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Environmental performance of second-life lithium-ion batteries repurposed from electric vehicles for household storage systems

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A R T I C L E I N F O

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A B S T R A C T

Repurposing lithium-ion batteries has proven to be a promising solution to address the rising number of end-of-life batteries that can be used for second-life energy storage systems and thus extend their service life. While previous research has provided valuable insights into the environmental benefits of battery repurposing, there is still a need to examine the repurposing process more thoroughly, in order to make well-informed decisions on the implementation of second-life battery storage systems. Therefore, this study examines the influence of different repurposing strategies on the environmental performance of second-life battery energy storage systems. A life cycle assessment was conducted, analysing four repurposing cases relating to the exchange of components, namely i) new battery management system and module casing (Base case), ii) new battery management system and reuse of module casing (Case 1), iii) new module casing and reuse of battery management system (Case 2) and iv) reuse of module casing and battery management system (Case 3). These impacts were compared to a storage system with new batteries, to determine the potential environmental benefits and identify the most suitable repurposing strategy. Our findings demonstrate significant environmental benefits of second-life battery energy storage systems across various impact categories and repurposing cases. The Base case and Case 1 resulted in environmental benefits across all impact categories. The highest benefits were observed for metal depletion with savings of 58 % and 61 %, respectively. Increased savings were obtained for Case 2 and Case 3. However, environmental drawbacks were identified for freshwater and marine ecotoxicity. In particular, Case 2 resulted in the highest drawbacks of −22 % and −16 %, respectively. These can be attributed to the allocation procedure, particularly affecting the recycling credits of battery management system recycling. The full allocation of end-of-life impacts and consequently the recycling credits to the second-life battery has not only led to a substantial increase in overall savings, but also resulted in impact categories that originally had disadvantages becoming those with the highest environmental savings. This study demonstrates the importance of carefully selecting repurposing strategies for second-life energy storage systems to maximize their environmental benefits and avoid drawbacks. Additionally, the results highlight the substantial influence of allocation procedures on overall environmental impacts, underscoring the need for clearer methodological guidance on addressing the multi-functionality of repurposed batteries.

1. Introduction

With the "European Green Deal", the EU has committed to reducing its carbon emissions by 55 % by 2030 compared to 1990 and becoming the first climate-neutral continent by 2050 (European Commission, 2019). Overall, net greenhouse gas emissions in the EU in 2021 have been reduced by 28 % compared to 1990. However, emissions from the transport sector have been increasing (Eurostat, 2023). Tackling the reduction of GHG emissions in the transport sector is therefore a crucial step. Consequently, the expansion of the electric vehicle (EV) fleet, and thus lithium-ion batteries (LIBs), has been seen as an important measure

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Glossary

BESS	Battery energy storage system
BMS	Battery management system
CC	Climate change
DoD	Depth-of-discharge
EoL	End-of-life
EP_{fw}	Eutrophication, freshwater
EP_m	Eutrophication, marine
ET_{fw}	Ecotoxicity, freshwater
ET_m	Ecotoxicity, marine
ET_t	Ecotoxicity, terrestrial
EV	Electric vehicle
FD	Fossil depletion
FLB	First-life lithium-ion battery
FLBESS	First-life battery energy storage system
FU	Functional unit
GHG	Greenhouse gas
HT_c	Human toxicity, cancer
HT_{nc}	Human toxicity, non-cancer
IR	Ionising radiation
LCA	Life cycle assessment
LCI	Life cycle inventory
LFP	Lithium iron phosphate
LIB	Lithium-ion battery
MD	Metal depletion
NMC	Lithium nickel manganese cobalt oxide
$PCOF_{ec}$	Photochemical ozone formation, ecosystems
$PCOF_{hh}$	Photochemical ozone formation, human health
PMF	Particulate matter formation
PV	Photovoltaic
SLB	Second-life lithium-ion battery
SLBESS	Second-life battery energy storage system
SOD	Stratospheric ozone depletion
SoH	State-of-Health
TA	Terrestrial acidification

(European Environment Agency, 2021). Besides EVs, the demand for LIBs in battery energy storage systems (BESS) has been increasing in recent years. BESS are crucial for the widespread integration of renewable energy (Bielewski et al., 2022) and have notable potential for frequency balancing (Kebede et al., 2022) and decentralised off-grid solutions (Asian Development Bank, 2018). Despite their advantages in reducing carbon emissions, various environmental impacts result from LIBs, particularly due to the mining of raw materials (Buchert et al., 2017; Peters and Weil, 2018). Considering the continuous demand for EVs and BESS, the number of LIBs reaching their end-of-life (EoL) will rise substantially, necessitating suitable EoL strategies (Systemiq, 2023). Repurposing EoL-LIBs from EVs for less demanding applications, such as BESS, can contribute to sustainability by prolonging their lifetime, reducing the strain on critical raw material logistics (Fischhaber et al., 2016; Kotak et al., 2021), and delaying the generation of waste (Kamath et al., 2020).

In July 2023, the EU introduced a new regulation concerning batteries and waste batteries, which addresses the need for guidelines on the second-life of batteries and formally defines repurposing as "*any operation that results in a battery, that is not a waste battery, or parts thereof being used for a purpose or application other than that for which the battery was originally designed*" (Regulation 2023/1542). Although repurposing is a promising solution to enhance the sustainability of LIBs and meet the growing demand for BESS, it still faces technical challenges due to the wide variety in battery designs (Börner et al., 2022; Prenner et al., 2024). An industrial standard procedure for repurposing has yet to be established (Bobba et al., 2018b), despite recent political and institutional progress, complicating efforts to accurately assess the process' environmental impacts.

In light of existing uncertainties surrounding the repurposing process and the introduction of the new EU battery regulation, this study presents a novel approach to assessing the environmental performance of SLBs. We conduct a cradle-to-grave LCA and incorporate regulatory aspects related to repurposing. Drawing on implications from the battery regulation, we formulate four repurposing cases, focusing on component replacement, to assess their potential impact on the environmental performance of SLBs. To evaluate the potential contribution of SLBs to enhancing the sustainability of BESS, we compare them to first-life LIBs (FLBs). The objectives are to identify the environmental benefits or drawbacks of SLBs under different repurposing assumptions and to pinpoint key factors that may enhance or diminish the environmental performance of repurposed LIBs in BESS applications.

2. Literature review

Several studies applied LCAs to evaluate the ecological sustainability and quantify the environmental benefits or burdens of second-life battery energy storage systems (SLBESS).

Ahmadi et al. (2017) compared the impacts of a conventional system, i.e., internal-combustion-engine vehicle and stationary power by natural gas, to a cascaded system, i.e., EV and SLBESS. The authors found significant environmental benefits of the cascaded system for most impact categories, except for metal depletion (MD), which they attributed to battery production. Only minor impacts were identified for the repurposing phase, which included the exchange of the cell container, module and battery packaging as well as electronics.

Bobba et al. (2018a) analysed the environmental performance of repurposed LIBs for increasing PV self-consumption in residential settings. They identified environmental benefits of SLBs, when replacing a FLB in a grid-connected house and when applied in an off-grid house. Environmental drawbacks were found when introducing a SLB to a grid-connected house without an existing battery. In their study, the authors found a non-negligible impact of the repurposing phase, which comprised the replacement of the battery tray and battery retention system.

Stolz et al. (2019) conducted an LCA analysing the GHGs of integrating a photovoltaic (PV) system with current, future, and SLBs from electric scooters for a cloakroom and club building. Both current and future LIBs refer to nickel manganese cobalt oxide (NMC111) cathode chemistry. However, optimisations in the manufacturing process were assumed for future LIBs, including reduced energy demand covered by a PV system, the use of secondary copper, and a reduction of required battery components. The best environmental performance was identified for the BESS equipped with future LIBs. Compared to current LIBs, the impacts of SLBs were 56 % lower. The production phase of the repurposed system contributed 32 % to the total GHGs including the allocated impacts of the battery as well as the addition of a printed wiring board.

Another study compared the environmental impacts of SLBs to new LIBs using a novel physical allocation approach (Wilson et al., 2021). They reported benefits in CC of 15 % for the repurposed system, while other categories such as HT_c, EP_{fw} and ET_{fw} resulted in higher impacts than the new battery. The manufacturing of exchanged components had only a minor influence, except for the categories HT_c and ET_{fw} mainly caused by the housing structure and module interconnects, respectively. Other components included in the study were the BMS and data cable.

Koroma et al. (2022) studied the environmental impacts of EVs considering different improvement scenarios including repurposing. Their results indicated that repurposing only leads to minor environmental improvements of EVs, whereas recycling was identified as a crucial step for reducing their overall impact. For instance, CC impacts were reduced by 1 % with repurposing, whereas recycling reduced them by 8.3 %. The authors assumed a full replacement of the BMS and cooling system, while 50 % of the battery packaging was reused.

Wang et al. (2022) conducted an LCA of second-life applications and recycling of LIBs in China. The study found that repurposing leads to environmental benefits compared to the impacts of avoided new lead-acid batteries. For the repurposing process the authors assumed the replacement of interconnects and added aluminium housings and connecting strips.

