12 (2021) 100116

Contents lists available at ScienceDirect

International Journal of Thermofluids

journal homepage: www.sciencedirect.com/journal/international-journal-of-thermofluids

Comparative environmental life cycle assessment of conventional energy storage system and innovative thermal energy storage system

Borbala Rebeka David [a], Sean Spencer [a], Jeremy Miller [a], Sulaiman Almahmoud [a], Hussam Jouhara [b,*]

[a] Spirax Sarco Engineering PLC, Cheltenham GL51 9NQ, United Kingdom

[b] Heat Pipe and Thermal Management Research Group, College of Engineering, Design and Physical Sciences, Brunel University London, UB8 3PH, United Kingdom

ARTICLE INFO

Keywords:
Thermal energy storage
Lithium iron phosphate battery
Life cycle assessment
Terrestrial ecotoxicity
Environmental impacts

ABSTRACT

As policies have been implemented globally to limit the production of greenhouse gases (GHGs) and the effects of climate change, the generation of electricity by renewable technologies has started to increase. The development of sustainable energy storage solutions has also become more important. The continued use of conventional chemical batteries presents environmental issues such as heavy metal pollution and the use of unsustainable resources.

An environmental Life Cycle Assessment (LCA) has been conducted to analyse the environmental impact of an innovative Thermal Battery (TB) and was compared with the impact of a Lithium Iron Phosphate Battery (LIPB) using a "cradle-to-gate" approach to establish the system boundaries. The study used the findings from existing literature to determine the environmental impact of the LIPB. The life cycle inventory for the TB was constructed based on a model and available literature. In this regard, the two products were compared on 10 impact categories, and the results indicated that the TB performed better in 8 categories on average. The highest impact observed from the TB was in terrestrial ecotoxicity, where it emitted above 7000 times more than the LIPB, amounting to approximately 0.0153 after normalisation. The highest normalised environmental load in the study was indicated to be in the category of marine ecotoxicity by the LIPB at 0.27, which was significantly higher than any load for the TB. Overall, the results obtained are encouraging for the TB, but it is recommended that a field study is completed to verify the assumptions made in this paper and to achieve a better comparability with studies conducted similarly.

1. Introduction

Climate change and the implementation of new environmentally focused legislation has shifted the focus for scientists and engineers towards more climate-neutral solutions. In this regard, many countries around the world have started collaborative projects to combat the effects of climate change caused by Greenhouse Gases (GHGs) and harmful carbon dioxide emissions [1]. For instance, the Paris Agreement is the first legally binding global agreement of its kind, which has been signed by almost 190 parties and influences the participating countries' internal policies considerably such as the European Green Deal [2,3]. Another significant driver for the development of sustainable energy storage is the increasing proportion of renewable energy generation. In 2019, the UK generated 37% of its electricity using renewable resources, and in Q3 of the same year, they exceeded the amount of energy produced by coal, oil and gas combined for two consecutive months [4,5]. The reduction in fossil fuel consumption within the UK forms part of a progression towards a low carbon economy with zero net GHG emissions by 2050, as per the Climate Change Act. Similar trends are being observed globally, such as the Kyoto protocol, which is a global legal agreement for industrialised countries to significantly reduce emissions. The binding global agreement further pushes industries to develop and install technologies, in a bid to meet emission targets.

The influence of legislation accelerates the development and the demand for climate-neutral solutions, for both existing and novel technologies. With the increase in electricity generation via renewable technologies, the need for sustainable storage methods is of great importance.

In general, energy storage solutions can be classified in the following solutions: electrochemical and batteries, pumped hydro, magnetic, chemical and hydrogen, flywheel, thermal, thermochemical,

* Corresponding author.
E-mail address: Hussam.Jouhara@brunel.ac.uk (H. Jouhara).

https://doi.org/10.1016/j.ijft.2021.100116

Available online 25 September 2021

Nomenclature

Acronym	description
TB	thermal Battery
LIPB	lithium Iron Phosphate Battery
NMP	n-methylpyrrolidone
PVDF	vinylidene fluoride homopolymer
CMC	sodium carboxymethylcellulose
SBR	waterborne styrene-butadiene latex
LCA	life Cycle Assessment
FU	functional Unit

compressed air, and liquified air solutions [6–8]. The most common solution of energy storage for heating applications is thermal storge via sensible and latent heat storage using phase-change materials (PCMs), and thermochemical storage [9–11]. It should be noted that thermal energy storage contributes to peak shaving [12]. Furthermore, latent heat storage offers larger heat storage capacity than sensible storage. However, one of the main issues with latent heat storage by PCMs is its low thermal conductivity which limits expanding its application [13]. Another solution for latent heat storage is using a steam accumulator coupled with direct steam generator solar-based systems [14]. In addition, the utilisation of steam accumulator in solar-based heat recovery addresses the issue of the mismatch between the steam production and demand rates and the intermittent solar irradiance [15].

Moreover, the current technologies for electrochemical storage are split into two main branches: capacitors and batteries, as shown in Fig. 1. Capacitors are a suitable alternative to batteries, by offering high efficiencies and an increased lifespan. The downfall of the capacitor lies within the capacity, as batteries can store 30 times more charge per unit per mass in comparison to a capacitor [16]. Although unsuitable for large scale applications, the application of capacitors has been applied in small-scale energy recovery such as nanogenerators [17], energy harvesting via vibrations [18], and piezoelectric technology. The application of capacitors has been investigated in personal electronic devices to reduce the application of batteries and the dependency of using a wired electricity supply to charge a personal electronics device.

However, the role of batteries has been widely noted in energy storage systems, with usage in multiple applications and integration within renewable technology systems [19,20]. A study conducted by Dhiman and Deb [21] shows the addition of a lithium ion based battery energy storage system to create a hybrid wind farm. The study investigated the optimisation of a wind farm turbine to improve the battery charge and discharge cycle. A key point to note from the study was the improved dynamic load due to the reduction of wake zones. The overall outcome of the study highlighted that the life span of the system increased by 12.6% and the study extended the cradle-to-grave design principle whilst maximising the output of the system.

A similar combination of technologies was investigated numerically by Yuan et al. [22]. The study investigated the addition of a battery energy storage system, to level the power mismatch as a result of the power fluctuations produced by a wind farm. The study presents the addition of a dual battery energy storage system to eliminate incomplete charge and discharge cycles experienced by a single unit battery energy storage system. By allowing complete charge and discharge cycles, the lifespan of the battery system is significantly increased and down time due to maintenance and wastage caused by frequent replacement of batteries is minimised. The outcome of the study shows the potential lifespan of the system being 832 weeks (16 years), although the study shows a further potential for optimisation via LCA analysis. The study could be taken further by including a thermal management system to prolong the lifespan to match the windfarm lifespan (20 years).

By limiting the mismatch between energy cycles, the amount of curtailed energy is reduced considerably, but optimisation still has not been fully achieved. Siddique and Thakur [23] presented a study on harnessing curtailed energy from wind farms by proposing mobile battery storage systems. The study highlighted an interesting point by stating that approximately 400GWh of curtailed wind energy is available in Germany per year, showing the potential for wind energy but also the need for a suitable storage system. The study was conducted via MATLAB Simulink to calculate the potential of mobile battery units, as a replacement for fossil fuel-based generators. The investigation compared an off-grid single unit battery with an estimated consumption of 1500 kWh against a diesel-based generator. The results showed that 1.13 GWh of curtailed energy was generated within 22 h but for the purpose of transportability, the results indicate the viability to charge a 1.5 MWh battery. The study compared relative environmental factors and highlighted a potential saving of 8.4 million tonnes in carbon

