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Life cycle assessment of lithium iron phosphate and electrochemical recuperator cells for city buses in Finland

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ABSTRACT

The study investigates the environmental impacts of electric city buses based on the storage technologies applied and the degree of electrification within the Finnish context. Lithium iron phosphate (LFP) and electrochemical recuperator (ECR) were selected as storage technologies. ECR can be an alternative to the lithium-ion battery; however, little is known regarding its environmental performance when applied to electrify city buses. The study focused on diesel buses, battery electric buses (BEB) and plug-in hybrid buses. Life cycle assessment (LCA) was used to assess the potential environmental impacts between storage technologies and the degree of electrification. Primary data from the industry was used to assess the impacts of manufacturing ECR. The results showed that manufacturing a kWh of ECR generated a global warming potential (GWP) of 178 kg CO_2-eq, higher than LFP. However, its application indicated that ECR performed better. The impacts of using ECR and LFP in BEB were 385 g CO_2-eq/km and 441 g CO_2-eq/km, respectively. The hybrid system generated 652 g CO_2-eq/km and 670 g CO_2-eq/km for ECR and LFP, respectively. The study also showed no consistent pattern between the degree of electrification and environmental benefits. Scenario analysis revealed that BEB provided the best GWP when assessed using a Finnish and Norwegian electricity mix, while the hybrid system performed the best when Polish electricity was applied. This study demonstrated that storage technologies, degree of electrification, fuel consumption, and electricity sources affect environmental performance. Careful assessment is needed before deciding to electrify the city's transport system.

1. Introduction

The transportation sector is one of the primary sources of greenhouse gases emission (GHG). Globally, it was reported that 23% of GHG emissions were from the transport sector, with 70% of it generated by road vehicles, while 1%, 11%, and 12% were generated by rail, shipping, and aviation, respectively (Jaramillo et al., 2022). In the European Union (EU), the emissions from transportation consistently increased between 2013 and 2019 due to the growth of passenger and inland freight transports, followed by a 13.6% decrease during the COVID-19 restriction (European Environment Agency, 2022b). In 2021, the carbon emission from the transport sector bounced back following the lift of pandemic restrictions, as indicated by around 8% growth compared to the 2020 level (European Environment Agency, 2022b; IEA, 2022). Road transport comprises the highest proportion of transport emissions in the EU. In 2020, 77% of GHG emissions in the transport sector accounted for road transport, whose trajectory showed a steady increase of about 28% between 1990 and 2019 (European Environment Agency, 2022a, 2022b).

Intercity and intracity road transportation are in the spotlight to mitigate climate change since built environment properties can affect the impacts of transportation (Arioli et al., 2020; Lim et al., 2019). On top of it, the electrification of vehicles is expected to play a more important role in reducing GHG emissions from road transport (Zhao et al., 2021b). Driving electric vehicles (EV) does not generate tailpipe emissions. However, the total emission of EV, hybrid electric vehicles (HEV) and plug-in hybrid electric vehicles (PHEV) is affected by the electricity sources used to power the vehicles (Onat and Kucukvar, 2022). Consequently, quantifying the impacts and benefits of transitioning toward electric vehicles requires a comprehensive tool that

Abbreviations: ADFF, Abiotic depletion fossil fuels; BEB, Battery electric buses; DoD, Depth-of-discharge; ECR, Electrochemical recuperator; EoL, End of life; EV, Electric vehicle; FAETP, Freshwater aquatic ecotoxicity; FU, Functional unit; GHG, Greenhouse gases emission; GWP, Global warming potential; HEV, Hybrid electric vehicle; HT, Human toxicity; LCA, Life cycle assessment; LFP, Lithium iron phosphate; LTO, Lithium titanium oxide; MAETP, Marine aquatic ecotoxicity; NMC, Lithium nickel manganese cobalt; ODP, Ozone layer depletion; PHEV, Plug-in hybrid electric vehicles (PHEV); SR, Sensitivity ratio.

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applies a lifecycle perspective so that upstream emissions from electricity production can be incorporated. Life cycle assessment (LCA) is still the most suitable tool to assess a wide range of environmental impacts from products or activities covering all the inputs and outputs throughout their life cycle (Baumann and Tillman, 2004). The tool was developed initially in the 60s, and its implementation has become more popular, especially in the United States, China, Italy, Spain, and Germany (He and Yu, 2020). By considering the inputs and outputs from multiple life stages, the impacts caused by upstream activities can be assessed.

Most LCA studies concerning electric mobility focus on private vehicles (Jakub et al., 2022). However, a growing number of studies are investigating the environmental impacts of heavy-duty road vehicles, such as electrifying city buses (e.g., Gustafsson et al., 2021; Huber et al., 2022). Public transport plays a key role in increasing trip efficiency, reducing traffic congestion, and improving overall air quality (Dillman et al., 2021; Wimbadi et al., 2021). It was also reported that it emits substantially lower GWP than private vehicles (European Environment Agency, 2022a). To improve its environmental performance, the government started setting a target to electrify city buses, where 90% of them in Europe are still powered by diesel (Glotz-Richter and Koch, 2016). In the city of Helsinki, 11% of the bus fleet is electrified to achieve a 30% electrification target by 2025 (Todorov, 2021). Norway has an ambitious plan to achieve zero tailpipe emissions and biogas by 2025, while Denmark and the Netherlands set the target by 2030 (Ager-Wick Ellingsen et al., 2022; Wappelhorst and Rodríguez, 2021). The situation shows the importance of understanding the potential impacts and benefits of electrizing the bus fleet.

In recent years, many LCA studies on electric public mobility started to grow, focusing on different types of batteries and levels of electrification. Most studies examined the use of lithium iron phosphate (LFP) batteries to power battery electric buses (BEB) and hybrid buses where the battery capacity for BEB ranges from 60 to 324 kWh (Ager-Wick Ellingsen et al., 2022; García Sánchez et al., 2013; Harris et al., 2018). Other studies applied lithium manganese oxide (LMO) (Bi et al., 2015), lithium titanium oxide (LTO), and lithium nickel manganese cobalt (NMC) (Ager-Wick Ellingsen et al., 2022; Harris et al., 2018). Some LCA studies also investigated the impacts of hybrid or plug-in hybrid city bus where it needs a smaller battery capacity of around 7–19 kWh combined with a different type of fuel such as diesel or vegetable oil (García Sánchez et al., 2013; Lajunen and Lipman, 2016; Nordelöf et al., 2019). These studies applied different assumptions for the depth-of-discharge (DoD) and charging cycles that affected the battery lifespan. Consequently, each study has a different frequency in changing the battery, from no replacement up to 5 times replacement during bus operation for 10–12 years. Furthermore, these studies also applied varying sources of electricity, such as Sweden, Norway, Spain, the EU average, the United States, and the United Kingdom resulted in varying degrees of environmental impact.

While there are a growing number of LCA studies concerning the impacts of an electrifying city bus, plenty of relevant aspects are still yet to be studied. Most studies only studied certain storage technologies, mainly batteries, such as LFP, LTO, LMO, and NMC. At the same time, recent studies investigated the potential of using different technology, such as ultracapacitors, to power city buses or implementing it as a hybrid system paired with batteries (İnci et al., 2021; Łebkowski, 2019; Lemian and Bode, 2022). On a practical level, an ultracapacitor has been used in a city bus in Italy, Serbia, Bulgaria, and Israel (Chariot Motors, 2021; Sustainable Bus, 2021, 2022). However, studies about the environmental impacts of storage technologies other than batteries are still rare. The search results on Scopus at the beginning of June 2023 using combining keywords “life cycle assessment” AND “capacitor” resulted in 29 documents, whilst a combination of “life cycle assessment” AND “battery” yields more than 1000 results. It indicates the need to expand the study about environmental impact assessment of various storage technologies.

Moreover, as far as the author's knowledge, LCA studies regarding capacitor-like technology to electrify public transportation have not been done. Hence, this study fills the gap by assessing the environmental impacts of electrifying city buses using ultracapacitor-like technology within the Finnish context. The results would be compared with mainstream battery technology such as LFP, providing knowledge about the environmental impacts of various energy storage system. The ultracapacitor-like technology is represented by an electrochemical recuperator (ECR) manufactured by a Finnish company, underlining the significance of this study that uses primary data input to manufacture the ECR. The ECR technology can also be considered a hybrid of battery and supercapacitor, as it can store electrical energy electrostatically and electrochemically. The aim of the study was attained by focusing on these specific goals: i). investigating the environmental impacts of electrified city bus which equipped with different storage technologies such as ECR and LFP, ii) examining how the degree of electrification is affecting the environmental impacts, iii) assessing the effect of the electricity mix used to charge the bus on the environmental impacts, iv) identifying the most critical input parameters. The main novelty of the study was in the assessment of ECR and its application on city buses, including the specific case covering the Finnish context.