Another study by Schulz-Mönninghoff et al. (2021) integrated energy flow modelling in a LCA of repurposed LIBs and compared them to other circular business model options, such as recycling and remanufacturing. From the perspective of an energy consumer, the study found environmental benefits for CC, while drawbacks were found for resource depletion. In the role of an automotive manufacturer repurposing yields the highest environmental benefits for CC, when compared to other circular business model strategies. The repurposing process had only minor contributions, as no replacement of parts was assumed due to repurposing on pack level.

While these studies provide valuable insights into the environmental performance of SLBs, their assumptions regarding the repurposing process vary considerably, leading to a wide range of reported impacts attributed to this stage and, consequently the overall life cycle. Furthermore, none of these studies have explicitly integrated aspects of the new battery regulation, as it only came into force 2023, nor have they focused on the influence of component replacement during repurposing. This study addresses these gaps by comparing new LIBs to repurposed LIBs under different repurposing assumptions derived from regulatory implications.

3. Methods

LCA is a standardized method that enables the quantification of a product's environmental impacts along its complete life cycle ("cradle-to-grave"), and thus was applied in this study. The software *LCA for Experts* (Sphera, 2024) was used to conduct the LCA, and *Ecoinvent 3.8* (Wernet et al., 2016) served as a database. *ReCiPe 2016* (Huijbregts et al., 2017) was chosen as an impact assessment method similar to previous studies focusing on the repurposing of LIBs (Koroma et al., 2022; Wang et al., 2022; Wrålsen and O'Born, 2023).

3.1. Goal and scope

The objective of this LCA was the analysis of the environmental performance of repurposed LIBs after their first use in EVs, compared to new LIBs applied in a household BESS. For this study, FLB refers to new LIBs produced for BESS, for which we assumed the same capacity of 14.4 kWh and battery chemistry (NMC111, where the atomic ratio of Ni:Mn:Co is 1:1:1) to make it comparable with the SLB.

The analysed midpoint impact categories were: Climate change (CC), particulate matter formation (PMF), fossil depletion (FD), ecotoxicity freshwater, marine, terrestrial (ET_{fw}, ET_m, ET_t), human toxicity cancer, non-cancer (HT_c, HT_{nc}), Ionising radiation (IR), eutrophication freshwater, marine (EP_{fw}, EP_m), metal depletion (MD), photochemical ozone formation ecosystems, human health, ($PCOF_{ec}$, $PCOF_{hh}$), stratospheric ozone depletion (SOD) and terrestrial acidification (TA). Data availability concerning water consumption or land use in the context of lithium-ion battery repurposing is currently scarce. Therefore, the impact categories freshwater consumption and land use were excluded from the analysis. This approach aligns with previous studies, which similarly excluded both categories due to the lack of available data (Bobba et al., 2018a; Cusenza et al., 2019a). However, by evaluating the remaining categories, comprehensive coverage of the three endpoint categories (damage to human health, ecosystems and resource availability) was guaranteed. The functional unit (FU) was 1 kWh of energy delivered by the battery to meet the electricity demand of an average German household until it reaches a lower capacity than the daily household demand. A two-person household with an electricity consumption of 8.7 kWh per day was considered (Statistisches Bundesamt, 2023a, 2023b). Due to possible deviations caused by the Covid−19 pandemic between 2020 and 2021, the electricity consumption was based on the year 2019. The FLB served as a reference scenario. Thus, the FU only involves the delivered energy according to the reference number of cycles of the new LIB while functional for BESS.

The lifetime of both LIBs in the BESS was determined by the number of cycles until they can no longer provide the average household electricity consumption of 8.7 kWh due to their capacity loss, calculated based on Quan et al. (2022) (see Section 3.2.3). Both batteries had a capacity of 14.4 kWh at the beginning of the stationary application. However, the State-of-Health (SoH) of the FLB was at 100 %, whereas the SLB was at 70 %. This value was based on battery warranties of various car manufacturers (ADAC, 2022). Given these assumptions, the FLB achieved 2934 cycles to the EoL while the SLB achieved 3739 cycles. This is due to the higher initial capacity of the SLB at the beginning of life resulting in a lower capacity loss at the beginning of the stationary application compared to the FLB. More modules are necessary to achieve an equivalent capacity as the new LIB, resulting in an increased weight of the SLB. Table 1 summarizes the characteristics of both batteries.

The BESS is installed in a single-family house equipped with a photovoltaic (PV) system in Germany. Only the battery modules were included in the analysis, as identical impacts are assumed for other system components, such as inverters or cables (Kim et al., 2015).

Two scenarios were developed to represent the life cycles of both battery types. Scenario 1 (Fig. 1) portrays the life cycle of the first-life battery storage system (FLBESS), which begins with the production phase, including raw material extraction, battery manufacturing and assembly. The use phase covers the electricity consumption of the BESS, while the EoL phase comprises the collection, sorting, dismantling and recycling of battery components. A pyrometallurgical cell recycling process was assumed, as it currently represents the prevalent recycling route in Europe (Bruno and Fiore, 2023; Windisch-Kern et al., 2022). In Scenario 2 (Fig. 2), the multifunctionality of the LIB had to be addressed, as the battery serves multiple functions during its lifetime, i.e., automotive and stationary storage application (Richa et al., 2015). This was solved by allocating the impacts of the production and EoL phases between both functions, as described in more detail in Section 3.1.1. The life cycle of the SLB also starts with the production phase, before the use phase in the EV. However, the EV use phase was excluded, as it does not represent the battery's second-life in the BESS, which is the function of the analysed product systems (Bobba et al., 2018a). Next is the repurposing process, which encompasses battery testing and the exchange of certain components. Four repurposing cases were analysed focusing on the replacement of battery components (see Section 3.2.4). The same steps included in the EoL phase of the FLB also apply to the SLB.

Table 1
Battery characteristics for BESS application.

	First-life battery (FLB)	Second-life battery (SLB)
Battery chemistry of cathode	NMC111	NMC111
State-of-Health (SoH)	100 %	70 %
Capacity	14.4 kWh	14.4 kWh[a]
Number of modules	6	9
Weight	162 kg	231.43 kg
Lifespan (until capacity = 8.7 kWh)	2934 cycles	3739 cycles

[a] The initial capacity at the beginning of life was 20.57 kWh. At 70 % SoH the capacity has decreased to 14.4 kWh.

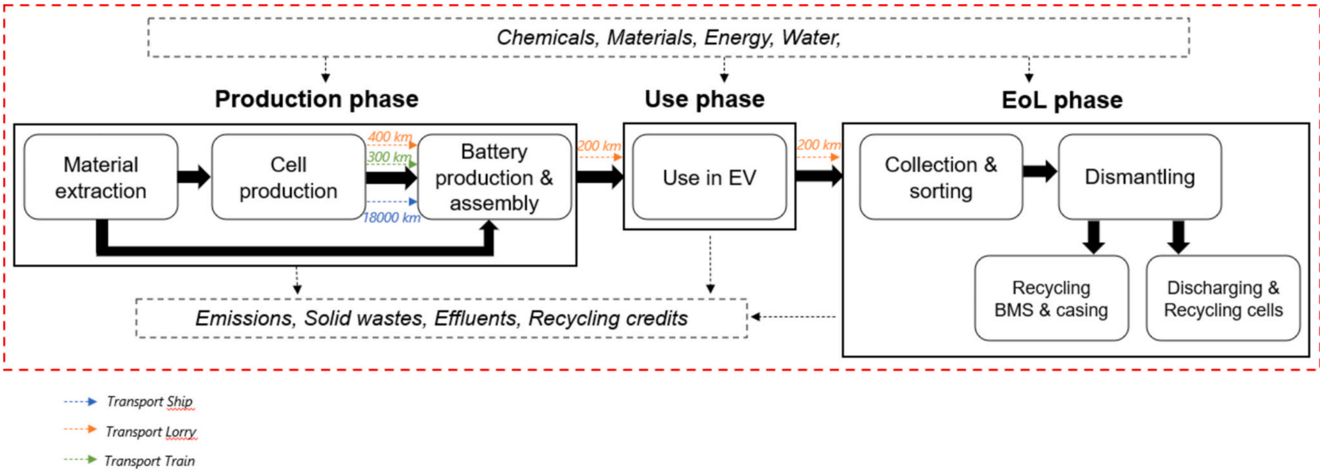

Fig. 1. Scenario 1: System boundary of a first-life lithium-ion battery storage system (FLBESS).