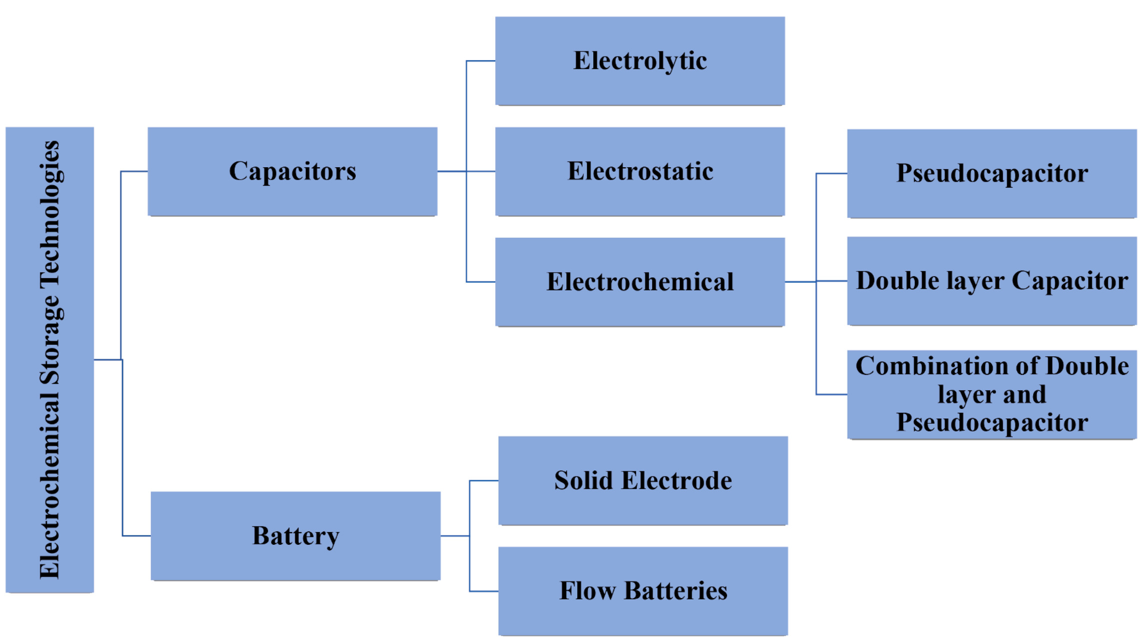

Fig. 1. Overview of electrochemical storage technologies.

dioxide emissions.

The use of battery energy storage systems has been branching out into hybrid systems, consisting of tidal, solar and wind, in a single network. The combination of wind, tidal and battery energy storage systems has been numerically investigated by Mousavi [24]. The study investigated the viability of an integrated system to provide energy to remote locations such as coastlines. The research suggested a framework to integrate both systems without the need for additional converters and controllers. The associated results highlight the successful theoretical integration of both technologies whilst satisfying the current economic investment in renewable technologies. In reality, the combination of tidal and wind energy networks highlights the potential problem in environmental impact and LCA analysis. As each technology is a separate entity, the life cycle of each technology differs and this mismatch can lead to an increase in replacement parts which can compound into larger carbon footprints [25,26]. Luerssen et al.[27]. studied the life cycle cost (LCC) of a PV-powered buildings with off-grid cooling system with thermal energy and battery storage. It was concluded that the batteries could reduce the LCC around 17% compared to a diesel-powered generator.

It is evident that the use of batteries enhances the potential of renewable technologies and solves a number of issues, such as power disparities, and actively reduces carbon dioxide emissions, but certain issues have not been addressed. The continued use of conventional chemical batteries presents further significant environmental issues, such as the consumption of resources, the method of production and the use of heavy metals [20,28,29]. Furthermore, the vast extraction of batterie raw materials and recycling processes are intensive energy and water processes [30]. As a result, there are additional negative impacts on the environment due to the expansive usage of chemical batteries [30].

The reality of batteries needs to be considered and investigated objectively from economic and environmental perspectives. The investigation needs to compare the level of emissions produced during manufacture and the reduction in emissions as a result of their application in battery energy storage systems. The use of batteries needs to be considered, as the lifespan and discharge cycles change with time. Various factors such as fast discharge conditions, thermal runaway, high current rates and unsuitable operating conditions can impact the efficiency, usability and output of the batteries [31,32]. In this regard, the influence of temperature in relation to the operation and performance of the cells has been widely noted in literature. The effect of temperature has a significant impact on the capacity and output efficiency from batteries. A study conducted by Jouhara et al. [33] presents a novel method to improve the efficiency of battery packs. The study highlights the design of an innovative thermal management technique using heat pipe technology, to form a base mat structure as shown in Fig. 2. The developed heat pipe was tested with an automotive lithium ion battery pack. The premise of the study involved the charge and discharge of the battery pack, to mimic real time operation and the associated thermal variations across the battery. The role of the heat mat aimed to cool the battery pack to the optimum operating temperature to maximise the efficiency and life span of the battery. The results from the study showed the successful cooling of the battery pack to maintain the optimum operating conditions. The study indicated the potential of maximising the life span of a battery whilst reducing the production of replacement batteries.

It is widely noted in many studies that the different chemical compositions of batteries offer different performance characteristics [34,35]. Nevertheless, waste heat generation from cells has been described as a major problem, as it is linked to several issues related to lifespan and capacity loss [36]. For instance, as concluded by Amine et al. [37], the chemical reaction which occurs in the $LiFePO_4$ electrode and the generation of high temperature is directly related to the dissolution of Fe^{2+} and the subsequent loss of capacity and life of the battery. Moreover, Song et al. [38]observed in a study, that batteries which are operating at high temperatures will degrade faster by nearly 25%. This subsequently makes the battery less effective in terms of efficiency when charging and discharging at a high rate.

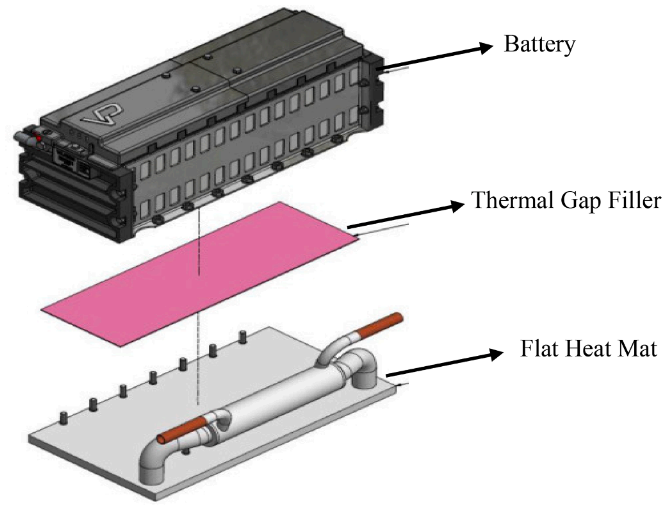

Fig. 2. Heat mat battery thermal management assembly [33].

Having considered chemical batteries and the need for an appropriate thermal management technique to increase their life expectancy and effective output efficiency, it is found that the use of thermal management techniques directly impacts the size of the carbon footprint of the life cycle of the battery. In this regard, several studies have been conducted to calculate the environmental impact of different types of batteries by conducting Life Cycle Assessments (LCAs). LCA is described as a methodology to assess the environmental impacts associated at all stages of a product life cycle [39]. By performing an LCA, the advantages or disadvantages of producing a certain type of product and the effect on the environment can be evaluated and compared with other technologies. Fig. 3 graphically illustrates the procedure of conducting LCA based on two scenarios of producing batteries, one is a recyclable scenario and the other is a non-recyclable scenario.

For instance, Yajun [41] conducted an LCA and discovered the toxicity impact of lithium-ion, nickel metal hydride, nickel cadmium and lead-acid batteries on human health. Zackrisson et al. [42] demonstrated the environmental impact of lithium-ion batteries used in electric vehicles, in comparison to two other solvent battery types, through performing an LCA. Majeau-Bettez [43] investigated several different impact categories by developing a comparison between nickel metal hydride, nickel-cobalt-manganese-lithium-ion and iron phosphate lithium-ion batteries in different functional units. Messagie et al. [44] investigated the availability and demand of lithium, and then considered the environmental performance of a lithium manganese oxide and a lithium iron phosphate battery through performing an LCA and by comparing the results obtained for the two technologies. In this study, it was concluded that the applicability of the batteries directly depends on the efficiency and the energy output provided by the technology.

It is widely noted, that performing an LCA not only helps to investigate the environmental impact of a product but also allows the investigation of other state-of-art technologies and what they can offer [45,46]. There are several studies which have provided a direct comparison of LCA and environmental impact between two different types of energy storage systems. This paper investigates the environmental life cycle analysis of an innovative thermal battery (TB) developed by Spirax Sarco Engineering Ltd. The innovative thermal battery designed to store thermal energy generated by an electrical heater. Furthermore, the LCA of the TB was compared with a conventional electric battery, more specifically a lithium iron phosphate battery (LIPB).