2. Materials and method

2.1. Electrochemical recuperator (ECR)

This section describes the energy storage system studied called the electrochemical recuperator (ECR), developed and manufactured by a Finnish-based company. The information was obtained from the company's documentation and interview with the company's representative. The description has been proofread; hence, it adheres to the company confidentiality. The technology can efficiently store energy both electrostatically and electrochemically by combining the advantages of both methods: a high number of high-power charging cycles (typical for electric double-layer capacitors EDLC) and good energy storage capability (typical for batteries in the high-power segment). Unlike hybrid supercapacitors, these batteries demonstrate electrochemical reactions at the surfaces of both electrodes. ECR cells consist of two carbon electrodes built in a bipolar design. The electrodes are so thin that each cell has a thickness of about 0.5 mm, allowing the highest possible discharge performance. The technology uses aqueous electrolytes, radically simplifying the manufacturing process and making it both low-carbon and low-CAPEX. Moreover, the ultimate safety of water-based electrolytes ensures simplicity of the recycling chain, whereas the bipolar design of the modules ensures simplicity of the recycling process as it allows for fast and exact separation of all cell components.

ECRs typically demonstrate relatively high specific power (up to 10 kW/kg) and battery-like specific energy (up to 30 Wh/kg, depending on an electrochemistry, use-case and design). Unlike traditional batteries, they can be charged very quickly, and unlike supercapacitors, they can accumulate substantial energy with a smaller footprint. It means they can provide massive bursts of energy to the system and efficiently absorb bursts of energy (e.g. the energy recuperated by a vehicle in every breaking event or by a crane from every event of lowering the weight). ECRs do not use metals such as nickel, cobalt, or lead for their electrode. It allows 100% depth-of-discharge (DoD) with a long lifespan of up to 1 million fast charge-discharge cycles, even at low ambient temperatures. The manufacturing is relatively less energy intensive because it does not require any drying process. The bipolar design offers a large cross-section for internal current in the whole device and uniform current density on the electrodes, contributing to extreme high-power capabilities, low heat generation and high cycle life. Typical use cases include hybrid powertrains for heavy-duty vehicles and stationary fast energy storage.

2.2. Goal and scope definition

The main goal of this study was to compare the environmental impacts and benefits of different energy storage technologies used in the 12-m city bus with the main focus on the global warming potential (GWP). The assessment was conducted following the standards for LCA covered by ISO 14040 and 14,044 (ISO, 2006a, 2006b). The comparison among the degree of electrification was also scrutinized along with different electricity mixes that powered the batteries. Electricity mixes affect its production system, influencing the emissions generated by electric vehicles (Gustafsson et al., 2021). The LCA was conducted on hypothetical cases covering different types of buses. These are battery electric buses (BEB) run by ECR and LFP, hybrid buses with a pantograph charging system (which is referred to in this document as 'hybrid bus') employing ECR and LFP, and conventional buses run by diesel (Table 1). Although the ECR is specially designed for the hybrid application, the hypothetical case included a fully electric system to investigate whether there is a consistent correlation between the environmental benefits versus the degree of electrification (e.g., whether BEB always provides higher benefits than hybrid). The LFP was chosen to represent the more mainstream energy storage system since its characteristics fall between NMC and LTO, especially concerning its energy density and lifetime, and offers versatile technology (Ellingsen et al., 2016). The BEB was powered by 200 kWh of LFP and 4.3 kWh of ECR, whereas the hybrid bus used 19 kWh LFP and 0.75 kWh ECR (Kempower, 2020; Łebkowski, 2019; Nordelöf et al., 2019). A pantograph charging system was assumed since it became more popular, and bus companies incorporated this technology into their electric bus (Volvo, 2020). Technological performances such as charging cycles, depth-of-discharge (DoD) and charging efficiency were considered due to their impacts on battery lifespan (Ager-Wick Ellingsen et al., 2022; Göhlich et al., 2018). Battery replacement was also integrated into the analysis so that the impacts could be captured. The functional unit (FU) was "the extraction of raw materials, manufacturing and use of the electricity storage system for 1 km of driving during 12 years of bus lifespan", and thus the LCIA results will be presented "per km" driving for 12 years of bus lifespan, considering its suitability in representing bus mobility.

The study focused on battery cells, considering the data availability of ECR and the importance of cells as the main technology of an energy storage system. The vehicle and the charging infrastructure were not included in the study. The system boundaries were specified as material extraction, component production, cell manufacturing and battery use in the city bus (Fig. 1). In the use phase, the inputs considered were energy consumption (electricity and diesel) and battery replacement if necessary. The average Finnish electricity mix was used in the baseline analysis in the production and use phase.

2.3. Life cycle inventory

Foreground and background data were collected to perform LCA. The data regarding ECR component production and cell manufacturing were obtained from the company that develops and manufactures the ECR as described in Section 2.1, using the baseline year 2022. Information regarding the GWP of LFP batteries on the production level was taken from Ager-Wick Ellingsen et al. (2022). Besides, the background system was complemented by Ecoinvent database version 3.8 (Ecoinvent, 2021). When the Finnish context was unavailable in the database, the European was applied, followed by the global context. Due to proprietary reasons, the detailed data cannot be disclosed; nevertheless, the material input can still be categorized into different components. Table 2 shows the components of LFP (Ager-Wick Ellingsen et al., 2022) and ECR cells.

The inventory data for the use phase was mainly obtained from literature supplemented by expert judgment. A comparison was conducted based on the degree of electrification and storage technology, where the LFP battery was compared to ECR because it is widely used and has decent properties. The battery has a relatively high charging cycle of up to 4000 times for DoD of 80% with a specific energy of 150 Wh/kg-cells (Ager-Wick Ellingsen et al., 2022; Saft, 2015). Meanwhile, the ECR used in this application has a low specific energy of 12.7 Wh/kg-cells with 100% charge-discharge capability for up to 1 million cycles. The ECR operating temperature ranges between −40 °C to 60 °C; meanwhile, the LFP technology taken from Nordelöf et al. (2019) was based on Volvo electric buses tested in extreme cold and heat in Nordic countries and Spain. Nevertheless, both storage systems were equipped with automatic systems to control temperature where the average operating temperature is around 23 °C. The air density was assumed to be 1.225 kg/m^{3,} and relative humidity ranges between 65 and 90% (Finnish Meteorological Institute, 2022). The inventory of the use phase can be found in Table 3, whereas the complete reference of each data is provided in the supplementary material Table 1.

The table presents information regarding the storage technology on the cell level since it was the focus of the study. The basis of the bus weight was the same, and the difference was from the weight of the storage technology installed. However, the pack's weight was applied for bus weight calculation to represent the actual situation. The pack-to-cell ratio of the storage system can range around 1.24–1.6 (Ager-Wick Ellingsen et al., 2022; Wieczorek et al., 2019), and a conservative estimation of 1.6 was applied in this study. The ECR was paired with a small capacity LFP of 10 kWh for the BEB as an emergency backup. It was estimated that the use of auxiliary LFP was about 0.2% of distance driving. The property differences between LFP and ECR, including ultra-fast charging for ECR, affect the sizing and replacement frequency. A few seconds of charging at the bus stop is sufficient, making it possible to install a small capacity of ECR. A Replacement may be necessary when the batteries reach their end-of-life cycle. This study showed the need for LFP battery replacement due to its life cycle. The replacement time was estimated by calculating the total cycle needed during 12 years of bus lifespan compared to the batteries' maximum charging cycle. LFP replacement was done once for the BEB and six times for the hybrid bus, respectively. In contrast, the ECR did not require replacement in the BEB and hybrid applications. Unlike ultracapacitors, the application of ECR will not require DC/DC converters.

The baseline of electricity needed per km driving of a bus with LFP and ECR were assumed to be similar (Ager-Wick Ellingsen et al., 2022; Nordelöf et al., 2019). The difference was accounted for by the bus weight; hence, an adjustment was made where every 10% of mass vehicle reduction correlates to a 4.5% electricity decrease (Bi et al., 2015). On average, the bus covers 65,000 km per year, resulting in a total of 780,000 km during the bus lifespan.