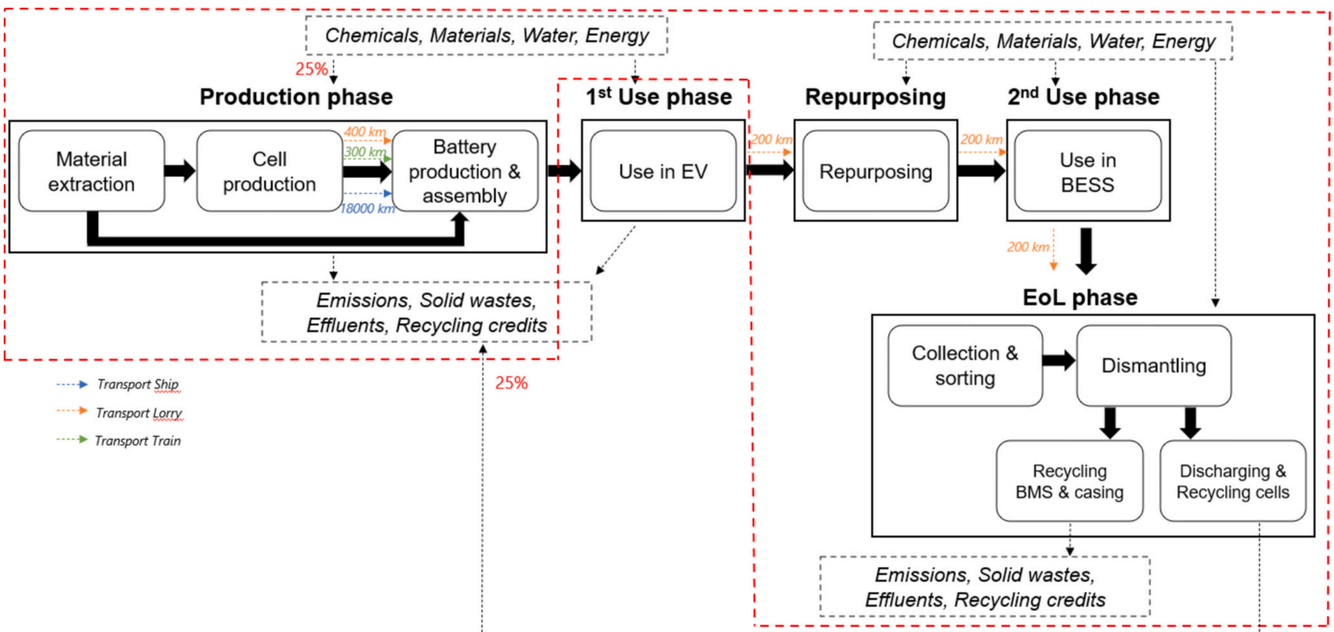

Fig. 2. Scenario 2: System boundary of a second-life lithium-ion battery storage system (SLBESS). Due to the multifunctionality of the SLB (EV and BESS), only 25 % of the production and EoL impacts were included in the assessment (allocation factors α = β = 0.25).

To deal with uncertainties in the model, sensitivity analyses were conducted on the allocation factors and the cell recycling method. Two hydrometallurgical processes were assessed (see Section 3.2.5). In addition, a Monte Carlo analysis was performed to address uncertainties concerning the impact distribution due to the LIB's different lifespan (i.e., number of cycles to reach EoL).

3.1.1. Allocation and impact calculation

As mentioned in Section 3.1, allocation was chosen to address the multifunctionality of LIBs. Yet, several approaches exist to deal with this problem, as was addressed by Schulz et al. (2020). The authors reviewed 11 studies on LIB repurposing and found that the following approaches were used: cut-off allocation, allocation with allocation factors and system expansion. Then these approaches were linked with circular economy considerations from the viewpoint of a LIB producer. Accordingly, with the cut-off criteria the LIB producer takes on the full burdens and acts therefore as a supplier of retired LIBs for the energy sector. Applying allocation would require a certain level of collaboration between sectors in the sense of sharing battery data, while a system expansion represents a full sector collaboration between the LIB producer and energy sector. Based on these insights and current developments of second-life LIBs, we selected allocation, as a full sector collaboration is not established yet, while a cut-off approach implies a non-existing market for second-life LIBs (Bobba et al., 2018a). Furthermore, multifunctionality has been solved by allocation by similar studies (Bobba et al., 2018a; Cusenza et al., 2019b; Wilson et al., 2021). For this study, it was assumed that the allocation factors applied for the components reused in the SLB are α = β = 0.25., whereas the impacts of the rest of the EV-battery components are fully allocated to the LIB's first-life (Bobba et al., 2018a; Wilson et al., 2021). SLBESS are currently being developed (UpVolt GmbH, 2023) and second-life battery packs or modules can already be purchased (Second Life EV Batteries Ltd, 2023), which indicates an emerging market for repurposed LIBs.

To ensure an equal study timeframe, only the impacts corresponding to 2934 cycles (FLB lifespan) were included when evaluating the SLBESS. This is represented by a proportioning factor γ. The calculation for this factor can be found in the Supplementary information in Section S 1. The impacts of Scenario 1 ($I_{Scenario1}$) and 2 ($I_{Scenario2}$) were calculated similar to Bobba et al. (2018a) for each impact category *i*. Eq. (1) represents the impacts of Scenario 1:

$$I_{Scenario1,i} = P_{FL,i} + U_{FL,i} + EoL_{FL,i} \tag{1}$$

- $P_{FL,i}$ = impact of category *i* for the production phase of the FLB
- $U_{FL,i}$ = impact of category *i* for the use phase of the FLB
- $EoL_{FL,i}$ = impact of category *i* for the EoL phase of the FLB.

The impacts of Scenario 2 are represented by Eq. (2) and calculated as follows:

$$I_{Scenario2,i} = \left(P_{SL,i}{}^{*}\alpha + Rep_{SL,i}\right)^{*}\gamma + U_{SL,i} + \left(EoL_{SL,i}{}^{*}\beta + EoL_{new,i}\right)^{*}\gamma \tag{2}$$

- $P_{SL,i}$ = impact of category *i* for the production phase of the repurposed EV-LIB components
- $Rep_{SL,i}$ = impact of category *i* for the repurposing phase of the SLB
- $U_{SL,i}$ = impact of category *i* for the use phase of the SLB
- $EoL_{SL,i}$ = impact of category *i* for the EoL phase of the repurposed EV-LIB components
- $EoL_{new,i}$ = impact of category *i* for the EoL phase of the new LIB components
- α = allocation factor for the EV-LIB production phase impacts to be allocated to the SLB
- β = allocation factor for the EV-LIB EoL phase impacts to be allocated to the SLB
- γ = proportioning factor for the different lifespans of the FLB and SLB.

To compare the environmental impacts of FLBs and SLBs, a normalization approach was employed using the FLB as a reference system. Eq. (3) was used to calculate the normalized results:

$$Normlized\ LCA\ results = I_{Scenario2,i} / I_{Scenario1,i}{}^{*}100\% \tag{3}$$

3.2. Life cycle inventory

The life cycle inventory (LCI) was based on secondary data from scientific literature. All life cycle stages of the FLB and SLB, except cell manufacturing, were assumed to take place in Germany. For both scenarios, the German electricity mix was presumed, unless otherwise specified in the applied Ecoinvent datasets. Cell manufacturing was assumed to occur in China, due to the country's significant position in the cell manufacturing market (International Energy Agency, 2022). The detailed inventories can be found in the Supplementary information in Section S 2.

3.2.1. Production phase

The LCI data of the module production was adapted from Jasper et al. (2022) according to the scope of this LCA. The adjustments comprised modifications of the transportation specifications and the adoption of the NMC111 cell chemistry instead of lithium-iron-phosphate (LFP) cells. NMC and LFP are the dominant cell chemistries for home storage systems (Figgener et al., 2022; Wilson et al., 2021), however, LFP does not play a significant role for EVs in the European market (Element Energy, 2019; Hill et al., 2019). Thus, EV batteries reaching their EoL and becoming eligible for a 2nd-life in the next years will most likely consist of NMC cells (Wilson et al., 2021). Besides the battery cells, the modules consist of a BMS and a module casing. Ecoinvent datasets were used for the battery cells and BMS, while the module casing was modelled according to Jasper et al. (2022).

3.2.2. Transport

The transportation modes and distances were obtained from Recharge (2018). It has to be noted that no specific manufacturing facility in China was considered for the transportation specifications. The battery cells were transported from the cell manufacturing facility in China to a container port. From there a container ship transported the cells to Germany. All other transportation processes were assumed to occur within Germany. An overview of transport assumptions is presented in Table 2.

3.2.3. Use phase

The use phase impacts are attributed to the electricity losses during charging and discharging, the electricity consumption while the system is in standby mode and the transport process to the use location. The parameters regarding the standby mode were based on Jasper et al. (2022), who assumed an energy consumption of 22.5 W and a standby time of 6000 h per year. 50 % of the time, the electricity demand is met by the PV system, while for the remaining 50 % electricity from the grid is required, as the battery is in a discharged mode. The standby electricity consumption $\boldsymbol{E_{sb,n=2934}}$ was calculated according to Eq. (4):

$$E_{sb,n=2934} = E_{del,n=2934}{}^{*}E_{sb} / \left(E_{avg} + E_{sb}\right) \tag{4}$$

where $E_{del,n=2934}$ represents the energy delivered by the battery for 2934 cycles. The number of cycles stands for the lifespan of the LIBs included in the assessment (see Section 3.1.1). E_{sb} denotes the standby electricity consumption per year and E_{avg} is the average electricity consumption of a household per year. The electricity losses (E_{loss}) represented by Eq. (5) were calculated according to Quan et al. (2022):

$$E_{loss} = \sum_{n=1}^{l_c} E^{*}(1-\xi)^{*}DoD^{*}(1-n_T{}^{*}n_{k2}) / (n_T{}^{*}n_{k2}) \tag{5}$$

where ξ denotes the capacity loss of the battery for each cycle, *DoD* denotes the depth of discharge, n_T represents the transmission efficiency and n_{k2} stands for the charge-discharge efficiency. The capacity loss (ξ) (Han et al., 2014) is expressed in Eq. (6):

$$\xi = A^{*}exp^{-(E_\alpha/RT)}{}^{*}n^{z} \tag{6}$$

where A denotes a constant, E_α is the activation energy, R is the gas constant, T represents the temperature, n denotes the number of cycles and z is the power law factor. $A^{*}exp^{-(E_\alpha/RT)}$ was simplified to a constant B (Li et al., 2016). A summary of the use phase parameters is provided in Table 3.