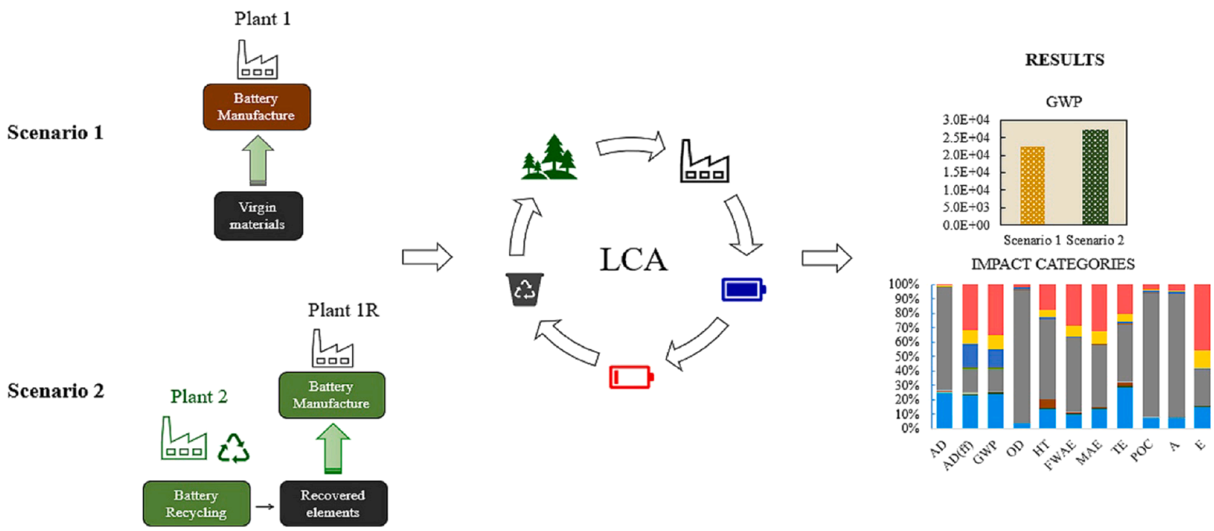

Fig. 3. Examples of life cycle assessment scenarios [40].

2. Methodology

As defined by the International Standard ISO 14,040 [47], LCA consists of goal and scope definition, inventory analysis, life cycle impact assessment and interpretation phase. The life cycle assessment was carried out using the open-source openLCA 1.10.3 software and the also open-source ELCD 3.2 database.

2.1. Thermal battery description

Spirax Sarco Ltd in conjunction with their sister company Chromalox Inc have invented a new type of thermal store that stores heat generated by an immersed electrical heater as high pressure hot water in an well insulated vessel. For the purposes of this paper this approach has been given the working title and generic name “Thermal Battery” (TB)

When heat (or steam) is required from the TB, steam is taken from the ullage (gas) space of the vessel and used directly as steam or indirectly through a heat exchanger to interface with a “wet” heating system and/or a domestic hot water system. Condensed steam is returned to the vessel. As heat is withdrawn the pressure gradually lowers until the TB is completely discharged. The TB is re-charged by the water immersed electrical heater which can utilise renewable electricity such as wind power or solar PV directly or in addition/alternatively can connect to the grid to harvest renewable low-cost electrical power when available. The TB can be discharging heat and be charged simultaneously allowing its use to be flexible for many diverse situations. In addition, it acts as a buffer storage. The basic principles of the TB is illustrated in Fig. 4.

2.2. Goal and scope

Based on the International Standard ISO 14,040 [47], the goal and scope definition phase describes the overall aims and objectives of the study being conducted. It also provides a picture of the boundaries and functional units which will be investigated. The aim of this comparison is to identify the environmental impact of the TB and compare it with that of the LIPB to assess its environmental competitiveness against traditional chemical battery. The outcome of this research can be used internally for the business case to aid decision making and for external use. The findings of the study are limited, as an analysis of the actual manufacturing process for the TB has not been conducted yet. The construction of the life cycle inventory was based on information published in existing literature.

2.3. Functional unit

The discharge potential of 1000 kWh was chosen as the functional unit (FU) on which to base the environmental impact. This aids the comparison of the results with the findings presented by Wang et al. [28]. Those findings were used for the environmental impact of LIPB.

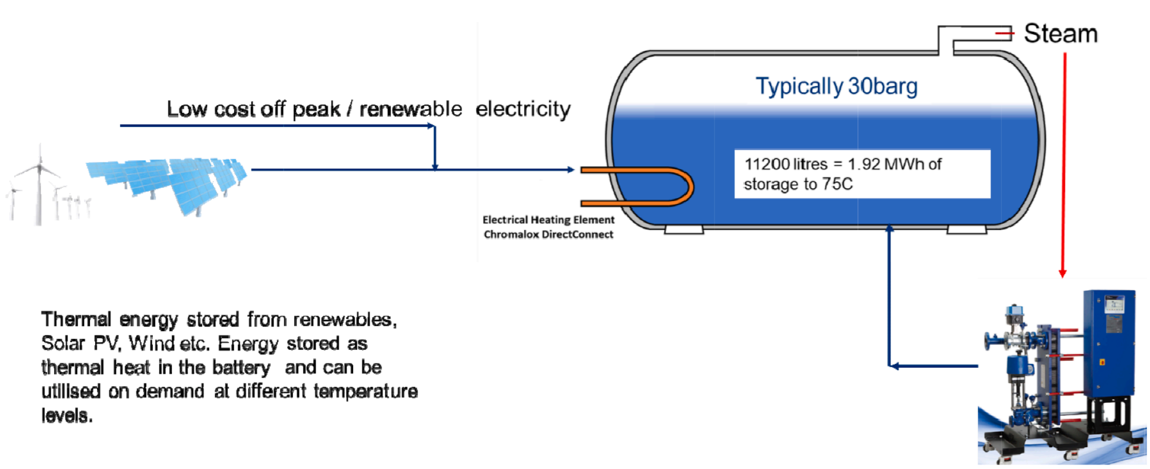

Fig. 4. An illustration of the thermal battery concept.

2.4. Case study scenario

A TB is compared to a LIPB . The manufacturing location of the LIPB was assumed to be in China, which was also assumed by Wang et al. [28] used data from an unnamed Chinese manufacturer and a document from Eastwood Park energy storage published by AlphaESS [48]. The database shows that the company has two manufacturing locations, one in China and one in Australia. The thermal battery was assumed to be fabricated in the UK, while the data used for steel manufacturing was obtained from a global manufacturer.

2.5. System boundary

As shown in Fig. 5, Wang et al. [28] used a cradle-to-gate approach for defining the system boundary, which means data were only included for obtaining raw materials without accounting for usage and disposal of the battery. Product quality allocation was not considered, as the processes only had one product. The impact of raw material transport was omitted as the modes of transport were various.

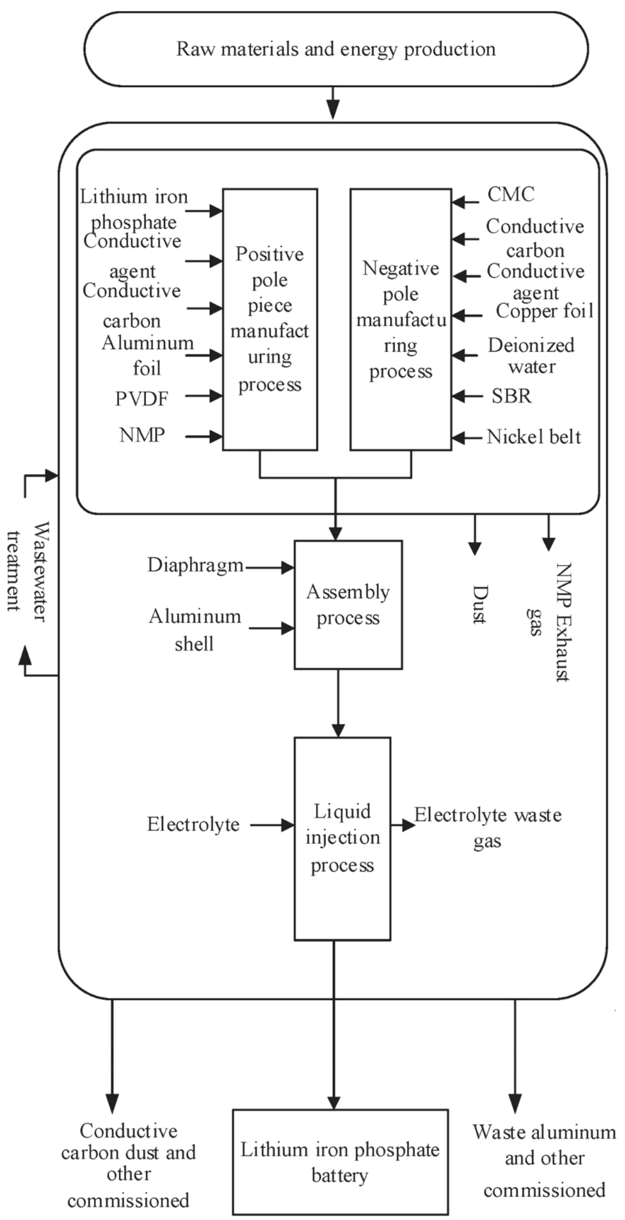

Fig. 5. System boundary for the LIPB as defined by Wang et al. [28].