2.4. Life cycle impact assessment

The impact assessment followed the CML baseline (CML Leiden University, 2016). The University of Leiden developed the impact assessment method, and it is divided into baseline and non-baseline. The baseline impacts were applied since it covers LCA's most common impact categories; hence, it is broad enough but will not overwhelm those unfamiliar with LCA. The CML-baseline impact categories include abiotic depletion, abiotic depletion fossil fuels (ADFF), acidification, eutrophication, freshwater aquatic ecotoxicity (FAETP), global warming potential (GWP), human toxicity (HT), marine aquatic ecotoxicity (MAETP), ozone layer depletion (ODP), photochemical oxidation. All

Table 1
Hypothetical cases assessed by LCA.

Electrification Storage system	BEB	Hybrid	Conventional (full diesel)
ECR	x	x	
LFP	x	x	
None			x

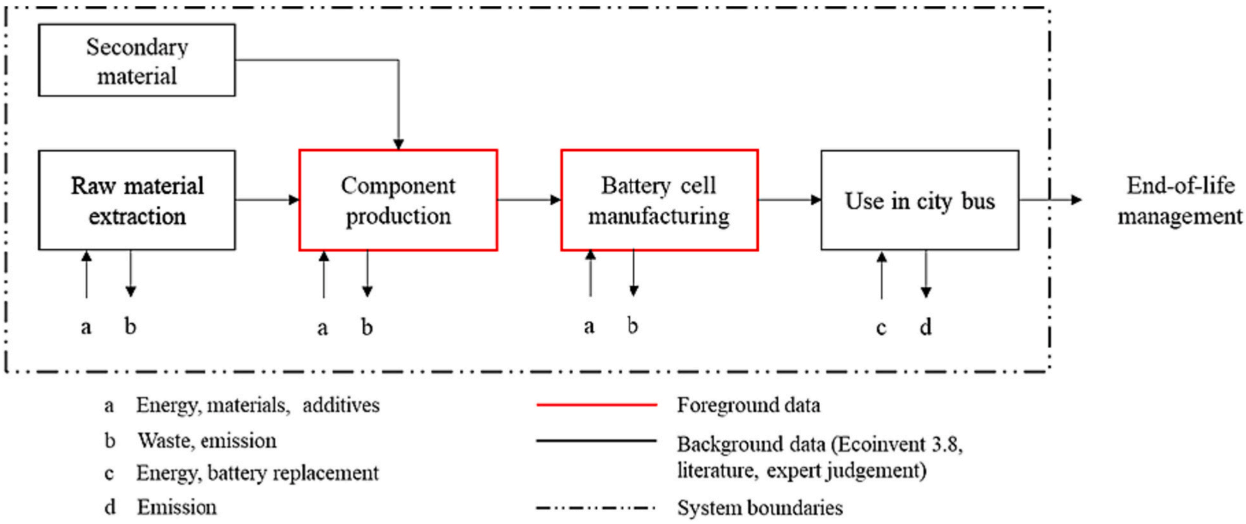

Fig. 1. System boundaries of the LCA on ECR cells.

Table 2
Components of ECR and LFP cells.

Inputs and outputs	ECR	LFP	Unit
Inputs			
Cathode	3	451.2	Gram
Anode	2	336.9	Gram
Separator	1	44.5	Gram
Current collectors	8	(Included in anode and cathode)	Gram
Electrolyte	9	61.4	Gram
Sealant	4.5	–	Gram
Pouch	–	3	Gram
Tabs		3.41	Gram
Water	–	8100	Gram
Electricity	0.00375	3.2	kWh
Heat	–	2.3	kWh
Outputs			
Cells	27.48	900.5	Gram
Liquid waste	0.018	8100	Gram

these impacts would be included in the ECR manufacturing stage, while GWP was the main emphasis in the subsequent stage, including comparing cases, scenarios, and sensitivity analysis. GWP was the focal point due to the urgency to reduce fossil fuel consumption through bus electrification.

Attributional and comparative approaches were applied in this study, where the impact accounting was conducted, and different cases were compared. Contribution analysis was employed to investigate the relative contribution of each process or input to the environmental impacts. The modelling used OpenLCA software version 1.11 (OpenLCA, 2022). No normalization was applied since the core of the work was to quantify environmental impacts and compare cases without the need to evaluate relative magnitude with reference situations.

2.5. Sensitivity and scenario analysis

Sensitivity and scenario analysis were employed to address the uncertainty issue. The global sensitivity analysis can provide insight on how the results change due to varying the input parameters, while

Table 3
Data of bus alternatives in the use stage.

Item	Unit	BEB		Hybrid		Conventional
		LFP	ECR	LFP	ECR	Diesel bus
Bus weight	kg	13,400	11,921	11,465	11,359	12,500
LFP cell installed	kg	1329.1	66.7	126.3	–	–
LFP capacity installed	kWh	200	10	19	–	–
LFP specific energy	kWh/kg	0.150	0.150	0.150	–	–
ECR cell installed	kg	–	337.6	–	58.9	–
ECR capacity installed	kWh	–	4.3	–	0.75	–
ECR specific energy	kWh/kg	–	0.0127	–	0.0127	–
LFP DoD	%	80	80	80	–	–
LFP charging cycle	times	4000	4000	4000	–	–
ECR DoD	%	–	100	–	100	–
ECR charging cycle	times	–	1,000,000	–	1,000,000	–
Auxiliary use of LFP	% of distance	–	0.2	–	–	–
Electricity consumption	kWh/km	1.430	1.423	0.530	0.530	–
Mileage	km/year	65,000	65,000	65,000	65,000	65,000
Bus lifespan	years	12	12	12	12	12
Charging efficiency	%	95	95	95	95	–
Diesel consumption	l/km	–	–	0.19	0.19	0.45
LFP replacement	times	1	0	6	–	–
ECR replacement		–	0	–	0	–
Total distance	km	780,000	780,000	780,000	780,000	780,000
LFP cycles	Cycles in 12 years	6971	277	27,197	–	–
ECR cycles	Cycles in 12 years	–	257,591	–	550,970	–

scenario analysis deals with the background system (Faraca et al., 2019).

2.5.1. Sensitivity analysis

Sensitivity analysis examines how outputs vary due to input parameter changes (Bisinella et al., 2016). This paper applied the "one-at-a-time" method, where each input parameter during the manufacturing and use phase was increased by 10% while keeping the rest at their baseline value. The sensitivity ratio (SR) was then calculated to offer insight into the model sensitivity against each parameter. If a parameter has an SR of 5, it indicates that a 10% increase of a certain parameter will increase the final result by 50%. SR can be calculated using formula (1):

$$SR_i^j = \frac{\left(\frac{\Delta result}{initial\ result}\right)^j}{\left(\frac{\Delta parameter}{initial\ parameter}\right)_j} \approx \frac{\partial z_j}{\partial x_i}\ \frac{x_i}{z_j} \tag{1}$$

2.5.2. Scenario analysis

How the model behaves was tested using scenario analysis. It was applied at the use phase of ECR, since the ECR is the main focus of the study, by assuming that the buses are operated in countries other than baseline, namely Norway and Poland. Scenario analysis employed the same data as the baseline, presented in Table 3; however, the electricity mix that powers the bus was adjusted according to respective countries. Distinct geographic locations have different average electricity mixes, so the effect of those differences can be evaluated. The data and impacts of the electricity mix in each country were obtained from Ecoinvent 3.8. The GWP results from scenario analysis were compared to the baseline situation where the effect of electricity mixes to operate electric buses was assessed. When the electric bus was found to be inferior to diesel, additional analysis was applied to investigate a possible electricity source composition to benefit from the electric bus.

3. Results

3.1. Impacts of ECR production

The CML-baseline impacts of ECR manufacturing were assessed per kWh capacity (Table 4). Every kWh of ECR equals 78.5 kg or 2857 cells due to its lower specific energy compared to mainstream battery technologies. Manufacturing ECR cells generated a GWP of 178 kg CO_2-eq per kWh of stored capacity. Contribution analysis showed that current collectors and sealant were the highest emitters, accounting for about 42% and 20% of total GWP, respectively. Fig. 2 reveals contribution analysis from each cell component with regards to GWP.

The impact contributions of material input did not always align with their weight and were subjected more to the material type. The proportion of the material inputs for the current collector and electrolyte was about 29% and 33%, respectively. Nevertheless, only current collectors showed a high impact; meanwhile, electrolytes generated one of

Table 4
Environmental impacts of manufacturing 1 kWh ECR.

