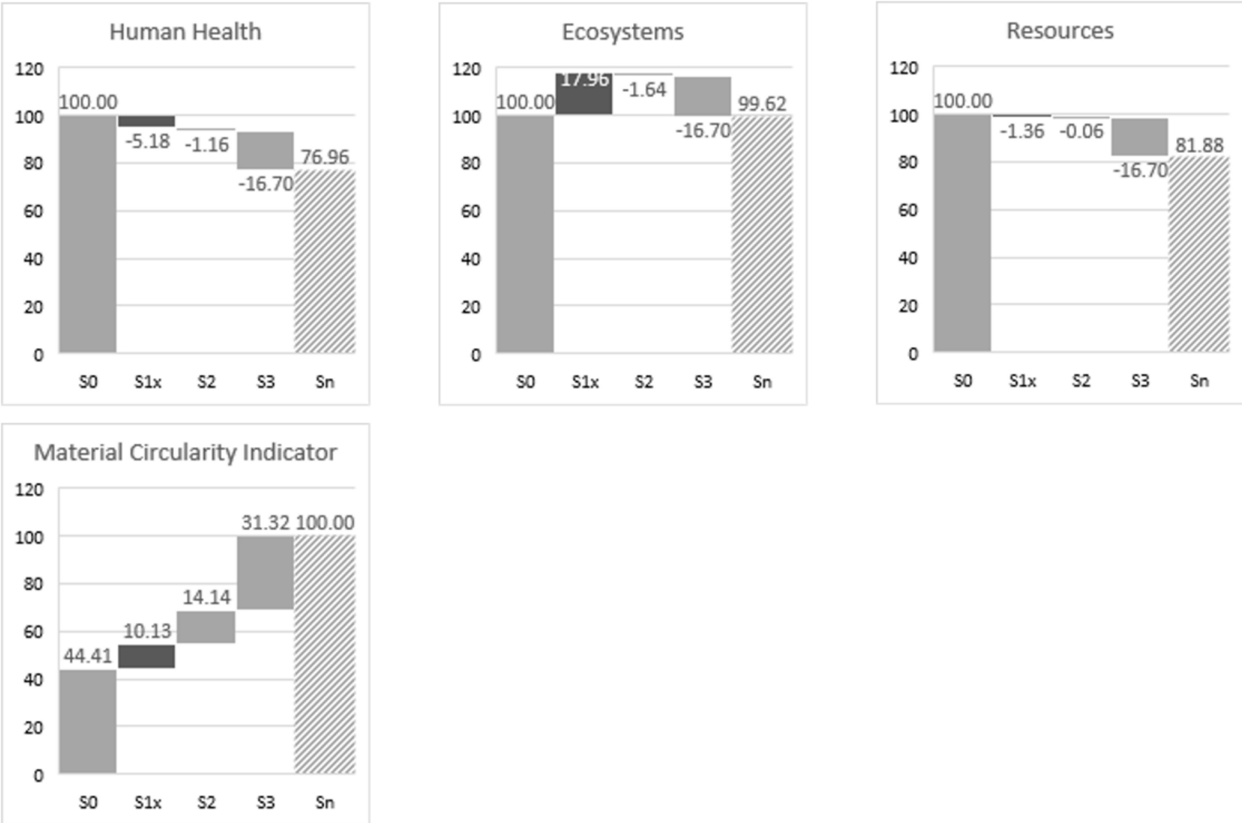

Figure 3. Relative performance of endpoint impact categories and MCI for circularity strategies of improvement route 2.