To maximise the comparability between the data for the LIPB and the TB, the system boundaries were drawn in a similar approach. As presented in Fig. 6, the following system boundaries were applied:

- Transportation was not accounted for when analysing the environmental impact of the TB
- The allocation method was not defined due to the lack of by-products
- Impact due to use or disposal was not included. Ancillaries were not included in the impact analysis for either of the product systems.

It is worth noting that methods are available to recycle LIPBs, but they are limited to laboratory scale applications. As the technology is labour intensive, an extensive amount of research is being conducted to address the challenges of upscaling these technologies [49]. On the other hand, both steel and rock wool are widely recycled which is highlighted by the fact that Rockwool International has its own recycling programme for customers [50].

2.6. Assumptions

Based on the study conducted by Wang et al. [28] several assumptions have been made and applied throughout the investigation, as listed below.

- 14 m3 of the TB has been used for the environmental impact assessment.
- The lifespan of the TB is estimated to be 20 years, during which it is charged to its capacity (1.14 MWh) each weekend and discharged during the week. The estimated lifetime capacity is 1140 MWh if a 50-week operation per year is considered (equating to 1140 functional units).
- The LIPB life cycle was assumed to be 2000 as stated by Wang et al. [28] (data in the life cycle inventory for the product is already expressed per functional unit).
- Data for the LIPB life cycle inventory was collected during an on-site investigation from a Chinese manufacturer [28].
- For the impact assessment, Wang et al. [28] used ReCiPe midpoint model (H) 2008, given the limitations of openLCA, ReCiPe midpoint model (H) 2016 was used for the TB. The differences between both models causes a deviation in the available comparable environmental impact categories.
- For assembling the TB, it was assumed that Gas Metal Arc Welding (GMAW) is used, as this technique is suitable for thick metal plates (>20 mm) [51]. This type of welding method is also approved by the KLM Technology Group [52] for pressure vessels. The TB consists of two heads and two cold-rolled cylinders made from steel sheets, each having a longitudinal welded seam. The cylinders are then joined by another weld and the two heads are also welded to the resulting cylinder. Their cumulative length is 23.47 m, but due to the addition of fittings this length was rounded up to 25 m.
- Sproesser et al. [51] noted that the shielding gas used for welding had no significant impact on any of the impact categories used in their study per 1 m of weld (Eutrophication Potential; Acidification Potential; Photochemical Ozone Creation Potential; Global Warming Potential), therefore the shielding gas was excluded in this study as there is only 2.20 cm weld/functional unit in the TB.
- The welding fume composition was established by Pires et al. [53], the study noted a fume composition consisting of: 82% Ar and 18% CO2 shielding mixture at 180 A. This specific mixture of gases was specified by Sproesser et al. [51].
- The filler wire composition was assumed to be as described by ESAB [54] for the Mn3Ni1CrMo wire specified for GMAW by Sproesser et al. [51].
- The steel used from openLCA is based on the production of hot rolled steel sections. The data set is based on average site-specific data (gate-to-gate) of global steel producers, although the electricity grid

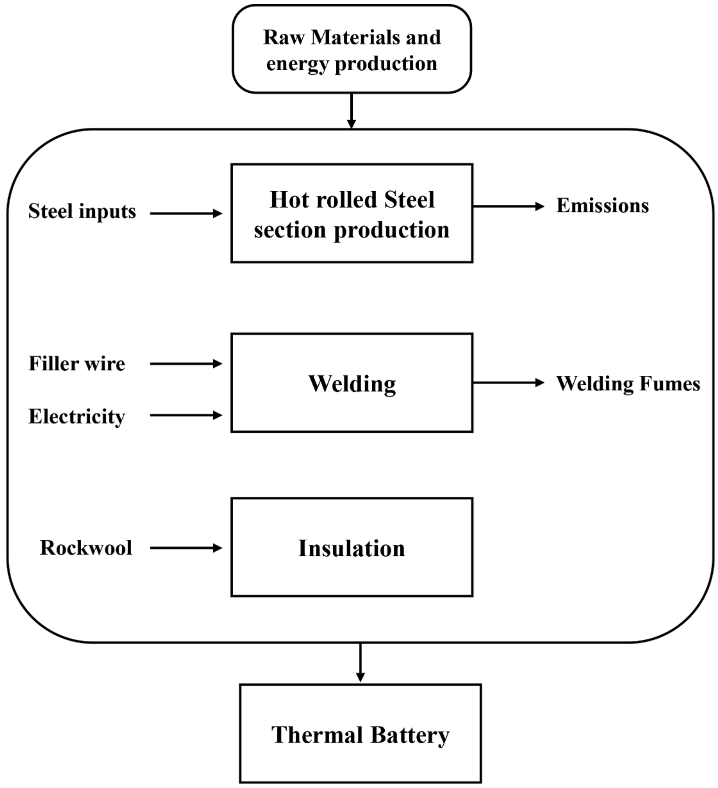

Fig. 6. System boundary for the TB [28].

mix is country specific. Other upstream data such as iron ore production are based on global averages, as well as openLCA ELCD 3.2 database, from the project: IISI Life Cycle Inventory Study for Steel Industry Products, 2000.

2.7. Evaluation method

After defining of the system boundary, a life cycle inventory was constructed for the product evaluated. The inventory contains the quantities of raw materials and energy needed along with all the emissions that are associated with the life cycle of the product within the system boundary. This inventory then was used in conjunction with a database and an evaluation model to quantify the environmental impact of the product in different categories, such as climate change and marine ecotoxity. The result of this process was the quantity of equivalent emission for each impact category. For example multiple gases can aggravate climate change including methane and carbon dioxide, but the emissions are measured in carbon dioxide equivalent. The last step of the analysis, was the dimensionless evaluation of the impacts which was done through a normalisation process. Normalisation is a process where average emissions are used to make emission values dimensionless and allow the comparison of a product's impact across multiple categories. The emission values after the normalisation are referred to as environmental impact load. The analysis of the environmental impact load is used to highlight in which environmental impact category does the product need to improve the most, or which product has the highest environmental impact across categories out of the two compared. i.e., Normalisation approach allows comparison of the severity of emissions between the impact categories. For instance, the comparison would reveal whether a product has a higher environmental impact on climate change or ozone depletion. This would not be evident from the mass quantities as the magnitude of the environmental impact of 1 kg CO_2 is not the same as 1 kg CFC-11.

2.8. Life cycle inventory construction and environmental impact assessment

The lifecycle inventory was constructed for the LIPB by Wang et al. [28]. based on data from an unnamed Chinese manufacturer. Their inventory was not adopted and re-evaluated due to some key items missing from the ELCD 3.2 database used in this study. Instead, the impact values per functional unit resulting from the evaluation using ReCiPe midpoint (H) model 2008 by Wang et al. [28] were adopted for the environmental impact categories as shown by Table 1. The model used by the Wang et al. [28] study evaluates environmental impact in 18 categories such as climate change, ozone depletion, human toxicity, photochemical oxidant formation, particulate matter formation, ionizing radiation, terrestrial acidification, freshwater eutrophication,

Table 1
Environmental impact results for the lithium iron phosphate battery (LIPB) by Wang et al. [28].