3.2.4. Repurposing

The repurposing phase starts with the collection and sorting of LIBs, which was based on Kallitsis et al. (2022), followed by a dismantling process to the module level. In the absence of an automated process (Rallo et al., 2020) a manual disassembly was presumed, assuming irrelevant energy consumption for tools. Thus, no environmental impacts result from this step. To evaluate their SoH and suitability for the designated application, the modules need to be tested (Hossain et al., 2019). The electricity inputs required for this step were derived from Wilson et al. (2021), who based their calculation on the cycle test regime

Table 2
Transport specifications.

Route	Location	Means of transport	Distance (km)
Cell manufacturing facility to harbour	China	Train	300
		Lorry	200
Harbour to battery production facility	China-Germany	Ship	18,000
	Germany	Lorry	200
Battery production facility to user	Germany	Lorry	200
User to repurposing facility[a]	Germany	Lorry	200
User to EoL facility	Germany	Lorry	200

[a] Applies only to Scenario 2.

Table 3
Use phase parameters.

Parameter	First-life battery (FLB)	Second-life battery (FLB)	Unit	Reference
Delivered energy ($E_{del,n=2934}$)	31,713.88	34,972.83	kWh	Own calculation
Avg. household electricity consumption (E_{avg})	3175.5	3175.5	kWh/year	See Section 3.1
Standby electricity consumption (E_{sb})	135	135	kWh/year	(Jasper et al., 2022)
Initial capacity (E)	14.4	20.57	kWh	See Section 3.1
Constant (B)	0.00362	0.00362		(Quan et al., 2022)
Cycles (n)	2934	2934	cycles	Own calculation
Power law factor (z)	0.588	0.588		(Quan et al., 2022)
Transmission efficiency (n_T)	90	90	%	(Quan et al., 2022)
Charge-discharge efficiency (n_{k2})	90	86	%	Own assumption[a]
Depth of discharge (DoD)	83	69	%	Own assumption[a]

[a] Average values based on Ahmadi et al. (2014); Bobba et al. (2018a); da Silva Lima et al. (2021); Hiremath et al. (2015); Jasper et al. (2022); Le Varlet et al. (2020); Quan et al. (2022); Wang et al. (2022).

provided by Cready et al. (2003). Next, for the exchange of battery components, we analysed four different cases, which are summarized in Table 4. First, it was assumed that the BMS and module casing need to be replaced (Base case). The BMS contains sensitive information and is not programmed for a stationary application and must therefore be exchanged (Börner et al., 2022; Canals Casals and Amante García, 2016), while a replacement of the module casing is required due to safety reasons (Bobba et al., 2018b).

The battery regulation states that "*the Commission should encourage the development of standards for design and assembly techniques that facilitate the maintenance, repair and repurposing of batteries and battery packs*" (Regulation 2023/1542). Hence, it was assumed that the module casing can be reused, given the less demanding stationary application and under the presumption that the safety is not compromised (Case 1). For Case 2, the reuse of the BMS was presumed considering the suggestion of a software reset function included in the BMS mentioned in the battery regulation. Considering these aspects, Case 3 assumed that the module casing and BMS are both reused in the stationary application. The multiple use of components in the automotive and stationary applications implies that their impacts need to be allocated between both functions (see Section 3.1.1).

Table 4
Summary of repurposing assumptions.

	Collection, sorting, dismantling	Testing	Replacement BMS	Replacement module casing
Base case	x	x	x	x
Case 1	x	x	x	
Case 2	x	x		x
Case 3	x	x		

3.2.5. End-of-life (EoL) phase

Once the batteries can no longer supply the electricity demand of the house, they are collected, sorted and dismantled into their components. Subsequently, the module casing and BMS are sent to a suitable recycling treatment facility. Ecoinvent datasets were applied for modelling the recovery of aluminium, steel, copper and precious metals. It was assumed that 90 % of the recovered aluminium and 100 % of all other metals could substitute primary resources in terms of material quality (Unger et al., 2017).

Before the metallurgical processes, the battery cells need to be discharged. The process is based on the immersion of the battery cells in a $FeSO_4$ solution (Kallitsis et al., 2022). Afterwards, the battery cells are fed to the pyrometallurgical recycling process, while in the sensitivity analysis a standard hydrometallurgical and an advanced hydrometallurgical recycling (Duesenfeld GmbH, 2022) were analysed. In contrast to the standard hydrometallurgical process, the advanced process enables the recovery of the electrolyte and graphite, the latter being considered a critical raw material (European Commission, 2023).

The inventory data is based on Mohr et al. (2020), with several adjustments made to the inventories. It was assumed that no aluminium or manganese is recovered by the pyrometallurgical process (Harper et al., 2019). Given the incineration of plastics (Windisch-Kern et al., 2022), the plastic waste fraction was replaced with a slag fraction. Discrepancies in the mass balance were identified by Mohr et al. (2020). To maintain an accurate mass balance, missing outputs were added to the slag fraction in this model concerning the pyrometallurgical and hydrometallurgical processes. Also, the inventory of the advanced process was adapted. Since data on waste streams and emissions were unavailable when Mohr et al. (2020) conducted their study, we added identical outputs as those assumed for the hydrometallurgical process. However, these presumptions may not accurately reflect the actual outputs of the process by the modelled recycler.

The recovery rates of all materials are displayed in Tables S4 and S5 of the Supplementary information. The calculation of the recovered material is presented in Eq. (7).

$$Output\ (kg) = cell\ input\ (1kg) * share\ of\ material\ in\ cell\ (\%) * recovery\ rate\ (\%) * stochiometric\ ratio \quad (7)$$

3.3. Sensitivity analysis methodology

3.3.1. Allocation factor

Given the lack of standardization on solving multifunctionality (Bobba et al., 2018a; Schulz et al., 2020), two allocation approaches were analysed. The Circular Economy Action Plan and the EU Battery Regulation emphasize facilitating the repurposing of batteries (European Commission, 2020; Regulation 2023/1542). Thus, we assume continuous efforts to enable a circular battery value chain, implying LIB production with an intended second use. Consequently, a sensitivity analysis was performed with $\alpha = \beta = 0.5$. The regulation also states that extended producer responsibility (EPR) applies to economic operators placing repurposed batteries on the market, highlighting that the original producer should not bear additional waste management costs resulting from the batteries' second-life. Therefore, the EoL impacts were fully allocated to the battery's second-life ($\beta = 1$).

3.3.2. Battery cell recycling process

Considering the imposed recycling targets for lithium, 80 % until 2031 (Regulation 2023/1542), and the economic superiority of hydrometallurgical recycling plants (Bruno and Fiore, 2023) it is anticipated that the EU will rely less on pyrometallurgical recycling plants. Thus, we analysed the influence of two hydrometallurgical recycling routes on the results. Details on the recycling processes are found in Section 3.2.5.

3.3.3. Proportioning factor

In order to determine, the influence of the proportioning factor on the results, a Monte Carlo analysis was conducted. The number of cycles a LIB can sustain until the capacity falls below 8.7 kWh was determined by the capacity loss of each cycle and is an uncertain factor in this analysis. Due to the static LCA model, the analysis could not be performed on the capacity loss of the LIBs and was hence limited to the proportioning factor, which is based on the adopted cycle life of both batteries (see Section 3.1.1). The Monte Carlo analysis comprising 3000 iterations was performed for the Base case on the proportioning factor, representing a parameter depending on the LIB's capacity loss and cycle life. A normal distribution was adopted, as indicated by previous studies addressing Monte Carlo simulations for LIBs (Barbers et al., 2024; Yu et al., 2012). More details are found in section S 5 in the Supplementary information.

4. Results

The environmental savings and drawbacks of the SLBESS compared to the FLBESS are presented in Table 5. Additionally, the normalized results of the environmental impacts of the repurposed battery and the new battery are displayed in Fig. 3, with the new battery serving as a reference scenario and, therefore, it is set to 100 %.

Considering the Base case, i.e., exchange of BMS and module casing, the SLBESS shows superior performance compared to the FLBESS in all impact categories. The highest savings of 58 % are observed for MD. Significant benefits of are also found for HT_{nc}, $PCOF_{ec}$, $PCOF_{hh}$ and TA, resulting in environmental savings of >40 %. These can be attributed to the reduced material requirements for the repurposed battery due to the reuse of the battery cells, despite the increased number of modules. The lowest environmental savings are identified for IR (13 %), resulting from minor impact reductions by cell recycling. The benefits for HT_c (17 %) also fall within the lower range, which can be attributed to the significant influence of the module casing, especially due to the higher weight of the SLB. The use phase had a more determining influence on the SLBESS, due to its higher electricity losses caused by inferior battery performance.

Reusing the module casing (Case 1) also results in environmental benefits in all impact categories. Similar to the Base case, the highest savings are obtained for MD (61 %) due to metal recycling. The most significant influence of reusing the module casing is observed for HT_c, leading to an increase in environmental benefits by almost 50 % compared to the Base case. In all other categories, the environmental benefits enhance, as these are only marginally affected by the production of the module casing.