Impact categories	Value	Unit
Abiotic depletion	0.0018	kg Sb-eq
Abiotic depletion (fossil fuels)	2785	MJ
Acidification	0.9707	kg SO_2-eq
Eutrophication	0.7290	kg PO_4-eq
Freshwater aquatic ecotoxicity	134	kg 1,4-dB-eq
Global warming potential	178	kg CO_2-eq
Human toxicity	164	kg 1,4-dB-eq
Marine aquatic ecotoxicity	224,514	kg 1,4-dB-eq
Ozone layer depletion	3.74E-05	kg CFC-11-eq
Photochemical oxidation	0.0607	kg C_2H_4-eq
Terrestrial ecotoxicity	0.3130	kg 1,4-DB-eq

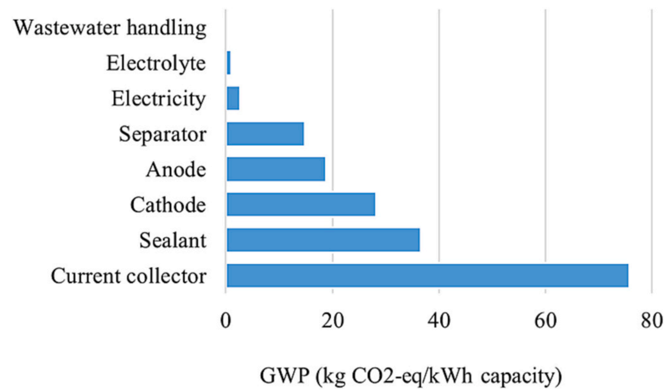

Fig. 2. GWP of the ECR based on the component inputs.

the lowest contributions toward GWP. A complete contribution analysis of each impact category can be found in the supplementary material Fig. 1.

3.2. GWP of ECR and LFP

The impact of storage technology in the BEB with ECR refers to the impacts caused by ECR and the small LFP installed as a power backup. Technology comparison was carried out between ECR, LFP, and diesel bus, as shown in Fig. 3. For the LFP, the impact from manufacturing the product per kWh was taken from Ager-Wick Ellingsen et al. (2022) while adjusting the result so that the boundaries were comparable with the ECR. Both ECR and LFP showed environmental benefits when compared to a diesel bus. Diesel buses caused about 1.2 kg CO2-eq per km driving. Operating BEB and hybrid instead of diesel buses could reduce GWP by around 63–68% (with ECR application) and 44–46% (with LFP application). The benefits were also affected by the degree of electrification, where BEB was 1.7 and 1.5 times better than hybrid buses when applying ECR and LFP technologies, respectively.

When the overall GWP were compared, the bus that applied ECR (0.385 kg CO_2-eq/km) performed 14.5% better than LFP (0.441 kg CO_2-eq/km). Whereas in the hybrid bus, the ECR application (0.652 kg CO_2-eq/km) provided a 3% improvement to LFP (0.670 kg CO_2-eq/km). These differences were observed based on the overall outcomes, which were dominated by the impact of energy consumption during the use phase (e.g., electricity and diesel consumption). A clearer comparison would be obtained when the focus was on the impact of the battery technologies per km driving during the bus lifespan. The GWP due to storage technologies per km driving for the BEB with ECR was 0.0024 kg CO_2-eq. (0.62% of total impact); meanwhile, BEB with LFP generated 0.0565 kg CO_2-eq. (12.8% of total impact). These results showed that the impact generated by LFP was 23 times higher than ECR. In the hybrid system, the GWP per km of bus applying ECR and LFP were 0.00017 kg CO_2-eq. (0.03% of total impact) and 0.01878 kg CO_2-eq. (2.8% of total impact), respectively. It indicated that GWP generated by LFP was about 110 times higher than ECR in the hybrid application.

The GWP of applying LFP was higher than ECR since LFP had fewer charging cycles. Deeper discharge in the LFP battery will cause the battery to wear off quicker than a shallow cycle (Göhlich et al., 2018; Saft, 2015). With a relatively high DoD of 80%, the LFP battery was assumed to last for about 4000 cycles; hence, battery replacement was needed in both BEB and hybrid. The calculation showed that BEB required one battery replacement, whereas hybrid systems needed six times replacement. On the contrary, the ECR could withstand 1 million cycles with DoD 100%, so replacement is unnecessary throughout the bus lifespan.

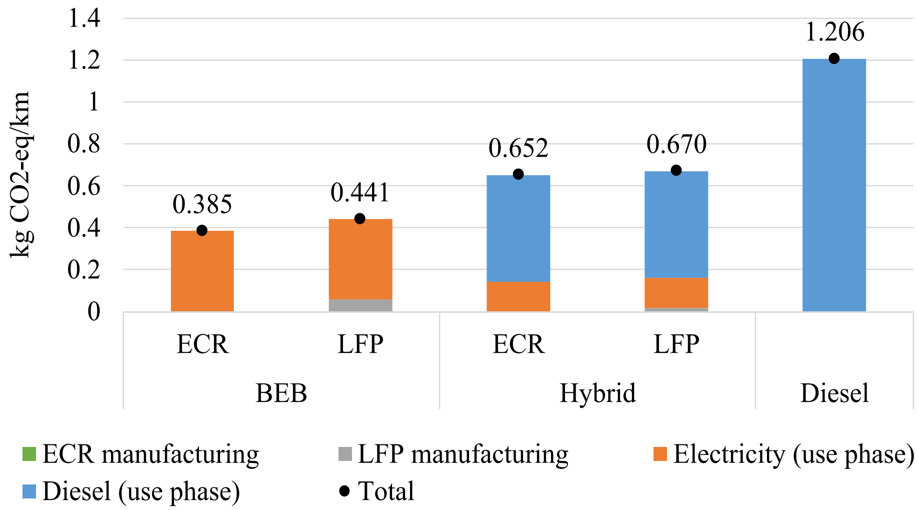

Fig. 3. GWP of BEB, hybrid, and diesel bus using ERC and LFP technologies.

3.3. Sensitivity analysis

Each parameter in the manufacturing and use stage was increased by 10% to assess its sensitivity to the model by calculating the sensitivity ratio (SR). The SR of the input parameters in the manufacturing stage ranged from around zero to 0.0013. Meanwhile, the parameters in the use phase resulted in SR between −0.94 and 1. It implied that changing some parameters in the use stages affected the overall results more significantly than in the manufacturing process. A change in the most sensitive parameter during the manufacturing stage would increase the overall GWP by about 0.013%. Fig. 4 shows the sensitivity ratio of seven input parameters in the manufacturing stages for both BEB and hybrid systems.

The BEB was more sensitive to the change in the material inputs compared to the hybrid bus. The most sensitive parameter was the current collector, while increasing the use of electrolyte did not affect the overall GWP (SR = 0). In the hybrid system, the only material input resulting in a change in the GWP was the current collector (SR = 0.000153). For BEB, the SR ranged between 0.00026 and 0.0013. The ECR materials were less influential in hybrid since their role in powering the bus was divided with diesel.

Fig. 5 provides information regarding the SR of parameters in the use stage for both BEB and hybrid. The most sensitive parameters for the BEB were electricity consumption, charging efficiency, and bus weight, as shown by SR of 1, −0.94, and 0.04, respectively. A 10% increase in the electricity needed to drive per km would raise the overall GWP value by 10% up to 0.423 kg CO_2-eq per km. Charging efficiency was increased from 95% to 100% since changing its value by 10% would result in efficiency above 100%. Negative SR was found when charging efficiency was increased since the energy consumed by the charging process and the energy saved by the battery were the same. The third most sensitive parameter was bus weight because the electricity consumed to run the bus was affected by its weight.

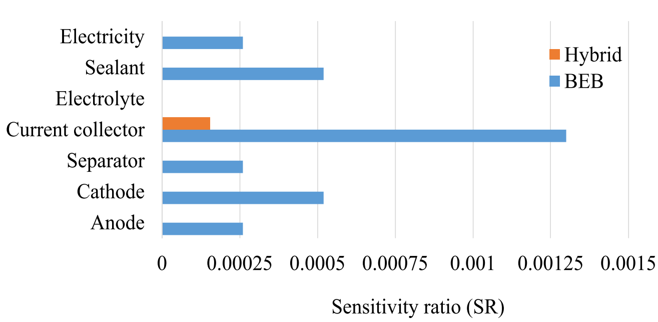

Fig. 4. SR of manufacturing input against GWP.

Differences and similarities were found in the hybrid system. The sensitivity of electricity consumption and charging efficiency showed similar trends. However, they were not as sensitive as in the BEB. Changing the hybrid system's electricity consumption and charging efficiency resulted in SR of 0.22 and − 0.11, respectively. The most sensitive parameter in the hybrid system was diesel consumption, where a 10% increase in diesel would raise overall GWP by about 8% (SR = 0.78).