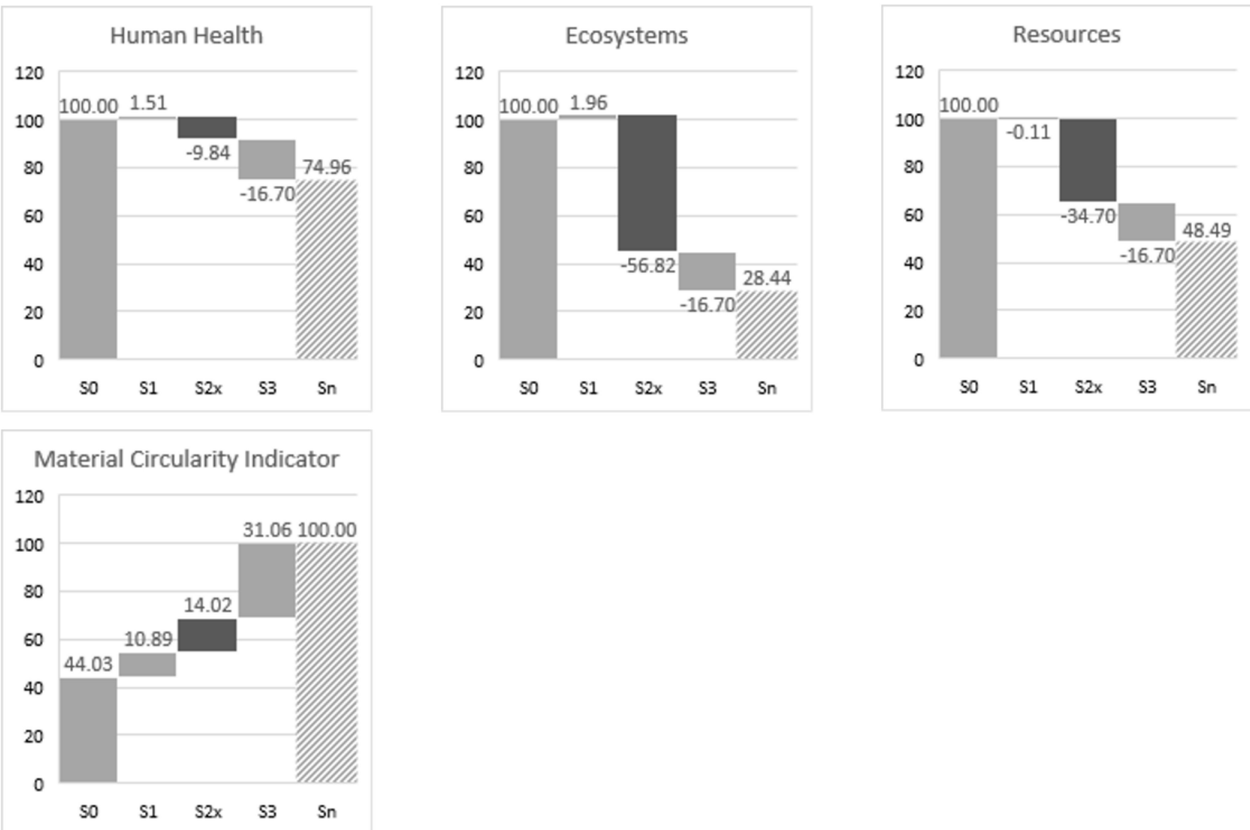

Figure 4. Relative performance of endpoint impact categories and MCI for circularity strategies of improvement route 3.

Table 4. Impact values for endpoint categories of ReCiPe endpoint (H) and MCI. Abbreviations: Human Health—HH, Resources—RE, and Ecosystems—EC.