Assuming the reuse of the BMS, the repurposed battery shows an inferior performance compared to the new battery in ET_{fw} and ET_m with drawbacks of −22 % and − 16 %, respectively. This can be attributed to the considerably lower reductions achieved through the EoL treatment of the BMS, particularly the recovery of precious metals, due to the applied allocation approach. Compared to other categories, the impact of BMS production remains relatively high, despite allocation. For the rest of the analysed categories, the SLBESS is more beneficial than the FLBESS. The highest benefits are observed for MD (59 %) and TA (50 %), mainly caused by the reduced impacts of the battery cells. The least benefits are observed for HT_{nc} (3 %) and EP_{fw} (7 %), due to reduced EoL reductions and high production impacts of the BMS. Compared to the Base case, increased savings are identified for Case 2 for all categories except for HT_{nc}, EP_{fw} and HT_c. When compared to Case 1, reusing the BMS is more beneficial for all categories, except for HT_{nc}, EP_{fw} and HT_c, EP_m and MD.

Environmental drawbacks of the SLBESS of −20 % and − 14 % are identified for ET_{fw} and ET_m, when assuming the reuse of the module casing and the BMS (Case 3). Similar to Case 2, the drawbacks and low benefits are caused by reduced recycling credits due to the allocation approach. The highest benefits are observed for MD (61 %), TA (52 %), $PCOF_{hh}$ (52 %), $PCOF_{ec}$ (51 %) and PMF (50 %). These can be attributed to reduced impacts of all analysed battery components due to allocation. Contrary to the aforementioned impact categories, these categories are less affected by the EoL treatment and thus the reduced recycling credits. In summary, Case 3 is the most beneficial option for 12 of 16 impact categories. Detailed results can be found in section S 3 of the Supplementary information.

4.1. Environmental impacts of a first-life battery energy storage system

The life cycle impacts of the new battery are displayed in Fig. 4. The production phase is responsible for most impacts, ranging between 64 % and 96 %. In particular, the manufacturing of the battery cells causes >70 % of the production impacts, except for HT_c. This category is mostly affected by the steel production for the module casing (46 %). Transportation processes play a relatively minor role, contributing 1 % to CC and a maximum of 8 % to $PCOF_{ec}$ and $PCOF_{hh}$ primarily due to the maritime transport of battery cells from China to Germany. Considerably lower impacts are identified for the use phase ranging from 4 % to 36 %. The highest influence during this phase is observed for EP_{fw} (36 %), IR (30 %) and CC (30 %). Despite the low share of required

Table 5
Environmental savings or drawbacks of a second-life lithium-ion battery storage system (SLBESS) compared to a first-life lithium-ion battery storage system (FLBESS) for each repurposing case. (Base case = New BMS and module casing; Case 1 = New BMS and reuse of module casing; Case 2 = New module casing and reuse of BMS; Case 3 = Reuse of BMS and module casing.)

Impact category	Unit	Base case	Case 1	Case 2	Case 3
CC	kg CO_2-eq.	2.85×10^{-2}	3.16×10^{-2}	3.48×10^{-2}	3.79×10^{-2}
PMF	kg CO_2-eq.	7.29×10^{-5}	7.80×10^{-5}	8.81×10^{-5}	9.32×10^{-5}
FD	kg oil-eq.	9.36×10^{-3}	1.04×10^{-2}	1.15×10^{-2}	1.25×10^{-2}
ET_{fw}	kg 1.4 DB-eq.	1.45×10^{-3}	1.60×10^{-3}	-1.65×10^{-3}	-1.50×10^{-3}
EP_{fw}	kg 1.4 DB-eq.	7.86×10^{-6}	9.05×10^{-6}	2.83×10^{-6}	4.02×10^{-6}
HT_c	kg 1.4 DB-eq.	1.64×10^{-3}	3.16×10^{-3}	2.71×10^{-3}	4.23×10^{-3}
HT_{nc}	kg 1.4 DB-eq.	5.07×10^{-2}	5.49×10^{-2}	4.33×10^{-2}	8.58×10^{-2}
IR	kBq Co-60 eq. to air	8.42×10^{-4}	1.08×10^{-3}	1.42×10^{-3}	1.66×10^{-3}
ET_m	kg 1.4 DB-eq.	1.88×10^{-3}	2.07×10^{-3}	-1.56×10^{-3}	-1.37×10^{-3}
EP_m	kg N-eq.	2.94×10^{-6}	3.28×10^{-6}	3.19×10^{-6}	3.52×10^{-6}
MD	kg Cu-eq.	1.26×10^{-3}	1.31×10^{-3}	1.27×10^{-3}	1.33×10^{-3}
$PCOF_{ec}$	kg NOx-eq.	9.55×10^{-5}	1.02×10^{-4}	1.08×10^{-4}	1.15×10^{-4}
$PCOF_{hh}$	kg NOx-eq.	9.44×10^{-5}	1.01×10^{-4}	1.07×10^{-4}	1.13×10^{-4}
SOD	kg CFC-11-eq.	1.03×10^{-8}	1.12×10^{-8}	1.27×10^{-8}	1.36×10^{-8}
TA	kg SO_2-eq.	1.98×10^{-4}	2.07×10^{-4}	2.29×10^{-4}	2.37×10^{-4}
ET_t	kg 1.4 DB-eq.	3.09×10^{-1}	3.23×10^{-1}	3.99×10^{-1}	4.14×10^{-1}

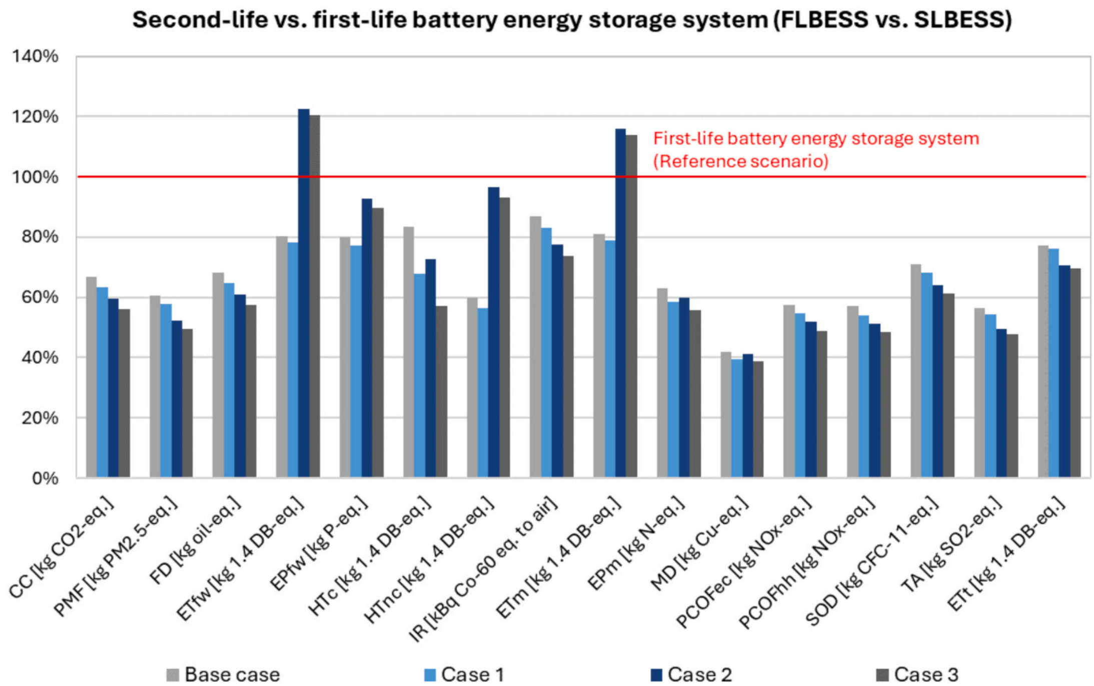

Fig. 3. Normalized environmental life cycle impacts of a second-life lithium-ion battery storage system (SLBESS) compared to a first-life lithium-ion battery storage system (FLBESS), which is the reference scenario. (Base case = New BMS and module casing; Case 1 = New BMS and reuse of module casing; Case 2 = New module casing and reuse of BMS; Case 3 = Reuse of BMS and module casing).