3.4. Scenario analysis

Fig. 6 showed the results of scenario analysis when the BEB and hybrid bus were assessed using Norwegian and Polish contexts. The scenario analysis focused on the electricity mix since the baseline analysis shows a significant impact from energy sources in the use phase compared to the production stage. This analysis demonstrated that the benefit of the electrifying bus depends on the electricity sources and did not always perform better than the conventional diesel-powered bus.

The same pattern was found in both BEB and hybrid systems, where Norway showed the best performance, followed by Finland and Poland. Using BEB in Norway was shown to be 11 and 42 times better than Finland and Poland, respectively. It is attributed to the GWP per kWh of electricity produced in each country. Norway, Finland, and Poland emit about 25 g of CO_2-eq/kWh, 250 g of CO_2-eq/kWh, and 990 g of CO_2-eq/kWh, respectively (Ecoinvent, 2021). Running a fully electrified bus in Poland demonstrated an inferior performance, as shown by 23% higher GWP than a diesel bus, whereas Norway and Finland obtained benefits, as shown by a GWP reduction of −68% and − 97%, respectively. The hybrid system provided different insights where the performance of three countries offered benefits than a diesel bus. The benefits of the hybrid system in Finland and Norway were lower than BEB, as indicated by GWP reductions of −46% and − 57%. Contrary to the BEB, the hybrid bus provided climate benefits to Poland, as displayed by a reduction of −12% compared to a diesel bus. Hybrid system attempts to maximize vehicle performance through propulsion configurations such as series, parallel, and power-split configurations (Pei et al., 2018). It determines fuel consumption that affects the vehicle's emissions. It is known that estimating fuel consumption in the hybrid system can be challenging due to the vehicle's operations and driving cycle (Sun et al., 2021). In this study, the data for fuel consumption in the hybrid system was for Volvo 7900, which used parallel configuration and showed an efficient fuel consumption of 19 l/100 km (Nordelöf et al., 2019). The empiric value of the Volvo bus in this study was relatively efficient compared to,

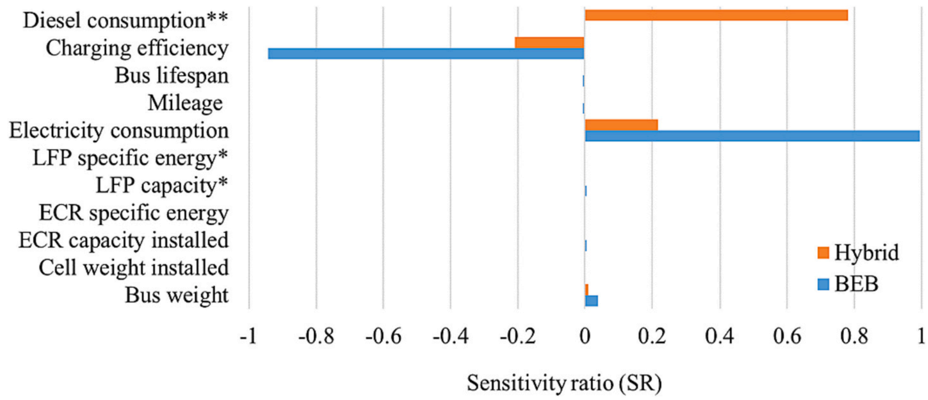

Fig. 5. SR of parameters in the use stage against GWP (** is for hybrid only, whereas * is for BEB only).

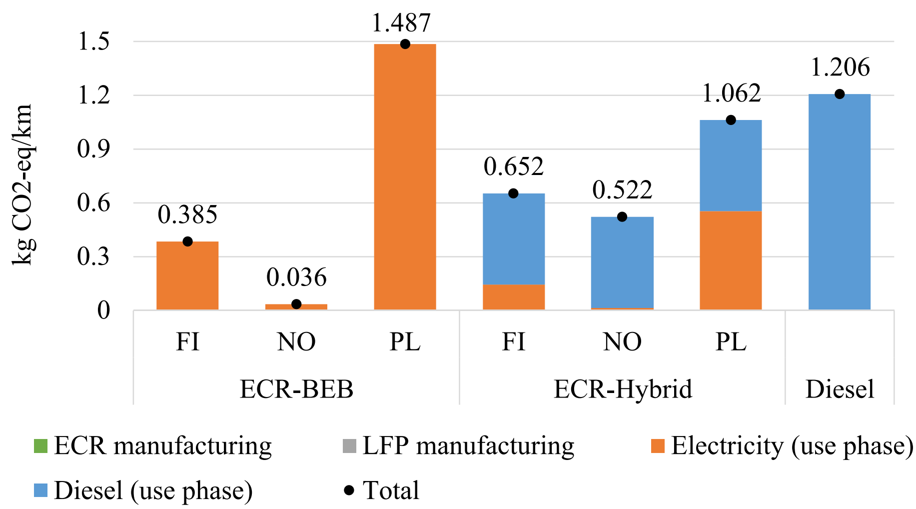

Fig. 6. Scenario analysis of BEB and hybrid in different geographic locations.

for example, a study done by Zhang et al. (2019), who applied a power-split configuration for a 12-m bus. Their results showed fuel consumption between 24 and 28 l/100 km with an average of 26.3 l/100 km. If the average fuel consumption from Zhang et al. (2019) were applied, the GWP would be higher than diesel buses, making diesel the more favourable option in Poland.

The calculation was also done to assess Poland's electricity composition, which would not result in worse performance from BEB than a diesel bus. Ecoinvent 3.8. shows that about 81% of total electricity sources in Poland are hard coal (45.7%), lignite (27.3%) and wind turbine (7.7%). The adjustment focused on the highest fossil fuel and renewable sources while maintaining the rest as the baseline. The breakeven point of the GWP from BEB and diesel bus occurred when hard coal use decreased to 33%, and the wind energy was doubled to 15.2%, resulting in GWP of 1.203 kg CO_2-eq/km for BEB where the impact from diesel bus was 1.206 CO_2-eq/km.

4. Discussions

4.1. Overall results

The study focused on determining the environmental impacts of different storage technologies to electrify city buses, comparing the degree of electrification, and assessing the effect of background system and input parameters. Comparing the findings across different LCA studies is often not straightforward since there are differences in boundaries, assumptions, methods, system model, background system, etc. Nevertheless, comparing previous studies is crucial to confirm this study's outputs. On the manufacturing level, where the boundaries cover cradle-to-gate, a more straightforward comparison can be made, with attention paid to the level of battery manufacturing (cell, module, or pack), FU, and geographical location.

In manufacturing, ECR cells emitted about 178 kg CO_2-eq/kWh capacity. The impact caused by materials accounted for about 98.5%, and about 1.5% of the GWP was from electricity consumption. Ellingsen et al. (2017) summarized the GWP per kWh of cell materials ranging from 22 to 110 kg CO_2-eq. ECR has a significantly higher impact per kWh cell manufactured due to its low specific energy. On the other hand, studies showed that the energy consumption to manufacture 1 kWh cell ranged widely from 2.7 to 1060 MJ, whereas the ECR cell required 38.58 MJ/kWh. It was also reported that 50% of the impact of cell manufacturing was attributed to energy consumption, whereas others showed energy consumption resulted in 0.9–2.3 kg CO_2-eq/kWh (Dunn et al., 2012; Ellingsen et al., 2014; Kim et al., 2016). This study showed that electricity consumption was insignificant, as indicated by the 2.7 kg CO2-eq/kWh emission. The impact of energy consumption during cell production is caused by the quantity consumed and the production locations, which reflect the mix of electricity sources. Currently, most battery cell manufacturers are in the United States, China, Japan and South Korea (Aichberger and Jungmeier, 2020).

The average GWP of 55 studies on lithium-ion battery packs per kWh was about 187 kg CO_2-eq. (Zhao et al., 2021a). Among others, 63% of the impacts were generated from cell manufacturing, equaling 119 kg CO_2-eq/kWh (Ager-Wick Ellingsen et al., 2022; Zhao et al., 2021a). On a more specific level, Ager-Wick Ellingsen et al. (2022) analyzed three different battery technologies: LTO, LFP, and NMC. The outcomes of their study were adjusted by excluding the impact of factory infrastructure; hence, the boundaries are comparable with this study, which showed GWP per kWh of cells were 277 kg CO_2-eq (LTO), 108 kg CO_2-eq (LFP), and 70 kg CO_2-eq. (NMC).