Impact category	Unit	LIPB
Climate change	kg CO_2 eq	16.10
Ozone depletion	kg CFC-11 eq	1.28×10^{-6}
Terrestrial acidification	kg SO_2 eq	0.12
Freshwater eutrophication	kg P eq	1.07×10^{-2}
Marine eutrophication	kg N eq	1.08×10^{-2}
Human toxicity	kg 1,4-DB eqa	10.73
Photochemical oxidant formation	kg NMVOC	6.11×10^{-2}
Particulate matter formation	kg PM_{10} eq	5.15×10^{-2}
Terrestrial ecotoxicity	kg 1,4-DB eq	2.21×10^{-3}
Freshwater ecotoxicity	kg 1,4-DB eq	0.30
Marine ecotoxicity	kg 1,4-DB eq	0.28
Ionizing radiation	kBq U235 eq	0.22
Agricultural land occupation	m^2a	3.40
Urban land occupation	m^2a	0.25
Natural land transformation	m^2	2.50×10^{-3}
Water depletion	m^3	0.60
Metal depletion	kg Fe eq	10.70
Fossil depletion	kg oil eq	4.87

marine eutrophication, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, agricultural land occupation, urban land occupation, natural land transformation, water depletion, metal depletion and fossil depletion.

For the TB, an inventory was constructed by taking into account the amount of steel needed for the vessel itself, the rock wool for the insulation, the filling wire, the electricity used for the welding and amount of fumes released during the welding process. An input and output inventory was constructed for the steel production process (see Table 2 for inputs and Table 3 for outputs), for welding process (see Table 4), and for the insulation (see Table 5). These were based on information about corresponding processes found in the ELCD 3.2 database and were scaled according to the amount of output required from each process for the thermal battery. The process inputs can be used to track resource use, while process outputs are indicative of emissions.

The welding process inventory was based on the inputs and emissions outlined by Sproesser et al. [51] for the Gas Metal Arc Welding process, but, as mentioned in Section 2.6, the shielding gas was not included as it did not have a significant enough impact. The exact welding wire was obtained from ESAB [54] and the fume compositions from Pires et al. [53]. The length of weld corresponding to one FU was 2.20 cm. Finally, the life cycle inventory for the insulation is shown in Table 5. For the calculation, the insulation was expected to be 150 mm thick on the TB and with the density of 22 kg/m^3, this resulted in 0.12 kg corresponding to 1 FU.

The impact of these cumulative emissions is compared to the impact of the LIPB emissions in the results section.

3. Results

3.1. Environmental impact analysis

The ReCiPe midpoint (H) 2016 model was used to analyse the impact of the TB production within the system boundary outlined in Section 2.5, whereas the life cycle assessment of the LIPB conducted by Wang et al. [28] utilised the ReCiPe midpoint (H) 2008 version. Due to differences between the categories in the 2016 and 2008 versions of the model, only 10 of the 18 environmental impact categories found in the original 2008 version provided directly comparable data for the TB and the LIPB. The results for these 10 categories are detailed in Sections 3.1.1 to 3.1.5.. Information on the results in the categories with no direct comparison are discussed in Section 3.1.6.

3.1.1. Greenhouse gas emissions

The climate change potential of the two products was one of the categories that allowed direct comparison. As mentioned in Section 2.7 all greenhouse gases emitted were benchmarked against carbon dioxide and the emissions were expressed in kg of carbon dioxide equivalent.

The TB was found to emit 8.58 kg/1000 kWh of energy stored throughout its lifetime, while the LIPB emitted 16.10/1000 kWh of energy stored throughout its lifetime. This showed that the TB emits nearly half as much carbon dioxide equivalent as the LIPB per functional unit.

Table 2
Process inputs of steel production based on data from openLCA.

Process inputs	Inventory value per FU	Unit
Hard coal; 26.3 MJ/kg	55.60	MJ
Water	26.16	kg
Natural gas; 44.1 MJ/kg	18.62	MJ
Crude oil; 42.3 MJ/kg	13.49	MJ
Iron	2.313	kg
Metamorphous rock, graphite containing, in ground	1.54	kg
Calcium carbonate, in ground	0.70	kg
Dolomite, in ground	0.18	kg
Zinc	3.17×10^{-2}	kg

Table 3
Emissions and process outputs of steel production based on openLCA software.

Process outputs	Inventory value per FU	Unit	Type of flow
Ammonia	1.20×10^{-3}	kg	Emission to water
Cadmium	4.90×10^{-7}	kg	Emission to air
Cadmium	2.86×10^{-8}	kg	Emission to water
Carbon dioxide	8.25	kg	Emission to air
Carbon monoxide	9.95×10^{-2}	kg	Emission to air
Chromium	6.63×10^{-6}	kg	Emission to air
Chromium	3.26×10^{-7}	kg	Emission to water
COD, Chemical Oxygen Demand	2.93×10^{-4}	kg	Emission to water
Dinitrogen monoxide	3.61×10^{-4}	kg	Emission to air
Dioxins, measured as 2,3,7,8-tetrachlorodibenzo-p-dioxin	2.99×10^{-11}	kg	Emission to air
Hydrogen chloride	2.95×10^{-4}	kg	Emission to air
Hydrogen sulfide	2.44×10^{-4}	kg	Emission to air
Iron	8.79×10^{-4}	kg	Emission to water
Lead	1.80×10^{-5}	kg	Emission to air
Lead	1.47×10^{-6}	kg	Emission to water
Mercury	8.22×10^{-7}	kg	Emission to air
Methane	5.40×10^{-3}	kg	Emission to air
Nickel	3.98×10^{-7}	kg	Emission to water
Nitrogen	1.24×10^{-4}	kg	Emission to water
Nitrogen dioxide	1.27×10^{-2}	kg	Emission to air
NMVOC, non-methane volatile organic compounds, unspecified origin	9.81×10^{-4}	kg	Emission to air
Particulates, > 10 um	5.09×10^{-4}	kg	Emission to water
Particulates, > 2.5 um, and < 10um	4.61×10^{-3}	kg	Emission to air
Phosphate	1.17×10^{-4}	kg	Emission to water
Steel hot rolled section	7.51	kg	
Sulphur dioxide	1.54×10^{-2}	kg	Emission to air
Waste (unspecified)	3.38	kg	Consumer waste
Zinc	1.01×10^{-4}	kg	Emission to air
Zinc	2.30×10^{-6}	kg	Emission to water

3.1.2. Ecosystem quality

The effect produced by the two products examined on ecosystem quality was investigated by comparing the equivalent emissions of the products in 6 environmental impact categories. The categories account for the ecotoxicity and eutrophication in marine and freshwater environments as well as the acidification and ecotoxicity in terrestrial ones. The emission figures for each one of these categories can be seen in Table 6. For the environmental categories related to freshwater and marine environments, the TB was estimated to reduce emissions by at least 95% compared to the LIPB. This was a significant enough

Table 4
Inputs and outputs of the welding process based on Sproesser et al. [42].

Process inputs	Inventory value per FU	Unit	Process outputs	Category	Inventory value per FU	Unit
Carbon	1.74×10^{-2}	g	Iron	Emission to air	1.25×10^{-2}	g
Chromium	5.05×10^{-2}	g	Manganese	Emission to air	1.95×10^{-3}	g
Electricity EU-28+3	0.00	kWh	Silicon	Emission to air	5.35×10^{-3}	g
Manganese	4.61×10^{-2}	g	Sodium	Emission to air	1.54×10^{-3}	g
Molybdenum	0.31	g	Steel vessel		7.51	kg
Nickel	4.69×10^{-2}	g				
Silicon	0.24	g				
Steel hot rolled	0.10	kg				

Table 5
Life cycle inventory of the insulation.

Process input	Inventory value per FU	Unit
rock wool	0.12	kg

Table 6
Equivalent emissions of TB and LIPB related to ecosystem quality.

Environmental impact category	TB	LIPB
Freshwater ecotoxicity (kg 1,4-DB eq)	1.11×10^{-3}	0.30
Freshwater eutrophication (kg P eq)	3.90×10^{-5}	1.07×10^{-2}
Marine ecotoxicity (kg 1,4-DB eq)	1.28×10^{-2}	0.28
Marine eutrophication (kg N eq)	3.31×10^{-4}	1.08×10^{-2}
Terrestrial acidification (kg SO_2 eq)	2.00×10^{-2}	0.12
Terrestrial ecotoxicity (kg 1,4-DB eq)	15.87	2.21×10^{-3}

difference that it could be seen by looking at the difference in magnitude between the emissions. For example, the TB emitted 3.90×10^{-5} kg of phosphorus equivalent while the LIPB was found to emit 1.07×10^{-2} kg of phosphorus equivalent. At least 50% of the emissions generated by the LIPB could be contributed to its cathode plate manufacturing process in the freshwater eutrophication, freshwater ecotoxicity and marine ecotoxicity [28]. The contribution of the process went as high as 69% in the case of freshwater eutrophication making it a significant area for improvement.