Impact Category	**Unit**	**S0**	**S1**	**S2**	**S3**	**S1x**	**S2x**
HH-Human toxicity	DALY	3.7×10^{-8}	3.6×10^{-8}	3.7×10^{-8}	3.1×10^{-8}	3.6×10^{-8}	3.5×10^{-8}
HH-Ozone depletion	DALY	7.2×10^{-12}	6.9×10^{-12}	7.1×10^{-12}	6.0×10^{-12}	7.4×10^{-12}	6.0×10^{-12}
HH-Ionizing radiation	DALY	6.6×10^{-11}	6.5×10^{-11}	6.6×10^{-11}	5.5×10^{-11}	5.1×10^{-11}	5.5×10^{-11}
HH-Particulate matter formation	DALY	2.6×10^{-8}	2.5×10^{-8}	2.5×10^{-8}	2.1×10^{-8}	2.3×10^{-8}	2.5×10^{-8}
HH-Photochemical oxidant formation	DALY	6.4×10^{-12}	6.5×10^{-12}	6.3×10^{-12}	5.3×10^{-12}	6.6×10^{-12}	4.6×10^{-12}
HH-Climate Change	DALY	4.4×10^{-8}	4.7×10^{-8}	4.3×10^{-8}	3.7×10^{-8}	4.1×10^{-8}	3.6×10^{-8}
HHtotal	DALY	1.1×10^{-7}	1.1×10^{-7}	1.1×10^{-7}	8.9×10^{-8}	1.0×10^{-7}	9.6×10^{-8}
RE-Metal depletion	$	1.6×10^{-2}	1.6×10^{-2}	1.6×10^{-2}	1.4×10^{-2}	1.6×10^{-2}	1.0×10^{-2}
RE-Fossil depletion	$	1.4×10^{-3}	1.5×10^{-3}	1.4×10^{-3}	1.2×10^{-3}	1.3×10^{-3}	1.1×10^{-3}
RE-total	$	1.8×10^{-2}	1.8×10^{-2}	1.8×10^{-2}	1.5×10^{-2}	1.7×10^{-2}	1.2×10^{-2}
EC-Climate Change	species.yr	2.5×10^{-10}	2.7×10^{-10}	2.4×10^{-10}	2.1×10^{-10}	2.3×10^{-10}	2.1×10^{-10}
EC-Freshwater eutrophication	species.yr	9.1×10^{-13}	9.7×10^{-13}	9.1×10^{-13}	7.6×10^{-13}	8.7×10^{-13}	1.0×10^{-12}
EC-Marine ecotoxicity	species.yr	3.7×10^{-13}	3.2×10^{-13}	3.7×10^{-13}	3.0×10^{-13}	3.2×10^{-13}	3.4×10^{-13}
EC-Natural land transformation	species.yr	9.1×10^{-11}	8.4×10^{-11}	9.1×10^{-11}	7.6×10^{-11}	7.9×10^{-11}	-7.3×10^{-11}
EC-Agricultural land occupation	species.yr	2.7×10^{-11}	2.5×10^{-11}	2.7×10^{-11}	2.2×10^{-11}	1.3×10^{-11}	2.2×10^{-11}
EC-Freshwater ecotoxicity	species.yr	1.6×10^{-12}	1.4×10^{-12}	1.6×10^{-12}	1.3×10^{-12}	1.3×10^{-12}	1.7×10^{-12}
EC-Terrestrial ecotoxicity	species.yr	3.7×10^{-12}	3.6×10^{-12}	3.7×10^{-12}	3.1×10^{-12}	4.1×10^{-12}	-1.6×10^{-12}
EC-Urban land occupation	species.yr	1.4×10^{-11}	1.4×10^{-11}	1.4×10^{-11}	1.2×10^{-11}	1.4×10^{-11}	1.2×10^{-11}
EC-Terrestrial acidification	species.yr	1.3×10^{-12}	1.4×10^{-12}	1.3×10^{-12}	1.1×10^{-12}	1.3×10^{-12}	1.3×10^{-12}
EC-total	species.yr	3.9×10^{-10}	4.0×10^{-10}	3.8×10^{-10}	3.3×10^{-10}	4.6×10^{-10}	1.7×10^{-10}
MCI	-	1.9×10^{-1}	2.4×10^{-1}	2.5×10^{-1}	3.3×10^{-1}	2.3×10^{-1}	2.5×10^{-1}

The highest value for MCI is achieved with the increased utility (31.06%), followed by recycling (14.02%) and the improvement in recycled content (10.89%). Given the similar material flow efficiencies between the two scenarios for recycling and use of recycled content, MCI values for each circularity strategy are similar for the three routes, which allows us to demonstrate the variability of the environmental impact scores. Overall, it appears that recycling and improved utility reduce the environmental impacts for all endpoint categories, and the use of recycled content creates trade-offs in some cases. Improvement of circularity by choice of the system boundary to consider valorization of the waste slag results in an improvement in circularity by 14.02%, while the influence on impacts is comparatively low (1.64–0.06). On the other hand, circularity improvement of a similar value (14.14%) employing recycling route 2 is accommodated by the more substantial impact reductions (9.84–56.82%). The more significant impact reductions come as a result of more functional recovery of zinc and manganese that are assumed for use in agriculture. The use of recycled content in battery manufacture to improve circularity by around 10% has more varied implications for the impact categories, with both reductions and increases in impacts in some cases. For recycled content sourcing through open-loop recycling of galvanized steel, impacts increase by 1.51% and 1.96%, for human health and ecosystems, and are slightly reduced for resources (0.11%). Considering the recycled content scenario obtained through the closed loop (S1x), impact burdens are reduced for the human health endpoint (5.18%) and resources (1.36%), but increase for the ecosystems (17.96%).