ECR generated higher GWP per stored kWh in cell manufacturing than LFP and NMC, while the climate-related impact from LTO was higher than ECR. The impacts could be attributed to the specific energy of different storage technologies. The specific energy of LTO, LFP, and NMC per kg cell was 85 Wh, 150 Wh, and 245 Wh, whereas ECR specific energy was about 12.7 Wh/kg cells. It indicated that the cell materials or utilities such as electricity or gas in the LTO production caused a high impact. On the contrary, the ECR is water-based and does not incorporate metal, including cobalt, nickel, or lead, in its electrode.

Besides its manufacturing stage, the battery was assessed based on its specific application. When the impacts covered the raw material extractions up to the use phase, higher uncertainties were found due to the introduction of more parameters. These parameters included distance travel, bus lifespan, bus weight, DoD, battery life cycle, electricity consumption, etc. The GWP per km driving for BEB using the ECR was 2.4 g CO_2eq/km. By adjusting the battery pack impact into cell level, the GWP per km driving of LTO, LFP, and NMC applied in BEB with pantograph system were 90.3 g CO_2-eq, 36.6 g CO_2-eq, and 24.4 g CO_2-eq, respectively (Ager-Wick Ellingsen et al., 2022). LTO had the highest impact among these three technologies due to its lowest energy specific. Comparable results were found where the adjusted GWP per km driving was gram 30 CO_2-eq (Nordelöf et al., 2019), 23 g CO_2-eq (de Souza et al., 2018) and 22 g CO_2-eq (Bi et al., 2015). For the plug-in system, previous studies showed a GWP of 10 g CO_2-eq/km (Nordelöf et al., 2019) and 15 g CO_2-eq/km (de Souza et al., 2018). Meanwhile, this study generated 0.17 g CO2-eq/km of ECR cell use in the hybrid pantograph system. ECR offered significant benefits, although it had substantially lower specific energy. The ECR applied in BEB and hybrid had a lower negative impact due to its high DoD and charging cycles. Therefore, battery replacement was not required, and a few thousand hundred cycles were still left after the end of the bus lifespan.

Electricity mix played a major role in affecting the GWP of electric bus deployment. Using the Finnish context, the GWP of BEB due to electricity consumption was 382 g CO_2-eq/km driving. The result aligned with previous studies that used different electricity backgrounds. It was reported that the GWP per km driving ranged between 470 and 570 g CO_2-eq for a mix of Norwegian and Hungarian (Ager-Wick Ellingsen et al., 2022). The impact of the study using Swedish and European electricity mix was around 170- and 505-g CO_2-eq, respectively (Nordelöf et al., 2019). The Spanish mix showed higher GWP, accounting for about 700 g of CO_2-eq (García Sánchez et al., 2013). An even higher result was obtained from this study when applying a Polish electricity mix where the GWP was around 1487 g CO_2-eq. In the hybrid system, GWP per km driving was 652 g CO_2-eq. In comparison, a previous study showed that the hybrid system emitted GWP of 576 g CO_2-eq (Sweden), 832 g CO_2-eq. (EU), and 912 g CO_2-eq (The United States) (Nordelöf et al., 2019).

The end of life (EoL) management of the battery was excluded. It is often found that this aspect of the life cycle has a relatively minuscule impact when compared to other stages (e.g., Ager-Wick Ellingsen et al., 2022; Grazieschi et al., 2023; Nordelöf et al., 2019). In the case of batteries used to electrify public buses, most environmental impacts tend to be concentrated in the use stage where the electricity is generated. Having adjusted the FU, a study by Nordelöf et al. (2019) showed that EoL management of the bus and battery of both hybrid and BEB systems were about 16–18 g CO2-eq/km. It can be translated into an estimation of 0.5 g CO2-eq/km and 2 g CO2-eq/km for BEB and PHEV, respectively. Similarity was found in another study where EoL management of 200 kWh LFP was about 3 g CO2 eq/km (Ager-Wick Ellingsen et al., 2022). Rough estimations based on studies mentioned above indicated that EoL management of LFP battery cells of BEB in this study would be about 2.5% of total GWP, whereas the EoL management of batteries used in the other bus types was about 0.3% or less, causing inconsequential change to the overall results.

Although including end-of-life management in the LCA system boundary is essential for a comprehensive assessment, it is unlikely to alter the overall results significantly. To drive meaningful improvements in the environmental performance of battery technologies, a more substantial focus should remain on reducing the environmental impacts in the earlier stages of the life cycle since battery replacement is often needed, optimizing battery life and efficiency, as well as clean electricity sources.

4.2. The effect of methods, assumption, and technology applications

This study applied an attributional system model using average value and common methods that can facilitate comparison across studies. The use of average value had a drawback that was anticipated by applying sensitivity analysis. The results of SR offered insights showing that not all parameters are equally influential. The differences in the value used or assumptions indicated the importance of transparency in LCA. Disclosing the methods, assumptions, and parameters could improve the accountability of the study.

Meanwhile, the results from scenario analysis where the bus was operated in Finland, Norway, and Poland emphasized how generalizations should be made carefully. Norway and Finland are in the Nordic region, where the energy mix is relatively clean, and the demographic can be similar. Poland has a much different energy mix, so generalizing should be done cautiously. The situation shows the importance of case-per-case analysis to understand whether certain cases contribute positively toward sustainability. A similar sentiment was voiced by Korhonen et al. (2018), who emphasized the importance of careful analysis in each case or project.

This study also showed the importance of selecting a proper energy storage system for maximum benefit. The application of ECR for city buses is apt considering the short distance to travel between each bus stop, mainly between 200 and 600 m (Na and Isma, 2013; Nordelöf et al., 2019), added by some buffer capacity to anticipate congestion and traffic lights. Consequently, the ECR in the BEB is equipped with a small LFP battery as a power backup. Using ECR in the city bus is especially appropriate with a pantograph charging system that allows the bus to charge at each stop (Volvo, 2020), at which the ECR can be fully charged in seconds. ECR with lower installed capacity makes the system lighter, so it consumes less energy per km driving or makes more room for passengers. Its high charging cycle numbers make battery replacement unnecessary and can be transferred when the bus is decommissioned.

4.3. Practical implications

Transitioning from diesel to electric buses carries practical implications, particularly for public transportation agencies and related stakeholders. These implications will have a profound impact on management practices and decision-making processes. The transition requires substantial capital to procure electric buses, charging infrastructure, and related technology. It is reported that the front-end cost of electric buses is high and can be a barrier to transitioning, although the operational costs are lower in the long run compared to diesel buses (Johnson et al., 2020). More stakeholders are involved in the procurement and operation of electric buses than conventional ones. The transportation agency, bus operator, bus manufacturers, the operators of the bus and the charging infrastructures, municipality, electricity providers, and grid owner should collaborate. The collaboration model may be adjusted depending on the stakeholder's experience handling the electric bus system and where the chargers are located (e.g., along the route or at the depot) (Borén and Grauers, 2019). A strategy for phasing out diesel buses and integrating electric buses into the existing fleet is also needed. Generally, it takes at least 18 months from the beginning of transitioning until the buses are in service (Volvo, 2021). This strategy encompasses decisions on the quantity and types of electric buses to purchase, considering route requirements, passenger demand, and environmental goals (U.S. Department of Transportation, 2023). The selection of reputable electric bus manufacturers and charging infrastructure providers is critical for long-term success, emphasizing the importance of supplier vetting and relationships.

Moreover, standardization in terms of bus models and charging equipment can significantly streamline maintenance operations, reducing complexity and operational costs. Future expansion of the electric bus fleet and associated infrastructure, as well as potential technology upgrades, should be considered in long-term planning to sustain the benefits of the transition. The transition to electric buses requires effective managerial strategies, forward-thinking approaches, and the ability to adapt to changing technologies and regulations. Responsible parties must carefully plan, execute, and monitor the transition to ensure the success of the electric bus fleet while delivering environmental, economic, and operational goals.

5. Conclusions

This research examined the influence of different energy storage systems, specifically LFP and ECR, on the environmental impacts of a pantograph city bus. The effect of the electrification degree of the city bus was also examined. The study implies the importance of choosing an appropriate energy storage system based on the application. Electric city buses with a pantograph charging system require energy to power the bus between bus stops. The study suggests that ECR offers a good solution for this specific application. Its small pack makes it lighter, and a high charging cycle increases the lifespan while equipped with frequent fast charging technology. Meanwhile, there is no consistent pattern between the degree of electrification and the environmental benefits because of the fuel consumption rate in hybrid systems and the variety in the electricity mix, which causes different environmental impacts. Hence, climate change impact reduction through electrifying city buses begins with decarbonizing electricity generation. The results presented here depend on the assumptions made in the present study and might not be universally applicable.