In terrestrial environments, the performance of the products showed a different figure than in the aquatic ones described above. The sulphur dioxide produced by the TB which causes terrestrial acidification was 83% less than the LIPB. However, based on the model TB emitted 15.87 kg of 1,4-DB equivalent, while the LIPB was only found to emit 2.21×10^{-3} kg of 1,4-DB-equivalent. This was a significant difference in emissions which is discussed further in Section 3.1.5.

3.1.3. Human health

Human health would normally be described by a number of environmental categories such as fine particulate formation, human toxicity (including both carcinogenic and non-carcinogenic) and ozone formation alongside the ozone depletion. Due to the differences between the 2008 and 2016 versions of the ReCiPe midpoint (H) model, the products could only be compared directly in the ozone depletion category. The reason ozone depletion is a problem for human health is that the increased exposure to UV radiation can increase the incidence of some types of skin cancers, cataracts and immune deficiency disorders. The life cycle assessment presented that the TB emitted four times as much CFC11 equivalent than the LIPB at 4×10^{-6} kg/1000 kWh energy stored throughout the life cycle examined. This was another significant relative difference and is evaluated further in Section 3.1.5.

3.1.4. Natural resource use

Similar to the human health, natural resource use is examined by more than one environmental impact category such as fossil depletion, water depletion and metal depletion. Out of these both fossil depletion and water depletion provided directly comparable data as presented in Table 7. In both categories the TB emitted at least 95% less than the LIPB. It is important to note that the TB's fossil depletion is likely higher than zero, a field study can provide a figure with higher accuracy, however, the low value is favourable. The TB also performs better in terms of water depletion using almost one magnitude less water per functional unit in the examined life cycle than the LIPB.

3.1.5. Cross-category evaluation

Due to the wide variety of units used for the different environmental impact categories, the impact values cannot be compared without making them dimensionless first. Customarily, this is done with normalisation. This case study used the World 2010 (H) dataset for normalisation which was provided with the impact assessment methods package in openLCA.

Fig. 7 shows the impact loads for both the TB and the LIPB in the categories where direct comparison could not be drawn. The largest impact load overall was found to be the marine ecotoxicity of the LIPB at 0.27, the cathode plate manufacturing process of the battery contributed the most to this impact [28]. The second highest load was in the category of freshwater ecotoxicity and belonged to the LIPB as well at 0.24, similarly inflated by the cathode plate manufacture [28]. In contrast, the highest impact load for the TB was 1.53×10^{-2} in the category of terrestrial ecotoxicty. If this impact load was only compared to the load of the LIPB in the same category, it would be a serious environmental performance issue for the TB, however once put in context of the impact loads of both products across the different environmental categories, it could be re-evaluated as an area for potential improvement instead. Another cause for concern highlighted previously was the performance of the TB ozone depletion category. However, Fig. 7 shows that the emissions in this category produced a significantly lower impact load than the others, therefore the large relative difference was less relevant than the dimensional emission values would suggest.

3.1.5.1. Impact categories with no direct comparison.

Table 8 shows the environmental impact values for categories that could not be directly compared. The categories were thematically grouped in each row, for example, fine particulate matter formation and particulate matter formation were in the same row, they were not easily comparable as LIPB particles were 10 μm or smaller, whereas TB particles are 2.5 μm or smaller. Based on this difference, it could be expected that the impact value for LIPB is higher than for TB, which was indeed the case.

The human toxicity category for LIPB did not separate carcinogenic and non-carcinogenic toxicity, but all three categories used the same unit (1,4-dichlorobenzene equivalent), theoretically just purely based on the units these carcinogenic and non-carcinogenic impacts could be added to compare the TB human toxicity value with the LIPB one. The

Table 7
Environmental impact related to natural resource use.

Environmental impact category	TB	LIPB
Fossil depletion (kg oil eq)	0.00	4.87
Water depletion (m^3)	2.63×10^{-2}	0.60

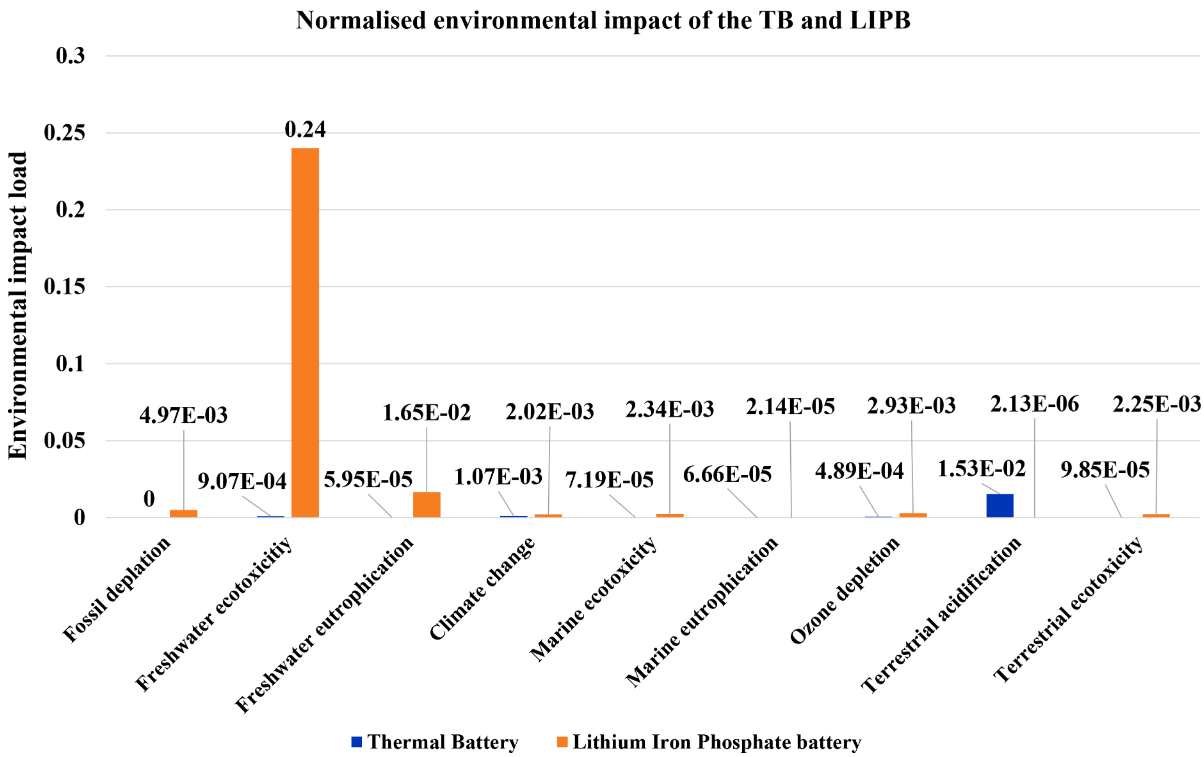

Fig. 7. Normalised environmental impacts in the directly comparable categories.

Table 8
Data for the environmental categories with no direct comparison.

TB			LIPB		
Fine particulate matter formation	kg PM2.5 eq	5.88×10^{-3}	Particulate matter formation	kg PM10 eq	5.15×10^{-2}
Human carcinogenic toxicity	kg 1,4-DCB eq	3.29×10^{-3}	Human toxicity	kg 1,4 DB eq	10.73
Human non-carcinogenic toxicity	kg 1,4-DCB eq	1.30			
Land use	m^2a crop eq	0.00	Urban and agricultural land occupation	m^2a	3.40
			Natural land transformation	m^2	2.50×10^{-3}
Mineral resource scarcity	kg Cu eq	7.00×10^{-3}	Metal depletion	kg Fe eq	10.70
Ozone formation, Human health	kg NO_x eq	1.77×10^{-4}	Photochemical oxidant formation	kg NMVOC	6.11×10^{-2}

summation would show that the TB has a lower impact on human toxicity overall, however, this would most likely not be a fair assessment, depending on which other elements were changed between the two versions of the ReCiPe impact analysis models.

Land use for the TB seemed to have correlated to two categories for LIPB: urban and agricultural land occupation, and natural land transformation. This group of categories presented an issue, because it contained a host of different units. However, it is worth pointing out that for the TB, the land use impact was valued at 0. This could be taken as a signal that if a field study was conducted on the TB rather than the evaluation based on the model, it would most likely find a very small value. Mineral resource scarcity and metal depletion were in the same group, but unfortunately due to the different units, the values could not be compared. The next group of categories labelled as photochemical oxidant formation, or ozone formation depending on the source, was an easier comparison. While on paper they have different units, during the inventory analysis for the TB, the amount of NMVOC emitted was characterised as 9.85×10^{-4} kg/ FU, this suggests that the TB would be more favourable in this category. The final category of ionising radiation also presented the issue of different units, which meant that the impact values could not be compared.