The results allow us to highlight how the reduction of material quality with recycling may influence variability between circularity and impact reduction and how this may affect the robustness of such a combined analysis. Valorization of manganese slag in the building industry (S2), motivated by the reduction of waste rather than creating a valuable product, has a small effect on impact mitigation but significant improvement of circularity. Conversely, small changes in circularity and high changes in impacts are observed when cross-comparing S1 and S1x. Variability among two sets of indicators is largely associated with material quality loss with recycling which is not captured in the quantification of MCI, while the LCA results reflect on this aspect indirectly through the avoided burden approach—i.e., given the quality of byproducts, the system is either credited for avoiding emission for producing the primary material (i.e., material moves in a closed-loop fashion) or for avoiding the production of lesser-quality products, thus signaling if the recycling is "functional" or the material is downcycled.

3.2. Opportunities and Limitations of the New Comparative Approach and Potential Implications for MCI Use and Development

The scope of characterization of downcycling should bear implications to combining MCI with LCA that increasingly sought in practice and considered among LCA software providers. Better capacity of LCA method to account for downcycling certainly favors combining methods, but full-fledged LCA may be too costly and difficult for companies. Therefore, the consideration of quality should be incorporated in the measure of circularity that is ultimately different from the protection of the environment [8,10,25,26] and since the recovery of materials of sufficiently high quality is critical in reaching higher recovery rates in the circular economy [27]. This shortcoming is likely exacerbated for the circularity assessment of already manufactured products, in which case circularity improvements are limited to end-of-life strategies. MCI could adopt more advanced designs to address this aspect, similar to the efforts made to respond to some other limitations of this method [11,28], or be combined with other circularity indicators better suited to account for end-of-life management practices and secondary material quality [10,29–36]. Multiple circularity indicators have been developed exclusively to support end-of-life management practices and secondary material quality that could be used to complement MCI given some additional data of end-of-life procedures, material pricing or market potential [10,29–35]. The circularity indicator proposed by Huysman et al. (2017) allows quality characterization using the proxy of exergy, while a proxy of price in a value ratio between input product and secondary (recycled or reused) material is also frequently used [10,29]. More indirectly, material quality can be grasped by evaluating the ease of disassembly, which evaluates product fraction separation [34], or by evaluating the reuse potential of material on the market [32,35].

The new comparative approach introduced in this study allows easier understanding of impact trade-offs when combined with LCA. While MCI results could be also represented in absolute terms in the same manner, the normalized measuring scale allows easier comparison with LCA given common measuring scale (0-100). In addition, it enables the estimation of trade-offs for different products irrespective to their specific limitations to achieve circularity. Therefore, in addition to its appeal for evaluating environmental trade-offs of circularity strategies, it could inspire future indicator use and approaches for combining circularity and environmental impact methods.

4. Conclusions

The improved use of resources in a circular economy could result in trade-offs to the environment. Therefore, it is important that these decisions are properly informed and supported with adequate analytical tools. In our study, we show how trade-offs between circularity and environmental impacts are influenced for several scenarios for the circularity of alkaline batteries. The comparison was carried out in order to highlight how trade-offs are affected given that the loss of material quality with recycling is differently captured by the two methods. The opportunity was also used to introduce a new comparative approach to express the results in a numerically and visually consistent manner. The results show that the improvement of circularity often leads to the reduction of environmental impacts, but that scores among two sets of indicator can vary strongly in instances of material downcycling. This aspect should have implications for whether these two methods should be combined to support recycling strategies. On one hand, LCA can complement MCI to better characterize instances of downcycling, but on the other, such a combination is not intuitive given the potentially large trade-offs between MCI and impact categories. While circularity and environmental assessment are posed to evaluate different characteristics of a product, the significant variability between MCI and LCA impact scores can affect a meaningful interpretation of the results and undermines hypothesized close-knit relationship between impact mitigation and circularity. Consequently, we advocate for better characterization of secondary material quality loss in the calculation of MCI and discuss how other circularity indicators could be useful.

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