Therefore, research emphasized cautious handling of generalizations, noting that while they may be applicable on a trend level, adjustments are often needed when using the actual value of the results. The adjustments are needed because of the study's focus on cells, the background system representing the Finnish context, and the technology's suitability for the city bus with the pantograph charging model. It also stressed the importance of evaluating green projects individually to validate their environmental benefits.

Data availability posed the main limitation of this study. While primary data informed the manufacturing stage of ECR, the LFP manufacturing and usage phases relied heavily on secondary data, with relevant values selected. Notably, information on various lithium batteries was more accessible than for ECR or ultracapacitor-like systems. Consequently, some LFP battery information was adapted for ECR with adjustments. For instance, the electricity consumption per km was influenced solely by bus weight, neglecting different battery technologies. Therefore, the study incorporated scenario and sensitivity analysis, highlighting the key parameters to account for variability and data uncertainty. Understanding these variations can aid decision-makers and stakeholders in making informed choices for electrifying public transport, including bus types, energy storage systems, and charging models.

Author's contributions

The author designed the research concept and did the LCA calculation, analysis, discussion, and manuscript writing. The ECR company has approved the manuscript adhering to the confidentiality issues.

CRediT authorship contribution statement

Bening Mayanti: Conceptualization, methodology, data curation, formal analysis, validation, visualization, writing - original draft, writing -review and editing.

Declaration of competing interest

The author has no conflicts of interest to declare.

Data availability

The authors do not have permission to share data.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eiar.2023.107413.

References

Ager-Wick Ellingsen, L., Jayne Thorne, R., Wind, J., Figenbaum, E., Romare, M., Nordelöf, A., 2022. Life cycle assessment of battery electric buses. Transp. Res. Part D: Transp. Environ. 112 https://doi.org/10.1016/j.trd.2022.103498.

Aichberger, C., Jungmeier, G., 2020. Environmental life cycle impacts of automotive batteries based on a literature review. Energies 13 (23), 1–27. https://doi.org/10.3390/en13236345.

Arioli, M.S., D'Agosto, M., Amaral, F.G., Cybis, H.B.B., 2020. The evolution of city-scale GHG emissions inventory methods: A systematic review. Environ. Impact Assess. Rev. 80 https://doi.org/10.1016/j.eiar.2019.106316.

Baumann, H., Tillman, A.-M., 2004. The hitch hiker's guide to LCA. In: Studentlitteratur Lund. https://doi.org/10.1065/lca2006.02.008.

Bi, Z., Song, L., De Kleine, R., Mi, C.C., Keoleian, G.A., 2015. Plug-in vs. wireless charging: life cycle energy and greenhouse gas emissions for an electric bus system. Appl. Energy 146, 11–19. https://doi.org/10.1016/j.apenergy.2015.02.031.

Bisinella, V., Conradsen, K., Christensen, T.H., Astrup, T.F., 2016. A global approach for sparse representation of uncertainty in life cycle assessments of waste management systems. Int. J. Life Cycle Assess. 21 (3), 378–394. https://doi.org/10.1007/s11367-015-1014-4.

Borén, S., Grauers, A., 2019. Stakeholder collaboration models for public transport procurement of electric Bus systems. Int. J. Sustain. Policy Pract. 15 (1) https://doi.org/10.18848/2325-1166/CGP.

CML Leiden University, 2016. CML-IA Characterisation Factors. Institute of Environmental Sciences (CML). https://www.universiteitleiden.nl/en/research/research-output/science/cml-ia-characterisation-factors.

Dillman, K., Czepkiewicz, M., Heinonen, J., Fazeli, R., Árnadóttir, Á., Davíðsdóttir, B., Shafiei, E., 2021. Decarbonization scenarios for Reykjavik's passenger transport: the combined effects of behavioural changes and technological developments. Sustain. Cities Soc. 65 https://doi.org/10.1016/j.scs.2020.102614.

Dunn, J.B., Gaines, L., Sullivan, J., Wang, M.Q., 2012. Impact of recycling on cradle-to-gate energy consumption and greenhouse gas emissions of automotive lithium-ion batteries. Environ. Sci. Technol. 46 (22), 12704–12710. https://doi.org/10.1021/es302420z.

Ecoinvent, 2021. Ecoinvent v3.8. https://ecoinvent.org/the-ecoinvent-database/data-releases/ecoinvent-3-8/.

Ellingsen, L.A.W., Majeau-Bettez, G., Singh, B., Srivastava, A.K., Valøen, L.O., Strømman, A.H., 2014. Life cycle assessment of a Lithium-ion battery vehicle pack. J. Ind. Ecol. 18 (1), 113–124. https://doi.org/10.1111/jiec.12072.

Ellingsen, L.A.W., Hung, C.R., Majeau-Bettez, G., Singh, B., Chen, Z., Whittingham, M.S., Strømman, A.H., 2016. Nanotechnology for environmentally sustainable electromobility. Nat. Nanotechnol. 11 (12), 1039–1051. https://doi.org/10.1038/nnano.2016.237.

Ellingsen, L.A.W., Hung, C.R., Strømman, A.H., 2017. Identifying key assumptions and differences in life cycle assessment studies of lithium-ion traction batteries with focus on greenhouse gas emissions. Transp. Res. Part D: Transp. Environ. 55, 82–90. https://doi.org/10.1016/j.trd.2017.06.028.

European Environment Agency, 2022a. Transport and environment report 2021: Decarbonising road transport - the role of vehicles, fuels and transport demand (issue 02). https://www.eea.europa.eu//publications/transport-and-environment-report-2021.

European Environment Agency, 2022b. Greenhouse gas emissions from transport in Europe. https://www.eea.europa.eu/ims/greenhouse-gas-emissions-from-transport.