4. Conclusions

This paper conducted an LCA of an innovative thermal battery solution and compared the environmental impacts with one of the state-of-the-art electrical storage technologies. It should be noted that only a few studies have analysed different types of thermal and electrical storage systems which was a lithium iron phosphate battery (LIPB). It could be included from the environmental life cycle assessment of the LIPB and the thermal battery (TB) developed by Spirax Sarco Engineering Ltd. that:

In dimensional terms, the LCA comparison showed the THB was more environmentally friendly than LIBP by producing almost half of carbon dioxide equivalent obtained by the LIPB LCA, and by reducing the environmental impact by at least 80% compared to the LIPB in all categories related to ecosystem health with the excepetion of terrestrial ecotoxicity. Furthermore, the TB had a smaller impact on natural resource use such as fossil and water depletion.

Furthermore, by evaluating the normalised environmental impact loads it was observed that the two highest loads at 0.27 and at 0.24 were both emitted by the LIPB in the marine ecotoxicity and freshwater ecotoxicity categories. The highest environmental load observed by the TB was 1.53×10^{-2} in the category of terrestrial ecotoxicty. The order of magnitude difference in the highest impact loads of the two products was a very encouraging sign for the environmental performance of the TB. There were environmental categories where the TB emitted more than the LIPB such as in terrestrial ecotoxicity and ozone depletion, but the normalisation showed that these were lower impact categories among the directly comparable categories

These results are promising for the TB technology, but it is recommended that the model is made more robust by incorporating data directly from the vessel manufacturer and by verifying the assumptions made in this study. A future work to be done is to conduct a thermodynamic performance and ecosystem assessment of an actual TB.

Declaration of Competing Interest

We wish to confirm that there are no known conflicts of interest associated with this publication and there has been no financial support for this work that could have influenced its outcome.

We confirm that the manuscript has been read and approved by all named authors and that there are no other persons who satisfied the criteria for authorship but are not listed. We further confirm that the order of authors listed in the manuscript has been approved by all of us.

We confirm that we have given due consideration to the protection of intellectual property associated with this work and that there are no impediments to publication, including the timing of publication, with respect to intellectual property. In so doing we confirm that we have followed the regulations of our institutions concerning intellectual property.

We understand that the Corresponding Author is the sole contact for the Editorial process (including Editorial Manager and direct communications with the office). He is responsible for communicating with the other authors about progress, submissions of revisions and final approval of proofs. We confirm that we have provided a current, correct email address which is accessible by the Corresponding Author and which has been configured to accept email from the IJTF Journal.

Acknowledgment

The work has been funded by Spirax Sarco Engineering PLC.

References

[1] J. Malinauskaite, H. Jouhara, The trilemma of waste-to-energy: a multi-purpose solution, Energy Policy 129 (2019) 636–645, https://doi.org/10.1016/j.enpol.2019.02.029.
[2] European Commission, 2050 long-term strategy, Eur. Comm. (2021). https://ec.europa.eu/clima/policies/strategies/2050_en#tab-0-0 (accessed August 1, 2021).
[3] United Nations, Paris Agreement, in: Proceedings of the 2015 United Nations Climate Change Conference, 2015. https://ec.europa.eu/clima/policies/international/negotiations/paris_en#tab-0-0 (accessed August 30, 2021).
[4] Department For Business E& IS, UK Energy Statistics, 2020, 2019 & Q4 2019.
[5] S. Evans, Analysis: UK renewables generate more electricity than fossil fuels for first time, Breaf Carbon (2019). https://www.carbonbrief.org/analysis-uk-renewables-generate-more-electricity-than-fossil-fuels-for-first-time (accessed August 30, 2021).
[6] S. Koohi-Fayegh, M.A. Rosen, A review of energy storage types, applications and recent developments, J. Energy Storage 27 (2020), 101047, https://doi.org/10.1016/j.est.2019.101047.
[7] H. Jouhara, A. Żabnieńska-Góra, N. Khordehgah, D. Ahmad, T. Lipinski, Latent thermal energy storage technologies and applications: a review, Int. J. Thermofluids (2020) 5–6, https://doi.org/10.1016/j.ijft.2020.100039.
[8] M. Ramadan, A review on coupling Green sources to Green storage (G2G): case study on solar-hydrogen coupling, Int. J. Hydrog. Energy 46 (2021) 30547–30558, https://doi.org/10.1016/j.ijhydene.2020.12.165.
[9] A. Amini, J. Miller, H. Jouhara, An investigation into the use of the heat pipe technology in thermal energy storage heat exchangers, Energy 136 (2017) 163–172, https://doi.org/10.1016/j.energy.2016.02.089.
[10] M.K. Koukou, G. Dogkas, M.G. Vrachopoulos, J. Konstantaras, C. Pagkalos, V. N. Stathopoulos, et al., Experimental assessment of a full scale prototype thermal energy storage tank using paraffin for space heating application, Int. J. Thermofluids 1–2 (2020), 100003, https://doi.org/10.1016/j.ijft.2019.100003.
[11] O. Okogeri, V.N. Stathopoulos, What about greener phase change materials? A review on biobased phase change materials for thermal energy storage applications, Int. J. Thermofluids 10 (2021), 100081, https://doi.org/10.1016/j.ijft.2021.100081.
[12] M. Mahmoud, M. Ramadan, K. Pullen, M.A. Abdelkareem, T. Wilberforce, A. G. Olabi, et al., Waste heat recovery applications incorporating phase change materials, Ref. Modul. Mater. Sci. Mater. Eng. (2021), https://doi.org/10.1016/B978-0-12-815732-9.00074-7.
[13] A.G. Olabi, T. Wilberforce, K. Elsaid, E.T. Sayed, M. Ramadan, S.M. Atiqure Rahman, et al., Recent progress on carbon-based nanomaterial for phase change materials: prospects and challenges, Therm. Sci. Eng. Prog. 23 (2021), 100920, https://doi.org/10.1016/j.tsep.2021.100920.
[14] V. Samson Packiaraj Raphael, R. Velraj, P. Jalihal, Transient analysis of steam accumulator integrated with solar based MED-TVC system, Desalination 435 (2018) 3–22, https://doi.org/10.1016/j.desal.2017.12.045.
[15] F. Bai, C. Xu, Performance analysis of a two-stage thermal energy storage system using concrete and steam accumulator, Appl. Therm. Eng. 31 (2011) 2764–2771, https://doi.org/10.1016/j.applthermaleng.2011.04.049.
[16] B.D. Boruah, Recent advances in off-grid electrochemical capacitors, Energy Storage Mater. 34 (2021) 53–75, https://doi.org/10.1016/j.ensm.2020.08.031.
[17] J. Briscoe, S. Dunn, Piezoelectric nanogenerators – a review of nanostructured piezoelectric energy harvesters, Nano Energy 14 (2015) 15–29, https://doi.org/10.1016/J.NANOEN.2014.11.059.
[18] E. Arroyo, A. Badel, F. Formosa, Y. Wu, J. Qiu, Comparison of electromagnetic and piezoelectric vibration energy harvesters: model and experiments, Sens. Actuators A Phys. 183 (2012) 148–156, https://doi.org/10.1016/J.SNA.2012.04.033.
[19] J. Boyle, T. Littler, A. Foley, Battery energy storage system state-of-charge management to ensure availability of frequency regulating services from wind farms, Renew. Energy 160 (2020) 1119–1135, https://doi.org/10.1016/J.RENENE.2020.06.025.
[20] A. Baroutaji, A. Arjunan, M. Ramadan, J. Robinson, A. Alaswad, M. A. Abdelkareem, et al., Advancements and prospects of thermal management and waste heat recovery of PEMFC, Int. J. Thermofluids 9 (2021), 100064, https://doi.org/10.1016/J.IJFT.2021.100064.
[21] H.S. Dhiman, D. Deb, Wake management based life enhancement of battery energy storage system for hybrid wind farms, Renew. Sustain. Energy Rev. 130 (2020), 109912, https://doi.org/10.1016/J.RSER.2020.109912.
[22] Y. Yuan, X. Zhang, P. Ju, K. Qian, Z. Fu, Applications of battery energy storage system for wind power dispatchability purpose, Electr. Power Syst. Res. 93 (2012) 54–60, https://doi.org/10.1016/J.EPSR.2012.07.008.
[23] M.B. Siddique, J. Thakur, Assessment of curtailed wind energy potential for off-grid applications through mobile battery storage, Energy 201 (2020), 117601, https://doi.org/10.1016/J.ENERGY.2020.117601.
[24] G.S.M Mousavi, An autonomous hybrid energy system of wind/tidal/microturbine/battery storage, Int. J. Electr. Power Energy Syst. 43 (2012) 1144–1154, https://doi.org/10.1016/J.IJEPES.2012.05.060.
[25] T. Lehtola, A. Zahedi, Solar energy and wind power supply supported by storage technology: a review, Sustain. Energy Technol. Assess. 35 (2019) 25–31, https://doi.org/10.1016/J.SETA.2019.05.013.
[26] E. Galvan, P. Mandal, Y. Sang, Networked microgrids with roof-top solar PV and battery energy storage to improve distribution grids resilience to natural disasters, Int. J. Electr. Power Energy Syst. 123 (2020), 106239, https://doi.org/10.1016/J.IJEPES.2020.106239.
[27] C. Luerssen, O. Gandhi, T. Reindl, C. Sekhar, D. Cheong, Life cycle cost analysis (LCCA) of PV-powered cooling systems with thermal energy and battery storage for off-grid applications, Appl. Energy 273 (2020), 115145, https://doi.org/10.1016/j.apenergy.2020.115145.
[28] Q. Wang, W. Liu, X. Yuan, H. Tang, Y. Tang, M. Wang, et al., Environmental impact analysis and process optimization of batteries based on life cycle assessment, J. Clean. Prod. 174 (2018) 1262–1273, https://doi.org/10.1016/J.JCLEPRO.2017.11.059.
[29] M.J.M Al Essa, Power management of grid-integrated energy storage batteries with intermittent renewables, J. Energy Storage 31 (2020), 101762, https://doi.org/10.1016/J.EST.2020.101762.
[30] J. Malinauskaite, L. Anguilano, X.S. Rivera, Circular waste management of electric vehicle batteries: legal and technical perspectives from the EU and the UK post Brexit, Int. J. Thermofluids 10 (2021), 100078, https://doi.org/10.1016/j.ijft.2021.100078.
[31] H. Jouhara, N. Khordehgah, N. Serey, S. Almahmoud, S.P. Lester, D. Machen, et al., Applications and thermal management of rechargeable batteries for industrial applications, Energy 170 (2019) 849–861, https://doi.org/10.1016/J.ENERGY.2018.12.218.
[32] N. Omar, M. Daowd, O. Hegazy, B.P.V. den, T. Coosemans, M.J. Van, Electrical double-layer capacitors in hybrid topologies-assessment and evaluation of their performance, Energies 5 (2012) 4533–4568, https://doi.org/10.3390/en5114533.
[33] H. Jouhara, N. Serey, N. Khordehgah, R. Bennett, S. Almahmoud, S.P. Lester, Investigation, development and experimental analyses of a heat pipe based battery thermal management system, Int. J. Thermofluids 1–2 (2020), 100004, https://doi.org/10.1016/J.IJFT.2019.100004.
[34] A. Burke, M. Miller, Performance characteristics of lithium-ion batteries of various chemistries for plug-in hybrid vehicles, in: Proceedings of the 24th International Battery, Hybrid and Fuel Cell Electric Vehicle Symposium and Exhibition 2009, EVS 24 2, 2009, pp. 816–828. EVS 24.
[35] N. Omar, J. Van Mierlo, B. Verbrugge, P. Van Den Bossche, Power and life enhancement of battery-electrical double layer capacitor for hybrid electric and charge-depleting plug-in vehicle applications, Electrochim. Acta 55 (2010) 7524–7531, https://doi.org/10.1016/J.ELECTACTA.2010.03.039.
[36] N. Omar, M.A. Monem, Y. Firouz, J. Salminen, J. Smekens, O. Hegazy, et al., Lithium iron phosphate based battery - assessment of the aging parameters and development of cycle life model, Appl. Energy 113 (2014) 1575–1585, https://doi.org/10.1016/J.APENERGY.2013.09.003.
[37] K. Amine, J. Liu, I. Belharouak, High-temperature storage and cycling of C-$LiFePO_4$/graphite Li-ion cells, Electrochem. Commun. 7 (2005) 669–673, https://doi.org/10.1016/J.ELECOM.2005.04.018.