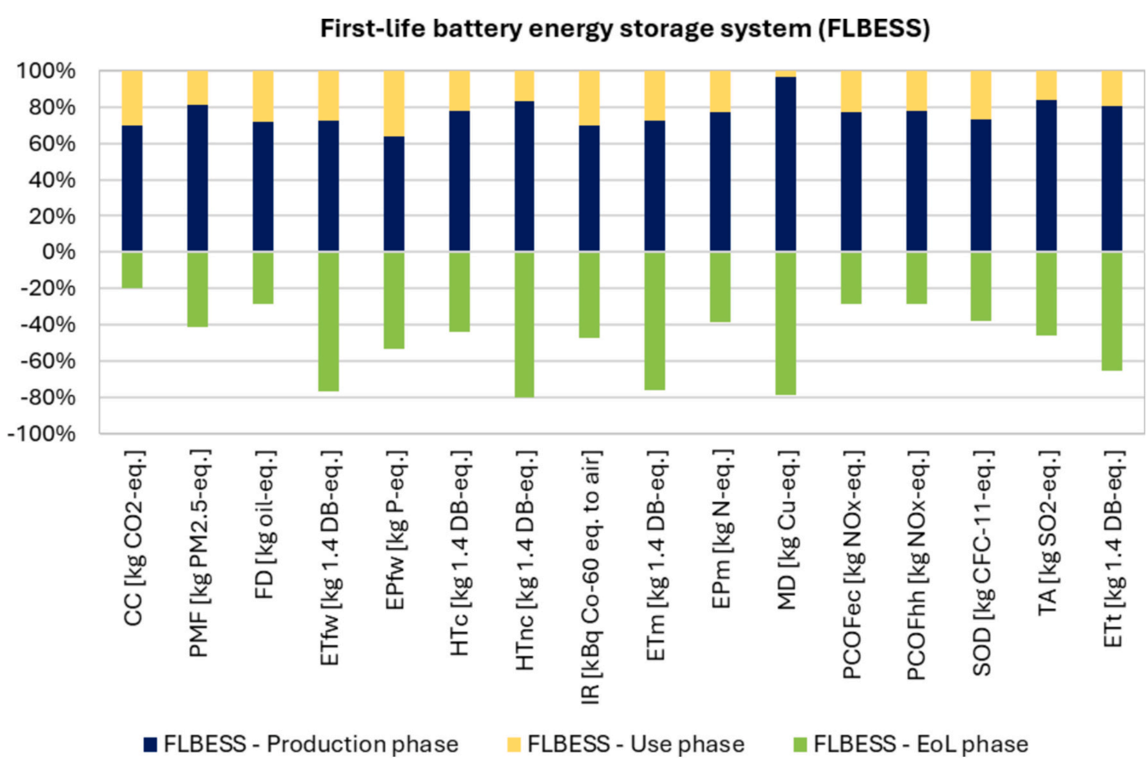

Fig. 4. Relative life cycle impacts of a first-life lithium-ion battery storage system (FLBESS) ($I_{Scenario1}$) for a household. Green bars indicate the end-of-life (EoL) phase, including battery recycling, resulting in environmental benefits (negative savings values).

electricity from the grid (only for 50 % of standby electricity consumption), >50 % of the use impacts of IR and EP_{fw} can be attributed to this input. Impact reductions between 20 % and 80 % are obtained for all categories due to the recovery of valuable resources through recycling, whereby cell recycling has the highest impact on the reductions. Specifically, the recovery of cobalt and copper play a crucial role. For HT_c the recycling of the module casing, i.e., the recovery of steel, has the highest influence, while the treatment of the BMS contributes significantly to the reductions observed for ET_{fw}, EP_{fw}, HT_{nc} and ET_m, particularly due to the recovery of precious metals.

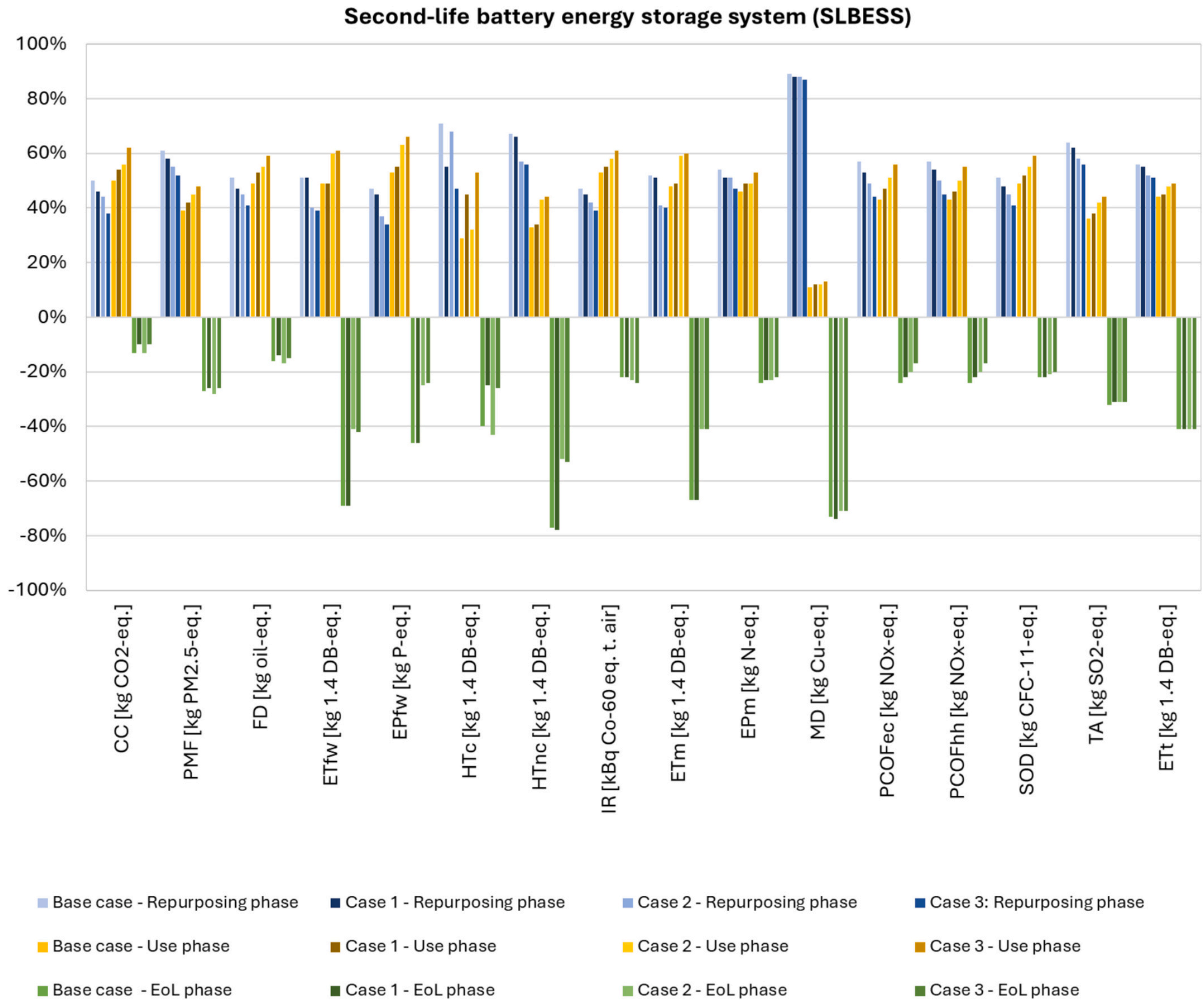

Fig. 5. Relative life cycle impacts of a second-life lithium-ion battery storage system (SLBESS) ($I_{Scenario2}$) for a household considering different repurposing cases (Base case = New BMS and module casing; Case 1 = New BMS and reuse of module casing; Case 2 = New module casing and reuse of BMS; Case 3 = Reuse of BMS and module casing). Green bars indicate the end-of-life (EoL) phase, including battery recycling, resulting in environmental benefits (negative savings values).

4.2. Environmental impacts of a second-life battery energy storage system

The relative environmental impacts of the life cycle stages of the second-life batteries are shown in Fig. 5 for each analysed repurposing scenario. The use phase will only be discussed for the Base case, as the absolute impacts remain unchanged across different repurposing scenarios. Considering the Base case, the use phase causes ≥50 % of the total impacts for EP_{fw}, IR and CC. Similar to the new battery system, a non-negligible part of these impacts is caused by the grid electricity required to cover half of the standby electricity consumption of the BESS. The rest of the use phase impacts is mainly caused by electricity losses during charging and discharging.

In the repurposing phase, assuming the Base case, the production of the module casing is responsible for a major share of the HT_c impacts (66 %). Accordingly, reusing the module casing (Case 1) highly affects this category and reduces the impacts of the repurposing process by 16 %. BMS production significantly contributes to ET_{fw} and ET_m with 48 %, to HT_{nc} with 47 % and to EP_{fw} with 46 %. Thus, reusing the BMS (Case 2) is the most beneficial solution for these categories. Other impact categories are highly influenced by the battery cells. For instance, the cells contribute 78 % to the MD impacts from repurposing, 76 % to ET_t and 64 % to TA. No significant reductions in the repurposing phase are identified when assuming different repurposing scenarios. Overall, Case 3 shows the lowest influence of the repurposing phase on the total impacts. Regardless of the repurposing case, transportation has only a minimal impact, contributing <1 % across all categories.

The EoL phase results in impact reductions for all categories ranging from 13 % to 77 % assuming the Base case. Cell recycling remains the decisive process in the EoL phase for most impact categories, despite reduced recycling credits due to the applied allocation approach. However, the BMS treatment became the determining process for impact reductions on ET_{fw}, EP_{fw}, HT_{nc} and ET_m (> 60 %). The recycling of the module casing proves to be particularly beneficial for HT_c contributing 79 % to the overall reductions. Compared to the Base case, the impact reductions obtained in Case 1 remain in a similar range, except for HT_c. For this category, the EoL reductions are substantially reduced due to the allocation procedure. This also applies to Case 2 and Case 3, where BMS recycling is the decisive factor for impact reductions.