Faraca, G., Martinez-Sanchez, V., Astrup, T.F., 2019. Environmental life cycle cost assessment: recycling of hard plastic waste collected at Danish recycling centres. Resour. Conserv. Recycl. 143, 299–309. https://doi.org/10.1016/j.resconrec.2019.01.014.
Finnish Meteorological Institute, 2022. Climate elements. https://en.ilmatieteenlaitos.fi/climate-elements.
García Sánchez, J.A., López Martínez, J.M., Lumbreras Martín, J., Flores Holgado, M.N., Aguilar Morales, H., 2013. Impact of Spanish electricity mix, over the period 2008-2030, on the life cycle energy consumption and GHG emissions of electric, hybrid diesel-electric, fuel cell hybrid and diesel Bus of the Madrid transportation system. Energy Convers. Manag. 74, 332–343. https://doi.org/10.1016/j.enconman.2013.05.023.
Glotz-Richter, M., Koch, H., 2016. Electrification of public transport in cities (horizon 2020 ELIPTIC project). Transp. Res. Proc. 14, 2614–2619. https://doi.org/10.1016/j.trpro.2016.05.416.
Göhlich, D., Fay, T.A., Jefferies, D., Lauth, E., Kunith, A., Zhang, X., 2018. Design of urban electric bus systems. Design Sci. 4 https://doi.org/10.1017/dsj.2018.10.
Grazieschi, G., Zubaryeva, A., Sparber, W., 2023. Energy and greenhouse gases life cycle assessment of electric and hydrogen buses: A real-world case study in Bolzano Italy. Energy Rep. 9, 6295–6310. https://doi.org/10.1016/j.egyr.2023.05.234.
Gustafsson, M., Svensson, N., Eklund, M., Fredriksson Möller, B., 2021. Well-to-wheel climate performance of gas and electric vehicles in Europe. Transp. Res. Part D: Transp. Environ. 97 https://doi.org/10.1016/j.trd.2021.102911.
Harris, A., Soban, D., Smyth, B.M., Best, R., 2018. Assessing life cycle impacts and the risk and uncertainty of alternative bus technologies. Renew. Sust. Energ. Rev. 97, 569–579. https://doi.org/10.1016/j.rser.2018.08.045.
He, X., Yu, D., 2020. Research trends in life cycle assessment research: A 20-year bibliometric analysis (1999–2018). Environ. Impact Assess. Rev. 85 (August), 106461 https://doi.org/10.1016/j.eiar.2020.106461.
Huber, D., Viere, T., Horschutz Nemoto, E., Jaroudi, I., Korbee, D., Fournier, G., 2022. Climate and environmental impacts of automated minibuses in future public transportation. Transp. Res. Part D: Transp. Environ. 102, 103160 https://doi.org/10.1016/j.trd.2021.103160.
IEA, 2022. Transport – improving the sustainability of passenger and freight transport. International Energy Agency. https://www.iea.org/topics/transport.
İnci, M., Büyük, M., Demir, M.H., İlbey, G., 2021. A review and research on fuel cell electric vehicles: topologies, power electronic converters, energy management methods, technical challenges, marketing and future aspects. Renew. Sust. Energ. Rev. 137 https://doi.org/10.1016/j.rser.2020.110648.
ISO, 2006a. ISO 14040: Environmental Management-Life Cycle Assessment-Principles and Framework. International Organization for Standardization.
ISO, 2006b. ISO 14044: Environmental Management - Life Cycle Assessment - Requirements and Guidelines. International Organization for Standardization.
Jakub, S., Adrian, L., Mieczysław, B., Ewelina, B., Katarzyna, Z., 2022. Life cycle assessment study on the public transport bus fleet electrification in the context of sustainable urban development strategy. https://doi.org/10.1016/j.scitotenv.2022.153872.
Jaramillo, P., Kahn Ribeiro, S., Newman, P., Dhar, S., Diemuodeke, O., Kajino, T., Lee, D., Nugroho, S., Ou, X., Hammer Strømman, A., Whitehead, J., 2022. Climate change 2022: mitigation of climate change. In: Transport, vol. 2022. https://doi.org/10.1017/9781009157926.012.
Johnson, C., Nobler, E., Eudy, L., Jeffers, M., 2020. Financial Analysis of Battery Electric Transit Buses. National Renewable Energy Laboratory, US Department of Energy, pp. 1–45. June. www.nrel.gov/publications.
Kempower, 2020. Volvo introduces the new Volvo 7900 Electric bus to Finland for a test tour. https://kempower.com/news/volvo-introduces-the-new-volvo-7900-electric-bus-to-finland-for-a-test-tour/.
Kim, H.C., Wallington, T.J., Arsenault, R., Bae, C., Ahn, S., Lee, J., 2016. Cradle-to-gate emissions from a commercial electric vehicle li-ion battery: A comparative analysis. Environ. Sci. Technol. 50 (14), 7715–7722. https://doi.org/10.1021/acs.est.6b00830.
Korhonen, J., Nuur, C., Feldmann, A., Birkie, S.E., 2018. Circular economy as an essentially contested concept. J. Clean. Prod. 175, 544–552. https://doi.org/10.1016/j.jclepro.2017.12.111.
Lajunen, A., Lipman, T., 2016. Lifecycle cost assessment and carbon dioxide emissions of diesel, natural gas, hybrid electric, fuel cell hybrid and electric transit buses. Energy 106, 329–342. https://doi.org/10.1016/j.energy.2016.03.075.
Łebkowski, A., 2019. Studies of energy consumption by a city bus powered by a hybrid energy storage system in variable road conditions. Energies 12 (5), 951. https://doi.org/10.3390/EN12050951.
Lemian, D., Bode, F., 2022. Battery-supercapacitor energy storage systems for electrical vehicles: A review. Energies 15 (15). https://doi.org/10.3390/en15155683.
Lim, J., Kang, M., Jung, C., 2019. Effect of national-level spatial distribution of cities on national transport CO2 emissions. Environ. Impact Assess. Rev. 77, 162–173. https://doi.org/10.1016/j.eiar.2019.04.006.
Motors, Chariot, 2021. 12 m ultracapacitor chariot e-bus. https://chariot-electricbus.com/cmproduct/12m-ultracapacitor-chariot-e-bus/.
Na, M., Isma, Y., 2013. Estimation of Bus stops spacing on public transport routes in Kano Metropolis using minibus stop time interval. Int. J. Eng. Sci. Invent. 2 (9), 36–44. www.ijesi.org.
Nordelöf, A., Romare, M., Tivander, J., 2019. Life cycle assessment of city buses powered by electricity, hydrogenated vegetable oil or diesel. Transp. Res. Part D: Transp. Environ. 75, 211–222. https://doi.org/10.1016/j.trd.2019.08.019.
Onat, N.C., Kucukvar, M., 2022. A systematic review on sustainability assessment of electric vehicles: knowledge gaps and future perspectives. Environ. Impact Assess. Rev. 97 https://doi.org/10.1016/j.eiar.2022.106867.
OpenLCA, 2022. https://www.openlca.org/openlca/new/.
Pei, H., Hu, X., Yang, Y., Tang, X., Hou, C., Cao, D., 2018. Configuration optimization for improving fuel efficiency of power split hybrid powertrains with a single planetary gear. Appl. Energy 214, 103–116. https://doi.org/10.1016/j.apenergy.2018.01.070.
Saft, 2015. Evolion Li-ion battery technical manual. https://www.saftbatteries.com/emailing/evolion/doc/Evolion_TM_en_0115.pdf.
de Souza, L.L.P., Lora, E.E.S., Palacio, J.C.E., Rocha, M.H., Renó, M.L.G., Venturini, O.J., 2018. Comparative environmental life cycle assessment of conventional vehicles with different fuel options, plug-in hybrid and electric vehicles for a sustainable transportation system in Brazil. J. Clean. Prod. 203, 444–468. https://doi.org/10.1016/j.jclepro.2018.08.236.
Sun, R., Chen, Y., Dubey, A., Pugliese, P., 2021. Hybrid electric buses fuel consumption prediction based on real-world driving data. Transp. Res. Part D: Transp. Environ. 91 https://doi.org/10.1016/j.trd.2020.102637.
Sustainable Bus, 2021. A pilot in Turin for the SmartBUS, the e-bus with supercaps in place of batteries. https://www.sustainable-bus.com/electric-bus/smartbus-turin-supercapacitors/.
Sustainable Bus, 2022. Sofia and Belgrade to operate Europe's first 18m ultracapacitor e-bus. https://www.sustainable-bus.com/news/sofia-belgrade-first-18m-ultracapacitor-e-bus/.
Todorov, Y., 2021. Successful electrification of city's transport system. VTT. https://www.vttresearch.com/en/news-and-ideas/successful-electrification-citys-transport-system-simulation-guiding-road-future.
U.S. Department of Transportation, 2023. Transitioning to zero-emission bus operation. www.transportation.gov/Momentum.
Volvo, 2020. New alternative for charging electric buses. https://www.volvobuses.com/en/news/2020/oct/volvo-buses-presents-new-alternative-for-charging-electric-buses.html.
Volvo, 2021. How to make a smooth transition to electric mobility. https://www.volvobuses.com/content/dam/volvo-buses/markets/master/insights/how-to-make-a-smooth-transition-to-electric-mobility/MobilityInsightsbyVolvoBuses_Howtomakeasmoothtransitiontoelectricmobility_April 2021.pdf.
Wappelhorst, S., Rodríguez, F., 2021. Decarbonizing bus fleets: global overview of targets for phasing out combustion engine vehicles. International Council on Clean Transportation. ICCT. https://theicct.org/blog/staff/hdv-fleets-phase-out-targets-dec21.
Wieczorek, M., Lewandowski, M., Jefimowski, W., 2019. Cost comparison of different configurations of a hybrid energy storage system with battery-only and supercapacitor-only storage in an electric city bus. Bull. Polish Acad. Sci. Tech. Sci. 67 (6) https://doi.org/10.24425/bpasts.2019.131567.
Wimbadi, R.W., Djalante, R., Mori, A., 2021. Urban experiments with public transport for low carbon mobility transitions in cities: A systematic literature review (1990–2020). Sustain. Cities Soc. 72 https://doi.org/10.1016/j.scs.2021.103023.
Zhang, F., Yang, F., Xue, D., Cai, Y., 2019. Optimization of compound power split configurations in PHEV bus for fuel consumption and battery degradation decreasing. Energy 169, 937–957. https://doi.org/10.1016/j.energy.2018.12.059.
Zhao, E., Walker, P.D., Surawski, N.C., Bennett, N.S., 2021a. Assessing the life cycle cumulative energy demand and greenhouse gas emissions of lithium-ion batteries. J. Energy Storage 43, 103193. https://doi.org/10.1016/j.est.2021.103193.
Zhao, J., Xi, X., Na, Q., Wang, S., Kadry, S.N., Kumar, P.M., 2021b. The technological innovation of hybrid and plug-in electric vehicles for environment carbon pollution control. Environ. Impact Assess. Rev. 86 https://doi.org/10.1016/j.eiar.2020.106506.