[38] H. Song, Z. Cao, X. Chen, H. Lu, M. Jia, Z. Zhang, et al. Capacity fade of LiFePO4 /graphite cell at elevated temperature n.d. 10.1007/s10008-012-1893-2.
[39] M. Raugei, P. Winfield, Prospective LCA of the production and EoL recycling of a novel type of Li-ion battery for electric vehicles, J. Clean. Prod. 213 (2019) 926–932, https://doi.org/10.1016/J.JCLEPRO.2018.12.237.
[40] L. Silvestri, A. Forcina, G. Arcese, G. Bella, Recycling technologies of nickel–metal hydride batteries: an LCA based analysis, J. Clean. Prod. 273 (2020), 123083, https://doi.org/10.1016/J.JCLEPRO.2020.123083.
[41] G. Yajun, Study on life-cycle assessment methods for waste secondary battery, Environ. Sanit. Eng. (2008) 46–48, https://en.cnki.com.cn/Article_en/CJFDTotal-HJWS200802018.htm.
[42] M. Zackrisson, L. Avellán, J. Orlenius, Life cycle assessment of lithium-ion batteries for plug-in hybrid electric vehicles - critical issues, J. Clean. Prod. 18 (2010) 1519–1529, https://doi.org/10.1016/J.JCLEPRO.2010.06.004.
[43] G. Majeau-Bettez, T.R. Hawkins, A.H. Strømman, Life cycle environmental assessment of lithium-ion and nickel metal hydride batteries for plug-in hybrid and battery electric vehicles, Environ. Sci. Technol. 45 (2011) 4548–4554, https://doi.org/10.1021/es103607c.
[44] M. Messagie, L. Oliveira, S. Rangaraju, J.S. Forner, M.H. Rivas Environmental performance of lithium batteries: life cycle analysis. Recharg. Lithium Batter. From Fundam. to Appl., Woodhead Publishing; 2015, p. 303–18. 10.1016/B978-1-78242-090-3.00011-0.
[45] K.D. Kepler, Y. Wang, H. Liu. Secondary battery material . US 2009/0208844 A1, 2009.
[46] K.Q. Chen, W.H. Pu, Q. Zhang, X.L. Xing, C. Xiong, G.M. Di, Thermodynamic and economic assessment on the supercritical compressed carbon dioxide energy storage system coupled with solar thermal storage, J. Energy Storage 41 (2021), 102959, https://doi.org/10.1016/j.est.2021.102959.
[47] ISO (International Organization for Standardization, Environmental management - life cycle assessment - principles and framework (ISO 14040:2006), Environ. Manag. Syst. Requir. 44 (2004).
[48] AlphaESS. An energy storage system integrator covering full power range 2021. https://www.alpha-ess.com/ (accessed August 30, 2021).
[49] F. Forte, M. Pietrantonio, S. Pucciarmati, M. Puzone, D. Fontana, Lithium iron phosphate batteries recycling: an assessment of current status, Crit. Rev. Environ. Sci. Technol. 51 (2021) 2232–2259, https://doi.org/10.1080/10643389.2020.1776053.
[50] D.T Conley. Building tomorrow today sustainability report. Hedehusene, Denmark: 2018.
[51] G. Sproesser, Y.J. Chang, A. Pittner, M. Finkbeiner, M. Rethmeier, Life cycle assessment of welding technologies for thick metal plate welds, J. Clean. Prod. 108 (2015) 46–53, https://doi.org/10.1016/J.JCLEPRO.2015.06.121.
[52] KLM Technology Group, Pressure Vessel Welding Requirements, 2013.
[53] I. Pires, L. Quintino, R.M. Miranda, J.F.P. Gomes, Fume emissions during gas metal arc welding, Toxicol. Environ. Chem. 88 (2006) 385–394, https://doi.org/10.1080/02772240600720472.
[54] ESAB. Welding filler metal handbook. 2021. XA00003797.