4.3. Sensitivity analysis results

4.3.1. Allocation factor results

Regarding the multifunctionality of the repurposed battery, the influence of the allocation approach was investigated by setting the allocation factors α and β to 0.50 and 1. Changing the allocation factors to $\alpha = \beta = 0.5$, i.e., 50 % of the environmental burdens of the production and EoL phase are allocated to the second-life, leads to a decrease in savings for all scenarios. Considering MD, the savings decrease by 20 % for the base case and Case 2, and by 21 % for Case 1 and Case 3. HT_{nc} and EP_{fw} show the lowest decrease in savings for Case 2 (3 % and 5 %) and Case 3 (4 % and 6 %). These can be attributed to the higher impacts of the repurposing stage, which are not compensated by the higher reductions through the EoL treatment, thus resulting in less savings. In contrast, the drawbacks identified for Case 2 and Case 3 for ET_m and ET_{fw} are reduced when applying an allocation factor of 0.5. This is due to the increased EoL reductions, resulting in lower life cycle impacts compared to when $\alpha = \beta = 0.25$.

Setting the allocation factor to $\beta = 1$, meaning that 100 % of the calculated environmental impacts of the EoL phase are allocated to the SLBESS, significantly increases the savings in all impact categories and scenarios. The highest increases are observed for MD with savings of 326 %, 335 %, 341 % and 349 % for the base case, Case 1, 2 and 3, respectively. Substantial savings (between 179 % to 225 %) are identified for ET_{fw} and ET_m, which previously resulted in environmental drawbacks. An overview of allocation factor differences is shown in Fig. 6.

4.3.2. Battery cell recycling method results

Assuming standard hydrometallurgical and advanced hydrometallurgical recycling, the benefits of the SLB are slightly reduced compared to pyrometallurgical recycling. This is due to reduced life cycle impacts of the FLB resulting from higher recycling credits. For all cases, the categories EP_m and MD are mostly affected by alternate recycling methods. Specifically, with advanced hydrometallurgical recycling, the savings for MD are reduced by 31 %, while with the standard hydrometallurgical recycling, the savings for EP_m are reduced by 24 % (Base case). This is because the EoL credits of the FLB increase notably with hydrometallurgical methods for these impact categories. The identified drawbacks for ET_{fw} and ET_m with pyrometallurgical recycling (Cases 2 and 3) increase when applying standard and advanced hydrometallurgical treatments. Assuming standard hydrometallurgical recycling, EP_{fw} and HT_{nc} show environmental drawbacks of −2 % and − 3 %, respectively for Case 2. This applies also when assuming advanced hydrometallurgical recycling with drawbacks of −5 % and − 16 %, respectively. These categories also show drawbacks of −2 % (EP_{fw}) and − 12 % (HT_{nc}) and relatively low environmental benefits assuming pyrometallurgical recycling. The reason for these drawbacks is the improved performance of the FLB. Despite the increased recycling credits from hydrometallurgical recycling processes, the SLBESS showed inferior results because of allocation (see Fig. 6).

4.3.3. Proportioning factor results

The variation of the proportioning factor affects most impact categories with noticeable deviations occurring within a short, but statistically relevant, range. The resulting variation in output flows for most impact categories is concentrated within a ± 6 % range of the mean results. CC, FD, EP_m and SOD are less affected by the uncertainty of the proportioning factor, with an 87–89 % probability of experiencing a variation of ±4.8 % in the parameter. In the Monte Carlo simulation, these categories showed a standard deviation of <0.06 % from the results, therefore indicating a high level of stability. The impact category

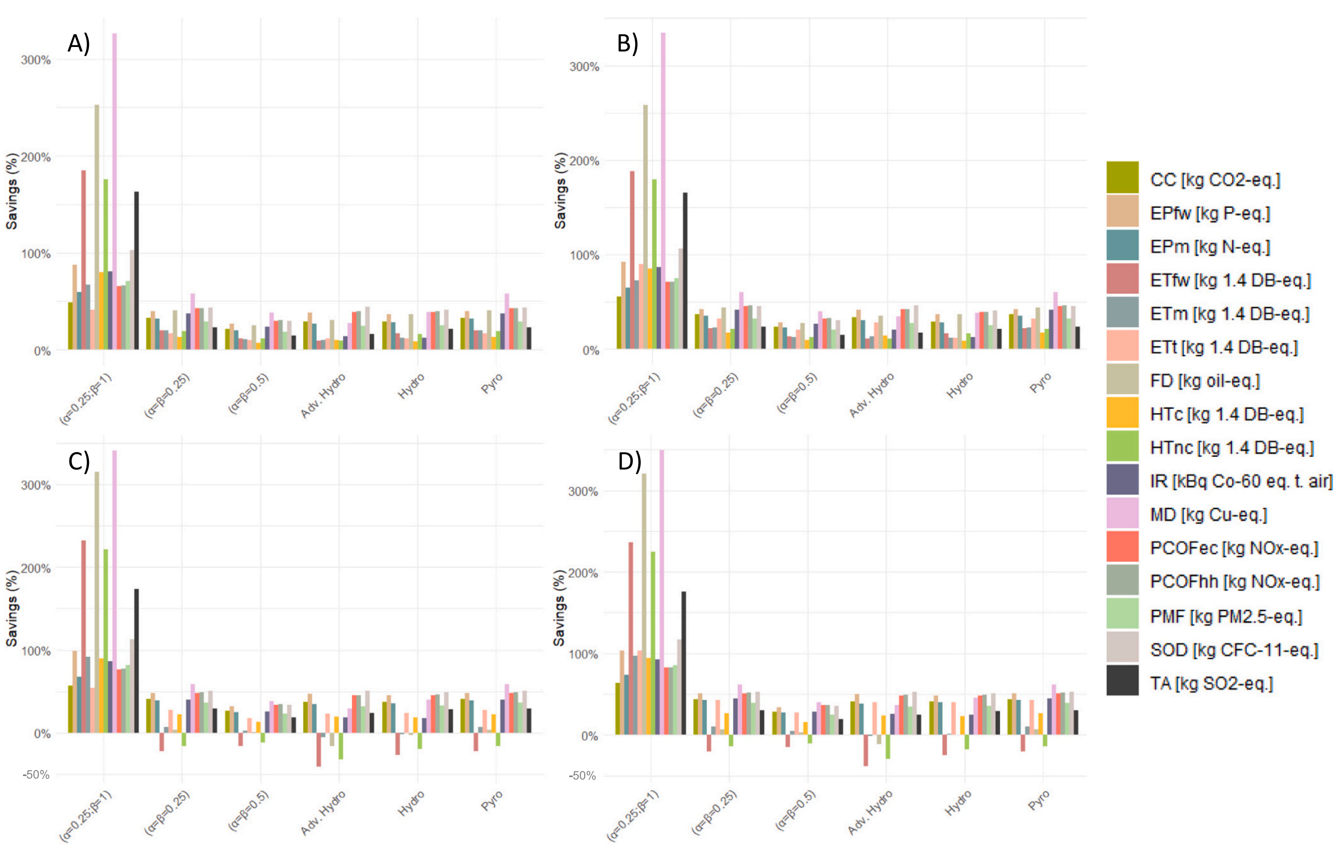

Fig. 6. Comparison of the sensitivity analysis results between 4 repurposing cases: A) Base case = New BMS and module casing, B) Case 1 = New BMS and reused module casing, C) Case 2 = New module casing and reuse of BMS D) Case 3 = Reuse of BMs and reused module casing, and two factors: allocation factors ($\alpha = 0.25$; $\beta = 1$, $\alpha = \beta = 0.25$, and $\alpha = \beta = 0.5$) and recycling technologies (advanced hydrometallurgy, hydrometallurgy, and pyrometallurgy).

mostly influenced by the uncertainty in this parameter is HT_{nc}, which exhibited a standard deviation of 12.3 % from the base LCIA results. In this category, approximately 70 % of the output flows are impacted by parameter variations exceeding ±4.8 %. Other impact categories highly affected by this parameter are ET_{fw}, ET_{m} and MD, showing a standard deviation between 6 and 7.5 %. Overall, in the Monte Carlo simulation, the average standard deviation of the impact categories was 4.6 %.

5. Discussion

Our study presents new insights into the environmental performance of SLBESS, by exploring various repurposing strategies. Notably, the SLBESS showed a better environmental performance than the FLBESS across most impact categories, underscoring the environmental advantages of applying repurposed LIBs in household energy storage systems. A key finding of this study is that varying repurposing strategies have a significant influence on the environmental performance of SLBESS, with the highest benefits obtained when both the module casing and the BMS are reused. However, reusing only one component also proved advantageous in most impact categories, highlighting the critical need for accurate assessments of the repurposing stage, when conducting LCAs of SLBESS. Otherwise, potential environmental improvements could be overlooked and the overall impact of SLBESS overestimated. To enable such repurposing strategies, standards for design for repurposing as suggested in the battery regulation would be essential.

Overall, it must be noted that the feasibility of such repurposing strategies faces some challenges. First, the module casing might be damaged during the disassembly process due to the use of adhesives connections (Graner et al., 2023), which should therefore be addressed when thinking of design for repurposing. Another aspect concerning the reuse of the module casing is the material composition. The benefits of reusing the module casing observed in this study can be attributed to the avoided steel. However, module casings are also made from aluminium alloys or plastics (E-Mobility Engineering, 2024), which could alter the environmental impacts and potentially reduce the benefits. Second, reusing the BMS necessitates a software reconfiguration, which requires access to the BMS software by the repurposing operator (Börner et al., 2022). As the BMS contains sensitive data, it is unlikely that original equipment manufacturers would share this information with independent operators. However, with the introduction of the battery passport, more information might become available to supply chain actors (Regulation 2023/1542), potentially reducing obstacles faced by repurposing operators. Additionally, design for repurposing could include the suggested reset function, enabling the reuse of the BMS without compromising data privacy. Another crucial factor influencing the environmental impacts of both, new and repurposed LIBs, is the electricity mix. This is highlighted by the significant impact of standby electricity consumption on certain impact categories, despite its low share, which underscored the importance of a decarbonized electricity mix. Electricity losses due to battery cycling were particularly relevant for the SLB. However, the calculation was based on secondary data and assumptions and might not represent a realistic battery behaviour. Hence, incorporating primary data would provide a more accurate representation of the use phase impacts. Transport processes have a minor influence on the environmental impacts of the battery life cycle. For SLBESS, transport contributed 1 % across all impact categories, while for FLBESS, the highest contribution was observed for $PCOF_{ec}$ and $PCOF_{hh}$ accounting for 8 %. While repurposing was assumed to occur in Germany, if China - a major hub for battery cell production - were to establish repurposing facilities, EoL-LIBs would need to be transported to and from China, increasing the impacts associated with maritime transport. This shift would also result in more domestic transport within China, raising overall transport impacts. Given these considerations, efficient and sustainable planning of transport routes is crucial to minimizing environmental impacts. Ideally, repurposing should take place close to where batteries are decommissioned or retired to reduce transport efforts. However, should China become a key player in battery repurposing, prioritizing efficient and sustainable transport infrastructure will be essential to ensuring the sustainability of repurposed LIBs.

Other authors also found SLBESS to be environmentally beneficial, although the extent of reported environmental savings varies. For instance, Wilson et al. (2021) reported CC reductions of 15 % compared to a storage system with new LIBs. Yang et al. (2020) studied the environmental performance of SLBESS compared to lead-acid batteries and found CC reductions of 20 %. Environmental savings between 10 % and 44 % were reported by Kamath et al. (2020) for a residential BESS in different locations. Depending on the assumed repurposing strategy, we identified CC savings within a higher range between 33 % and 44 %., which underscores the importance of optimized repurposing strategies. However, it has to be noted that these differences can also be attributed to several other aspects, such as battery characteristics, choice of system boundaries, LCIA method and multifunctionality approach. Therefore, ongoing harmonization efforts, such as the Product Environmental Footprint (PEF) (European Commission, 2021), have the potential to improve the comparability of LCA results of repurposed LIBs.

Despite various benefits, our study identified two impact categories, relating to ecotoxicity, where SLBESS showed higher impacts than the new batteries, when the BMS is reused. The sensitivity analysis proved that this is caused by the allocation procedure, especially due to reduced EoL credits with significant variations in results (−22 % to 233 %), depending on the assumed allocation factors. This highlights the substantial impact of the chosen allocation approach, which has also been reported by previous studies (Bobba et al., 2018a; Cusenza et al., 2019a; Wilson et al., 2021). A system expansion approach could provide significantly different results, as it would comprehend both uses of the battery. Thus, the reference system would consist of two independent LIBs for each use, with their respective production and disposal impacts. These results underline the need for clearer guidance on solving the multifunctionality of LIBs, as discussed by Schulz et al. (2020). The 'Circular Footprint Formula' (CFF) constitutes another approach to address this issue, however its application to repurposed LIBs presents methodological challenges. Specifically, determining how second-life LIBs should be classified in terms of their function and lifetime extension, as outlined in a recommendation by the European Commission (European Commission, 2021), is not a straightforward task. While the CFF is more comprehensive than other approaches, its complexity increases due to the number of parameters involved (Malabi Eberhardt et al., 2020; Yang et al., 2024). It would be beneficial for future studies to have more detailed instructions on how to apply the CFF to products such as repurposed LIBs with a complex material composition of the battery cells and the necessary components for a module (e.g., battery management or thermal management systems).

Another factor influencing the results is the chosen recycling route, although the variations were much smaller compared to the allocation factor. Both hydrometallurgical recycling pathways improved the overall environmental performance of FLBESS and SLBESS. However, when comparing the two battery systems, the SLBESS achieved fewer benefits with hydrometallurgical recycling processes than with pyrometallurgical recycling. Nevertheless, it should be noted that the EoL phase is highly influenced by allocation. Despite the achieved impact reductions, the pyrometallurgical process cannot recover lithium, manganese or graphite, which are classified as critical raw materials (European Commission, 2023). This is specifically relevant considering the recycling targets set in the EU battery regulation (e.g., 80 % for lithium by 2031). Therefore, recycling processes should be optimized to fulfil the requirements set by the EU.

Besides the recycling route, also the proportioning factor affects the results, which was addressed through Monte Carlo simulations. Low uncertainties were identified for CC, while the highest were found for toxicity related categories and MD, indicating that the cycle life of the battery, represented by the proportioning factor, highly affects the environmental performance in these categories. This can be attributed to the EoL phase, as the recycling reductions highly impact these categories. Recycling was identified as an important measure to reduce the environmental impacts of LIBs in these categories by previous studies (Duarte Castro et al., 2022; Feng et al., 2022; Koroma et al., 2022). Philippot et al. (2023) also found robust results for CC in their LCA of a third-generation LIB. Still, they identified high uncertainty for HT_c, which they attributed to limitations in evaluating toxicity impact categories mainly because of the large number of associated substances.

Our study also encountered several limitations. First of all, general uncertainties of LCAs need to be considered, when interpreting the results. For instance, methodological aspects like the choice of functional unit or system boundaries as well as data availability and quality highly affect the outcome of the study (European Commission - Joint Research Centre, 2010). Second, our model assumed that a household uses the full capacity of the LIB each day until the average electricity demand can no longer be met. However, this assumption does not reflect realistic household consumption patterns. Thus, including primary data on a household's electricity consumption over time would deliver more robust results. The expected increased electricity demand due to EV charging and heat pumps (International Energy Agency, 2024) is another influencing factor requiring further research. Second, the exclusion of the impact categories land use and freshwater consumption presents another limitation of this study, as it effects the comprehensiveness of the environmental assessment. While the potential contributions to all three endpoint categories were addressed through the assessment of other impact categories, it is crucial to acknowledge that land use and water consumption can affect the overall sustainability of BESS. Thus, future studies could benefit from the collection and integration of LCI data for these categories, particularly for reuse, remanufacturing, and repurposing processes, to capture the sustainability of repurposed LIBs more accurately.

6. Conclusions

This study analysed the environmental benefits of applying repurposed batteries in a household BESS. In addition, the influence of varying repurposing inventories on the environmental performance of SLBESS was investigated. The results of the LCA demonstrate that using SLBESS can provide significant environmental benefits. Particularly, reusing both the module casing and BMS in the second-life application proved to be beneficial. However, the technical and economic feasibility of these approaches remains uncertain. Thus, original equipment manufacturer should be encouraged to consider potential SLB applications when designing new battery packs. By doing so, potential damages during the disassembly process could be avoided in the sense of the design for repurposing principle. Also, integrating a reset-function in the BMS, as suggested in the battery regulation, would enhance the sustainability of SLBESS without compromising data confidentiality. Despite various advantages, two impact categories, freshwater and marine ecotoxicity, resulted in environmental drawbacks. However, the drawbacks were mainly caused by the allocation procedure, which underlines the need for clearer instructions and rules on handling the multifunctionality of SLBs, to reflect realistic scenarios. Battery recycling was found to be a crucial step in reducing the environmental impacts of both new and repurposed LIBs. Therefore, recycling and repurposing should not be viewed as opposing strategies, as SLBESS producers are currently competing with recyclers for cheap EoL-batteries. In such cases, the objective of extending the operational lifespan through a battery's second-life should take precedence over recycling and ultimate disposal.

While our findings highlight the potential of applying SLBESS and employing enhanced repurposing strategies, further research is needed in several aspects. First, the technical feasibility of reusing battery components needs closer investigation. Expert opinions could provide valuable insights in this regard and could help to establish realistic repurposing scenarios. Second, future studies should consider changes in the electricity demand and consumption patterns of households as well as the electricity mix by applying dynamic LCA modelling. In future LCA studies, a system expansion approach or the 'Circular Footprint Formula' (CFF) could be employed to address multifunctionality and to consider all externalities, particularly the environmental impacts of the LIB's first-life in EVs, and the environmental benefits using recycled, reused, remanufactured, or repurposed battery components. It is also important to take future technological developments into account, as next-generation batteries such as sodium or solid-state batteries are already on the market.

CRediT authorship contribution statement

Anna Spindlegger: Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Lyubov Slotyuk:** Writing – review & editing, Validation, Formal analysis, Data curation. **Aleksander Jandric:** Writing – review & editing, Supervision, Project administration, Methodology, Investigation, Formal analysis, Conceptualization. **Ricardo Gabbay De Souza:** Writing – review & editing, Validation, Supervision, Software, Methodology, Data curation. **Stefanie Prenner:** Writing – review & editing, Investigation, Conceptualization. **Florian Part:** Writing – review & editing, Visualization, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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